Ecosystem Services and the Political Economy of Watershed Governance

by
Matthew Retallack

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School of Public Policy and Administration
Carleton University

Ottawa, Ontario

Abstract

Water is a multi-use resource, with governance being shaped by a range of interacting institutional and economic imperatives. These many uses comprise a joint system where water and economy are linked by varying degrees of dependency on water-based ecosystem services. The production of ecosystem services is contingent on underlying ecological processes. However these processes can be affected by our actions, creating a sustainability dilemma. Institutions are in place to manage the impacts of our actions; however, institutions are subject to a range of pressures including actor preferences, historical factors, socio-cultural narratives, the influence of ideas and experts, and political context. Thus, we have multiple co-existing and competing resource regimes, drawing on an underlying resource system (water), across a shared landscape, with interrelated institutional mechanisms that are shaped by an array of factors. Moreover, water resources governance is highly normative as multiple actors engage in political contestation and seek to privilege their individual interests within institutional outcomes.

Through three separate but interrelated studies, my thesis argues that more attention needs to be paid to the values and normative dimensions that underlie water governance. The first study develops a decision-support tool that helps planners understand the critical linkages between economic activities and ecological factors at the watershed level. The analysis uses Ontario's Mississippi Valley as an illustrative case. The second study draws on natural resource economics and models of participatory governance in order to examine how cultural values affect water management in the case of Chelsea, Quebec. The third study treats the Ontario *Clean Water Act* (2006) as a case study. It traces actor involvement in the development of the legislation, examining subsequent institutional changes, and how

these changes were then interpreted and transformed by decentralised planning boards that were empowered to develop rules to control activities within local watersheds.

Together, these analyses demonstrate how system effects combined with actor-institution interactions can lead to a range of social, environmental and economic risks. Conceptualising ecosystem services as a boundary object, the concept is used to examine how the interrelationships between social and ecological subsystems condition the challenges and outcomes of water resources governance. A systems-based approach and broad institutionalist analysis, are used to define the main contours of the evolving political economy of water governance. A novel environment-economy model and empirical case study demonstrate how exercising property rights in one location may impinge on the ability of others elsewhere to do the same. This results in a degree of conceptual slippage such that private property may participate in what amount to open access conditions. Issues of fit may exacerbate these effects where local political agendas do not consider wider ecological implications. Tensions between competing values and normative dimensions of water governance were observed to contribute to a range of social, environmental and economic outcomes. This feeds into a critical discussion about the role of watersheds and watershed communities in water resources governance. Local input and participation in governance processes is valuable however this has to be balanced against concerns around capture and the privileging of selected ecosystem services at the expense of broader ecological health and social-ecological resilience.

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As an engineer who has transitioned into the social sciences I am struck by the nature of the work. Certainly the work of engineers involves significant human consequences, both positive and negative, potentially even fatal. However these concerns are generally in the background. In the social sciences the human face is often front and centre. The policy questions we ask affect real people's lives, and so it was with my research. The locations of my case studies are all places I have visited. In the case of Chelsea we even lived there for a while. Thus the meaning of the work is not abstract but rather direct and visceral. Beyond a general appreciation for the faces and places that are interwoven through my doctoral research, I would extend particular gratitude to those who participated in this work directly, giving of their time to share their story. All of these individuals were engaged with making their communities better, either in a professional capacity or as engaged citizens. That they chose to give even more of themselves, to share their experiences so as to advance the wider project of human progress, is humbling and gratefully received.

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Chapter 1 – Introduction

Water is essential for life. It surrounds us. Its presence, or absence, shapes the living and non-living landscapes that constitute our environment. Historically, we have settled on shorelines and river banks, drawing water to drink and to grow the food we need, to produce goods, and to enable trade with others elsewhere. Over time our societies have become more complex. Our uses of water have increased in number as our economy and the technologies that drive it have grown in size and complexity. Societies have formed increasingly sophisticated systems of government, which in turn have sought to resolve disputes and collective action dilemmas that relate to our expanding use of water in an effort to advance social and economic development. The management of water resources has been subsumed by governments and put to work for the causes of development (Huitema & Meijerink, 2014). Great gains have been made. Industry and society flourished in many parts of the world. However the cumulative effects of our growing population, increasing wealth, and burgeoning technology have increased the pressure on the environments that sustain us (Ehrlich & Ehrlich, 1981), leading to a general level of concern over the sustainability of our water resources.

This did not happen all at once, and over time an institutional framework and depth of knowledge has developed to help understand and protect these precious resources. Nevertheless, even with increasing knowledge and institutional safeguards we still struggle to manage water resources sustainably. It is paradoxical. What are the forces that underlie these management processes? How do we understand and value our water resources relative to other concerns? The basic challenge is simple. We draw on water in various ways to support numerous social and economic activities, and in doing so we can also

undermine the health of the water resources and ecosystems that provide those very services. While there are technical challenges involved with untangling and understanding the many interwoven ecological and economic factors, striking a balance and finding the appropriate level and type of use is ultimately rooted in social values, implying normative dimensions beyond the simple technical facts. It becomes a moral question of what is right and how we *ought* to manage water. There are both technical and democratic dimensions and these have to be balanced both internally and against one another.

In Canada, the Constitution states that the manner in which water is used as a resource falls under provincial jurisdiction. Many other uses of water, such as habitat for fish or as a medium for transport, are federal concerns, and both levels of government have a stake in protecting and managing the environment, broadly speaking, within which water constitutes a key and cross-cutting concern. Municipal governments do not have a constitutionally defined mandate but do overlap with water resources governance through municipal functions such as land-use planning. Municipal water and wastewater services are regulated by provincial legislation (Hill et al., 2008). Following largely from these divisions of mandate, water governance in Canada comprises a patchwork of federal and provincial policies (Bakker, 2007). It follows that many government actors have a stake in water governance, as do water users across the spectrum of different uses. Johns et al. (2008) identify the most significant uses as commercial fishing, commercial shipping, hydroelectricity, forestry and pulp and paper, mining and resource extraction, municipal water use, wastewater disposal, agricultural and irrigation uses, the bottled water industry, recreational water use, and cultural and spiritual water uses. Defining uses of water in this way serves to make a straightforward but important conceptual point: while social and

economic activities depend on environmental inputs, these relationships are mediated by policy and governance systems. Three general statements follow from this conceptual point. First, all of these factors are interconnected in a systems sense: ecological and socio-economic systems are internally coherent and woven together through these various uses. Second, we define these relationships from a societal perspective, and policy is focussed on social activities rather than the underlying ecological supports. Third, and following the previous two points, there may be key connections within this larger system of interrelated elements that are critical for understanding system dynamics and observable social, economic and environmental outcomes.

As a multi-use resource, conflict over water is possible as a result of scarcity (Schindler & Donahue, 2006), but also as a result of institutional failure and the larger social, political and economic forces that structure power relationships and fundamentally shape resource governance (Johns, 2008a). Paradigms are prone to shift when anomalies and failures raise questions about core principles and theories (Khun, 1970), ushering in a period of innovation. Bakker (2007) suggests that systemic weaknesses in Canadian water governance have brought it to the point of potential crisis, and that weaknesses need to be exposed and challenges confronted. The time for innovation and new approaches to water governance has come.

My research seeks to contribute to innovation on water resources governance by employing the concept of ecosystem services - defined as the benefits provided to people by the environment (Fisher et al., 2009) - as an analytical lens to better understand technical and institutional aspects of how social and ecological systems are linked. Specifically, and as I explain further below, I ask how does demand for the services water generates through

its role in ecosystems affect water governance? Normatively, I also ask whether integrated approaches to water management that employee a broader conception of ecosystem services can help enhance attention to the ecological integrity of ecosystems, and associated socio-economic implications. The main aim of taking this analytic approach is to offer a new perspective on water resources governance; an ecosystem approach has not been given sufficient attention to date, which is one contribution my dissertation seeks to make. A secondary aim is to contribute to the literature on ecosystem services. Through these services, we can trace direct linkages between our socio-economic and cultural activities and the environment, and we can describe a degree of dependence these human social and cultural systems have on the environment. As such they provide an ideal unit of analysis for exploring the paradox that lies between an increased understanding of the threats, on the one hand, and an increasingly threatened resource, on the other.

My research findings show that competition over access to needed ecosystem services is linked to the formation of actor strategies, and that patterns of demand across shared landscapes are useful for tracing how factors within the political economy affect institutional development and implementation in the context of water resources governance. In this way, ecosystem services are not a simple one-way flow of benefits from ecological to social systems but are instead the subject of political contestation, the results of which shape ecological outcomes and the subsequent availability of ecosystem services. In addition to economic motivations, cultural values can also affect actor strategies, and by extension the relative value of different ecosystem services. Taken together these findings challenge current approaches to water governance. Political contestation between actors and coalitions of varying power and influence underscores the importance of understanding

water governance in a systems context. In this broader scope "water problems" and not just about water and new actors may come into focus, for instance when their interests are not water-specific but are affected in some manner by water governance outcomes. This in turn challenges conventional notions about the appropriateness of watersheds as a basis for water resources governance. Ecosystem services were found to be of instrumental value for untangling the web of interrelationships between social and ecological systems in order to identify causal mechanisms and explain institutional outcomes.

The following section will discuss the history and evolution of ecosystem services as a concept, providing the reader with some conceptual guideposts in preparation for subsequent application of the concept. This will be followed by a brief discussion of theories falling within the domain of "new institutionalism" alongside a review of several key institutionalist theories focussed on natural resource management. These two areas of scholarship will then be brought together with a brief discussion of systems theory and perspectives on managing joint social and ecological systems. Finally, all of these elements will be brought together in a conceptual framework that provides the rationale and theoretical orientation for three individual research studies. This introductory chapter will close with a brief introduction and discussion of these three studies.

1.1 Ecosystem Services

Originally a pedagogical concept aiming to compel society toward a new relationship with the natural environment, ecosystem services have been instrumental for disaggregating society's dependence on nature and organising it in a discrete manner, thereby identifying how specific activities draw on specific services. The central concept of ecosystem

services, that human welfare is interrelated with the natural environment, is ancient in its origins. For the last 10,000 years humans have gone from taking shelter from nature to drawing on and manipulating their environmental surroundings to increase production and improve their lives (Ehrlich & Ehrlich, 1981). Plato connected deforestation with soil erosion and the drying up of springs, and in doing so captured the relationship between ecological change and societal interests, as well as human capacity to introduce such self-defeating changes (Mooney & Ehrlich, 1997).

In 1864, George Perkins Marsh renewed these ideas for modern times in his prescient book, *Man and Nature*. Our dependence on nature and its interconnectivity was "evident" for Marsh, and it was "equally obvious that all agricultural and all industrial operations tend to disturb the natural arrangements of this element" (Mooney & Ehrlich, 1997). Tansley (1935), coined the term "ecosystem" to encompass both the biological community and the abiotic conditions from which they could not be separated but instead form one physical system. Nevertheless, while humans have long understood their dependence on nature and have made limited efforts to manage pollution and conserve select wild spaces, opportunities for industrial activity and economic development have been the dominant preoccupation of society. This changed with the publication of Rachel Carson's *Silent Spring* in 1962, which ushered in the dawn of the modern environmental era and triggered a concern over ecosystem health. This prompted large-scale assessments such as the International Biological Program in the late 1960s, and the Study of Critical Environmental Problems in 1970. Ecosystems were found to be in jeopardy and linkages were drawn between impacts on ecosystem functions and a decline in the level of benefits available to society.

While the original concept was largely pedagogical in its orientation, conceived to convey the importance of conservation and biodiversity, "ecosystem services" was confronted by a society that was deeply embedded in global economic concerns. Over time the language of conservation gave way to an increasingly utilitarian approach. Schumacher (1973) is credited with first characterising ecosystems as containing a stock of "natural capital" from which benefits flow to society. The intention was to raise awareness and concern around the indiscriminate use of non-renewable resources. However, framing environment and human economy as a stock-flow scenario made it possible to discuss ecosystem change in terms of spent capital, where a loss of capital equated readily with a reduction in the flow of related benefits. In 1977, Westman suggested that "nature's services" could be enumerated in order to inform policy and management (Fisher et al., 2009). By moving from abstract concepts to more concrete characterisations of the interrelationships between society and the environment, ecosystem services were made policy relevant. Conceptually this amounted to a shift from a conservation metaphor to an economic model (Norgaard, 2010).

As the concept moved closer to policy circles and questions were being asked about how it could be applied and affect decision making, the source, language and composition of these questions were informed by dominant thinking of the time. Environmental economics, operating within a neoclassical framework, sought to address market failures by incorporating environmental externalities. Ecosystem services that delivered discrete use and non-use benefits could be individually valued through standard economic methods and aggregated to arrive at a Total Economic Value (Gómez-Baggethun et al., 2010). Ecosystem services had entered the economic lexicon. The evolution of the ecosystem

services concept toward increasing integration with policy, primarily through monetisation, has given rise to a series of concerns. While monetisation has enabled the concept to gain traction in the policy arena, it does little to challenge the values and objectives of society that are embedded within decision making, and the dominant economic paradigm is left largely intact. Supporting an economic model whose success is predicated on the assumption that people "will never be their better selves, but always be greedy social idiots" (Schumacher, 1973), is arguably out of step with the original formulation of the concept, tied as it was to the fundamental importance of maintaining biodiversity by acknowledging limits and conserving natural capital. Furthermore, integration of ecosystem services into utilitarian models that seek only to improve aggregate welfare, and not equitable distribution, do not ensure that social equity dimensions of ecosystem services, such as access, availability and preservation, are adequately addressed in the present day or in the long term.

Concerns over the long-term implications of monetary valuation overlap with the debate on sustainable development. Here there are two main conceptualisations. Strong and weak sustainability both have regard for intergenerational fairness, and both encompass natural capital. However the manner in which they do so leads to different policy conclusions. Weak sustainability calls for the maintenance of total capital stock in order to ensure non-decreasing welfare for future generations. Implicit in this is the substitutability of human and natural capital. Natural capital which is used to produce human capital of an equal or greater amount has not reduced total capital and thus is consistent with sustainability (Common & Stagl, 2005). Those who align with strong sustainability call for separate consideration of natural capital and human capital, generally

arguing that they are complements, not substitutes (Daly, 1996). Arguably neither position can be fully observed. There are ecological features for which there is no substitute (Dobson, 1998), and total preservation of aggregate natural capital would seriously curtail use, severely challenging the practicality of this approach (Simpson et al., 2005). Those seeking middle ground raise the notion of critical natural capital: natural systems and features that are considered vital for human wellbeing, and for which there is zero substitutability. However what exactly constitutes critical natural capital is not well defined. An important next step will be to disaggregate the abstract notion of "natural capital" to separate out critical natural capital from that which may be converted into human capital without significant repercussions for future generations (Neumayer, 2010).

There are other more pragmatic, technical limitations to monetising ecosystem services for the integration of environmental values within policymaking and the analysis of costs and benefits. The Millennium Ecosystem Assessment (MEA, 2005) provided a highly cited and commonly deployed typology for different categories of ecosystem services (see Appendix A). While there are some debates around aspects of how ecosystem services are defined and classified (deGroot et al., 2002; Fisher et al., 2009; Landers & Nahlik, 2013), it is nevertheless much easier to assign a monetary value to some ecosystem services (e.g. provisioning services that have a market price, such as food crops), than to others (e.g. cultural services, such as sense of place, for which values are differentiated across society). Ecosystems themselves are complex and non-linear, and there are real thresholds that, once crossed, lead to new ecological arrangements (Kremen, 2005). Utilitarian economic thinking that reduces complexities in order to optimise economic efficiency does so, potentially, at its own peril. Monetary approaches may not capture the

value of non-market services including the primary services that sustain the ecosystems themselves, thereby supporting the ongoing provision of secondary, or final ecosystem services (Jenerette et al., 2006, Grey et al., 2003). Over time, the erosion of ecological underpinnings can negatively impact the provision of ecosystem services as well as undermine social-ecological resilience and environmental sustainability.

Finally, in addition to being complex systems that are typified by non-linear response to disturbance, ecosystems are also idiosyncratic and vary in composition and function from location to location (Chapin III et al., 2000). As such the supply, and hence relative scarcity, of different ecosystem services changes from one setting to the next, depending on the underlying ecological factors and how these interact with the broader ecosystem. It is therefore important to understand the mechanisms and extent to which ecosystem services are locally produced (Kremen, 2005).

These fundamental shortcomings have prompted closer scrutiny around the types of valuation that may be undertaken, including ecological, socio-cultural and ethical valuation, in addition to economic valuation (Hackbart et al., 2017). This includes renewed efforts to reconnect social values with the valuation process (Costanza, 2001; Farber et al., 2002). Centred on notions concerning the nature of "value", non-monetary approaches to valuation of ecosystem services seek to involve society in the valuation process. Where the value of something is a function of its contribution to user specified goals, such approaches can inform social actors about the values-perspectives of others, and associated underlying needs. As a result, this can bring attention to social equity dimensions, requiring reconsideration of goals and objectives in light of these factors.

The challenges associated with valuing ecosystem services illustrate how the

relationship between social and ecological systems is more reflexive than a simple one-way flow of benefits from the environment to society. Indeed the relative value of different benefits is in part a social construction. Policy choices in either domain can have an impact in the other, and where each system is to some degree determinant of conditions in the other, feedback can continue to reverberate within this joint system.

Often policy instruments do not have the provision and use of ecosystem services as an explicit goal. However they may nevertheless work toward ecosystem services outcomes. For instance, pollution control mediates the production of numerous ecosystem services while regulations on water allocation, or licensing instruments such as catch limits, effectively manage access to the benefits of ecosystem services. My research is framed relative to this argument, with the ecosystem services of interest being those that are embedded within the policies and policy processes that are the focus of my empirical case studies (see Section 1.5). I seek to interpret how existing policy systems and actors work to protect and privilege certain ecosystem services over others, and illuminate how ecosystem services might be undermined by a given course of action, thereby introducing secondary effects.

Furthermore, the use of water by humans is not confined to extraction or consumption, or use as a receiving medium for various waste effluents. In an ecological sense, water is a core structural and functional component that is essential to ecological processes, including the production of a wide range of resources. These resources, alongside various uses of liquid water, comprise the water-based ecosystem services of interest to people. It follows that the governance of water resources can potentially intersect a large number of ecosystem services that are of direct interest to humans, prompting a

response from affected actors. Clarifying these ecosystem service-based interests provides an important perspective on political contestation that may be manifest within governance processes. More importantly, as actors seek to preserve ecosystem services of direct interest they can shape the underlying ecology, having a deterministic effect on the set of future options in terms of available ecosystem services, and the relative resilience of ecological systems. Scholars argue that a failure to account for these linkages can result in a loss of resilience, which in turn undermines the sustainability of society going forward (Berkes et al., 2008). Generally speaking, the focus of the empirical chapters is on ecosystem services that provide benefits to people, as these are assumed to most directly influence the motivation and strategies of involved stakeholders and governance actors. However the overarching purpose of identifying and analysing the effect of these interests is not necessarily to increase the production of ecosystem services of human interest, but rather to diagnose how existing demands can shape institutions in ways that run counter to ecological health and resilience. From a systems perspective, ecological, social and economic conditions are co-dependent. A failing ecological subsystem, characterised by a loss of resilience or transcendence across some irreversible ecological threshold, can put social and economic wellbeing at risk. Ecological change is thereby of material interest to the identification and management of social and economic risk. The three empirical studies will be introduced in Section 1.5 at which point the specific ecosystem services of interest in each study will be discussed in more detail. Individually, the three studies examine: 1) the ecological and governance implications of shared landscapes populated by a range of economic actors, inclusive of upstream-downstream considerations, 2) the importance of accounting for cultural values within policy processes that have ecological implications,

and, 3) the manner in which institutions are determinant of, but also shaped by, actor effects in the context of political contestation over access to ecosystem services. Taken together these three studies speak to the importance of geo-spatial, socio-economic and ecological context, and the influence of policy processes and institutional design, for natural resource governance. The next section will briefly outline current theory as it pertains to processes of institutional development and change, with a particular focus on natural resources governance.

1.2 Institutions

Institutions can be both the outcome of governance processes in a given socio-economic, cultural and historical context, as well as the means through which we govern our actions. Institutions are generally taken to comprise a set of interrelated and contingent rules; a framework of self-replicating social practices rather than discrete, semi-permanent agencies (North, 1990). Institutions include policies but are not limited to policies, instead comprising both formal and informal rules relative to which actors develop their strategies (Thelen, 2004). Institutions constrain and permit certain behaviours, and as a result there are winners (those whose interests are privileged by the institutions), and losers (those whose interests are not) (March & Olsen, 2006). Because of this functionality, institutions are highly contested with actors forming coalitions and squaring off with rival interests (Thelen & Steinmo, 1992). Power and political influence are often used to pursue institutional objectives. However, institutions are ultimately socially sanctioned with a level of adherence constituting a key source of legitimacy (Streeck & Thelen, 2005). Institutions are not static but instead are subject to change by a number of means, sometimes incrementally and sometimes more dramatically (Baumgartner & Jones, 1993;

Mahoney & Thelen, 2010). These changes can be the result of actor effects however institutions are also subject to the force of history and prevailing social beliefs (Hall, 2010). This research is concerned with institutions as they pertain to the management of the environment and natural resources, specifically water resources, and the interplay between these institutions and the network of interrelationships within and between the natural world and the social and economic order.

The management of water resources is unavoidably complex. This is because water is not simply a resource but readily fits Ostrom's (1990) definition of a resource system: it is the resource base from which numerous discrete resources are produced. Each resource has its own governance regime, including stakeholders, governing agencies and institutions. Young (2002a) identified three core components of institutional design that contribute to the rise of observable environmental challenges: fit, interplay and scale. These have particular relevance for water resources governance.

Issues of institutional "fit" occur when biophysical systems are geographically incongruent with the administrative jurisdictions by which they are governed (Young, 2002a). It is because of issues of fit that watersheds are commonly held up as ideal units within which to manage water resources. Comprising a quasi-closed system, watersheds enable the integration of water flows, uses and impacts in a relatively comprehensive manner (Rahaman & Varis, 2005; Warner et al., 2008). In addition to issues of spatial fit, natural and human systems may operate at different temporal scales. As a result, institutional response may be of an inappropriate speed and duration to comprehend and effectively manage changing ecosystems (Vatn & Vedeld, 2012).

Institutional interplay occurs when separate institutions have overlapping or

interacting elements within the broader undertaking of managing human-environment interactions, such that one institution may affect the circumstances of another through normal operation. Interplay can be horizontal across the same level of social organisation, or vertical between levels of government, and it can be either functional or political in nature (Young, 2002a). Dependence on a common resource system means that some degree of interplay between institutions governing different issue areas that draw on water-based ecosystem services is essentially unavoidable. Political interplay is understood to result from deliberate action to link institutions in order to achieve some defined objective.

The final element in Young's framework, scale, acknowledges that the dynamics of ecological systems may be similar enough to allow scaling up (Young, 2002b). The nesting of watersheds, subwatersheds and sub-subwatersheds provides one example. Hydrological processes and associated ecological functions are scalable across these different levels. However governance institutions are generally more sophisticated at higher levels of social organisation in terms of processes and agencies, as well as human and other resources (Vatn & Vedeld, 2012). This results in varying capacities to manage issues across scale.

There are a range of perspectives on how these elements fit together and this has implications for policy design. For Young, institutions are largely an independent variable, with institutional design being an important determinant of environmental performance. Close examination of institutional arrangements can thereby help explain negative environment outcomes (Young 2002a). Others question the degree to which institutions are decisive in bringing about environmental outcomes relative to the extent to which they reflect underlying power structures (Mann et al., 2015). A third perspective takes the more

reflexive position that the institutions themselves need to be adaptive, and that they need to respond to and in some sense be managed by environmental conditions rather than the other way around (Folke et al., 2007).

Institutions help manage collective action problems that would otherwise continue to work against our shared interests (Olson, 1971). While this is a well substantiated theoretical perspective there are several factors that can lead to sub-optimal outcomes. Those involved may seek to advance their specific interests above all else, for instance seeking to privilege their individual interests during processes of institutional design. The historical legacy of similar actor-institution interactions can result in a degree of embedded institutional bias toward the preferences of earlier governance arrangements. Furthermore, institutional arrangements and issues of fit, interplay and scale, may lead to negative environmental outcomes (Young, 2002a). Additional insight into factors underlying observable outcomes may be gained through examination of policy development processes. Extending Hall's (1993) work on levels of policy change, Cashore & Howlett (2007) argue that every policy is a mix of means and ends, and that disaggregating policy into its component parts can help identify which variables are deterministic of observable outcomes. Finally, once institutions are formed they are not static fixtures but are subject to a range of forces. The manner in which institutions are interpreted, observed and enforced also matters (Mahoney and Thelen, 2010; Streeck & Thelen, 2005), with processes of institutional change potentially underlying observable environmental outcomes.

My research is concerned with linkages between institutions and the outcomes of water resources governance, including the effects of different variables on the institutions

themselves. Examining institutional design in terms of process and structure is complemented by an analysis of function and implementation. Accordingly, I am focussed on the relative balance between institutions as tools of governance and institutions as outcomes of governance processes involving numerous actors of varying power, resource and influence. The relevant theoretical perspectives discussed in this section have a common appreciation of the importance of history and are rooted in historical institutionalism. Nevertheless, while historical context is vital, I follow Hall's (2010) arguments that these considerations on their own are not sufficient to explain institutional outcomes. While actors develop strategies relative to institutional structures that have taken shape over time, they do so in conjunction with expectations of how others will act. Furthermore, it is also important to consider how symbols and interpretation influence the decisions of these actors (Hall & Taylor, 1996). These themes are woven through my research design and analysis.

1.3 Social Ecological Systems, Systems-Thinking and Resilience

Ecological structure, such as the distribution of species or abiotic environmental conditions, determine ecological functions, for example growth rates or migration patterns. These in turn underlie the provision of ecosystem services (Kremen, 2005). When ecological structures are under pressure there is the potential for impacts on the provision of ecosystem services. Moreover, ecosystems are linked across scale such that effects are not isolated within any given ecological subsystem. Approaches that seek to optimise production often privilege some ecological processes by controlling others (Walker & Salt, 2006). This approach ignores natural variation and can result in overuse and reduced biodiversity. This in turn undermines redundancy of ecological function (Schindler et al.,

2010). Where ecosystem response is often non-linear, potentially irreversible and may involve transcending ecological thresholds, such approaches can, in extreme cases, lead to ecological collapse, as observed with the Atlantic cod fishery (Folke et al., 2007). The resulting impact on society can be devastating (Diamond, 2005). The interrelationships between ecological factors, ecosystem services, and human beneficiaries of these ecosystem services, weave together social and ecological subsystems into a larger, joint, social-ecological system (SES). In this introduction I will focus on two specific approaches to understanding and managing SES: resilience scholarship on the one hand, and Elinor Ostrom's research on institutional dimensions of SES on the other. Ostrom's approach will be discussed next, leading into a brief discussion of systems theory. Drawing on resilience scholarship I then use systems theory to challenge one of Ostrom's core assumptions, providing an alternative approach as the basis for my conceptual framework (Section 1.4).

Due to the large number of often competing interests within a SES there exists a need for institutions to manage this competition. These include institutions governing rights and entitlements for use, called property rights. Ostrom's work on common pool resource management led to a number of design principles addressing collective-choice arrangements, rights entitlements, and monitoring, compliance and conflict-resolution mechanisms (Ostrom, 1990). Over time the theory was expanded to encompass multiple resources regimes and associated institutional interplay. Recognising that feedback at a systems level can result in largely separate governance systems exerting some effect on one another, Ostrom (2007) conceptualised individual resource regimes - the institutional structures that pertain to a single resource - as being linked in a systems sense, with myriad feedback mechanisms between society and the environment. Resource governance was

positioned within a larger social-ecological system, and the interactions between resource regimes came into focus (Ostrom, 2009). The potential for these linkages to introduce changes across the system motivated a need to embrace complexity and understand institutions in a systems context. This is the same basic argument that motivates my research. However using ecosystem services as a lens through which to examine SES illuminates a key distinction within this basic argument, in terms of the manner in which SES are conceptualised.

In order to elaborate this distinction it is helpful to have some conceptual tools, and for these I turn to complex systems theory. The relationships between elements are the focus of complex systems theory and analysis (van Gigch, 1978). Rejecting reductionist approaches, this theory seeks to simplify complexity by identifying elements, characterising the strength of the relationships between them in terms of the transfer of mass, energy and information, and then organising these elements within hierarchies. A system is comprised of a number of subsystems, with the components of individual subsystems interacting more intensely with one another than with components in other subsystems (Simon, 1962). Natural systems are all open systems, meaning that a given system will relate, exchange and communicate to some degree with the other systems that comprise its surroundings. Establishing the system boundary is guided both by the defined goal of the system, and by the nature of the relationships between its component parts. These criteria determine which elements are included within the system, and which are determined to be minimally connected, and may therefore be held constant for the purposes of the analysis (Limburg et al., 2002).

Applying complex systems theory to Ostrom's work, I find that her account of SES

are appropriately conceptualised as open systems, comprising interactions between users, resource systems, resource units and political settings. These core subsystems are located inside the system boundary. Related ecosystems and external social, economic and political settings are shown as connected but outside the system boundary for the purposes of analysing the SES (see Figure 1.1). In this conceptualisation, resource systems are defined as the "facility or stock that generates the flow of resource units" (Dietz et al., 2002). In other words they are defined relative to the resource units, which are the substantive focus of the institutional arrangements and associated users with a given SES. There is some confusion within Ostrom's work about how resource systems are defined. For instance, watersheds are identified as resource systems but so are fishery grounds (Ostrom, 2009). The relevant distinction here is that watersheds are a natural phenomenon whereas fishery grounds are arguably a human construct. Fish are present in nature but the delineation of a fishery ground is made relative to human use. Elsewhere Ostrom writes, "[r]esource systems are best thought of as stock variables that are capable, under favourable conditions, of producing a maximum quantity of a flow variable without harming the stock or the resource system itself" (Ostrom, 1990). Again, the flow of resource units is used to define the system of interest, with system sustainability being one consideration for institutions that are primarily focussed on (maximising) production of the flow variable.

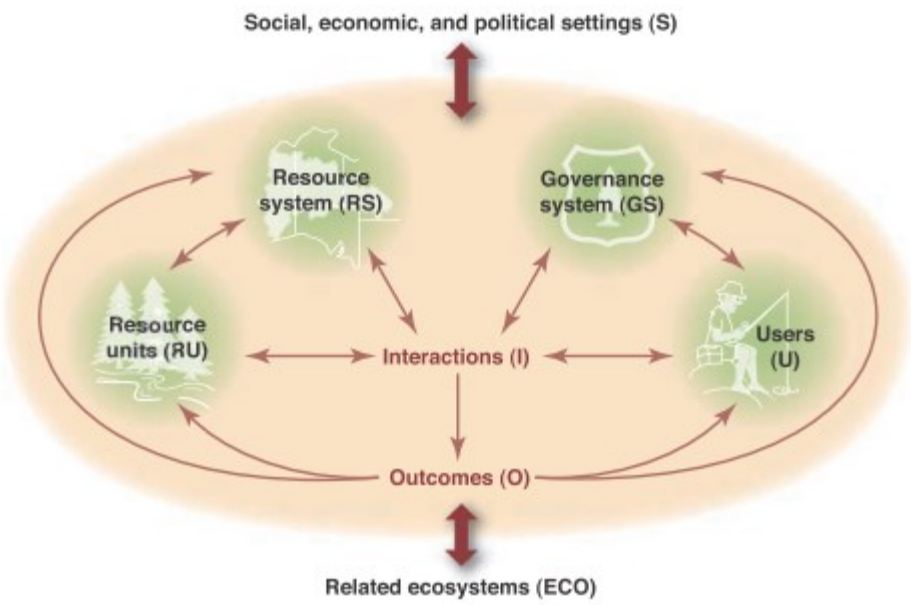


Figure 1.1 – The core subsystems for analysing social-ecological systems (Ostrom, 2009)

Ostrom's conceptual model appears to have an "institutions first," user-centric orientation. At its core are the resource units and the institutions needed to extract value from those resource units. This is not surprising given that this work is an outgrowth of her work on institutions for common pool resource management. Further, it is consistent with Young (2002b), who highlights this importance of examining the interactions between diverse, multilevel institutions, for understanding environmental outcomes. Resilience scholars are also concerned about the linkages between social and ecological systems. However they are fundamentally orientated relative to ecological imperatives. From this perspective SES are seen to encompass multiple ecological scales (Walker et al., 2006), with linkages between scales and sectors often driving changes in the particular (resource) subsystems being managed (Walker & Salt, 2006). Systems are understood as complex and adaptive (Folke et al., 2007), with system resilience being a function of diversity and the prospects for "mutualism, recombination, and processes at higher levels of organisation

that allow genomic lines of decent to adapt to an ever-changing environment" (Levin, 1999). These perspectives make two key points: ecosystems cannot be managed in isolation, and resource management approaches that seek to maximise the sustainable yield of a given resource do so with a degree of disregard for ecological realities that could result in a loss of social-ecological resilience (Walker et al., 2006).

My research draws on systems theory and conceptualises resource systems as overlapping at an ecological level such that ecosystems related to a given resource system (identified as ECO in Figure 1.1), are not external to the SES but are woven together in a common ecological system. In this way the main linkages are the overlapping resource systems. Conceptualised as subsets of ecological structure and function, multiple interrelated resource systems give rise to units of multiple resource types. Institutions and governance processes associated with these resource systems, and their accompanying resource units, are interrelated at an ecological level. In this way, while I do not dispute that institutional factors can shape environmental outcomes, I would add that ecological factors can similarly induce institutional outcomes. Examining these mechanisms is a core objective of this work.

1.4 Conceptual framework

This section draws together the core propositions from the preceding three sections to develop a conceptual framework within which I will orientate the three studies that comprise the core of my dissertation research. The framework was developed through the research process to highlight how the three papers speak to the larger aims of the dissertation as a whole. In this respect, it acts as a conceptual heuristic that intends to help the reader understand how the three independent research papers I conducted relate to each

other.

My research is orientated relative to three key reference points: institutions, social-ecological systems, and ecosystem services as a boundary object. First, institutions are conceptualised within broad terms encompassing rational choice, historical and sociological institutionalist traditions. Although these three institutionalist perspectives emphasise different causal factors, they are not mutually exclusive but instead can be seen as quite complementary (Hall, 2010). The second important concept is that of a highly interrelated, joint social-ecological system. This system encompasses, reacts with, and is to some extent determinant of observable institutional features that are the subject of my research. The third axis for this work conceptualises ecosystem services as a boundary object, linking not only policy makers from different fields (Abson et al., 2014), but also social and ecological phenomenon.

A boundary object is an analytical concept that can be used to examine intersecting social worlds (Star & Griesemer, 1989), and to foster transdisciplinary research on socially relevant problems "characterised by a permeable science-society boundary" (Schröter et al., 2014). The boundary in question is not conceptualised as the edge of a discipline's domain; it is not a border. Rather it is a shared space where disciplines overlap, "where exactly that sense of here and there are confounded" (Star, 2010). It is robust enough to be used by different disciplines, and go from a more open and ill-defined meaning between groups to a very well-structured meaning within a given group (Star & Griesemer, 1989).

That dynamic between ill-structured and well-structured use is a distinguishing feature, and vital to facilitating cooperation between disciplines (Star, 2010). Ecosystem services exhibit such a dynamic between vague and specific meaning. Conceptual

consensus on a vague meaning (that ecosystem services are benefits humans derive from nature), gives way to very specific understandings when scholars interact within their respective disciplines. Ecologists see ecosystem services as rooted in the complexities of ecological structure and function, while economists largely seek to reduce complexity and classify services in preparation for integration within valuation and decision-making processes. The evolution of ecosystem services as a concept presents a largely bifurcated history along these lines, with interdisciplinary research declining in recent years (Abson et al., 2014). However the intensely interconnected social and ecological elements that underlie the highly normative undertaking of water governance call for a transdisciplinary analysis that engages ecology and economics in an examination of policy and governance.

These three conceptual themes – institutions, social-ecological systems, and ecosystem services as a boundary object – motivate and unite the three studies that comprise my doctoral research, and will be used to define the key contours of an evolving understanding of the political economy of water governance. This research is largely focussed on governance processes and the political contestation that occurs among actors that are involved in governance within a given institutional context. Rather than engaging any particular theory of the state, political economy in this research is understood to comprise the social, economic and political sources of power and influence within governance processes (Johns, 2008a). In addition, following from the historical nature of institutional development and change, certain actors may also enjoy more or less power as a result of embedded institutional favour. Figure 1.2 conceptualises the relationships between natural resources and the actors, institutions, and political economic structures that affect the outcomes of water resources governance. Within the SES, denoted by the

large outer circle, are two subsystems: the social and the ecological. Ecosystem services define the boundary between these two subsystems. Within the social subsystem, formal and informal institutions largely define the context for the political economy. They affect actor strategies, some of which seek to engender institutional outcomes. Power is both constrained by institutions while also acting upon them through both actor effects and political processes. There is considerable exchange between the two subsystems with the social subsystem both drawing on ecosystem services provided by water resources, while also affecting the ecological subsystem that give rise to ecosystem services.

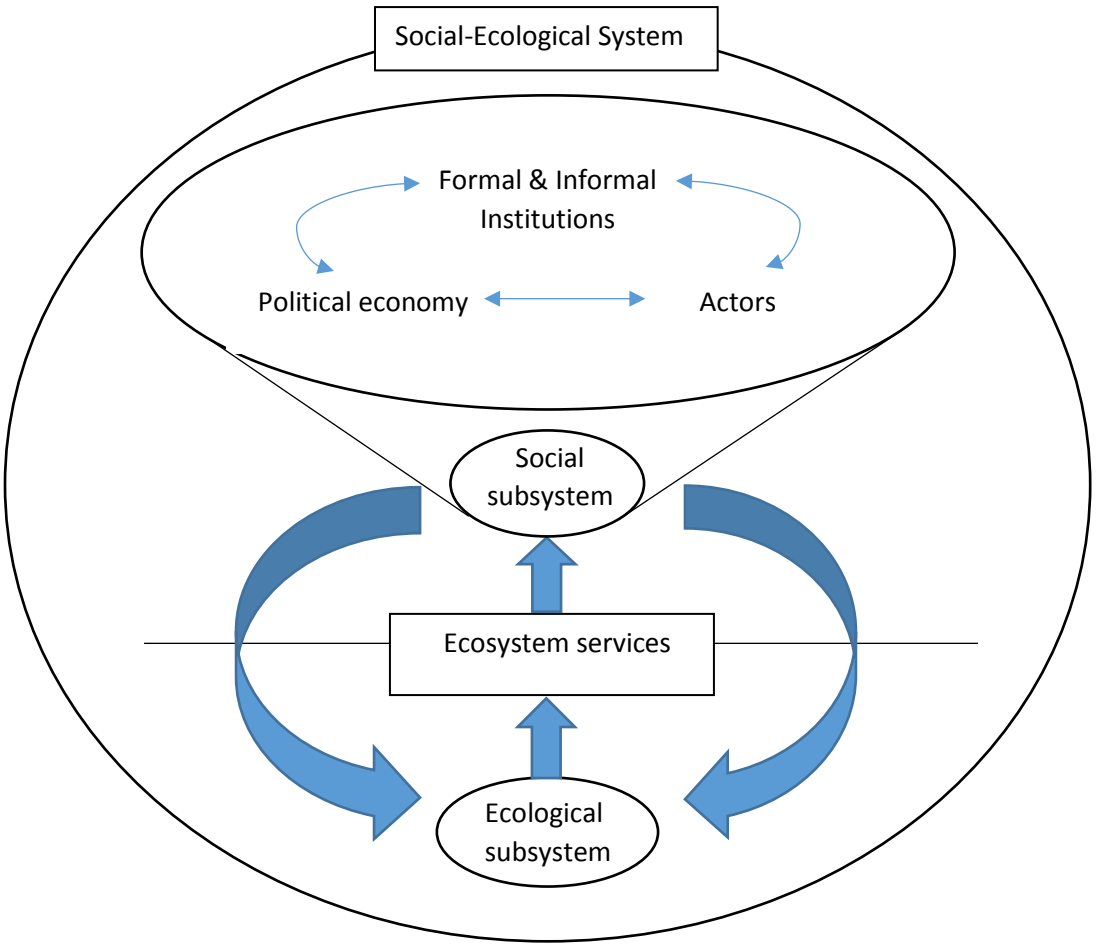


Figure 1.2 – Conceptual Framework

This conceptual framework presents a simplified account of a highly complex joint system. Embedded within this account are a number of assumptions. Fundamentally, the social and ecological subsystems are held to be co-determinant, meaning that conditions in one can induce change in the other. Thus, to analyse water resources institutions and governance processes it is essential to take into account the ecological dependencies and interests of governance actors, as well as any pressures that ecological change (such as climate change), might induce through these linkages. Similarly, ecological conditions are subject to change as a result of activity in the social world. This is not a particularly bold claim, but it has implications for how we seek to ensure more resilient social-ecological systems, and the importance of identifying embedded institutional barriers in order to transition toward more sustainable approaches. Engaging values and norms will be critical in that regard.

The broad institutionalist approach taken in this research offers flexibility and the capacity to engage a wider range of causal explanations. Certainly, rationalist explanations are deployed when analysing actor strategies. However the institutional environment of rules and incentives within which rational actors devise their strategies (relative to expectations of what others will do), is itself a historical construction. These institutions were created relative to different political issues and reflect compromises made among earlier and other governance actors. Furthermore, institutions change over time through implementation, with "rules as written" transforming into "rules in use." These historical factors are assumed to advantage some actors over others within current political debates. Finally, in addition to strategic calculation, actors are assumed to be guided by logics of appropriateness such that social norms and scripts also affect policy preferences.

The central research question for this thesis is as follows. How does demand for ecosystem services affect water resources governance and associated institutional outcomes? This research also asks whether an integrative approach, built around a more encompassing definition of ecosystem services, can overcome the force of vested interests operating within a political economy and institutional context that privileges socio-economic development over ecological sustainability. A systems-based approach to developing new perspective on this overarching question will involve a truly multidisciplinary effort, engaging ecological considerations and a diversity of social and political theory. To advance our understanding of water governance challenges stemming from systems effects this thesis will build on existing research through three separate but interrelated studies: (1) a characterisation and analysis of environment-economy linkages in a manner that encompasses and preserves a degree of ecological complexity; (2) an integration of cultural factors into the valuation of ecosystem services; and, (3) an examination of institutional dynamics stemming from contestation within the political economy over water resources and related services. These three studies will serve to provide a richer understanding of the paradoxical challenges of water governance noted above.

In the next section these three studies are introduced and discussed in more detail. This includes the specification of additional theoretical and methodological approaches specific to each of the studies. The scope of each study is defined relative to the preceding conceptual framework (Figure 1.2), and key research questions illustrate how each study contributes to an improved understanding of how demand for ecosystem services affects the governance of water resources and associated institutional outcomes, including

implications for social-ecological resilience

1.5 Three Studies

In this section I will introduce the three studies that comprise the core of my doctoral work. In addition to introducing empirical and methodological components, key conceptual foundations and relevant areas of scholarly research will be briefly discussed in order to highlight specific theoretical contributions. Case study selection and the specific ecosystem services of interest to each study will be defined, as will a guiding research question. Each study will then be connected back to the conceptual framework shown in Figure 1.2, thereby linking the individual contributions of each study and positioning them as aspects of a larger argument. A table summarising research questions, methodology and connection to the conceptual framework will follow this introduction to the three individual studies.

1.5.1 Projecting Economic Risks Stemming from Ecosystem Change at the Subwatershed Level

Ecosystem services are typically thought of as falling into four categories: provisioning services, regulating services, cultural services, and supporting services (MEA, 2005). However, ecosystem services are not independent from the environment; instead, they arise from a complex ecological system, which when disturbed can affect the flow of ecosystem services (Farber et al., 2002). While efforts are being made to understand the ecological underpinnings of ecosystem services (Kremen, 2005), current approaches to their integration within economic modelling do not take up this task (Fisher et al., 2008). This represents a significant challenge and opportunity.

This study is grounded in the assumption that social and ecological systems are

highly interrelated, both by the demand from society for ecosystem services and by the impact of society on ecological structures and processes (with implications for future availability of ecosystem services). Mapping linkages between ecological attributes, ecosystem services, and users of those services, helps to clarify the manner in which different actors draw on the environment and may in fact be in competition with one another for underlying ecological resources. This in turn provides a basis for better understanding how the demand for ecosystem services can affect water resources governance and associated institutional outcomes. Moving from an abstract argument to a robust implementable method is the focus of this study. Specifically, how can these linkages, including the demand for and impact on ecosystem services, be determined while also accounting for ecological complexity?

A new approach was needed in order to answer this question. Drawing on the United States Environmental Protection Agency's Final Ecosystem Goods and Services (FEGS) Classification System (Landers & Nahlik, 2013), I developed a novel environment-economy model that maps linkages between economic actors and the ecological subsystems that produce the ecosystem services they need. Using readily available georeferenced ecological and economic data, the model was used to analyse the spatial demand for multiple FEGS in Ontario's Mississippi Valley watershed. The Mississippi Valley watershed case study location was chosen due to an abundance of existing research on the observed and expected impacts of climate change. In addition to ecological pressure from users of various ecosystem services, climate change is a pressing concern for water resources governance. Using the model, areas of elevated demand for ecosystem services were identified. This enabled a determination of the potential for competition between

users of different ecosystem services over underlying ecological resources. Existing policy was assessed to ascertain where this competition could go unchecked, amounting to an embedded source of economic risk. My analysis also incorporated a consideration of climate change impacts. Natural resource managers were interviewed to assess the capability of the model, to better understand how existing policies were enforced, and to ascertain how political and economic considerations affected resource governance in the watershed, within and between political jurisdictions, and relative to the joint social-ecological system.

Among the three studies that comprise the core research contribution of my doctoral work, this study places the greatest emphasis on developing a general account of linkages between ecological and social subsystems, inclusive of a prioritisation of ecological considerations. With a focus on systems diagnostics, the ecosystem services of interest to this study are not determined by a given policy process but rather are a function of existing demand as determined by existing economic activities. Economic and ecological data were used to model current conditions. These results were then integrated with an analysis of the projected impacts of climate change, and qualitative data from interviews with local natural resource managers concerning changes and trends in resource use throughout the watershed. Developing a spatial description of how economic actors draw benefits from the ecological subsystem, and how demand and ecological conditions are expected to shift as a result of climate change, enabled close examination of the social subsystem in terms of policy coverage and implementation, as influenced by local political and economic imperatives. These factors help shape the level and manner of use, thereby affecting underlying ecological conditions and the subsequent ability of ecological systems to

provide a range of ecosystem services. If policy protections are absent or not adequately enforced this increases the potential for ecologically derived economic risks to develop. The study concludes with a discussion of property rights considerations.

1.5.2 Cultural Values and Ecosystem Management at the Subwatershed Level

This study examines the role of culture in preference formation and extends an existing governance framework by adding a cultural component. While ecosystem services define the manner in which society draws on and depends on the environment, the degree of dependence is not absolute; rather, it is a matter of interpretation. This is in part due to the large number of services rendered, with different people using different bundles of services. Similarly, the way we value services depends on how we prioritise one objective relative to another, and the nature of the service itself, for instance, whether or not it has a readily identifiable market value (Farber et al., 2002). Consultative approaches to the valuation of ecosystem services are grounded in these realities. However the allocation of water to one purpose may mean that less is available for something else, and with so many varied users competing for a common resource it becomes important to incorporate social factors within allocation decisions (Wilson & Horwarth, 2002). How we as individuals prioritise one objective relative to another is a matter of preference. However, preferences are subject to change under an array of influences (Farber et al., 2002), including association and bonding within a given socio-cultural context (Chee, 2004).

As discussed in Section 1.1, the concept of ecosystem services originally had a normative focus. It sought to reframe how society understood its relationship with the environment. This study is aligned with those earlier aspirations. Specifically, it focusses on exploring how culture and values affect the valuation of ecosystem services and

decision-making. The illustrative case study tests the hypothesis that cultural beliefs and values-based preferences for specific bundles of ecosystem services may underlie political contestation over preferred policy options, thereby comprising an important determinant of water resources governance outcomes.

Methodologically, this study takes stock of existing approaches to monetary and non-monetary valuation of ecosystem services, as well as frameworks for integrating valuation outputs within decision-making. An existing ecosystem services valuation framework was extended to better understand how social and ethical values affect the valuation process. Drawing on theory pertaining to deliberative processes, the extended framework elicits cultural values and integrates them within a closed loop decision-making process. I then applied the revised framework to a policy consultation process in Chelsea, Quebec, and assessed the relative influence of cultural values on local environmental governance. This illustrative case was chosen due to the availability of original data that was generated within an active policy process. These circumstances were expected to result in high-quality data due to the opportunity for participants to represent their interests to decision makers.

The ecosystem services of interest in this study are those that are embedded within the substantive focus of the policy process under investigation, i.e. municipal infrastructure and development. Moreover, given the plurality of perspectives on which arrangement of social-ecological system elements is preferable, this study seeks to help understand and manage political contestation between different bundles of preferred services, as embedded within policy preferences. The focus of the framework is on how actors interact with one another and debate different policy preferences, enabling culture and values perspectives

to be made explicit and integrated within policy making processes. To the extent that governance actors inhabit separate but overlapping social worlds, as determined by differing values and goals, this study is focussed on the internal workings of the social subsystem with an emphasis on values and resolving normative differences. The integrated decision-making framework is designed to approach this task with reference to the social-ecological systems context. For instance, ecological experts are tasked with providing information on system linkages, such as the manner in which ecological processes provide ecosystem services, and how the provision of these services could be impacted by various policy choices.

1.5.3 Enacting and Implementing Water Governance: The Role of Decentralisation in Transforming Policy Goals in Three Ontario Watersheds

Policy processes and institutional design are central to effective resource management. Since water resources, and by extension the water-based ecosystem services they produce, are subject to increasing competition among a wide range of users, we may expect these users to bring their resources to bear on acquiring the services they need (Johns et al., 2008). This competition between actors, in current and historical timeframes, underlies institutional outcomes and may structure sustainability challenges.

In this study I document extended actor pressure through the legislative process and subsequent implementation and institutionalisation of Ontario's 2006 *Clean Water Act* (CWA), observing conflicting agendas, the formation of bargains between involved actors, and the impacts of these factors on institutional outcomes. In doing so, I determine how key governance actors engage with the development and implementation of the CWA, and the extent to which policies at the source protection level were motivated by concerns over

the availability of ecosystem services other than safer source water. As such, this study directly takes up the central research question within a specific policy development process as it seeks to determine how the demand for various ecosystem services affects water resources governance and associated institutional outcomes. This study will explore a second paradox as well. When open governance processes are designed to encompass the concerns of actors whose activities may actually undermine the stability of water resources, there is the potential for contradiction between policies that seek environmental outcomes, and the governance processes used to develop these policies.

While the other two studies touch on institutions either in terms of substance and implementation, or as an eventual outcome of the policy process (Sections 1.5.1 and 1.5.2, respectively), institutions comprise the primary unit of analysis for this study, in particular the CWA, Source Protection Plans created under the CWA, and formal and informal institutions that influenced actor strategies throughout the legislative and policy processes. It follows that institutions were understood to be both tools of governance, and the outcome of governance processes involving numerous actors of varying power, resource and influence. The study consists of a two-stage qualitative analysis, first focussed on how different actors engaged with and affected the development of the CWA. This is followed by a case study analysis of three Source Protection Regions. Document research and semi-structured interviews with key individuals and stakeholder representatives were used to understand how different interests were balanced against one another in the development of the CWA. Additional interviews, a scan of local and farm media, document research and an email questionnaire were used to interpret differences between Source Protection Plans in the three case study watersheds.

As with the previous study, the ecosystem services of interest are those that are embedded within the substantive focus of the policy process under investigation, in this case source water protection. Providing a detailed examination of two stages of policy development, this study documents competition between governance actors during the development and implementation of the CWA in Ontario. This legislation seeks to increase drinking water safety by protecting the sources of municipal drinking water. However implications for land use and the relative distribution of costs and benefits led key governance actors to perceive it on dramatically different terms. For some the CWA equated with a loss of autonomy and a marginalisation of rural lifestyles, while for others it was a potential threat to their livelihood. In all cases there was a concern that the CWA would change current relationships between society and the environment, and that these changes were being imposed upon them to the benefit of others elsewhere. Source water protection thus connects multiple overlapping social worlds, with different actors seeking to advance their particular preferred policy options according to perceived implications for human-environment relationships, and the associated provision and access to ecosystem services.

1.5.4 Limitations of Qualitative Approaches and Generalisability

The three studies that comprise the core of my empirical research draw on qualitative methods. The generalisability of research findings is the subject of much debate among social scientists. On the one hand, studies drawing on a large number of samples often involve simplifying assumptions that can attract criticism around variable selection, challenging the strength and persuasiveness of causal explanations (Ragin, 1987).

Qualitative case studies on the other hand can incorporate a more nuanced account of complex phenomenon, but in doing so critics argue that they can have the opposite problem, encompassing more variables than can be resolved with a small number of cases (Lijphart, 1971). Reducing variables through use of comparable cases can help address these challenges however this may also introduce concerns around over-determination (Collier, 1993). Echoing an uncertainty about the appropriateness of law-like generalisations in the social sciences (Lincoln & Guba, 2000), Gomm et al. (2000) argue that "any social phenomenon in a specific social context is likely the product of multiple causal processes." Yin (2015) suggests a different approach. Instead of seeking to generalise to a population-level based on statistical analysis, the objective of "analytical generalisation" is to improve our understanding of particular concepts or theory. Rather than filling a theoretical gap, this approach sees generalisation and theory-building more in terms of working hypotheses. Theory-confirming or theory-infirming case studies can draw on techniques such as process tracing to see whether different cases follow a common causal pattern (Collier, 1993), in order to contribute to a wider intellectual purpose. My research follows this model, embracing complexity and positioning research findings relative to current theory in order to build confidence, in particular around processes of institutional change, issues of fit, interplay and scale, and the importance of variable selection.

1.5.5 Summary of Study-Specific Research Questions, Methodology and Connections to the Conceptual Framework

This thesis argues that a systems-based approach is essential to unlocking the central paradox of increased water resource sustainability challenges coinciding with increased

knowledge and institutional safeguards. This foundational argument led to the development of a broad conceptual framework (Figure 1.2) that encompasses the key social and ecological systems elements, and defines their interrelationships at a high level of abstraction. Three separate but interrelated studies were undertaken to shed light on aspects of the central research question of how demand for ecosystem services affects water resources governance and associated institutional outcomes. Table 1.1 briefly summarises the main research question for each study, and the methodological approaches used to answer these questions. I have also indicated which parts of the conceptual framework are most relevant to each study, thereby positioning each study as part of a larger argument.

	Research Question	Methodology	Connection to Conceptual Framework (Figure 1.2)
Study 1: Projecting Economic Risks Stemming from Ecosystem Change at the Subwatershed Level (Chapter 2)	How can linkages between social and economic systems be defined, and how do these linkages affect water resources governance in the context of climate change, and the politics and policy of resource governance in the Mississippi Valley watershed?	- Reformulation of the Final Ecosystem Goods and Services Classification System (FEGS-CS), and model development - Geo-spatial analysis of economic and land cover data - Review of key policies - Semi-structured interviews with local natural resource managers	This study focussed on delineating the relationships between social and ecological subsystems, describing how ecological subsystems generate ecosystem services that provide benefits to specific economic actors. This characterisation of the joint social-ecological system was used to identify key aspects of the social subsystem for further analysis. Interactions among subsystem components (actors, institutions and political economic conditions), were then assessed for their potential to result in ecological change, and associated socio-economic impacts.

Study 2: Cultural Values and Ecosystem Management at the Subwatershed Level (Chapter 3)	How do culture and values affect valuation of ecosystem services and subsequent decision making, with an application to water and sewage infrastructure in Chelsea, Quebec?	- Review of ecosystem services valuation and related decision-making frameworks - Integration of cultural theory with an existing valuation framework to develop a closed-loop decision-making framework - Assessment of illustrative case using the modified framework	This study was primarily concerned with the social and ecological subsystems and how cultural factors and values affect actor strategies. In particular, it examines how these effects could be resolved, and how they influence feedback from the social subsystem to the ecological subsystem, in terms of policy decisions.
Study 3: Enacting and Implementing Water Governance: The Role of Decentralisation in Transforming Policy Goals in Three Ontario Watersheds (Chapter 4)	How did key governance actors engage with the development and implementation of the *Clean Water Act* (2006), and to what extent were policies at the source protection level motivated by concern over the availability of ecosystem services other than safer source water?	- Semi-structured interviews with key governance actors and stakeholder representatives involved with the development of the act - Document research - Biographical description of Source Protection Committee (SPC) members - Semi-structured interviews with SPC Chairs and Source Protection Authority liaisons - Email questionnaire - Scan of local and farm media	Formal and informal institutions figure prominently in this study, both as the rules by which actors formulate their respective strategies, but also as the object of political contestation within a legislative and subsequent policy development process. Drawing on existing frameworks of institutional analysis and change, this study presents an in-depth examination of social subsystem dynamics that are largely motivated by contrasting policy images among governance actors. These images reflect a range of preferences for how human-environment relationships should be managed.

Table 1.1 – Summary of Research Questions, Methodology and Connection to the Conceptual Framework

Issues pertaining to scale comprise another cross-cutting concern that is not addressed within the table but is relevant to the overall cohesiveness of this thesis. There is an immediate tension with regards to scale when taking a systems-based approach to understanding water resources governance. On the social side, political jurisdictions are a key determinant of relevant scale. However ecological systems do not align with political boundaries, and hydrological systems are different again. Given the historical nature of institutional development and change there are temporal considerations that need to be accounted for as well. Approaching the larger arguments embodied within the conceptual framework through three smaller studies was helpful for managing these issues of scale, and for selecting a unit of analysis for each individual study. Selecting a subwatershed case study (i.e. the Mississippi Valley watershed in Chapter 2), or a municipal governance process (i.e. Chelsea in Chapter 3), constrained the spatial extent and total number of relevant societal elements. This allowed for a systems-based approach that engaged multiple scales (political, ecological and hydrological). Methodological choices also served to focus the analysis. For instance, although Source Protection Committees and the *Clean Water Act* (2006) were watershed-based, the policy regime was the primary unit of analysis for my institutionalist examination of legislative development and implementation in the case of source water protection in Ontario.

The remainder of the thesis will proceed as follows. The three studies are found in Chapters 2 to 4. These will be followed by a discussion chapter that draws out cross-cutting themes and observations concerning ecosystem services as a concept, including the degree to which it may be used to trace linkages between joint social and ecological systems, and its analytical value as focus of political contestation. The final chapter will take these

observations, and the findings from the three individual studies, and distil implications for water policy and governance. This will be followed by a thorough synthesis of the work, and reflections on how new theoretical insights and methodological approaches developed through the research can be brought to bear on enduring challenges facing water governance. The final chapter will conclude with a discussion of limitations of the work, thoughts on future research and some closing remarks.

Chapter 2 - Projecting Economic Risks Stemming from Ecosystem Change at the Subwatershed Level

Regional economies, to varying degrees, depend on water-based ecosystem services. With growing populations and other pressures increasingly stressing the underlying ecosystems, it is imperative to understand the associated economic risks. Monetary valuation of ecosystem services using aggregate-value, general-equilibrium and partial-equilibrium models provides some insight. However, these approaches inadequately consider ecological factors. Shifting the focus from marginal value or expected returns of total economic value toward outcome scenarios, this study explicitly considers and prioritises ecological complexity, interconnectivity, and local variation in the production of and demand for ecosystem services. It develops a novel organising framework for mapping linkages between complex interconnected ecological and economic systems in order to: (1) clarify how economic activities draw on and draw down ecological resources, and, (2) reveal competition for underlying ecological resources between users of different ecosystem services. These interconnections have the potential to introduce wider ecological and hence economic risk. Examining the Mississippi Valley watershed in Ontario, Canada, I used the framework to elucidate the vulnerability of individual economic activities to the ecological impacts of climate change, in conjunction with competition between users for underlying ecological resources. Comparing these risks with the extant policy regime highlights policy gaps that would allow projected risks to materialise.

2.1 Introduction

The economy is a complex system that is made more complex by ongoing change and the societal interpretation of those changes (Beinhocker, 2006). Modeling the economy requires understanding the complex factors that drive change in order to focus in on the crucial elements (North, 2010). The natural environment is a similarly complex system marked by continuous change driven by, among other things, human activities. Ecosystem services frequently connect these two systems, with environmental changes potentially causing ripple effects through the local, regional, and sometimes global economy, and vice versa. However, different ecosystem services (Lester et al., 2013), or bundles of services (Raudsepp-Hearne et al., 2010), may not exist concurrently, and their respective provisioning may arise from management choices, either deliberately or in unanticipated ways (Rodriquez et al., 2006). Thus, where demand for ecosystem services affects ecological integrity that in turn can affect the supply of different services (Brukhard et al., 2012), it becomes important to understand the dynamics of demand between different users.

Current approaches to representing the interrelationships between environment and economy often translate environmental factors into economic terms. Aggregate-value methods present a range of total values for all ecosystem services provided by different types of ecosystems (Russi et al., 2013). This range of values is often very large due to a number of factors, including the inherently local nature of environmental production functions (Brander et al., 2006). Risk projections based on aggregate values yield all or nothing claims. If an ecological unit is maintained its worth lies somewhere within the given range per unit area. If it is lost those are the stakes. However, many intermediate

outcomes will be of interest to decision makers, and will make efforts to integrate environment and economy more relevant.

Input-Output (I-O) models have been broadened to incorporate environmental factors. They have been used to allocate environmental harms (Leontief, 1970), and the supply of environmental goods to different economic activities (Grêt-Regamey & Kjtzia, 2007). However partial-equilibrium approaches are criticised for operating with fixed prices (Opocher, 2003). There are also concerns that findings are highly sensitive to how key ecosystem services are identified, and that using different services could change model outputs.

To better represent economic complexities such as fluctuating prices and substitution effects, computable general-equilibrium (CGE) models have been expanded to incorporate ecosystem services to varying degrees (Patriquin et al., 2003; Smajgl, 2006). Already a highly complicated approach to economic modelling, applications of CGE to the integration of environment and economy are purpose built, and require simplification of the environmental dimensions that are represented. Aside from criticisms relating to the realism of perfectly competitive markets, perfect information and frictionless transactions (Ackerman, 2004), there are additional challenges associated with accounting for substitution effects when environmental goods or services do not have a market price (Carbone & Smith, 2010). For instance, existence values for ecosystem services can be orders of magnitude larger than typical market prices for related ecosystem services (Loomis, 1996; Ackerman, 2004).

The discussion so far has raised several important considerations. Ecosystem production functions are inherently local, and ecosystem services arise from a complete

ecology, meaning that they cannot be fully understood in isolation (Erwin, 2009; Harrison et al., 2014). This requires that their ecological interdependencies be kept in mind as decisions are made concerning 'key' ecosystem services (Kremen, 2005). Economic models such as CGE or I-O models that are extended to include only a selection of environmental factors may be fruitful for very specific applications, but may equally overlook significant aspects of the scenario under question, due to an underrepresentation of highly complex and interrelated ecological systems. Indeed, theory on social-ecological resilience cautions against management approaches that focus on the production of a small number of services of particular interest without consideration of wider ecological linkages (Walker & Salt, 2006). Narrow conceptualisations of natural resources have resulted in management decisions that have brought a range of unintended consequences in terms of negative human health and socio-economic outcomes (Tallis & Polasky, 2009). Further, the very nature of ecological systems – interconnected, non-linear, and irreversible beyond certain thresholds (Chaplin III et al., 2000) – implies real economic risks that may otherwise go unidentified if we do not explore new ways of conceptualising the environment and the economy as two interrelated subsystems of a larger whole. We need to develop tools that are able to work across scale to accurately diagnose and discuss these challenges (Cumming et al., 2006).

To advance this line of inquiry, this study provides an approach to systematically identify economic risks that lie beyond the limits of current approaches. The central objective is to model the relationships between economic activities, the water-based ecosystem services upon which these economic activities depend, and the underlying ecological conditions that give rise to these valuable ecosystem services. The framework

is designed to use existing data to generate a spatial representation of ecological resources, identify associated economic activities, and assess areas of economic vulnerability arising from ongoing competition for scarce ecological resources. The use of existing data will enable wider use by resource constrained local governments and policy makers at the subwatershed level.

The study proceeds in three parts. The first part describes the theory underlying the framework as well as specific methods employed in the case study analysis. The second part applies the framework to Ontario's Mississippi Valley watershed using existing GIS data, census data on economic activities, and interviews with local stakeholders. Locations with elevated levels economic activity are identified and the extant policy regime is examined to assess the potential for ecologically-derived economic risk to develop. The study concludes with a discussion of the importance of considering patterns of demand for ecosystem services, and the implications of systems effects for property rights.

2.2 Theory and Methods

The complexity of the linkages between non-linear ecological and economic systems makes full specification of a mathematical model very difficult if not impossible. For this reason trade-offs need to be made between generality, precision and realism (Costanza et al., 1993). The organising framework presented below allows for the intensity of relationships between system components to be established and assessed with moderate generality and precision. Model resolution and relationships are simplified to answer basic questions about the overall magnitude and direction of expected ecological changes (Costanza et al., 1993), and their economic impact.

Two general principles are the basis for the organising framework presented in Figure 2.1. First, numerous ecological functions may be the basis for any given ecosystem service, and any given ecological function may affect multiple ecosystem services (Maynard et al., 2010). Second, numerous ecosystems services may affect any given economic activity, and any given ecosystem service may affect multiple economic activities (MEA, 2005). These principles provide a structure for organising various economic and ecological factors. Once these factors have been identified two similar matrices may be constructed. The first relates economic activities with their supporting ecosystem services, while the second relates those ecosystem services with underlying ecological structure and functions.

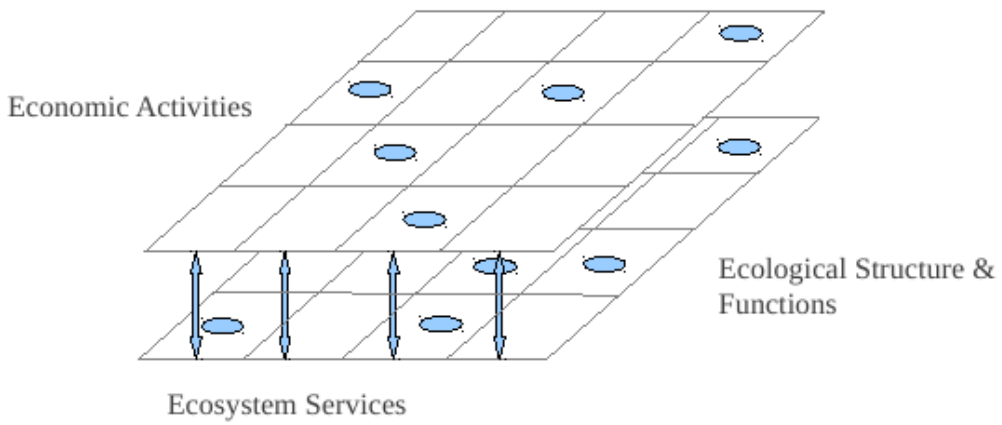


Figure 2.1 – Ecology-economy organising framework

Emphasising synthesis over analysis, the organising framework helps define the joint ecological-economic system under consideration. The system's boundary is determined by the defined goal of the system and the nature of the relationships between its component parts (Limburg et al., 2002). This dictates which components are included within the system and which are determined to be minimally connected and may therefore

be held constant for the purposes of the analysis. To reduce complexity, component parts may be organised within a hierarchy such that: a) more complex systems consist of simpler subsystems; b) relationships within subsystems are stronger than relationships between subsystems (van Gigch, 1978); and, c) the relative position in time or space has a bearing on the strength of relationship between elements (Urban et al., 1987). It follows that there are important dimensions of temporal and spatial scale that need to be considered, and in fact are useful for organising system components into semi-autonomous subsystems (Simon, 1962).

The idealised approach outlined here is confronted by several fundamental challenges, largely rooted in the complexities of ecology. Identification of key ecological factors requires an understanding of the functional importance of different species as ecosystem service providers (Kremen, 2005), as well as compensatory mechanisms that help stabilise ecosystem response (Elmqvist et al., 2003). Biodiversity offers a portfolio of genes and species that can provide insurance value and buffer ecosystem response (Schindler et al., 2010, Admiraal et al., 2013). However, actual ecological function can exhibit spatial and temporal variability (Koellner & Schmitz, 2006). The production and consumption of ecosystem services have similar spatial and temporal dimensions (Brauman et al., 2007), underscoring scale as an important consideration for application of the organising framework.

Furthermore, while detailed investigation of ecological mechanisms would provide an ideal foundation for land-use and ecosystem-based planning, to develop a comprehensive understanding would be an ambitious undertaking (Kremen & Ostfeld, 2005), often beyond the reach of resource constrained local governments where many of

these decisions are being made. In recognition of these tensions, an empirical application of the theoretical arguments presented above has been undertaken using the Final Ecosystem Goods and Services Classification System (FEGS-CS), developed by the United States Environmental Protection Agency (Landers & Nahlik, 2013). Final ecosystem goods and services (FEGS) arise from the interrelated elements of ecological structure and function, and directly connect intermediate services to the benefits humans receive from the environment (Fisher et al. 2009).

The FEGS-CS defines linkages between beneficiaries and 20 different FEGS for 15 different ecosystem types: rivers and streams, wetlands, lakes and ponds, estuaries and near coastal marine, open oceans and seas, groundwater, forests, agroecosystems, created greenspace, grasslands, scrublands/shrublands, barren/rock and sand, tundra, ice and snow, and atmosphere (Landers & Nahlik, 2013). From a systems perspective, ecological structure and function are considered in aggregate as they come together to constitute different ecosystem types. FEGS included in the FEGS-CS are: water, flora, fauna, fungi, fish, pollinators, depredators and (pest) predators, fibre, timber, natural materials, presence of environment, viewscapes, sounds and smells, open space, soil, substrate (for building on), air, weather, wind and atmospheric phenomenon. Beneficiary categories include: agriculture, commercial/industrial, government, municipal and residential, commercial/military transportation, subsistence, recreational, inspirational, learning, and non-use. Within each beneficiary category a number of sub-beneficiaries are also provided. North American Industry Classification System (NAICS) codes are provided for some sub-beneficiaries. The FEGS-CS provides a series of tables for each ecosystem type, which

specify sub-beneficiaries, FEGS of interest to these sub-beneficiaries, and a short description of the importance of a given FEGS, essentially the nature of use.

In order to model total users by ecosystem type, the FEGS-CS tables were reconfigured to list sub-beneficiaries according to ecosystem type and ecosystem service. Rather than having FEGS allocated on a beneficiary-by-beneficiary basis for each ecosystem type, the reformulated system is a matrix of ecosystem type-FEGS pairings. Sub-beneficiaries that have an interest in a given pairing are grouped together, thereby clarifying which specific sub-beneficiaries may be in competition with one another. Six ecosystem types and nine FEGS were included in this reformulation, as shown in Tables 2.1 and 2.2. Where NAICS codes were provided, economic data on active businesses and agricultural operations was used to determine the total number of unique users across all ecosystem services within a given ecological setting (i.e. ecosystem type). Additionally, users were enumerated for specific FEGS-ecosystem type combinations, where possible, to characterise the competition over individual ecosystem services. Where NAICS codes were not provided, FEGS beneficiary requirements, as indicated by the FEGS-CS, were used to qualitatively describe the competition, for instance the impact of increased recreational users in the upper Mississippi watershed.

ECOSYSTEM TYPE

FINAL ECOSYSTEM GOODS AND SERVICES

	Agroecosystems	Grassland/Herb	Barren/ Rock & Sand
water			
flora	experiencers and viewers	livestock grazers experiencers and viewers food pickers and gatherers	experiencers and viewers
fauna	experiencers and viewers hunters	livestock grazers experiencers and viewers food pickers and gatherers	experiencers and viewers hunters
presence of the environment	electric and other energy generators resource dependent businesses residential property owners experiencers and viewers	electric and other energy generators resource dependent businesses residential property owners experiencers and viewers	electric and other energy generators resource dependent businesses residential property owners experiencers and viewers
open space	Livestock grazers farmers	livestock grazers farmers	CAFO operators
fish			
soil	livestock grazers farmers	farmers	
pollinators	farmers	farmers	
depredators and (pest) predators	farmers	farmers	

Table 2.1 – Reformulated Final Ecosystem Goods and Services Classification System (FEGS-CS)

ECOSYSTEM TYPE

FINAL ECOSYSTEM GOODS AND SERVICES	Freshwater	Wetlands	Forests
water	livestock grazers CFAO Operators industrial processors industrial dischargers electric and other energy generators municipal DW plant operators WWTP operators water subsisters boaters	livestock grazers industrial dischargers WWTP operators boaters	
flora	livestock grazers experiencers and viewers food pickers and gatherers	livestock grazers experiencers and viewers food pickers and gatherers	experiencers and viewers food pickers and gatherers
fauna	experiencers and viewers food pickers and gatherers hunters	experiencers and viewers food pickers and gatherers hunters	experiencers and viewers food pickers and gatherers hunters
presence of the environment	industrial dischargers electric and other energy generators resource dependent businesses residential property owners experiencers and viewers waders, swimmers and divers boaters	industrial dischargers resource dependent businesses residential property owners experiencers and viewers waders, swimmers and divers boaters	resource dependent businesses residential property owners experiencers and viewers
open space		Livestock grazers farmers	foresters
fish	anglers	anglers	
soil		farmers	foresters
pollinators		farmers	
depredators and (pest) predators		farmers	

Table 2.2 – Reformulated Final Ecosystem Goods and Services Classification System (FEGS-CS)

Focussing on final ecosystem goods and services does constrain the ecological analysis. Yet, it also offers advantages by connecting ecological factors directly with beneficiaries, thereby avoiding much of the ambiguity around ecosystem services (Nahlik et al., 2012). NAICS codes proposed by the FEGS-CS for various sub-beneficiaries are also used by Statistics Canada's Canadian Business Patterns survey. Ecosystem types, as defined by the FEGS-CS, are consistent with existing GIS land cover databases in Ontario, as well the European Union's Mapping and Assessment of Ecosystems and their Services initiative (Maes et al., 2013). Compatibility with existing sources of georeferenced environmental and economic data should make the empirical application undertaken by this study relatively easy to replicate at the local level. This in turn could support a wider application and assessment of existing policies and their implications for the resilience of social-ecological systems.

FEGS-CS Sub-beneficiary categories	Agricultural survey/Canadian Business Patterns categories
Grazers	Grazers
CAFO	CAFO
Farmers	Farmers
Foresters	Forestry (maple syrup) Forestry Support for agriculture and forestry
Fur/hide trappers and hunters	Fish, Hunt & Trap
Industrial dischargers	Oil and gas extraction Mining Support for mining, oil and gas Manufacturing (list below)
Industrial processors	Oil & gas extraction Mining Support for mining, oil and gas Manufacturing

Electricity/other energy generators Municipal drinking water plant operators WWTP operators Water subsisters Residential property owners	Utilities Real estate Rental and leasing
Resource dependent businesses	
Experiencers and viewers Food pickers and gatherers Boaters Waders, swimmers, divers Anglers	Sightseeing transportation Heritage institutions Amusement, gambling, recreation industries Accommodation Food and drink

Table 2.3 – Key for Translating Economic Data into the Reformulated FEGS-CS Framework

The reformulated FEGS framework was used to investigate the impacts of climate change on competition between existing users of FEGS in Ontario's Mississippi Valley watershed. Ecosystem types were assumed to constitute subsystems in terms of the strength of relationships within and between system components, while relative upstream-downstream position was also used to determine key linkages. Economic data was gathered at census subdivision (CSD) level for total agricultural operations by type (Statistics Canada, 2011), and number of active businesses across sectors (Statistics Canada, 2008). This data was then translated into sub-beneficiary categories as defined by the FEGS-CS using the translation key shown in Table 2.3. Quantum GIS (QGIS) and base layers from Land Information Ontario (Ontario, 2015a) were used to delineate census subdivisions and determine the portion of each CSD that lay within the Mississippi watershed (CSD_{WS}), as illustrated in Figure 2.2. QGIS was then used to calculate land cover area by ecosystem type within these CSD_{WS}, enabling analysis of individual ecological subsystems and identification of locations with relatively high levels of economic activity,

both in terms of absolute number of active economic entities and the geographical intensity of demand (number of entities per square kilometer). Additional layers for land title, dams, rivers and other water bodies, were added to assist in analysis.

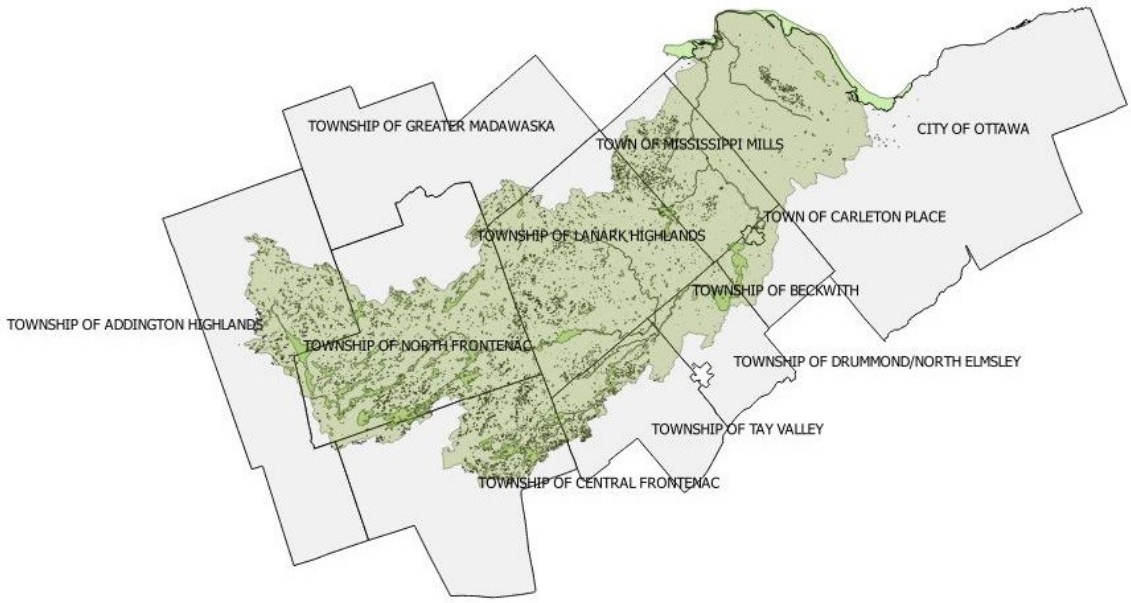


Figure 2.2 – Census subdivisions and major waterbodies in Mississippi Valley watershed

Maps were used to contextualise the quantitative analysis of economic activities by looking at patterns of land cover and spatial relationships between ecological subsystems. By visually communicating the relationships between numerous variables in a single image, maps can help us make sense of complex phenomenon (Burkhard et al., 2012). They have been used to determine the value of ecosystem services flowing from landscapes in Southern Ontario (Troy & Bagstad, 2010), and to relate overarching development pathways to spatially specific predictions of land cover change (Swetnam et al., 2011). Maps are of particular value for assessing ecosystem services where services are produced in one area but consumed elsewhere (Vrebos et al., 2015).

The approach outlined here produces sub-regional information on the level of usage experienced by six ecosystem types across a range of FEGS, differentiated among CSD within the watershed. The following section will introduce the case study region and outline the observed and expected impacts of climate change. An application of the reformulated framework is then used to identify locations in the watershed with elevated levels of economic activity. In these locations economic risk may arise due to competition between sub-beneficiaries for ecological resources, and as a result of the impacts of climate change. I then assess the capacity of the existing policy regime to manage competition and conflict between economic actors in these areas. To ground the analysis, fill in missing information and better understand its limitations, two local natural resource managers and two municipal officials were interviewed about economic activities within the watershed and the potential for competition between users of different ecosystem services[1].

2.3 Case Study – Mississippi River, Ontario

Encompassing most of Lanark County, the northern portion of Frontenac County, and the western portion of the City of Ottawa, the Mississippi Valley watershed covers an area of 3,750 km^2 from its headwaters at Mazinaw Lake in the west to its confluence with the Ottawa River in the east (MVCA, 2014). The effects of climate change are being observed in the Mississippi Valley watershed. Minimum summer stream flows have decreased since 1970, while average winter stream flows have increased over the same period. Both also

[1] Interviews were held confidentially due to the potential for negative repercussions from local stakeholders. Due to the small number of potential respondents and the specificity of their individual roles, direct quotations from the semi-structured interviews were not used due to the high potential for deductive disclosure. Results from the model and GIS analysis were not discussed prior to the interview in order to limit any unintended bias.

exhibit greater yearly variability (Lehman, 2012). Precipitation is predicted to increase by 10-20% overall and involve heavier precipitation events. Flooding will be possible in any season, with longer dry conditions between events. Spring flows are expected to be lower and earlier, and decreased lake levels are likely with more frequent and longer low water levels in summer. Average temperatures have risen by 1.5˚C over historic levels, and it is expected that they will continue to rise (Lemmen et al., 2008). Water temperatures are expected to increase accordingly, leading to increased evaporation (Egginton & Lavender, 2008). Combined with temporal changes in precipitation, the existing reservoir system in the Mississippi Valley is projected to be unable to meet an increasing need for flow augmentation during dry periods (Lehman, 2012). At the same time the demands from many water users are likely to increase.

These abiotic changes will be accompanied by a host of biotic changes. As a result, ecosystems will be fundamentally altered. Migration rates and pathways, over-wintering, multiple nestings, predator-prey dynamics, and the presence of certain pollinators all affect the numbers and distribution of species (Varrin et al., 2007). Moreover ecosystems do not move as a single unit, resulting in fragmented ecosystems that can affect functional groups and the production of ecosystem services (Chapin III et al., 2000; Mooney et al., 2009). As the ecology of the Mississippi Valley continues to change, land use, economic development, agriculture and a number of other policy domains will be directly affected.

2.3.1 Application of the Reformulated FEGS Framework

Economic data from two government databases was used to identify organisations operating in the nine CSD that span the Mississippi River's watershed boundaries[2]. Farm operations fell into 42 categories (Statistics Canada, 2011). These were aggregated into three sub-beneficiary groups: farmers, livestock grazers and concentrated animal feeding operations (CAFO). Unfortunately CSD-level data on irrigators was not available. The total number of potential beneficiaries was determined for each ecosystem type in the nine CSD. These numbers were then adjusted using the ratio of total area of the CSD to that of the CSD_{WS}. Available information on business size was used to conduct a parallel investigation into the level and intensity of demand for FEGS when accounting for business size. While the absolute numbers for total possible users and user intensity were affected by this adjustment, the relative ranking of CSD_{WS} at the high end did not change. A visual analysis of QGIS mapping showed a close association between freshwater, wetlands, agroecosystems and grassland/herb ecosystems. These four ecosystem types were flagged for closer investigation.

Maximum values, mean values, and number of standard deviations from the mean, were determined for the total number of unique users, and user intensity (i.e. users per square kilometer), inclusive of all FEGS in a given ecosystem type. This was done for each ecosystem type in all CSD_{WS}. Source data for each CSD_{WS} can be located in Appendix B.

[2] The northeast extent of the watershed includes land that belongs to the City of Ottawa. While this land is primarily rural, available economic data is given for the city as a whole. Including this data would skew economic analysis by including economic activities that were not being undertaken within the watershed. Since the majority of this area does not drain into the Mississippi River, but rather drains directly into the Ottawa River via the Carp River, this area does not participate in the upstream-downstream dynamics of the Mississippi watershed and was omitted.

The results of this analysis are found in Table 2.4 and describe the level of overall demand placed on the ecosystems in a given CSD_{WS}. This encompasses competition between users of the same FEGS within a given ecological subsystem, as well as between users of different FEGS within a given ecological subsystem. Forests were omitted from Table 2.4 due to the sheer abundance of forested land and comparatively insignificant levels of competing demand.

CENSUS SUBDIVISION	ECOSYSTEM TYPE							
	Freshwater		Wetlands		Agroecosystems		Grasslands/Herb	
	Absolute	Intensity	Absolute	Intensity	Absolute	Intensity	Absolute	Intensity
Addington Highlands	8	0.31	8	1.46	8	4.52	8	3.21
North Frontenac	46	0.30	61	2.41	58	7.56	58	6.87
Central Frontenac	43	1.07	61	3.03	59	5.39	59	6.08
Greater Madawaska	4	2.49	4	5.46	4	111.72		
Lanark Highlands	145	2.99	219	4.15	205	3.45	205	5.32
Tay Valley	49	3.68	87	11.66	85	2.65	85	7.10
Drummond/ North Elmsley	69	4.15	110	15.81	108	2.25	108	14.12
Carleton Place	414	1549.32	269	893.56	119	119.52	119	731.03
Beckwith	29	2.09	47	9.99	46	1.65	46	29.47
Mississippi Mills	422	60.03	514	45.96	447	3.20	447	9.41
Max	422	60	514	46	447	112	447	29
Max	M. Mills	M. Mills	M. Mills	M. Mills	M. Mills	G. Mad.	M. Mills	Beckwith
Next highest	Lanark H.	Drum.	Lanark H.	Drum.	Lanark H.	N. Front.	Lanark H.	Drum.
Mean	91	9	124	11	113	16	113	9
Standard deviation	124	18	151	13	131	34	133	7.89
Max (σ from mean)	2.68	2.82	2.59	2.65	2.55	-0.37	2.52	2.59
Next highest (σ from mean)	0.44	-0.24	0.63	0.36	0.70	-0.24	0.70	0.64

Note: Carleton Place Census Subdivision is comprised primarily of developed land. As a result there are relatively small areas of other ecosystem types, resulting in disproportionately large user intensity values. These were excluded from additional analysis for reasons outlined below.

Table 2.4 – Number of economic entities by ecosystem type and census subdivision

Mississippi Mills had the maximum total number of users across ecosystem types. In every instance the total number of users was more than two standard deviations from the mean value. It also had the largest intensity values for freshwater and wetland ecosystems. Lanark Highlands was second for total numbers in freshwater, wetland,

agroecosystems, and grassland/herb ecosystems. Although statistically less significant, the total number of users in Lanark Highlands was on the same order of magnitude as in Mississippi Mills.

Analysis of user intensity indicated elevated levels of economic activity in freshwater, wetland and grassland/herb ecosystems in Drummond/North Elmsley. Elsewhere, notably in agroecosystems in Greater Madawaska and grassland/herb ecosystems in Beckwith, and across ecosystem types in Carleton Place[3], relatively small areas resulted in very high intensity values. Rather than face this level of competition it is expected that users would locate the needed FEGS elsewhere, to the degree possible. For the three ecosystem types in Drummond/North Elmsley this was not the case. Here the ratio of the area of a given ecosystem type to total area of the CSD_{WS} was greater than, or on par with, that of other census subdivisions that were found to have high absolute levels of economic activity.

Direct competition between users for individual FEGS was assessed for all FEGS-ecosystem type pairings where NAICS coded economic data was available (Table 2.5). As before, Mississippi Mills was found to have the largest total number of individual economic entities in direct competition across quantifiable FEGS-ecosystem type pairings (shown in parentheses). Lanark Highlands had the second largest number of competitors across quantifiable FEGS, with the exception of 'presence of environment'. Here North Frontenac

[3] Carleton Place is unique among CSD within the watershed in that it is notably smaller and consists primarily of developed land (the town of Carleton Place). The level of economic activity is not appreciably larger than its downstream neighbour, Mississippi Mills, but it has considerably less land and water to provide the needed FEGS. For example, Carleton Place has 0.27 km^2 of freshwater compared with 48.47 km^2 in Mississippi Mills. For wetlands and agroecosystems the numbers compare as follows: 0.3 km^2 to 52.73 km^2, and 1.0 km^2 to 139.76 km^2, respectively. As such Carleton Place is likely drawing many of the needed ecosystem services from areas outside CSD boundaries. For these reasons Carleton Place was not included in statistical calculations, and was excluded from consideration for further analysis.

had higher levels of competition. This was likely driven by resource dependent businesses associated with a relatively strong tourism sector.

FEGS		ECOYSTEM TYPE							
		Freshwater		Wetlands		Agroecosystems		Grassland/Herb	
		CSD_{WS}	σ from mean	CSD_{WS}	σ from mean	CSD_{WS}	σ from mean	CSD_{WS}	σ from mean
Water	Maximum	M. Mills (302)	2.60	M. Mills (185)	2.55				
	Next Highest	L. High (124)	0.63	L. High (83)	0.72				
Open Space	Maximum			M. Mills (327)	2.39	M. Mills (327)	2.39	M. Mills (327)	2.39
	Next Highest			L. High (183)	0.94	L. High (183)	0.94	L. High (183)	0.94
Soil	Maximum			M. Mills (208)	2.41	M. Mills (327)	2.39	M. Mills (208)	2.41
	Next Highest			L. High (114)	0.91	L. High (183)	0.94	L. High (114)	0.91
Presence of Environment	Maximum	M. Mills (187)	2.75	M. Mills (187)	2.75	M. Mills (120)	2.70	M. Mills (120)	2.70
	Next Highest	N. Front (39)	0.10	N. Front (39)	0.10	N. Front (36)	0.36	N. Front (36)	0.36

Table 2.5 – Competition for individual FEGS by ecosystem type and census subdivision

The economic analysis has identified four CSD_{WS} with relatively high levels of economic activity: Mississippi Mills, Lanark Highlands, Drummond/North Elmsley and North Frontenac. Mississippi Mills and Lanark Highlands have been selected for further analysis due to their consistently high levels of absolute economic activity and intensity of economic activity across ecosystem types.

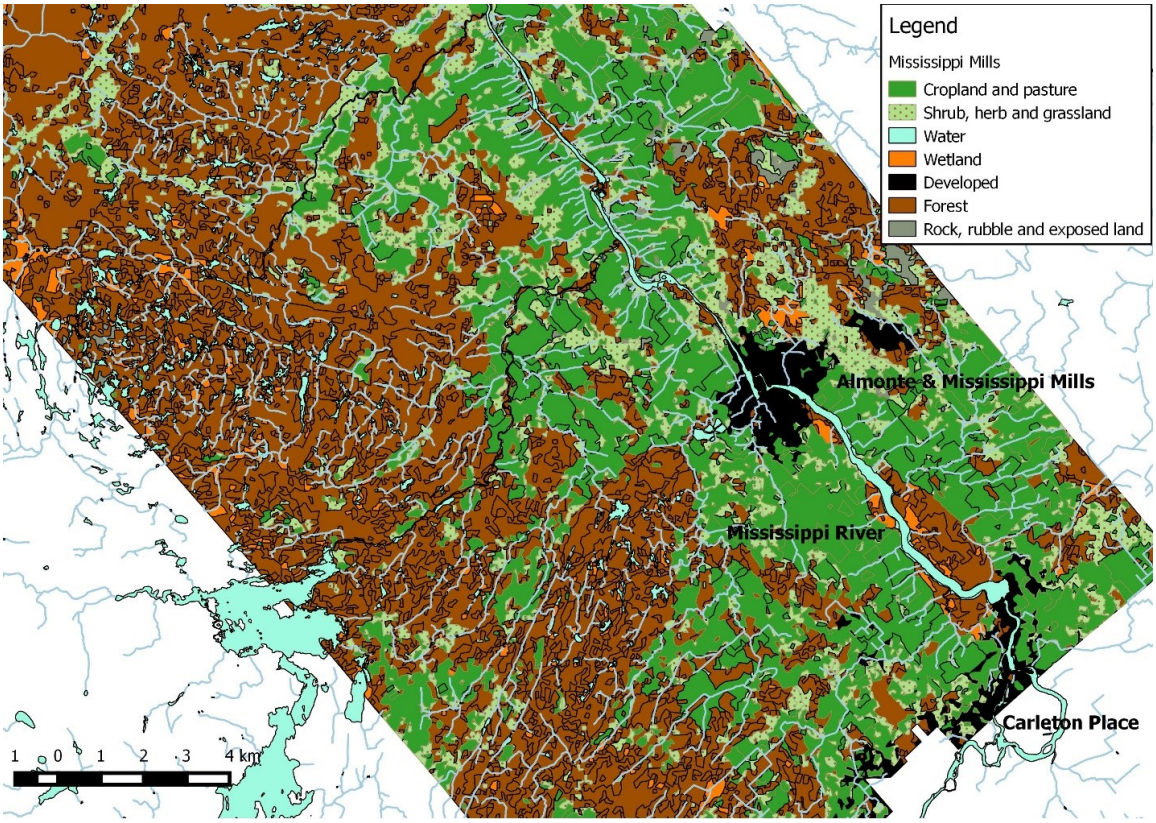
Shifting our attention to the natural features of these two CDS_{WS}, most wetland ecosystems in Lanark Highlands are located on private land with the exception of several along the Clyde River in the vicinity of the Village of Lanark. Meanwhile, almost all of the remaining wetlands in Mississippi Mills are on private land. Property rights are generally expected to eliminate direct competition for FEGS on private land. FEGS provided by freshwater ecosystems and wetland ecosystems not on private land may be better understood in terms of public goods (non-rivalrous and non-excludable), or common pool resources (rivalrous but non-excludable) (Costanza et al., 2014). However, exercising of one's property rights can affect ecological processes (Reeve, 1997), effectively shaping the provision and subsequent use of other ecosystem services. Externalisation through

ecological linkages in this manner may result in what functionally amounts to open-access conditions such that resources are rivalrous, non-excludable and openly accessible to any user (Lockie, 2013). In this way, a full definition of property rights is contingent on the entitlements of others as develop through systems effects stemming from geographical proximity and ecological linkages between different uses.

Conflicts between different claims are generally managed by relevant resource management institutions. For example, recreational activities such as fishing and hunting require a permit, as do large water withdrawals, the discharge of municipal and industrial effluents, and the operation of hydropower facilities. These instruments are in place to manage environmental resources thereby minimising conflict between competing uses. However these instruments may not be adequate, depending on systems effects arising from competition across ecosystem types and upstream-downstream factors. To assess whether economic risks may develop as a result of this competition, extant policy will be reviewed relative to a qualitative analysis of geographical patterns of demand and ecological linkages between users.

There is a substantial amount of agricultural activity along both sides of the Mississippi River in Mississippi Mills (Figure 2.3). This is comprised of largely perennial crop and pasture land, but includes annual cropland as well, with agricultural activities often situated immediately adjacent to the river. Grassland/herb ecosystems link agroecosystems and remaining wetlands to create a largely continuous structure. There are two larger communities in the CSD_{WS}, – Almonte & Mississippi Mills and Pakenham (not shown) – as well as observable spillover from Carleton Place, in terms of developed land. A handful of wetlands remain along the Mississippi River between Almonte and Carleton

Place. The location of these wetlands makes them available to recreational users, and potentially impacted by water quality and quantity conditions in the Mississippi River.

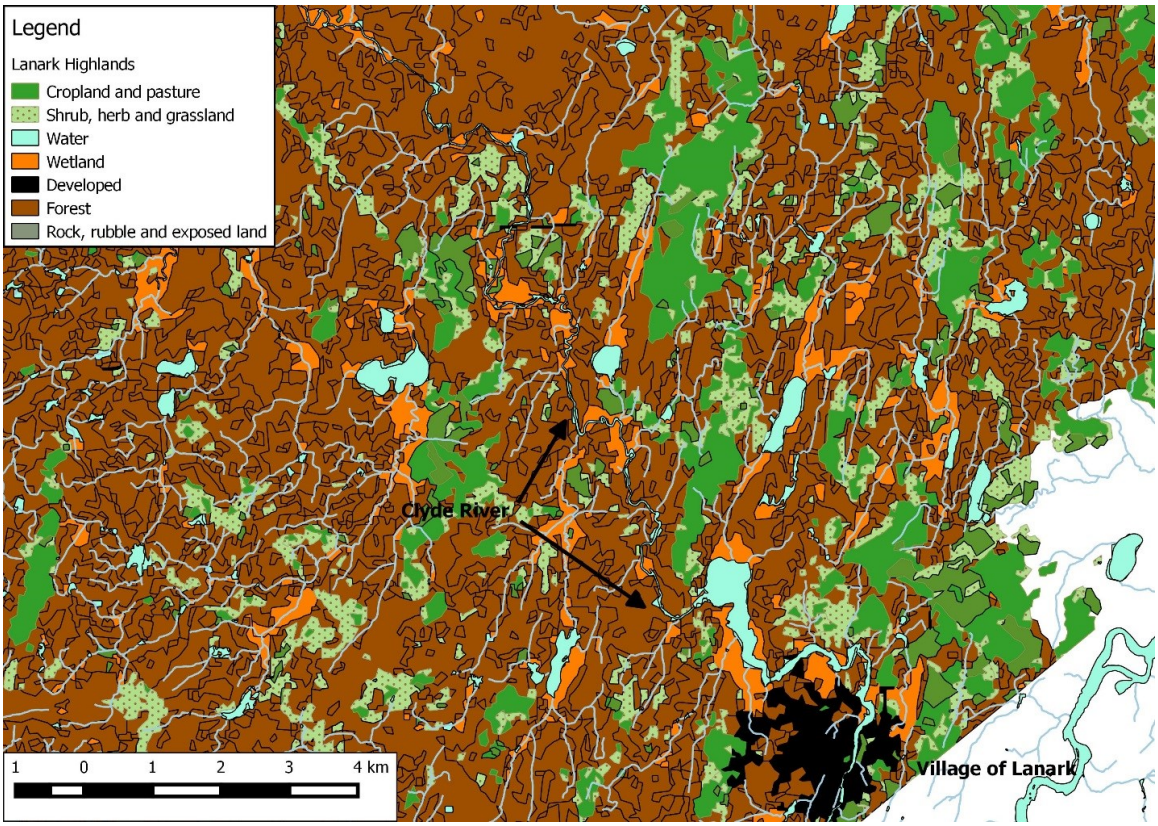


Grazers	133	Industrial dischargers	75	Residential property owners	54
CAFO	63	Industrial processors	68	Resource dependent businesses	81
Farmers	234				

Figure 2.3 – Portion of Mississippi Mills CSD_{WS} showing wetland, freshwater, grassland and agroecosystem clusters, and active economic entities

The Clyde River is a dominant feature of Lanark Highlands (Figure 2.4), flowing from the northwest corner of the CSD_{WS} to join the rest of the Mississippi River system just south of the CSD boundary. The total drainage area of the Clyde River is 614 km^2 with numerous wetlands located along its length. The Village of Lanark is the most significant settlement in the CSD and is located at the foot of the Clyde River. There is little reservoir storage within the Clyde River subwatershed and the Village of Lanark has annual flooding

and low flow problems (MVCA, 2006). Predominantly perennial crop and pasture land are located on tributaries upstream from the Village of Lanark, north of the Clyde River. As can be seen from Figure 2.4, most of these areas drain through wetlands, with numerous wetlands in close proximity to the Village of Lanark.



Grazers	78	Industrial dischargers	15	Residential property owners	10
CAFO	33	Industrial processors	12	Resource dependent businesses	15
Farmers	128				

Figure 2.4 – Portion of Lanark Highland CSD_{WS} showing wetland, freshwater, grassland and agroecosystem clusters, and active economic entities

Due to the potential for interference between beneficiaries, and relatively high levels of usage, publically accessible freshwater and wetland ecosystems were selected for further analysis in Mississippi Mills and Lanark Highlands, respectively. Further, since ecological systems are continuous across subsystem boundaries, and production of

ecosystem services does not necessarily coincide geographically with consumption of those same services (Naidoo & Ricketts, 2006; Vrebos et al., 2015), patterns of demand and competition effects that span ecosystem types were assessed, as were linkages between upstream and downstream users.

2.3.2 Existing Policy Regime and Risk Identification

The potential for economic risk to arise from competing demands for ecosystem services is curtailed by the protections put in place by existing policy. Conversely, gaps in policy coverage could allow such competition to go unchecked, increasing the likelihood that economic risks develop. Although Ontario's legislative and regulatory framework does not explicitly address ecosystem services (Miller & Lloyd-Smith, 2012), extant policy does provide some support for the provision of ecosystem services of societal importance.

Storage reservoirs in the upper watershed are operated according to protocols prescribed by the 2006 Mississippi River Water Management Plan (WMP). Specified uses include augmentation of minimum downstream summer flow conditions. However during drought events in 2001 and 2002, Crotch Lake, the largest and most important reservoir in the system, provided 100% of downstream flow (MVCA, 2006), and there are concerns that the system of reservoirs may not be able to meet minimum flow targets in the future (Casselman et al., 2011). Although the current plan is committed to maintaining minimum flow conditions in downstream reaches, natural resource managers indicated that there has been an increase in development in the upper watershed, and that this has resulted in mounting pressure to maintain water levels in populated reservoirs. This pressure is expected to continue to grow with increasing total value of real estate assets. Furthermore,

there is some indication that hydro-electric operators are seeking to revise the WMP, and that the process would likely be less inclusive of other interests than it was during development of the current plan. This could result in a revised WMP that less well informed and subsequently less able to balance the diversity of interests in the watershed.

Importantly, the WMP recognises that climate change will affect the future operating constraints for reservoir management. However, stating that current science is not able to predict future precipitation patterns, the WMP uses historical data as the basis for developing protocols for managing water levels in the reservoir system. While understandable, historical data will be increasingly less relevant to conditions going forward (Milly et al., 2007), raising concerns about management approaches that rely solely on this data. Meanwhile, natural resource managers have observed an increase in the number and intensity of precipitation events. These storms have had a tendency to flush landscapes in the Mississippi Valley, and have resulted in increased pressure for additional tile drainage to move excess water off flooded fields, potentially resulting in non-point source pollution of adjacent ecosystems. Vegetative filter strips are required under the *Nutrient Management Act* (2002) for all agricultural lands. However, drainage and grazing on agricultural land can reduce infiltration. Combined with more extreme precipitation events this can result in accelerated runoff with a higher pollutant load (Erwin, 2009). These flows may overwhelm the capacity of engineered filter strips. In Mississippi Mills, polluted runoff leaving agricultural lands will flow largely unimpeded into the Mississippi River. Climate change further exacerbates the potential for elevated levels of nutrients, pesticides and pathogens due to an increased frequency of low flow conditions throughout the watershed.

Thus, water quality in the Mississippi River is at risk in the Mississippi Mills CSD_{WS} due to the potential for increased runoff of a more polluted nature during peak events, compounded by a reduction in dilution resulting from lower stream flows. The sub-beneficiaries at risk are primarily recreational users of freshwater and the economic interests that cater to these users. These risks arise from competing demands for ecological resources between ecological subsystems. The use of ecosystem services on agricultural lands may affect ecological conditions in such a way as to inhibit the availability of ecosystem services in another ecological subsystem. The situation would be further compounded by upstream-downstream system dynamics should recreational users and land owners in the upper watershed successfully press for more water to remain in the reservoir system. Considering the increase in recreational development in the upper watershed, alongside climate projections of persistent low summer water levels in the lower Mississippi River, these upstream-downstream pressures remain a substantial concern in their own right.

Agricultural runoff in Lanark Highlands will be largely filtered by wetlands, protecting water quality in the Clyde River, though potentially at the expense of wetland health. This constitutes an important risk for Lanark Highlands, especially Kerr Lake, which is a provincially significant wetland and popular recreation and tourism destination, located immediately upstream from the Village of Lanark. Local officials confirmed that tourism is an important sector for Lanark Highlands, and this is expected to increase with longer summer recreational seasons. Kerr Lake and the Clyde River provide excellent boating and fishing opportunities, and the nearby Kingston & Pembroke Trail brings many visitors to the area.

Wetlands are covered by a number of legislative provisions. The drainage of wetlands in order to dewater agricultural lands and improve productivity is regulated under the *Drainage Act* (1990). Such actions are subject to approval by the Mississippi Valley Conservation Authority. Their mandate under the *Conservation Authorities Act* (1990) is to prohibit development in wetlands, or where it could interfere with the hydrologic function of a wetland. The Ontario Wetland Evaluation System (OWES) assesses ecological and human-use values (Ontario, 2014), and provides this information to municipalities for land-use planning purposes (Ontario, 2015b). Taken together these provisions do provide a level of protection. Actions that could affect the hydrological function of a wetland are regulated, whether they occur within the wetland or beyond. Ecosystem services are protected from development within provincially significant wetlands (as determined by an OWES assessment), to the extent these wetlands are identified in the relevant municipality's official plan. However ecological and human-use values are not safeguarded by either instrument from activities that occur beyond the boundary of the wetland. Moreover, municipal sources indicate that by-law variances are relatively easy to obtain. This has the potential to undermine municipal provisions, especially where there is a permissive municipal council. Municipalities for their part depend on new development to increase revenue. This has led to the overdevelopment of lakefronts in the watershed, with decisions pertaining to carrying capacity often being determined by the availability of land rather than the ecological capacity to absorb associated by-products.

In Lanark Highlands wetland services are at risk due to a general drying of wetlands resulting from lower soil moisture and longer dry periods in summer. Additional pressure

on wetlands may result from agricultural non-point source pollution breaching containment measures during peak events. The sub-beneficiaries at risk include recreational users of wetlands, as well as farmers affected by impacts on wetland habitat and pollinator populations (Zedler et al., 2005). From a systems perspective there is the potential for feedback between the construction of dugouts to provide water to agricultural operations, the drying of neighbouring wetland habitat, and reduced provision of pollinator services that are vital to agricultural productivity.

The preceding examination of existing policy has identified a number of critical gaps through which specific economic activities or interests could undermine the provision of ecosystem services elsewhere, constituting a potential economic risk. Pressure from property owners in the upper watershed combined with increased streamflow fluctuations and elevated potential for failure of vegetative filter strips, could result in poor water quality conditions and associated economic impacts in Mississippi Mills. In Lanark Highlands there is a strong policy framework protecting wetlands, in particular those that are designated as provincially significant. However the level of protection is a function of the degree to which these designations are reflected in the relevant municipality's official plan. Moreover, these protections do not extend to encompass impacts on ecological and human-use values resulting from activities that occur outside the wetland. I also found that reasonably comprehensive protections may be undermined by local political and fiscal concerns.

2.4 Discussion and Conclusions

This research extends existing ecosystem services literature by examining different uses of common underlying ecological resources, and assessing trade-offs in the form of potential economic risks resulting from competition between these different uses. Much of the existing literature focusses on supply-side trade-offs between different bundles of ecosystem services, whereas the approach taken here is essentially concerned with bundles of demand, and does so in a spatially specific way relative to existing policy safeguards.

Using ecosystem services to identify and map relevant linkages between social and ecological elements within the joint social-ecological system enabled the identification of competing interests and key systems attributes. For example, upstream and downstream social and environmental conditions were found to be determinant of how systems effects influenced underlying ecological processes and the ongoing provision of specific ecosystem services, potentially resulting in economic risks. Increased seasonal water use in the upper watershed coincides with a downstream need for water to maintain minimum flow conditions in summer, with climate change being a key driver of both increased upstream demand and downstream seasonal need for water. The degree to which the provision of services may be threatened by ecological impacts is in part a function of the aggregate pressure on the ecosystem. A resource base that is under little stress may exhibit normal functionality up to a point, but the ecological capacity to absorb the externalities of human activities decreases as these activities increase (Elmqvist et al., 2003; Mooney et al., 2009). Accordingly, the governance of water resources becomes doubly complex as landscapes are increasingly populated. As the competition between users for needed ecosystem services increases, ecosystems that are under pressure due to increased demand

and associated externalities may be simultaneously, and as a result, less able to supply those services. This in turn leads to institutional and economic pressures which can feedback through governance processes, further impacting ecological conditions. For instance, as recreational use in the upper Mississippi Valley watershed continues to grow we expect increased pressure from economic interests (such as residential property owners and recreational businesses) at a municipal level. While institutional safeguards exist, these conditions also present local governments with attractive opportunities for development and associated revenues. Although the combined interests of industrial, municipal and recreational users in Mississippi Mills comprise a formidable coalition, if upstream recreational users were to align with hydro-electric producers this would also constitute a powerful coalition. The Mississippi River Water Management Plan (WMP) guides management of water levels throughout the reservoir system. However the central role of hydro-electric operators in developing and implementing the WMP gives this coalition the institutional high ground, and could set the stage for increased tension around the management of water levels and flows, and the associated availability of ecosystem services. If bolstered by upstream land owners and governments, with little opportunity for resistance from dissenting interests, it is possible that hydro-electric operators might privilege upstream interests in a revised WMP where they overlap with those of hydro-electric production. This would almost certainly impact downstream provision of needed ecosystem services. In this example, climate change has caused changes in the ecological subsystem that would result in a periodic increased need for downstream flow augmentation. At the same time climate change has resulted in ecological changes upstream that have shifted the balance of power in the watershed. Where one party in the

upstream coalition has direct control over critical aspects of resource governance in terms of management of reservoir water levels, the system effects have come full circle: ecological changes have induced change in the social subsystem, which have the potential to feedback through resource governance mechanisms and result in additional negative downstream ecological impacts.

Methodologically, the reformulated FEGS-CS integrates ecological data with economic data at a local level. This supports a characterisation of current environment-economy linkages, and a systems-based assessment of extant policy protections and the potential for competition between users of ecosystem services to result in ecological change that could in turn result in economic risks through system feedback. The approach presented here does however have inherent limitations.

The complexity of ecological change at a population and species level in terms of varying species mobility, shifting composition and the fragmentation of ecosystems, remains an outstanding challenge. Complementing the model with an in-depth analysis of ecological processes could illuminate variation in the level of impact, and hence risk, to individual ecosystem services within a given ecosystem type. This would enable more detailed specification of economic risks, and it would feed into hierarchical analyses of ecological effects across scale, which in turn would open up new parts of the policy framework to assessment using these methods. This could include policy areas concerned with ecological populations including fisheries, biodiversity and endangered species legislation and regulations. However, there would clearly be trade-offs in terms of the feasibility of a more ecologically nuanced and comprehensive approach for resource constrained local governments.

More accurate georeferencing of economic data on business activities would improve quantification of competing economic actors and help limit the potential for false negatives. Detailed investigation subsequent to initial identification of areas of potentially elevated risk should limit concerns around the generation of false positives (i.e. where the model erroneously suggests an ecologically derived economic risk might exist). However false negatives could occur in the absence of georeferenced business data. For instance, a small number of closely clustered businesses competing for overlapping ecological resources could constitute an economic risk scenario that would not necessarily be captured by the approach presented here. This is primarily a matter of resolution. This study investigates risk at the level of the census subdivision in conjunction with a qualitative assessment of land cover to describe patterns and variation within the census subdivision. The size, nature and exact location of businesses would further increase the resolution of the analysis. Large, heavy users of a given FEGS that could have significant impacts locally within the watershed constitute another possible source of false negatives. Their impacts could be lost in the process of aggregation of their activities with others within their sub-beneficiary class, and subsequent averaging over available area of ecosystem type. However, where existing federal and provincial environmental assessment regulations are applicable this would effectively capture and manage these situations. In this regard the approach discussed here is seen as complementary to existing environmental assessment legislation. Natural resource managers and municipal officials are also able to provide valuable information about the specific nature of local economic activities that could help address limitations inherent in the model.

Using a systems-based approach, this study modelled linkages between two complex systems, the environment and the economy, and identified Mississippi Mills, Lanark Highlands, Drummond/North Elmsley and North Frontenac as locations with heightened use of ecosystem services at a local subwatershed level. Mississippi Mills and Lanark Highlands were selected for further analysis. Subsequent contextualisation of the economic analysis relative to patterns of land cover was used to assess two scenarios where ecologically derived economic risk could develop due to gaps in existing policy coverage. While these risks were exacerbated by climate change it is important to note that their origins are rooted in the existing social-ecological system, in terms of land cover, economic activities, existing policy and the extent to which it is implemented.

The approach taken in the empirical case study achieves two purposes. First, it presents a novel systems-based approach to modelling the complexities of joint ecological-economic systems at a local level using readily available data. This may be replicated elsewhere to produce new information and close gaps in existing policy regimes. Secondly, it helps mobilise the theoretical propositions of this study, engaging limitations with current approaches to modelling environment-economy linkages in order to lay down the empirical groundwork for further research into systems-based approaches to managing environment-economy linkages for increased social-ecological resilience. When observed through a systems lens, we may expect current approaches and institutional defaults to continue to arrive at sub-optimal outcomes due to implicit and fundamental shortcomings.

Depletion of a resource is often the result of rational action relative to institutional structures, property rights and incentives (Feeny et al., 1990). It is important to look beyond human greed as the driving factor and examine the institutional safeguards that are in place

(Holling et al., 1998). Simplified conceptions of ecosystems, and institutions that are orientated toward intensive production of individual resources, have led to degradation of the resource base that provides a multiplicity of resources (Folke & Berkes, 1995). To support sustainable use, property rights regimes must limit access to the resource system and the withdrawal of diverse units from the resource system (Ostrom & Hess, 2007).

Where property rights fundamentally begin with property and claims to it (Sproule-Jones, 2008), they are institutional arrangements that are distinctly of human origin. They have been described as pertaining to the relationship between the rights holder and others that have a claim on the rights of the holder (Hohfeld, 1919). When the provision of a needed ecosystem service is affected by ecological impacts resulting from the activities of others, adjacent or elsewhere, then at an ecological level affected rights holders arguably have some claim on the rights of those other actors, including private property owners.

Where property rights can be understood as bundles of entitlements, a duty of care is often in place as a condition for access. For instance, riparian rights are subject to reasonable use, and rights entitlements change alongside understanding about what constitutes "reasonable use" (Sproule-Jones, 2008). Ecological linkages that connect actions taken upstream with economic outcomes downstream in Mississippi Mills have implications for what constitutes "reasonable use" in the upper watershed.

This research illustrates how the exercising of one's property rights can exert ecological externalities which in turn affect the capacity of others to exercise their own property rights. The degree to which this occurs is a function of land cover, land use and extant policy and regulation. For instance, vegetative filter strips are required under the *Nutrient Management Act (2002)*. However climate change introduces a number of

additional factors that can result in agricultural activity on private land affecting the provision of ecosystem services elsewhere, and this can introduce an element of economic risk while also undermining associated property rights. Minimising economic risk and sustainable use are compatible objectives for institutional design. These require understanding the nature of the resource (Feeny, 1990), and how it is governed. It is landscape dependent, a function of land cover and patterns of land use, and effective management will require moving away from simplifying assumptions toward embracing systems-based approaches.

Chapter 3 - Cultural Values and Ecosystem Management at the Subwatershed Level[4]

The valuation of ecosystem services is often centred on the proper integration of ecological and economic values but overlooks the social and cultural dimensions of ecosystem changes. These factors are essential for sustainable outcomes because they underlie preference formation, affect ownership over decision-making, and may help ensure that primary non-market services needed to maintain ecosystem functions are sustained. In this study, we propose a conceptual framework for ecosystem service valuation at a local subwatershed scale that employs participatory democracy and discourse-based methods to elicit and develop values systems while at the same time serving as a decision-making process. Incorporating cultural theory and perceptions of risk tolerance, the framework discusses the importance of cultural considerations to achieving intergenerational equity, a precondition for sustainable development. The framework is applied to the case of Chelsea, Quebec, and used to examine the community consultation component of a policy development process.

3.1 Introduction

Regional groups of municipalities and their surrounding rural areas have shared and interconnected interests that are linked through the water-based ecosystem services upon which regional economies depend. Two key challenges for sustainable water resources management are the integration of decision-making between neighbouring jurisdictions,

[4] Retallack, M., & Schott, S. (2014). Cultural values and ecosystem management at the subwatershed level. *Environmental Policy and Governance*, 24(5), 351-363.

and between water-based ecosystem services and related economic activities. Although not meeting our expectations in terms of effectively integrating and balancing the range of interests (Medema et al., 2008; Jeffrey & Gearey, 2006; Saravanan et al., 2009), Integrated Water Resources Management (IWRM) does provide a framework for more coordinated and holistic water management, while developments in valuation of ecosystem services have improved the representation of ecological factors in traditional decision-making processes.

Valuation processes result in either a monetary or non-monetary measure of value for a given ecosystem service, set of services, or ecosystem, with monetary values more readily applicable to traditional cost-benefit type analysis. However monetary approaches may not capture the broader dimensions of social value (Farber et al., 2002); the value of non-market services (Jenerette et al., 2006); or ecological values (Vatn & Bromley, 1994), including primary services that sustain the ecosystems themselves, thereby supporting the ongoing provision of secondary services (Gray et al., 2003). As such they often underestimate the actual value of services provided. Fisher et al. (2008) address the importance of valuing non-market ecosystem services and suggest that a mix of formal and informal mechanisms should be used to ensure the continued delivery of these benefits.

Non-monetary approaches to valuation of ecosystem services are often centred on questions concerning the social nature of 'value,' for instance the manner and extent to which we assign importance to various ecosystem services. In its simplest and most straightforward form this may entail ranking and prioritisation (Ananda, 2007; Randhir & Shriver, 2009). However this does not necessarily address the central criticism of monetary approaches, that valuation through aggregation of individual values and preferences is

inappropriate for public goods where groups of people have a shared interest (Farber et al., 2002). Where the value of something is in its contribution to the realisation of a user-specified goal, and with societal goals flowing from the normative and moral frameworks that guide our beliefs and actions, there is a relationship between the act of valuation and the values upon which this activity is based. This speaks to the importance of social involvement in the process (Farber et al., 2002). However this in turn may limit "value" to being a measurement of worth relative to human goals, largely excluding the interests of ecosystems and non-human species from the calculation, except where their interests intersect our own. It is particularly important to incorporate ecological dimensions of value whenever a given ecosystem approaches a critical ecological threshold. In the context of irreversibility (Bockstael et al., 1995) and ecological uncertainty we need to shift from utility valuation to a risk avoidance perspective (Limburg et al., 2002). Finally, these values are subject to change and are not always obvious to decision makers and stakeholders.

The next section will review the methodological approach taken in this study. This will be followed by the elaboration of a novel decision-making framework which is then used as an evaluative tool to assess a policy development process in Chelsea, Quebec, against the theoretical arguments that are embedded within the framework. The study will conclude with a discussion of the broader relevance of the framework and empirical findings for environmental governance.

3.2 Methodology

Building on the discussion so far, this study examines key concepts of preferences and cost benefit analysis and integrates these into a conceptual framework for a values-based

approach to sustainable water resources management. Encompassing ecological and cultural dimensions of risk, the framework integrates deliberative democratic processes with ecological and socio-cultural expert input, to inform and embed valuation within decision-making in a closed-loop iterative learning process. The process enables values convergence through exposure to new information, while establishing vital channels for communication between stakeholders and decision makers. The conceptual framework is then used to assess the case of water and wastewater infrastructure planning in Chelsea, Québec, and highlight challenges and opportunities for the integration of cultural values within a realistic political process at the municipal level.

3.3 Public Participation and Small Group Processes

Public participation, as a core component of IWRM (Bandaragoda & Babel, 2010), offers a range of benefits. Meaningful inclusion can bring forward knowledge and information instrumental for issue identification, problem definition, and for more accurate and sophisticated policy responses (Watson, 2004). Further, participation in the decision-making process gives rise to the legitimacy of its outcomes and may lead to greater ownership of subsequent decisions (Fisher et al., 2008). Drawing on the work of Rawls (1971), and his assertions of the importance of public deliberation and access to the political agenda, Cohen (1997) suggests that democracy, when properly conducted, "involves *public deliberation focussed on the common good*, requires some form of *manifest equality* among citizens, and *shapes the identity and interests* of citizens in ways that contribute to the formation of a public conception of common good." It is through these means, he argues, that members of a pluralist democratic society will come to

understand themselves and their own legitimate interests. This approach to public participation is consistent with a broader shift from organisation-centred, hierarchical, process-orientated institutions to ones that are citizen-centred, participatory, consultative and results-orientated (Bourgon, 2007; Pal, 2010).

However, in practice deliberative approaches have distributive dimensions and can lead to conflict as well as cooperation (Lane et al., 2004). Asymmetries in power and social status (Shaw, 1981), capacity to participate, and past experience can all undermine the process (Ansell & Gash, 2008), and may entrench power differentials (Neef, 2009), while a focus on consensus may lead to politically workable solutions in the place of ideal outcomes (Nowlan & Bakker, 2007). Facilitation and mediation are therefore essential; however they may lead stakeholders to suspect government coercion (Singleton, 2002). If processes are not seen as legitimate this can create the conditions for conflict and widespread non-compliance (Adams et al., 2005). Furthermore, there can be accountability issues if decisions are made outside of democratically elected government bodies. This is of particular concern where a robust civil society does not exist and deliberative approaches may amount to a transfer of power over public and private resources to private interests (Lane et al., 2004). Finally, while deliberation can create social capital, potentially strengthening democratic processes, it also takes time (Gunton et al., 2003). For water resources management such inclusive and consultative processes may undermine adaptive capacity by inhibiting the ability to be decisive and respond rapidly to new information (Engle et al., 2011).

Nevertheless, small group processes are increasingly of interest for the formulation and elicitation of values for ecosystem services (Costanza, 2001; Farber et al., 2002).

Group processes are known to provide a corrective function where there are asymmetries or incomplete information (Kaplowitz, 2000; Wilson & Howarth, 2002), with information exchange in a small group setting exposing individuals to new ideas (Raymond et al., 2009). This is an important mechanism for the shifting and changing of values (Costanza, 2001). As Buchanan (1954) puts it, "individual values can and do change in the process of decision-making." Through discourse-based valuation of ecosystem services small groups are able to exchange information and move past individual interests to where they are valuing public goods in terms of widely held social values (Wilson & Howarth, 2002). Furthermore, since existing inequalities and asymmetries in access to benefits have a strong effect on the shaping of future patterns (Martinez-Alier & O'Connor, 1996), the achievement of intra-generational fairness is important for achieving equitable outcomes over longer time scales (Wilson & Howarth, 2002). Cohen (1997) provides four principles for an *ideal deliberative procedure*:

1. It should be free – consideration of proposals is not constrained by the authority of prior norms or requirements.
2. No force except that of the better argument is exercised – reasons are offered with the objective being to bring others to accept one's position given their different views and commitment to the process.
3. Parties are formally and substantively equal in voice and access to agenda – existing power distributions do not shape opportunities to contribute nor do they play an authoritative role.

4. It aims to arrive at a rationally motivated consensus – the process may end in voting, but if so it still benefits from committed effort to pursue a reasoned path to consensus.

While these four principles are crucial for informing democratic decision-making they do not guarantee that the deliberation process has any impact on, or link to, the official decision-making process. Participants need to know in advance if a given deliberation process is merely a town hall information session or if it is an integral part of the decision-making process. If stakeholders feel betrayed they will lose confidence in the deliberation and decision-making process, and this could seriously impede the other four principles. Most importantly, any given consultation process is not independent from previous processes. Individuals will assess the benefits of participation relative to the perceived utility of prior participation and the degree to which their participation impacted decisions made. We, therefore, propose to add the following 5th principle:

5. The consequences of the deliberative process for decision-making must be clear to all participants from the outset.

By ensuring a level of accountability, the additional principle is crucial for a constructive, meaningful process that works to guarantee that decision-making is transparent and consistent with the other four principles.

With institutional support and effective facilitation, small group processes can enhance effectiveness and legitimacy, help inform and guide guide collective decisions, and elucidate common interest within decision settings that are inherently complex and normatively charged. This will develop valuation outcomes in a way that systematically departs from standard approaches involving the aggregation of individual preferences

(Webler et al., 1995; Howarth & Wilson, 2006). Accepting that a range of tensions and trade-offs have been associated with deliberative approaches, by leading decision making toward the common interest, small group processes may serve to enhance intra- and intergenerational fairness, ultimately underpinning more equitable natural resource use, a core component of sustainable development.

3.4 A Cultural Bottom Line

Viewing sustainability as an inherently long-term proposition, the importance of understanding the shifting nature of preferences comes into focus (Norton et al., 1998). Looking at values and behaviours in the context of social institutions and cultural theory, O'Riordan & Jordan (1999) have shown how an individual's preferences are attached to and influenced by their worldview. Further, fairness considerations have been found to affect the behaviour of individuals, as well as decision makers, and therefore have direct implications for environmental valuation and the handling of policy issues with distributive dimensions (Johansson-Stenman & Konow, 2010). Syme et al. (2000) argue that, "there is a dynamic and complex association between temporal, interpersonal, and environmental fairness philosophies [and that] these are tempered by group association and self-interest." Where cultural preferences are unstable and endogenously defined through association and bonding, the formulation of an individual's values and preferences is influenced by socio-cultural context and learning through information sharing and social discourse (Chee, 2004).

Perceptions of risk and subjective risk are increasingly important for risk assessment (Williamson et al., 2005). How a problem is defined has embedded

assumptions about how the problem was caused, who or what played a role in this, and how the problem can be solved. Aspects of the various perspectives on a problem are often tied to shared cultural understandings (Dietz et al., 1989). This can be very influential among undecided members of a cultural group and shape their preferences toward different solutions. Environmental problems are inherently complex, full of uncertainty and are hard to understand. To a certain extent detailed technical analysis is required to assess conditions and evaluate risk. In the absence of information and understanding people may be prone to unreasonable fears or expectations, or undue influence.

In the context of our evolving preferences, those of future generations are not independent from current preferences, but rather are a continuation and a product of them. Culture stores these preferences and carries them forward through successive generations (Norton et al., 1998). Furthermore, landscapes can have strong cultural significance and are the result of complex interactions between cultural, political and ecological processes. In this way landscapes are in part a result of culture and at the same time a determinant of culture (Adger et al., 2009). Simply put, a change in landscapes can cause a change in culture. It is therefore important for culture to have a role in valuation and decision-making that will affect these landscapes. The loss and transformation of ecological systems is often irreversible, resulting in not only environmental but also cultural and social implications.

Culture is constantly changing under an array of influences, and these changes are reflected in shifting values systems. For individuals and their societies, actions are shaped by cultural and societal norms and beliefs. These in turn shape knowledge, experience, preferences and perceptions of risk, and the landscapes and environmental spaces that in turn help shape culture. Through this cycle and interconnectedness of cause and effect,

values and culture, the relevance and importance of cultural dimensions to the valuation of ecosystem services comes into crisp focus.

3.5 Conceptual Framework

Valuation of water-based ecosystem services is an important mechanism to clarify the environmental dimensions of decision-making. However existing approaches based on traditional economic analysis and the aggregation of individual preferences are flawed in three critical ways:

- They assume exogenous preferences and do not allow for values to shift towards shared societal interests.
- The role and importance of culture to the formulation of preferences is not well understood.
- Where environmental thresholds are involved these limits are often not adequately understood and accounted for in the formulation of individual preferences, thereby undermining ecological sustainability.

Taken together these factors can lead to issues of intra- and intergenerational social inequity where decision criteria, established through valuation processes, fall short of collective interests and set the direction for continued erosion of equitable and sustainable access to environmental goods and services. It is essential to incorporate a culturally sensitised, values-based component within an integrated valuation – decision-making process. Our suggested framework (see Figure 3.1) builds on the ecosystem function-analysis model of de Groot et al. (2002, 2006). We add an iterative process that shapes

and identifies social and cultural preferences, and perceptions of risk, and at the same time becomes part of a decision-making process that satisfies Cohen's revised five principles for an *ideal deliberative procedure*.

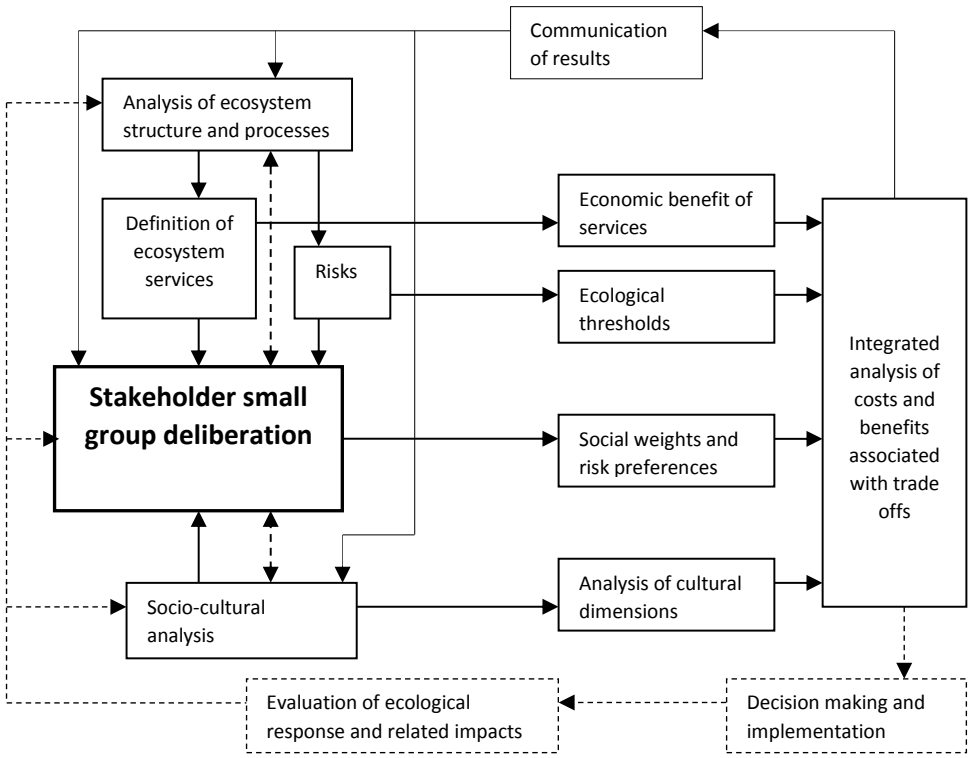


Figure 3.1 – Conceptual framework for culturally sensitised, values-based environmental decision making

Beginning with small group deliberation by all involved stakeholders about their priorities, cultural values and what they recognise as important ecological services and systems, this process enables the exchange of information that is essential to moving away from making decisions based on raw opinion toward some level of mutual understanding. In the context of watershed management, upstream and downstream users have different preferences, as do private sector and industry stakeholders, while individuals residing within different political jurisdictions most directly connect with different democratically

elected decision-making bodies. Where small groups are generally defined in the social psychology literature as consisting of between two and twenty individuals (Wilson & Howarth, 2002), encompassing this diversity presents certain challenges that will need to be addressed in a manner appropriate to the context of a specific application of the framework.

Since cultural groups often migrate toward a shared perspective, this limits the diversity of information concerning projected risks, and could lead to a culturally shared and subjective perception of actual risk (Davidson et al., 2003). Depending on the circumstances, historical factors may have shaped existing trust and power dynamics, resulting in barriers to information exchange. Recognising that culture is a determinant of values, and that decision-making outcomes can affect culture, expert socio-cultural analysis provides stakeholders and decision makers with information concerning potential risks, both to culture and from potential bias in decision-making.

Valuation processes often begin with a functional analysis**.** Coupled with the complexity of ecological processes, the connection between human activity and ecosystem integrity speak to the need for expert input into valuation and decision-making processes (Randhir & Shriver, 2009). Looking first at ecological structure and processes, these give rise to ecosystem functions, which in turn support specific services of interest (de Groot, 2002; Posthumus et al., 2010). This information is communicated to small groups of stakeholders to provide context for more informed deliberation of values and priorities. The manner in which these services support various economic activities within the watershed is subsequently characterised to highlight areas of key economic interest. In addition, expert analysis of the interconnectedness of ecological structures and processes

can illuminate ecological thresholds and the potential for irreversible effects associated with specific activities and impacts (Fisher et al., 2008).

With the support of expert analysis, small group deliberation produces a values-based assessment of local priorities. This statement of social weights and risk preferences, along with information concerning economic benefits, ecological thresholds and cultural factors, are communicated to decision makers to inform an integrated qualitative and quantitative analysis of the benefits and costs associated with different policy options. Drawing on a more complete set of information, this integrated analysis ensures more balanced decision-making and supports sustainable use of natural resources (de Groot et al., 2002). The framework requires that results from the integrated analysis be communicated back to the small group and expert bodies. This accomplishes a number of goals. The act of communicating results back to input agents makes bare the trade-offs involved and presents an opportunity for experts and stakeholders to challenge the relative weighting of different factors. The analysis will have incorporated economic aspects, potentially making new information available to stakeholders. Presented within discrete policy bundles, experts can respond directly to specific ecosystem scenarios and analyse ecological structure and processes in the context of possible actions, illuminating potential socio-cultural and ecological risks, while providing a contextualised description of the projected availability of ecosystem services. The latter informs a reassessment of economic aspects and environmental thresholds. Alongside revised social weights and preferences, subsequent government analysis incorporates expert predictions concerning ecological response to selected policy options, and how these changes could affect society, its economy, and both natural and cultural landscapes. This iterative process of government

analysts communicating back to stakeholders and experts may continue, or the government may move forward with decision-making. Following decision-making and implementation, actual ecological response and related impacts are evaluated. These now describe the current condition of the human-modified, semi-natural ecosystem (MEA, 2005; Oikonomou et al., 2011), and provide an ecological baseline and starting point for subsequent decision-making cycles, supported by ongoing monitoring and learning.

From a process perspective, stakeholders should see their contributions affecting the course of decision-making. In accordance with our Principle 5, the role that the outcome of the stakeholder deliberation process plays in higher level decision making needs to be specified upfront. For instance, the outcome of this process could go as far as being binding, to the limit of its analysis, unless overruled by higher-level authorities (e.g. municipal council, regional governments or other governing bodies). This amounts to a higher and more meaningful level of engagement that can result in more ownership of decisions made and smoother policy implementation (Ananda, 2007). Further, a genuine sense of empowerment, influence over decisions made and legitimacy of outcome may reinforce democratic ideals and result in social capital and a willingness to participate that will benefit subsequent processes. This moves a participatory process beyond the limitations of consultation to a more complete and collaborative partnership between government, expert communities and watershed citizens, resulting in an improved IWRM governance structure (Watson, 2004), as well as, potentially, increased adaptive capacity (Engle et al., 2010).

Alongside an evaluation of socio-economic, cultural and environmental impacts, the process itself should be evaluated. Cohen's (1997) *ideal deliberative procedure* will be

instrumental in this regard and can be used to assess whether consideration of proposals was constrained by prior norms or requirements, whether deliberation was free and reasoned, whether all participants had a free voice, if there was a reasoned path to consensus or an acceptable voting process, and whether there was transparency and accountability at each step of involvement (our 5th principle). Recalling that environmental governance is in part perceived to be a matter of justice, a fair process of small group deliberation should result in a convergence of values toward shared societal objectives. Tempered with environmental and economic insight, the outputs of decision processes that embody these values will generally be seen as just and fair, and by extension legitimate. In particular, it is essential to assess cultural impacts. While culture is a vessel for our values, it is also influenced by changes in the world around us brought on by the effects of decisions made. The next section will illustrate the framework, in particular the cultural dimensions of value formation, and the importance of the principles for an *ideal deliberative procedure*, by looking at social preference formation and sub-watershed management in the community of Chelsea, Québec.

3.6 Case Study

Many rural communities eventually come to a crossroads where major decisions on future development must be made. Chelsea, Quebec, is a typical case of a rural community on the brink of becoming a town with different infrastructure costs, alongside cultural and environmental implications. We will use our framework to illustrate how deliberative small group stakeholder consultations could have been better integrated in the overall visioning and decision-making process. In particular, we will focus on data acquired from one

specific workshop in October 2011, and how it relates to the framework presented in this study. Recognising that municipal decision making is part of a complex political process, we cannot say whether the outcome would have been any different had the framework been deployed. However this analysis does provide insight for other communities, and for Chelsea, in terms of how to better structure the sequence and iteration of decision making differently in the future.

Chelsea is a rural municipality of 6703 people (Statistics Canada, 2010) located in the Gatineau River watershed, a major tributary to the Ottawa River. Bordered on the west by a protected area (Gatineau Park), and on the east by the Gatineau River, there is a small town centre called Old Chelsea. Chelsea has ample outdoor recreational opportunities, which constitute a core interest for many residents, as well as an active arts and music community (see www.culturechelsea.ca). The community is known for its strong stand on environmental issues and has gained a reputation for being a progressive, environmentally friendly community. It was the second Canadian community to ban pesticide use on all residential properties in 1998, and it started a campaign in 2003 (known as H_2O Chelsea) to test groundwater for contaminants. The community is situated within a fragile subwatershed ecosystem that provides important ecological habitat. Until recently development was limited to one-acre or two-acre lots in order to avoid high-density developments, and to limit the obstruction of wildlife corridors and pressure on groundwater resources. However the town is located 15 minutes from Ottawa, a city of nearly one million inhabitants, and has attracted the attention of developers.

In 2008 a group of citizens approached the municipality for funding to develop a vision for Old Chelsea. The municipality supported the initiative and in May 2010 the

summary report, "Vision Chelsea," was released. The Municipal Council subsequently initiated the development of a Special Planning Program (SPP) and accompanying regulations to implement the vision. The message from the visioning process was clear: the residents of Chelsea wanted to maintain the rural character of the village. There was wide consensus on the need for better traffic control and enhanced alternative transportation networks, including better public transit and improved walking, bicycling and skiing trails. The details of how this vision would be implemented were to be outlined in a more specific plan for this area called a Plan Particulier d'Urbanism (PPU).

In December of 2010 the municipal council decided to examine three specific policy issues: water and sewer infrastructure, the development of a major community centre that included a hockey rink, and the PPU process. Although these three issues are interrelated, three separate meetings were scheduled to consult the public. In February of 2011, the Vision Chelsea Committee officially resigned stating that, "water infrastructure, not the vision of residents, [was] now driving Council's decisions on the future of Chelsea's centre core" (The Low Down, 2011a). The municipality went ahead with the planning process, and the resulting PPU had sections of Chelsea where the intensity of planned development was at least four times higher than previously permitted, seriously challenging the rural character of the community. The final version of the PPU and regulations were adopted on October 3, 2011, and entered in force on December 5, 2011 (Municipalité de Chelsea, 2011).

On March 14, 2011, Chelsea Municipal Council selected Hydro-Québec site No 2 on Mill Road to construct the future wastewater treatment plant (Municipality of Chelsea, 2011a). Ten days later 250 residents gathered at the municipal building and conducted a

first-ever protest march to demonstrate against water and sewage projects under the motto "Smart Water, Smart Waste" (The Low Down, 2011b). On June 21, 2011, Chelsea Municipal Council established an Ad Hoc Committee on Sustainable Infrastructure (SIC). Their role was to examine criteria proposed for site and technology selection, review studies provided by a consultant, and examine mitigation measures for the impacts of water and wastewater treatment facilities, and ultimately make recommendations to Council for criteria to be included in subsequent proposal documents for these projects (Chelsea Municipal Council, 2011). On October 12, 2011, a public workshop was held to gather input on site and technology selection criteria. The purpose of the meeting was to add to existing criteria and weight priority areas of focus and importance through plenary and voting (SIC, 2011a). The format of the public meeting included breakout groups formed according to numbers that were randomly given to individuals upon entry. Voting assigned priority to criteria with the overall results reviewed in plenary. Voting results were to be used by the SIC to provide recommendations to Council and assist the consultant in their ongoing engineering study. Tables 3.1 and 3.2 provide top priorities along with weighting data from the public meeting.

SITE CRITERIA – FINANCIAL		Votes	Percentage
New	Exploration of all options (costs & benefits)	51	31 (S,R)
New	Costs of piping and pumps, operating costs	49	30 (S,R)
	Cost of necessary site improvement	30	18 (R)
SITE CRITERIA – ENVIRONMENTAL			
	Impacts on significant ecosystems (e.g. streams, wetlands, nearby water courses, significant forests)	78	44 (S,R)
New	Impacts on health and safety	63	36 (S)
	Impacts on wildlife habitat, habitat connectivity or wildlife movement	15	8
SITE SELECTION – NEIGHBOURHOOD			
	Distance to residences (possibility of proximity influencing effects of noise and odour)	86	48 (S,R)
New	Minimising delays of implementation	21	12
	Impact of increased traffic and dust on quality of life	20	11

Note: "New" indicates preferences that originated at the October 12th workshop; "S" identifies criteria mentioned in the summary section of the SIC report to Council; and "R" identifies criteria addressed specifically in recommendations section.

Table 3.1 – Top preferences for site selection criteria (SIC, 2011b)

TECHNOLOGY CRITERIA – FINANCIAL		Votes	Percentage
	Effluent quality that meets more stringent proposed federal standards	57	31 (S,R)
	Operating costs (including maintenance, repair, effluent compliance monitoring, etc)	38	21 (S,R)
	Track record/ reliability	27	15 (R)
TECHNOLOGY CRITERIA – ENVIRONMENTAL		Votes	Percentage
	Effluent quality that meets more stringent proposed federal standards	65	31 (S,R)
	Track record	39	19 (S,R)
	Urgency of connecting Old Chelsea due to water quality issues	32	15 (S,R)
TECHNOLOGY CRITERIA – NEIGHBOURHOOD		Votes	Percentage
New	Noise/odour/aesthetics/light pollution	51	35 (S,R)
	Traffic (as related to frequency of maintenance, sludge removal, etc)	36	25
	Real estate values	28	19

Note: "New" indicates preferences that originated at the October 12th workshop; "S" identifies criteria mentioned in the summary section of the SIC report to Council; and "R" identifies criteria addressed specifically in recommendations section.

Table 3.2 – Top preferences for technology selection criteria (SIC, 2011b)

As a result of the small group process a number of new concerns came forward, many of which were weighted heavily by citizens in attendance. Specific examples include: exploration of all options; operating costs; impacts on health and safety; and, noise, odour, aesthetics and light pollution concerns. By supporting further analysis of costs and benefits of all options, over 30% of those in attendance do not appear to have accepted the municipality's presented alternatives. These were limited to a proposed wastewater

treatment facility at the Mill Road site, with a primary focus on technical requirements of the proposed treatment technology while not encompassing social or environmental factors. The addition of operating costs to the selection criteria identified a vitally important issue. The SIC overlooked health and safety in their environmental criteria and were instead focussed exclusively on the wellbeing of ecological features and wildlife. However, once identified and supported by workshop participants, this concern was not referred to specifically in the recommendations to Council. Noise, odour, and aesthetics were identified on the long list of considerations presented by the SIC, however light pollution was added and it was concluded that the four concerns were all important and should not be prioritised separately (SIC, 2011b).

In reaction to the SIC report the municipality held an information session on July 5, 2012, and made their recommendations available online in the form of a power point presentation (Municipalité de Chelsea, 2012). According to the municipal council, selection criteria were weighted relative to the importance (presumably based on the workshop results and recommendations by the SIC), and cost (not benefits) of different options. The presentation identified two major options for the siting of the potential treatment site: the original proposed site on Mill Road, and a site on Hudson Road near Hydro-Québec lands. The municipality recommended the Hudson Road site due to its proximity to the Gatineau River and lower socio-economic impact, constituting a change in direction that may have been influenced by group deliberation at the October 12th workshop.

In 2012 Council adopted four borrowing by-laws for the construction of wastewater and drinking water treatment systems for the built and unbuilt sectors of the Centre-Village

sector. Later in July of 2012 the municipality approved all four borrowing by-laws for $22.455 million in order to go ahead with the water and sewage infrastructure projects. As of October 2013 the Municipality of Chelsea was waiting for approval of its borrowing by-laws by Quebec's Ministry of Municipal Affairs (MAMROT). In addition it requires approval from the Ministry of Environment for the construction of the wastewater treatment and drinking water plants, Hydro-Québec for the water intakes and wastewater outlets, and the National Capital Commission (NCC) for the installation of pipes and the pumping station (Municipalité de Chelsea, 2013).

3.6.1 Application of Conceptual Framework

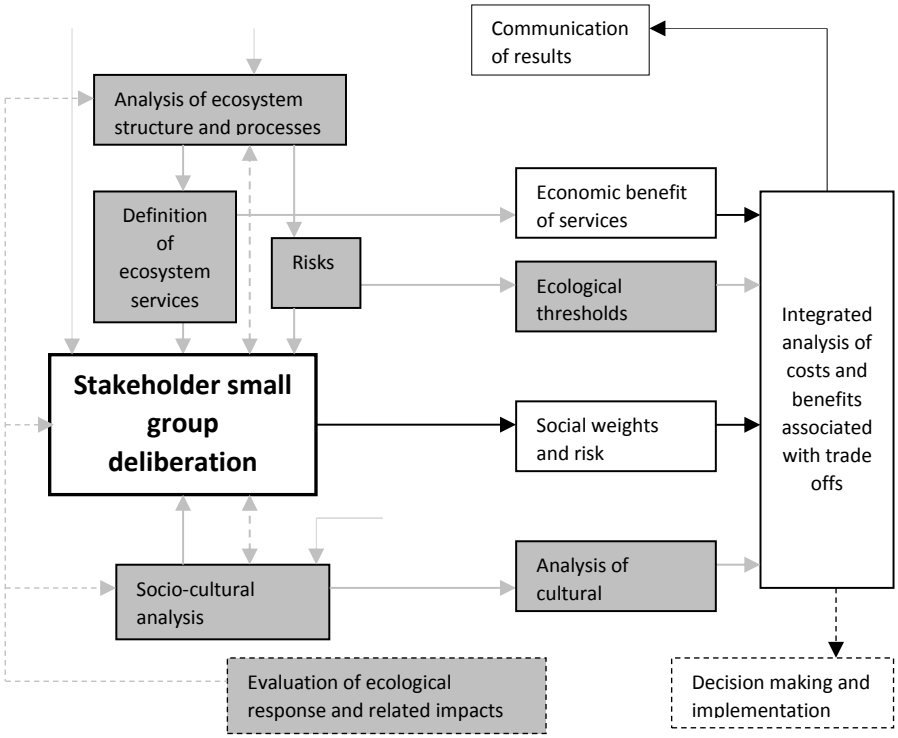


Figure 3.2 – Application of framework to sustainable infrastructure deliberation process in Chelsea, Québec

Although the October 12th workshop occurred late in the planning process some aspects were in general alignment with the logic of our conceptual framework. Having

been primed with relevant information, albeit limited in focus to support discussion of selection criteria, small group breakout discussions explored the issues. This allowed for the exchange of information and perspectives between individuals, with results being fed back into and informing the larger group prior to a weighting exercise. This created an opportunity for values to shift and converge, ultimately leading to a more equitable and community-orientated prioritisation of preferences. Small group processes implemented on October 12th generated new information in all criteria categories (see Tables 3.1 and 3.2). Community members with environmental expertise participated in the process, and helped characterise ecological concerns as well as relevant ecosystem services (beyond the provision of drinking water and the receiving of treated wastewater). This informed small group discussions, but no systematic analysis of ecosystem structure, processes or potential risks was presented. Consequently no ecological thresholds could be identified, for example the amount of water that could be extracted and treated without any significant harm to recreation and other ecosystem services. Only the economic benefits of piped water services were considered. No cultural analysis was conducted and as a result social and cultural implications of the different options were not assessed. For example, the location of the water treatment plant would have significant effects on future development in the vicinity of the chosen site, and the adoption of different sized systems would have implications for the overall growth and density of the community. Available information suggests that there was little consideration of community culture and how decisions being discussed could impact the character of the community. No connections were made between the Vision Chelsea process, the PPU, the new community centre development (which requires new water infrastructure), and consultations concerning water and

wastewater infrastructure. Had Vision Chelsea been more wholly embraced and integrated with the analysis and decision making around development and the subsequent need for centralised municipal water and wastewater infrastructure, the results may have looked quite different. The collected information from this group deliberation workshop could not be used in an integrated cost-benefit analysis as characterised in the framework because key inputs and information were not made available to the process, and the decision-making rationale was not clearly communicated to residents. Council made decisions in closed-door meetings and announced their decision at the July 5, 2012 information session. It is unfortunate that the rationale for deciding on the new site was not communicated back to stakeholders. This would have clarified considerations and trade-offs involved decision-making, and allowed for reconsideration of values and preferences by all parties.

Furthermore, some of the principles of an *ideal deliberative procedure* were not followed. With City Council members being allocated to each small group in the October 12th workshop the process was heavily exposed to existing power distributions, contravening Principle 3. The process was not free since the option of not undertaking infrastructure development was constrained by decisions outside the process, in violation of Principle 1. This was largely a result of the community workshop taking place too late in the process. Further, while the results of the meeting were supposed to inform next steps, this was done in an incomplete manner, with no attempt to report back to the community in a transparent fashion, thereby violating Principle 5.

The community will change, and it is not clear that it will change in ways that align with the interests of the majority of current residents. Furthermore, individuals who move into new developments may have different preferences from current residents. They could

be more inclined toward higher density housing, perhaps indicating a different relative value for space, and relationship with nature. These new residents will shift the collective culture as well as the balance of power in the community. The relative influence of earlier residents will diminish thereby affecting their future ability to represent their cultural interests.

From an IWRM perspective, decisions are being made that will affect water-based ecosystem services that are shared with watershed residents living outside the municipality of Chelsea. Any negative water quality effects will impact people downstream and on the other side of the Gatineau River, as well as individuals from upstream and downstream who may enjoy recreational activities on this common waterway. To some extent potential new residents of the watershed were given more of a voice than current and surrounding residents that share this watershed[5]. This can only further entrench patterns of insular decision making between neighbouring jurisdictions that share a fundamentally important resource, taking away any legitimate claim Chelsea may have to access decision-making processes elsewhere that could impact their interests. A non-inclusive process that violates principles outlined in our study and framework, therefore, could not only jeopardise sustainability principles within the subwatershed but also at the larger watershed level.

3.7 Conclusions

Based on deliberative democracy and empirical findings from social psychology, a new conceptual framework was proposed for a values-based approach to ecosystem

[5] Four landowners/developers and two public institutions were the only ones allowed to vote on the new water and sewer infrastructure plan for the unbuilt sector (borrowing by-law 781-11 (Municipality of Chelsea, 2011b).

management at the subwatershed level. Drawing on current best practices from IWRM, a literature review of monetary and non-monetary approaches to valuation, and frameworks for the integration of valuation within decision-making processes, a novel conceptual framework was developed that utilises an inclusive participatory approach and argues for the importance of incorporating cultural considerations within valuation processes. Culture has been shown to be both an input and outcome of a values-based approach, as such making its inclusion a precondition for cultural sustainability.

Participatory processes allow us to capture and incorporate the values of citizens within decisions made. This leads to more democratic, legitimate and equitable outcomes. Local knowledge and engagement can better inform decision making, including identification of risks and potential conflicts, which results in decisions that are better orientated within the broader array of social and environmental interests. Furthermore, meaningful and ongoing engagement through public participation can result in more flexible and dynamic decision-making processes. These are characteristic of adaptive management approaches, which are becoming increasingly important as global ecosystems become more variable due to climatic change. While participatory processes may take longer and could be more resource intensive, values convergence and local information generation is vital for sustainability and could form the basis of a new long-term governance partnership. It is essential that we value not just the services provided, but also the ecosystems themselves, from an ecological and cultural perspective. Where problem definition helps shape societal response, there is a central role for expert and stakeholder input into decision-making processes.

While non-monetary values are receiving increased attention from ecosystem services scholars, the influence of culture on ecosystem services valuation, and the impacts on culture from subsequent policy making, is largely overlooked. Ecosystem services, as embedded in policy discourse, are highly normative, with culture being a powerful determinant of how actors align, and form and contest their preferred policy options. Associated threats to culture and cultural perceptions of risk are highly influential in this regard. This study is illustrative of how dynamic processes within the social subsystem can be determinant of ecological outcomes (see Figure 1.2). In particular, the study focusses on the role and influence of culture in determining the relative values of different ecosystem services, bringing these concerns to the forefront and illustrating their central relevance to decision making.

Methodologically, the conceptual framework that is presented in this study can be used to support policy development, and as a guide for the design of policy processes where there are environmental implications. It can also be used to evaluate policy processes. For example, the framework was applied to the case of Chelsea, Quebec, to explore how the policy process in Chelsea, concerning the provision of municipal water and wastewater services, fits within our framework and revised principles of an *ideal deliberative procedure*. Participatory approaches were employed in a limited way, alongside an expression of interest by the community to develop a vision to support planning and growth based on community values and culture. In the end, the decision process could have been better aligned with the values of the community by using derived preferences, integrated cost-benefit analysis and more transparent justification of decision-making rationales. Employing a more integrated localised decision-making framework would have been

favourable to the continuation of a crucially important long-term partnership between the municipality, other levels of government, and its residents in an ongoing closed-loop process.

Chapter 4 - Enacting and Implementing Water Governance: The Role of Decentralization in Transforming Policy Goals in Three Ontario Watersheds

In May 2000, E. coli contaminated the municipal water supply of Walkerton, Ontario, leaving seven dead and over 2000 people gravely ill. In response to this crisis, source water protection emerged as part of a multi-barrier approach for the provision of safe drinking water because of its capacity to address a range of potential threats, many of them tied to land use. Source water protection was a fundamental conceptual shift in Ontario's policy approach to the provision of safe drinking water, and its introduction mobilised powerful actors to defend their interests against this change. Thus, to understand policy change in this case, one must look at both the enactment and implementation of Ontario's *Clean Water Act* (2006). Examining three case-study watersheds, I show that the choice to devolve responsibility over source water protection plans to watershed-based multi-stakeholder committees transformed the original policy design in fundamental ways. Close examination of the membership and function of three Source Protection Committees revealed how science-based risk assessment became subordinate to largely external historical and political forces, in particular as culminated in a shared sense of rural marginalisation. Hence, this study illustrates the value of disaggregating policy and accounting for implementation in efforts to measure and explain processes of policy change.

4.1 Introduction

In May 2000, the water supply of Walkerton, Ontario, was contaminated by E. coli. The source of the E. coli was a local farm where manure had been spread on fields during late April (according to best management practices). Five days of heavy rain resulted in contaminated runoff permeating into the underlying soil and geological structure. There, fractured bedrock conveyed E. coli-contaminated water to Well 5, the primary source of municipal drinking water at the time (O'Connor, 2002a). The resulting outbreak took seven lives and left over 2000 people ill. Government reaction was swift. An investigation revealed several factors had caused the contamination. These included physical factors, improper treatment by utility operators, as well as shortcomings and failures in government oversight mechanisms (O'Connor, 2002a). The proposed solution was a multi-barrier approach for the provision of safe drinking water, with source water protection comprising the first barrier. The *Clean Water Act* (CWA) was passed in 2006 to pursue this policy objective. Employing a watershed-based approach, multi-stakeholder source protection committees (SPCs) were to be established in source protection regions (SPRs) where they were to develop Source Protection Plans (SPPs) for municipal water supplies in their respective watersheds.

Managing water on a watershed-basis is widely considered the most effective way to incorporate the many factors that affect water resources within the watershed's quasi-closed hydrological system (Agarwal et al., 2000, Bandaragoda, 2010). However, devolving power to watershed stakeholders comes with its own consequences, which can

operate to offset the expected benefits of watershed management. Economic actors, for instance, may intervene to bend watershed-level institutions toward their interests. Through consultation processes and other channels, special interests may directly affect the policy frameworks that are intended to check potentially harmful activities and maintain the wellbeing of water resources (Winfield, 2011).

As a social process, source water protection in Ontario is influenced by ecological, social, economic, institutional and political factors. Diverse stakeholders populate the source water protection policy process, bringing with them a plurality of values and expectations (Ferreyra & Beard, 2007). Agricultural interests are particularly sensitive to the authority of the CWA. Surveys of farmers have shown that they oppose regulatory controls over their land by government (Filson et al., 2009, Lamba et al., 2009), and they see efforts to protect source water as local arenas for regulation rather than venues for collaborative multi-stakeholder dialogue (Ferreyra et al., 2008). This challenges the notion of cohesive watershed communities from which different actors may come together for a common interest. Indeed communities can be sites of conflict as well as collaboration (Lane et al., 2004), in particular where their uses of the resource are in conflict (see Chapter 3), or where different actors have dramatically different cultural frames and imperatives. For instance, civil society actors and other proponents of source water protection that are primarily interested in ecological and public health see the watershed as the appropriate scale for environmental governance. Farmers' core values, by contrast, are those of economic wellbeing and self-determination, leading them to see the farm and county as the appropriate scale for environmental governance (Ferreyra et al., 2008). Tensions between

different groups, grounded in different core values and perspectives on the objectives and impacts of the CWA, are expected to affect the institutionalisation of the Act.

This study examines changes to safe drinking water policy following the Walkerton tragedy. It considers two distinct stages: the development of the CWA and its implementation in three Ontario watersheds. Specifically, I was interested in the degree to which institutional outcomes are coherent with legislative intentions, and what factors led to these results.

Accordingly, this study follows two sequential lines of inquiry. First, I examine how actors engaged with the development of the CWA, their motivations, and how the various concerns came together to result in the decision to implement source water protection by devolving power to watershed-based SPC. The second part examines the implementation of the CWA in three SPR, documenting the source protection policies created by these three SPC in their specific socio-economic context, and how observed institutional outcomes were structured at the SPR-level, in part through prior actor interactions with the formulation of the CWA. Thus, the study combines an institutional analysis of the changing drinking water protection regime in Ontario with a case analysis of implementation within three watersheds. Following work in policy change, the study highlights the critical importance of carefully specifying what is changing (Cashore & Howlett, 2007), how these processes of change can unfold at the implementation stage, and what this means in terms of institutional outcomes (Mahoney & Thelen, 2010). Source water protection in Ontario involved devolution of decision-making to watershed-based, multi-stakeholder source protection committees. Disaggregation of changes to drinking water policy exposed the central role this transfer of policy making authority had in

determining institutional outcomes. Moreover, decentralisation in the context of sustained political pressure led to a concentration of power, thereby actually eroding local control.

The analysis proceeds in four parts. First, it provides a brief history of the Ontario case, sketching the policy changes that followed the Walkerton incident. Second, it reviews theories of policy analysis from historical, rational choice and sociological institutionalist perspectives, with a particular focus on work examining policy change. The third section describes the policy process that led to the CWA, key institutional objectives, and how these were modified as the Act was implemented at the subwatershed level across the province. In particular, it examines how powerful incumbent actors influenced the institutionalisation of source water protection in Ontario. Case study data follows in an analysis and discussion of how historical factors and actor effects, in the context of powerful social narratives, shaped the implementation of source protection planning in three Ontario watersheds. The study closes with a discussion of the findings before offering some concluding observations on the relevance of this work for policy studies and for water governance.

4.2 Source Water Protection and the *Clean Water Act* (2006)

Source water protection (SWP), as a policy innovation, was widely taken up across Canada with a diversity of regulatory and voluntary approaches in place. In many instances the source, treatment and distribution of drinking water is regarded as one system, with the municipality or the utility being responsible for ensuring the security of the source water

(British Columbia[6], Quebec[7], Prince Edward Island[8], Yukon[9]). Another approach finds SWP nested within an integrated water management framework where the provincial government takes responsibility (New Brunswick[10]), or provides oversight (Manitoba[11], Saskatchewan[12]). Other provinces employ community-led voluntary approaches (Alberta[13], Northwest Territories[14]). However consultation with local stakeholders, in some form, is common to all these approaches.

In Ontario, the *Clean Water Act* (2006) goes beyond public consultation to formally devolve policy making to the watershed level. Supported by experts and scientific input

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[6] British Columbia (2001). Drinking Water Protection Act. SBC 2001, Chapter 9, Government of British Columbia. Retrieved from BC Laws: http://www.bclaws.ca/civix/document/id/complete/statreg/01009_01#section18

[7] Quebec (2012). Stratégie de protection et de conservation des sources destinées à l'alimentation en eau potable. Gouvernement du Québec.

[8] National Collaborating Centre for Environmental Health (2014). Small Drinking Water Systems: Who Does What in Prince Edward Island?. Retrieved from http://www.ncceh.ca/sites/default/files/SDWS_Who_What_PEI.pdf

[9] Public Health and Safety Act (2007). Drinking Water Regulation. O.I.C. 2007/139. Retrieved from: http://www.gov.yk.ca/legislation/regs/oic2007_139.pdf

[10] New Brunswick (2017). Watershed Protection. New Brunswick Department of Environment and Local Government. Retrieved from: http://www2.gnb.ca/content/gnb/en/departments/elg/environment/content/land_waste/content/reference_manual/watershed_protection.html

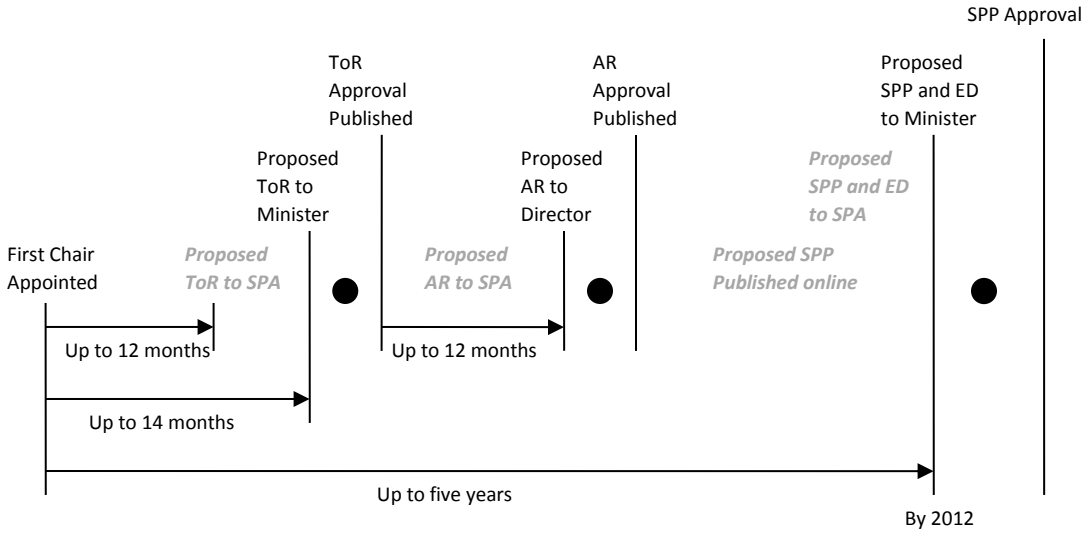
[11] Manitoba (2017). The Water Protection Act. C.C.S.M.c W65. Retrieved from Manitoba Laws: http://web2.gov.mb.ca/laws/statutes/ccsm/w065e.php

[12] Wang, H., & Patrick, R. J. (2014). Implementing source water protection plans in Saskatchewan: Local watershed perceptions. *Prairie Perspectives: Geographical Essays,* 17, 1-10. http://pcag.uwinnipeg.ca/PP-Vol17.html

[13] Alberta (2015). Draft Guide to Source Water Protection Planning in the South Saskatchewan Region (Alberta). Alberta Environment and Parks.

[14] NWT Water Stewardship (2017). Source Water Protection. Retrieved from http://www.nwtwaterstewardship.ca/swprotection

from the Ministry of the Environment and local Conservation Authorities, multi-stakeholder source protection committees (SPC) oversee the development of Terms of Reference (defining the scope of source protection planning in their watershed), Assessment Reports (identifying and assessing threats to drinking water), and Source Protection Plans (specifying policies to address those threats) (CWA, 2015a). As with SWP in other jurisdictions, the approach taken in Ontario was intended to incorporate a high level of public consultation. The idealised timeline for developing Source Protection Plans is presented in Figure 4.1. Formal opportunities for public input are identified in grey italics. SPC meetings were also open to the public (CWA, 2015b), and it was intended that SPC members would establish channels of communication with their respective stakeholder communities. The overall intention was that the process would be local, transparent and highly inclusive.



Note: Abbreviations for the above figure are follows: Terms of Reference (ToR); Assessment Report (AR); Source Protection Plan (SPP); Explanatory Document (ED); and, Source Protection Authority (SPA).

Figure 4.1 – Timeline for the development of a source protection plan

Previous studies have found that SWP in Ontario has resulted in the integration of land use and water management (Plummer et al., 2011), which is a key factor in the protection of water resources (Hill et al., 2008). However the CWA has a degree of paramountcy over other acts, and this has raised concerns that drinking water might be prioritised over other interests (Hill et al., 2008). Coupled with appointed committees developing policy outside of a democratic process, the CWA has been described as uni-functional and potentially undemocratic (Glenn 2009). Further, with devolution of power to the watershed level there were concerns that Conservation Authorities might become too powerful (Hill et al., 2006). Given the relative strength of the Act and lack of democratic oversight, the CWA could present opportunities for actors to develop policies that privilege interests other than safe drinking water. Indeed, the SPCs have strong

representation from actors whose interests and activities could threaten source water (Winfield, 2011); formally, one-third of committee members represent agricultural, commercial, and industrial interests.

4.3 Theoretical Framework and Methodology

The policy impact of the Walkerton crisis appears to be a straightforward case of punctuated equilibrium. Drinking water policy was moving along in an incremental fashion when it was interrupted by an exogenous shock – the events of May 2000. These brought about dramatic policy action and a significant departure from incrementalism (True et al., 2007), in this case toward watershed-based SWP. However public understanding can affect policy processes and policy development such that differences in these policy images can be determinate of observed outcomes (Baumgartner & Jones, 1993; Crow, 2010). Devolving policy making to multi-stakeholder committees was a distinct departure from the usual approach to policy development in Ontario. However this was consistent with wider trends that find policy development through decentralised and democratised processes becoming increasingly commonplace (Howlett, 2014). Indeed, for water resources governance, where integrated water resource management approaches cut across policy subsystems, decentralised and inclusive policy making is commonly seen as best practice (Rahaman & Varis, 2005; Warner et al., 2008). By involving all interested and affected stakeholders, different policy subsystems are effectively brought together to form a policy regime focussed on water management. While this can help ensure all concerns are accounted for, larger policy regimes may also undermine governance outcomes when involved actors pull in different directions (Jochaim & May, 2010). Moreover, historical

and institutional context, in terms of involved actors and associated preferences for various policy options, can further impact policy design (March & Olsen, 2004), with instrument selection ultimately being a strong determinant of policy outcomes (Linder & Peters, 1988).

This investigation draws on a broad institutionalist approach. While institutional structures affect actor strategies (Thelen, 2004), both historical factors and self-interest influence the formation of preferences that underlie political contestation between key actor groups (Thelen, 2004). Furthermore strategic choices depend on interpretation as well as calculation, making the influence of key social narratives central to the analysis. Although institutions are socially derived and endorsed through use (Streeck & Thelen, 2005), the nature of compromise means that some interests will be privileged by these institutions while others are not. It is this advantage that actors seek to acquire through competition in the institutional domain. Further, the process by which policy is developed has implications for policy implementation (Stokes, 2013). Stakeholders that are highly engaged early on can become disenfranchised with the process if outcomes do not meet expectations. Should key stakeholders abandon the process this can have implications for the manner in which institutions distribute power across groups.

Noting that policy change can take place at different rates and with different consequences, Hall (1993) described three orders of change: calibration of instruments, the instruments used to achieve policy goals, and the goals themselves. Cashore and Howlett (2007) extend this typology, arguing that "every 'policy' is in fact a complex regime of ends- and means-related goals, objectives, and settings" (see Table 4.1). Disaggregating policy change into its component parts facilitates the consideration of a more complex

interplay of possible causal factors (Cashore & Howlett, 2007, Howlett & Cashore, 2009), helping to ensure greater conceptual precision around the exact phenomenon being measured (Dupuis & Biesbroek, 2013).

		Policy Content		
		Goals	Objectives	Settings
Policy Focus	Means	What types of ideas govern policy development?	What specific requirements are operationalised into formal policy?	What are the specific on-the-ground aims of policy?
	Ends	What norms guide general implementation preferences?	What specific types of instruments are utilised?	What are the specific ways in which the instrument is used?

Table 4.1 – A modified taxonomy of policy measures following Hall (Cashore & Howlett, 2007)

I used Cashore & Howlett's framework to disaggregate two stages of policy change that sought to establish a multi-barrier approach to the provision of safe drinking water in Ontario (Table 4.2). The first stage is the 2002 enactment of the *Safe Drinking Water Act* (SDWA). This legislation is focussed on water utilities and the assurance of effective treatment and distribution of drinking water. The second stage of policy change is the addition of SWP to the multi-barrier approach with the passage of the *Clean Water Act* in 2006 (CWA). While provision of safe drinking water remained the end goal, the CWA introduced SWP as a new policy objective with decision making being devolved (means-goal) to multi-stakeholder, citizen source protection committees (means-objectives) operating at the watershed-level. Nominally these committees comprised one-third municipal representatives, one-third representatives of the agricultural, commercial and industrial sectors, and one-third other interests, in particular those representing environment and health, and the general public (CWA, 2015b). The members of these

committees were to develop Source Protection Plans on behalf of their respective stakeholder communities to identify and manage risks to the security of drinking water sources in a balanced way, taking into account the range of affected interests in the watershed (means-setting).

Policy Focus		Policy Content: Goals	Objectives	Settings
Ends	SDWA	Provision of safe drinking water	Effective treatment, safe delivery	Regulation of water utilities
	CWA	Provision of safe drinking water	Source Water Protection	Risk identification & management based on wellhead protection areas (WHPA), and intake protection zones (IPZ)
Means	SDWA	Preference for regulation, including monitoring and enforcement	Regulation of drinking water utilities Training of operators Drinking water testing	Reporting requirements Accreditation and licensing Inspections
	CWA	Devolution to watershed Consultation	Multi-stakeholder SPC Source Protection Plans	Inclusive development of balanced Source Protection Plans

Table 4.2 – Drinking water policy regime in Ontario at two historic intervals

As a decision-making body, the SPC was central to the delivery of SWP, making its structure and function of direct interest to explaining any divergence between legislative goals and institutional outcomes. However SWP was not the only policy option, nor was

the SPC model the only way to achieve this end. How a problem is defined often shapes the range of possible solutions (Hanberger, 2001; Pal, 2010), as reflected in the diversity of approaches for SWP across Canada. This study will trace how the problem was defined in Ontario, and by whom, establishing a continuum of actor-institution interactions that spans the development and implementation stages of the CWA to observe how different preferences carried forward through the devolution of decision making to the watershed level.

The first part of this study seeks to understand the development of the *Clean Water Act* (2006) and Ontario's approach to protecting sources of drinking water. Originating with the O'Connor recommendation that a multi-barrier approach be taken to the provision of safe drinking water, the development of the CWA was informed by three advisory committees, and a succession of scoping documents.[15] Key stakeholder groups presented to the O'Connor Inquiry, with many subsequently participating directly on the advisory committees and in the development of scoping documents. All of them had the opportunity to provide comment and influence the content and direction of successive steps toward the development of the CWA.

Supported by document research, semi-structured interviews were conducted with ten individuals from stakeholder groups and government agencies to ascertain how actors engaged with the process, and how their combined efforts influenced the institutionalisation of SWP, as a concept, within the CWA. The policy subsystem experts

[15] These include: The Advisory Committee on Watershed-Based Source Protection Planning's report, "Protecting Ontario's Drinking Water: Toward a Watershed-Based Source Protection Planning Framework" (2003); the Government of Ontario's White Paper on Watershed-Based Source Protection Planning (2004); and a piece of draft legislation, the *Drinking Water Source Protection Act* (2004).

that were interviewed included representatives from government, watershed management, agriculture, environment, and municipal stakeholder groups.[16] The majority of these interviews were held in-person in Toronto from March 8th-11th, 2016. However not all subjects were available during that time period, and in two cases the individuals did not reside in Toronto. These interviews were conducted over the phone. All interviews were recorded, with the permission of the interview subjects. Notes were made during and after the interview to record key points, and the audio recordings were revisited during analysis.

The second part of this study examines the implementation of the CWA. Institutional outcomes were observed at the source protection region (SPR) level through examination of Source Protection Plans. Controlling for the presence of a strong agricultural sector and for the absence of large municipalities, three case study SPR were selected: Saugeen, Grey Sauble, Northern Bruce Peninsula (SGSNBP), Ausable Bayfield Maitland Valley (ABMV), and Raisin-South Nation (RSN). SGSNBP was selected in part because Walkerton is located within its watershed boundaries. This was expected to have resulted in a strong shift in social values. ABMV, located immediately to the south, has been identified as a region where agricultural interests are resistant to additional regulation brought on by the CWA (Filson et al., 2009). RSN is geographically removed from Walkerton but has an agricultural sector of comparable size to the other two case study regions.

The cases were selected with an eye to understanding the processes playing out within each source protection region. The aim was to generate causal process observations

[16] Interviews were held confidentially due to the sensitivity of information being discussed and the potential for negative repercussions.

related to the role of local incumbent interests in shaping the implementation of a new policy, the CWA, given decentralised implementation. I examine the context within which Source Protection Plans were developed, encompassing an analysis of how economic interests, existing institutions, ongoing debate over water management, political climate and broader cultural dialogue, affected the strategies of various actors. In turn, I examine how actors engaged in the policy process, either independently or within coalitions, to affect the direction of policy making, and how this may be attributable to economic factors, SPC composition, or cultural perspectives in terms of broadly held social values.

Ontario Ministry of Agriculture and Rural Affairs (OMAFRA) statistics were consulted to develop a more detailed account of the agricultural sector in each SPR (see Appendix C), and County and Conservation Authority websites were consulted to characterise the sector-wise breakdown of watershed economies and identify additional economic interests. Policies pertaining to drinking water threats as specified by the CWA comprise the institutional outcomes of interest. Policies were selected for comparison based on descriptions of watershed economies for the three SPRs. A common set of policies was used in order to identify relatively more stringent or more lenient measures, and to check for correlation with economic interests in the watershed.

SPC Chairs and Source Protection Authority liaisons were contacted to participate in a semi-structured interview. These officials were chosen due to their depth of knowledge and experience with SPC proceedings. Moreover, they were positioned at the interface between the SPC and Ontario's Ministry of Environment. As such they could offer unique insight into the interplay between the province and the SPC. Interviews were held in-person when possible. From February 9th-11th, 2016, I visited Ausable Bayfield Maitland Valley

SPR, and Saugeen, Grey Sauble, Northern Bruce Peninsula SPR, however not all subjects were available during that timeframe. These individuals were interviewed over the phone, as was the Source Protection Authority liaison in the Raisin-South Nation SPR. All interviews were recorded, with the permission of the interview subjects. Notes were made during and after the interview, recording key points, and the audio recordings were revisited during analysis. Communicating through Source Protection Program managers in the three case study SPRs, all members of the three SPC, past and present, were contacted to take part in an email questionnaire to canvass individual stakeholder representatives regarding the degree to which they perceived SPC deliberations as being representative of watershed communities, or conversely, subject to undue influence from any individual group.

In total, two SPC Chairs and three SPA Liaisons were interviewed, and nine SPC members took part in the email questionnaire.[17] The questionnaire in particular was under-subscribed with 9 of 69 possible candidates taking part (a 15% response rate), despite numerous attempts to recruit participants. This made it difficult to do more than offer qualitative observations. However, seven out of nine respondents were involved in the source water protection process from the outset (in 2007), with the other two respondents having started in 2009 and 2010. As such those who did participate in the survey were involved in all stages of the source protection process, and were well positioned to observe any instances of undue influence or other irregularities.

[17] Interviews and surveys were confidential due to the potential for negative repercussions from local stakeholders.

Available biographical information on the composition of SPC membership was examined to determine how consistent actual committee composition was with the three stakeholder categories prescribed by regulations under the CWA (one-third municipal, one-third agricultural and commercial, and one-third other). In particular, while a given stakeholder may legitimately represent a certain category, their broader interests could span into other areas. For instance, a municipal representative in a rural setting might also own a farm. This would amount to a degree of "leakage" between categories. Finally, I scanned local and farm media sources in the three SPR to develop a sense of how SWP was being discussed in the public discourse. In total 15 sources were scanned. Searching for keywords including, "source protection," "Walkerton," and "Clean Water Act," a total of 374 articles were identified. Since media influence could affect the positions taken by SPC members, these articles were reviewed to assess whether the tone was either positive or negative, and the substantive focus of public discourse. The degree to which there was support for source water protection efforts, and the specific nature of any concerns present within the three SPRs, was expected to influence discussions among SPC stakeholder representatives. A detailed content analysis was not conducted.

4.4 Findings and Analysis

This section describes how drinking water policy changed following the events in Walkerton with a particular focus on the political contestation over the institutionalisation of SWP. This contestation occurred at two distinct stages. Section 4.4.1 discusses how the interests and efforts of three main stakeholder groups influenced the CWA itself, and the design of the source protection committees that were ultimately empowered with policy

making authority. This leads into Section 4.4.2 and a discussion of how devolution of decision making to the source protection committees affected subsequent implementation at the watershed level.

4.4.1 Part I - Development of the *Clean Water Act* (2006)

In many ways the die was cast during the O'Connor Inquiry. For some the problem was primarily one of poor treatment and quality assurance mechanisms: contaminated water should not have entered the distribution system. For others the presence of E. coli in the source water that entered the treatment plant was the fundamental problem, exacerbated by poorly trained operators and inadequate oversight mechanisms. Moving from an account of how the problem developed to how public policy could prevent future incidences, different interpretations of what went wrong led to a range of preferred policy options, as will be discussed. The need for improved training and oversight of water utilities and their operators was widely supported. A more contentious issue was whether the provision of safe drinking water required that action be taken at the source (Gov; Envi[18]). Justice O'Connor concluded that to effectively protect sources of drinking water, government action would be needed to address non-point source pollution (O'Connor, 2002a). In large part this meant managing land use. The following paragraphs summarise the concerns and policy preferences of three main stakeholder groups: Conservation Authorities, municipalities and agriculture.

[18] Information obtained from confidential interviews will identified by the interviewee's stakeholder category: government (Gov), municipal (Mun), environment (Envi), agriculture (Ag), watershed (WS). Source Protection Authority liaison (SPA-Liaison), or Source Protection Committee Chair (SPC-Chair).

Embedded within the concept of SWP are questions around how best to achieve this goal. Ontario has a long history of watershed management through the Conservation Authorities. Mandated to manage the hydrological integrity and operation of watersheds, Conservation Authorities and their representative umbrella organisation, Conservation Ontario, advocated for a watershed approach to SWP (WS; Ag; Envi). Indeed, the source of municipal water could lie outside of municipal boundaries, challenging the ability of municipalities to implement SWP. A watershed approach would also provid a framework to integrate broader water management efforts on a landscape basis (WS). Conservation Ontario saw this as being particularly important for southern Ontario where cites are encroaching on highly developed agricultural lands, prompting a move toward more intensive agriculture (WS). These factors combined with climate change are resulting in an increasingly stressed water resource system that they argued would best be addressed through more integrated water management (WS). From an operational perspective many Conservation Authorities also saw watershed-based SWP as an opportunity to rebuild (WS; Mun). Decimated by budget cuts in the 1990s, Conservation Authority capacity was reduced to a point where the bulk of Conservation Authorities' operational funds came from payments made by member municipalities.

For their part Municipalities resisted the whole concept of SWP. Anticipating that implementation costs would ultimately fall on municipalities, they preferred an engineering approach that focussed on upgrading utilities and infrastructure, where project costs would typically be split three ways with federal and provincial governments (Mun; Gov). As support for SWP continued to consolidate, municipalities repositioned to resist the idea that SWP be led by Conservation Authorities (Mun; WS). Since the provision of

drinking water falls under municipal control, and since they would likely be paying for implementation, they wanted to manage the process (Gov). Moreover, the relationship between some municipalities and their respective Conservation Authority may be strained due to friction between their respective mandates (Gov; Mun; SPA-Liaison). While Conservation Authorities manage water levels and flows, providing flood control and other ecosystem services to municipalities, they also have a planning function. Municipalities want the freedom to make land-use and economic development decisions, but the mandate of Conservation Authorities can impede if not prevent certain developments. The concern was that SWP administered by Conservation Authorities would result in stronger Conservation Authorities, and that this could further undermine municipal autonomy while at the same time requiring the provision of additional funds for implementation.

The agricultural community did not support SWP for three reasons. First, the geographic extent of associated regulatory actions was unclear. Some stakeholders were pushing government to protect all water supplies, existing and future, private and public, while others supported a more limited mandate, focussed on protection of municipal water sources (SPA-Liaison). Second, the agricultural community felt that it had a history of environmental stewardship and the organisational capacity to manage these issues through on-farm means, primarily using Environmental Farm Plans prepared by individual farmers (Armitage, 2001) and nutrient management. In the years leading up to Walkerton, agricultural representatives were working closely with OMAFRA to develop a province-wide approach to nutrient management (Ag). Their primary objective was to avoid a patchwork of municipal bylaws and ensure a level playing field (Ag). However during the O'Connor Inquiry the government retreated from working collaboratively on nutrient

management to take a more regulatory stance through the *Nutrient Management Act* (2002) (NMA), which was positioned as a response to Walkerton (Envi). Any perceived loss of control was likely exacerbated by government removing itself from collaboration on nutrient management issues (Ag). Finally, the notion of landscape-scale integrated water management, using SWP as a vehicle (as advocated by the Conservation Authorities), caused additional concern for the agricultural community due to the perceived possibility that land use management for the purposes of protecting source water might morph into broad, general, non-point source pollution controls, greatly increasing the regulatory reach of government over agricultural lands (Ag; SPC-Chair). The imposition of external regulatory control of an undetermined extent was a threatening proposition that resulted in considerable pushback from farmers and farmer organisations, including farm media.

After Justice O'Connor released his recommendations to resounding support from all political parties (Envi), watershed-based SWP was effectively enshrined and the challenge became finding a way for these three major stakeholder groups to work together. In deciding how SWP would be implemented, government policy makers took note of the concerns of agriculture and municipalities, both in terms of autonomy and potential financial impact (Gov). However there was obvious strategic value in having Conservation Authorities already in place in the more populated and heavily developed watersheds where SWP would be most difficult to achieve and also most needed (WS, SPA-Liaison), particularly since budget cuts had reduced internal capacity, effectively eliminating regional offices of the Ministry of Environment as potential delivery agencies.

Justice O'Connor had supported Conservation Authorities taking a lead role in implementation[19]. He further concluded that Conservation Authorities would "be the appropriate bodies to integrate a broader program of watershed planning if and when such a program [was] implemented" (O'Connor, 2002b). However, both municipal and agricultural stakeholders were uncomfortable with Conservation Authorities taking such a central role. Ultimately Conservation Authorities were given a supporting and administrative role, managing funds and providing the SPC with technical information. This required that source protection programs be set up and staffed to develop the necessary science and information, which in turn resulted in Conservation Authorities benefiting from a substantial influx of human, financial and material resources.

The distributed nature of the potential sources of contamination, such as on-farm practices or leaky fuel storage tanks at gas stations or in private homes, is a fundamental challenge for managing non-point source pollution. It is very difficult to enforce related regulations without some degree of voluntary participation (Lubell, 2002). As part of this challenge, SWP requires a shift in how utility operators, policy experts and the general public think about water: that the condition of water resources is in part a function of what happens on land (Envi), and this means that the actions of individuals may be part of the problem (Gov). Accordingly it was important that people understand the impact they can have (SPC-Chair). A perceived level of fairness (O'Connor, 2002b), community 'buy-in' and municipal support were identified early on as critical to the success of SWP (Advisory Committee, 2003). Further, the very nature of the policy, with cities and towns benefiting

[19] In Part Two, Chapter 4, of the Walkerton Inquiry O'Connor states: "I am satisfied that where Conservation Authorities exist and have the necessary capacity, they are the organisations best positioned to bring about effective source protection planning." (O'Connor, 2002b)

from regulatory action taken elsewhere, meant there were strong urban-rural dimensions that needed to be considered. Agriculturalists in particular expressed concerns that the Source Protection Plans could not be seen as originating (from the Ministry of Environment offices) in Toronto, far removed from the realities of rural life (SPC-Chair). These concerns were heightened by the CWA coming out under a Liberal government, which is often perceived to have an urban bias (Ag; SPC-Ag). Alongside knowledge and capacity building, collaboration with local leaders and forging an effective partnership with local stakeholders were deemed essential (Ontario, 2004). It was decided that SWP would not only be based on watersheds as a unit of management, but that decision making would be devolved to a local committee. This concerned agricultural representatives due to a perceived likelihood that it would once again result in a patchwork of different regulations at a local level (WS; Ag). Instead, the Ontario Farm Environmental Coalition lobbied for provincial authority over SPC (Ferreyra et al., 2008). The provincial government maintained that decisions should be made locally; however steps were taken to ensure a degree of policy coherence between adjacent source protection regions (Gov).

The Advisory Committee on Watershed-Based Source Protection Planning (2003) initially proposed the basic three-category formula for SPC membership[20]. In addition to sending a liaison to attend SPC meetings to provide expert input as required (they did not have a vote), Conservation Authorities had a central role in determining committee composition. They identified candidates for Chair and submitted these to the Ministry of

[20] The final formulation deviated somewhat from the Advisory Committee's recommendation that committees be comprised of one-third municipal representatives, one-third provincial, First Nations and federal government representatives, and, one-third public health and others. However the three equal allocations remained. They did recommend that agricultural and industry representation be mandated.

Environment who made the final selection. They also selected participants to represent industrial, commercial and "other" interests. Initially they were slated to identify agricultural representatives as well. However the Ontario Federation of Agriculture (OFA) was concerned that Conservation Authorities would not pick their preferred candidates and lobbied to have the right to select agricultural representatives (Ag). These individuals were identified through local elections in the farm community. There was some suggestion that the OFA had a certain amount of control over the outcome of this process (SPC-Ag). Although their citizens would be the ultimate beneficiaries, municipalities were already experiencing fiscal burdens associated with additional requirements under the *Safe Drinking Water Act* (2002). They were understandably concerned about implementation costs for source protection planning in addition to a loss of control over municipal affairs. Initially they wanted to lead the source protection process but this aspiration fell with the recommendation of a watershed-based approach. Ultimately municipalities were empowered to identify their representatives to a SPC which was at arms-length from Conservation Authority control. As illustrated in Figure 4.1, draft terms of reference, assessment reports and source protection plans would first go to the Source Protection Authority (the lead Conservation Authority in a given source protection region), where they would be reviewed by the Conservation Authority Board. This board is made up of representatives from member municipalities, thereby giving municipal stakeholders an additional opportunity to ensure institutional outcomes do not unduly affect their interests.

The final formulation for watershed-based SPC emerged as a result of recommendations from the O'Connor Inquiry, Advisory Committee Reports, and political pressure from key stakeholder groups. Agricultural and municipal stakeholders had direct

control over the appointment of representatives for their respective stakeholder groups, and their concerns around a Conservation Authority-led process were also reflected in the outcome. For their part Conservation Authorities were given a mandate to develop technical information, and this contributed to a rebuilding of internal capacity. They were also given selected powers of appointment and administrative responsibilities. Actual decision-making would be undertaken by SPC members with informational support from the Source Protection Authority and provincial liaisons. Resolution of any additional contestation between actor groups was thus devolved to citizen representatives at the local watershed level.

4.4.2 Part II - Implementation and Institutionalisation at the Watershed-Level

At the outset SPC members had different preconceptions about what the process would entail, including the scope of SWP. Some members sought a broad-based environmental approach with the protection of private wells being one motivation. Agriculture in particular was quite defensive at first (SGSNBP[21]), and was a target for several committee members (SGSNBP). Maps and other information helped clarify the scope of their mandate (ABMV, SGSNBP, RSN). This alleviated the concerns of agricultural representatives (ABMV), although they were not able to fully communicate this information to the broader agricultural community (SGSNBP) who were receiving information from a variety of sources. Despite substantial effort by the SPCs and Source Protection Authorities, the general public and local media were not generally engaged with the process except around

[21] Survey respondents were identified either by SPC, e.g. ABMV, SGSNBP or RSN, or by stakeholder category, e.g. SPC-Ag.

niche issues, such as nitrates in SGSNBP (ABMV; SGSNBP; RSN; Thompson, 2010). The farm community did follow the process closely and farm media publications were actively reporting on SWP throughout (ABMV). However, this reporting was not always accurate. For instance, there were reports that more legislation and restrictions on agriculture were forthcoming (RSN). Any such additional measures beyond expected regulatory components of the CWA have yet to materialise. Local media coverage that did exist often reinforced the claims of farm media. These factors had a divisive effect on committee work (SPC-Chair), and questions remain around whether devolution will ease enforcement considering a general lack of engagement by the wider public.

All three source protection regions (SPR) are major agricultural producers having comparable areas of land under production (See Appendix C). Huron County, comprising the bulk of Ausable Bayfield Maitland Valley SPR (ABMV), is the most agriculturally productive county in Ontario (Huron County, 2017). The world's largest salt mine is also located within this SPR in the town of Goderich. In terms of total farms by farm type, crop producers have a slight majority. In Saugeen Grey Sauble Northern Bruce Peninsula (SGSNBP), livestock operations comprise a substantial majority of farms. Nearly 60% of Bruce County is dedicated to agriculture (OMAFRA, 2016b), while recreational properties and tourism, including recreational boating, are another significant industry of note. Raisin-South Nation (RSN) has roughly the same number of livestock and crop operations. It is located between two major Canadian urban centres, Ottawa and Montreal, and just north of the United States border. There are almost 5 million people within a one-hour drive, making it strategically placed to offer shipping and warehousing services. Tourism is a key area of focus for economic development (Millier Dickinson Blais Inc., 2011).

Selected policies from Source Protection Plans for the three watersheds are presented in Table 4.3. Policies pertaining to crop production (fertilizer application) are most permissive in ABMV, while those pertaining to livestock (confinement lots, grazing and pasturing) are most permissive in SGSNBP. New policies in RSN involving prohibitions or risk management requirements apply only to activities not covered by the *Nutrient Management Act* (2002) (NMA). Policies on road salt handling and storage are substantially more permissive in ABMV while those pertaining to fuel storage were least restrictive in SGSNBP, particularly for designated lakeshore communities. From this analysis there is an apparent correlation between source protection policies and prevailing economic activities. Since source protection plans are supposed to account for local economic conditions this does not necessarily indicate that the SPC process has been captured. However Hall & Taylor (1996) argue that institutionalisation of power asymmetries would precede capture, leaving the question open pending closer examination of committee structure and practices.

Threat	ABMV	SGSNBP	RSN
Outdoor confinement lots	Existing and future confinement lots are prohibited in WHPA-A (100m around the wellhead), and future confinement lots are prohibited in WHPA-B (the distance for which local geological conditions would result in a two year travel time to the wellhead)	Only future confinement lots in WHPA-A are prohibited. All other activities are subject to risk management plans (RPM).	No prohibitions and risk management is only required for activities that are not subject to the requirements of the *Nutrient Management Act* (2002) *(*NMA*)*
Grazing and pasturing	Grazing and pasturing is prohibited in WHPA-A when density is greater than 1 NU/acre (see below); activities in WHPA-A and WHPA-B are subject to risk management when livestock density is below 1 NU/acre,	Risk management plans (RMP) for all vulnerable areas where activity is a significant threat	No prohibitions and risk management is only required for activities that are not subject to the requirements of the NMA
Commercial fertilizer application	Risk management plans for fertilizer application are only required when the density of livestock is above 1 NU/acre, or for highly developed lands	Future fertilizer application in prohibited in WHPA-A. Otherwise risk management plans are required for all vulnerable areas.	Existing and future fertilizer application is prohibited in WHPA-A; other application or storage activities, not subject to the NMA, require risk management plans be put in place
Road salt handling and storage	No prohibitions; RMP above 5000 tonnes	Prohibits existing and future storage of road salt (exposed) over 500 tonnes	Prohibition of future storage where it would be a significant threat; RMP for existing storage
Fuel storage	Prohibition of future installations > 250 l if partially below grade, or > 2500 l above grade; RMP for existing facilities at these levels	Specific policies in SGSNBP prescribing higher thresholds for the municipalities of Kinkardine, Meaford, Wiarton, Lion's Head, Owen Sound, Southampton and Thornbury (see Appendix C)	Prohibition of future storage of liquid fuel other than at private outlets or farms; RMP for existing facilities other than a private outlets or farms

Note: NU, or "Nutrient unit" has the same meaning as in Section 1 of O. Reg. 267/03 (General) made under the *Nutrient Management Act* (2002); specifically, the amount of nutrients that give the fertilizer replacement value of the lower of 43 kilograms of nitrogen or 55 kilograms of phosphate as nutrient annually.

Table 4.3 – Comparison of source protection policies from three case study SPR

Information on committee composition serves to further unpack the apparent correlation between economic interests and source protection policies across the three regions. Selecting case study locations where agriculture is a dominant industry means that many municipalities in the three SPR are small farm service communities, and individuals who are involved in municipal politics or other pursuits often have a close connection to the agricultural sector. An analysis of available biographical information found that agricultural actors served as representatives in all three stakeholder categories in ABMV. The SPC also included a representative from the company that owns the salt mine in Goderich. In SGSNBP, while some municipal representatives have agricultural backgrounds, environmental interests were also well represented in the agricultural-industrial-commercial category. In RSN two longstanding members of the SPC switched categories late in the process, moving from representing environmental organisations to representing agriculture. Both of these individuals are farm owners. Hence we see that the three basic categories that are used to compose SPC membership are not insulated from one another but rather there can be a substantial amount of "leakage" between categories, with some SPC members arguably having interests that overlap more than one stakeholder category. This could distort the relative level of representation for a given sector, as is the case for agriculture in these three SPR.

Following guidance from the Ministry of Environment the three committees adopted a consensus-based decision-making model. Respondents in all three locations spoke to the importance of having a strong Chair to ensure that all parties had an opportunity to be heard. There were however instances where different stakeholder groups were crowded out, as the discussion was frequently dominated by a small number of people

(ABMV, SGSNBP, SPC-Chairs). In ABMV agricultural interests generally carried the discussion. In SGSNBP agricultural interests and municipal representatives took turns leading the discussion. Industry representatives expressed difficulty representing the diversity of business interests found in the watershed (ABMV, SGSNBP). Having a vested interest, municipalities also represented commercial interests. One case in particular is that of marinas in SGSNBP and policies affecting fuel storage.

Beyond a general alignment of source protection policies with prevailing economic interests, an apparent overlapping of stakeholder categories with agricultural concerns, and a tendency for committee dynamics to result in a degree of crowding out of certain stakeholder interests within SPC deliberations, a strong sense of rural community was also at play. This resulted in a degree of permissiveness and protectiveness toward agricultural concerns, effectively constituting a broad coalition of support. From the outset there was concern that SWP was going to be at the expense of the rural community (RSN). Indeed, SWP was characterised in the media as evidence of Liberal "disdain" for rural populations (SGSNBP; Merriam, 2012), while the CWA was described as the culmination of a series of legislative steps designed to give towns power over rural areas. A local opinion piece implicated bureaucrats and academics, claiming that any stakeholder committees would be "window dressing along the way" (Zandbergen, 2006). Terms like "agriculture" and "rural" had special meaning (ABMV), and SPC members felt it was important that the needs and challenges of agriculture were understood by all (SGSNBP). For their part, farm media outlets pushed back hard on SWP and the CWA. They started early and were active throughout. Their reporting was often speculative bordering on fear-mongering and caused a good deal of concern in the agricultural community (SPA-liaison; Meagher, 2005).

Concerns over SWP impacting livelihoods and the "right to farm" led to strong support for landowner compensation in all three SPR, something that was ruled out by the province. Survey respondents from across stakeholder categories referred to a perceived urban-rural divide, where urban areas benefit without expense while rural areas bore the burden of the CWA. Moreover concerns about SWP affecting farm freedoms and even property values never really went away (ABMV).

Interestingly, in terms of symbolic meaning and associated influence, the name Walkerton itself evoked different reactions depending on location. In SGSNBP, where the town is located, Walkerton was a major issue. Some SPC members used it as a platform to target agriculture, while those directly involved with the incident preferred not to discuss it. In RSN and ABMV Walkerton was not particularly symbolic, instead prompting backlash. Walkerton was not about agriculture, it was about people not doing their jobs (ABMV). A number of respondents expressed the view that regulations were in place (through the SDWA) to deal with that issue and that the work of the SPC was not going to prevent another similar event (ABMV, RSN). This perspective is in direct contradiction to the findings of the O'Connor Inquiry and strikes at the very heart of SWP. To have SPC members question the fundamental motivation for their work suggests an underlying unwillingness to fully accept that agricultural activities can introduce risks for drinking water supplies.

Since source protection regions (SPR) were specifically selected based on the strength of their agricultural sector, it is not surprising that an analysis of source protection plans in these areas would place agriculture under close scrutiny. It is equally unsurprising that there would be widespread concern for the wellbeing of the sector, and that this could

have a bearing on the manner and direction of policy making within these watersheds. In other SPR this may be different. Nevertheless, in almost all SPR agriculture will be a key player and core concern. Similar dynamics will likely be at play, if not as dramatically as with these three SPR. It is true that other economic sectors (e.g. salt mining and recreational boating) receive similar consideration without the same level of tacit support. This makes it difficult to say that policies pertaining to agriculture are relatively more lenient than those pertaining to other economic activities. However where SPC deliberations were highly politicised around rural and agricultural issues, with these two concerns being largely synonymous for many SPC members, source protection plans were subject to political influence, made possible through devolution to citizen-based, multi-stakeholder SPC. Moreover, this influence was not contained as intended by stakeholder categories prescribed by regulations under the CWA.

4.5 Discussion

The decision to devolve policy making to multi-stakeholder citizen bodies had a fundamental impact on observed institutional outcomes. However devolution may have been less a rational design choice made relative to policy goals than it was emergent from a democratised process, and a reflection of prevailing thought among water governance experts at the time. Given the historical and institutional context, including the ideas and preferences of involved actors, it may have been inevitable.

Beginning with the O'Connor Inquiry, water governance experts argued for and were successful in embedding a watershed approach within his recommendations. This direction was further entrenched by the Conservative government making the design choice

to send the matter to a committee for study rather than addressing SWP head-on. The multi-stakeholder Advisory Committee comprised many of the same experts that advocated for a watershed approach at the O'Connor Inquiry, and only strengthened the path toward devolution. A SWP regime was established through deliberation and consultation during the formulation of the CWA. Bridging agricultural, watershed management, and municipal water supply policy subsystems, this regime comprised a number of politically powerful actors. These groups had different policy preferences from the outset. A compromise took shape in the SPC, with key players' interests in tension with the fundamental goal of protecting source water.

Devolution to local, multi-stakeholder committees was essentially a bargain among these powerful actors. It broke open a policy monopoly, incorporating a broader group of actors and associated values that were previously external to the process. Decision making was thereby subject to a range of new imperatives that were now endogenous to the process. These values were not fixed but subject to influence within small group settings (Buchanan, 1954; Wilson & Howarth, 2002). For instance, we have observed how an informal, tacit coalition formed around agricultural issues, effectively expanding the representation of these issues beyond the allotted number of stakeholder representatives. Not surprisingly municipal representatives in highly rural agricultural areas had an interest in agriculture, and in many cases were farmers themselves. What was unexpected was the powerful narrative that was shared by stakeholders across categories, that being "rural" had special meaning and that collective action was needed to protect themselves from the advance of urban control over rural landscapes: they were all in it together.

Nevertheless, while municipalities and farmers both felt like they were losing control, their specific concerns were divergent to some extent, and in some instances in opposition. Municipalities were concerned about implementation costs, while farmers were concerned about lost livelihood opportunities, property values and compensation. However towns were also the beneficiaries, and so were the very urban centres that were gaining control over rural land use. Yet on the SPC, when tasked with creating policies to protect source water by managing agricultural (and other) activities, they were largely in alignment.

Moving policy making closer to those affected by these decisions was supported by land owner representatives through local media (Thompson, 2006b; Van Dussen, 2006). This devolution of authority meant that decisions would be made locally with appropriate municipal oversight. However, once the policies are in place, the municipality is no longer in control. Developed by a multi-stakeholder SPC according to political and normative imperatives, SWP policies can only be appealed through Ontario's Environmental Review Tribunal (CWA, 2006). Whereas many land use decisions would have been previously subject to appeal at a municipal level, the CWA has created a situation where contesting interests no longer have the same rights of appeal, reshaping the overall institutional framework and expectations of involved actors. In our case study SPR these institutions were heavily influenced by agricultural stakeholders and those SPC members who were sympathetic to rural concerns and perceived these as interlinked with agriculture. In short, agricultural interests were to some extent empowered and privileged by the CWA.

During the O'Connor Inquiry there were concerns that the protection of drinking water could be compromised by political influence at the local level (O'Connor, 2002b)[22]. Once again Walkerton provides a compelling example. High levels of nitrates have been a persistent issue in a monitoring well that had previously been a municipal drinking water production well. Put forward and pressed by the Conservation Authority (SGSNBP-Ag; Mun), SPC concern resulted in the provincial government listing nitrates as a special threat. In the proposed SPP there was a set of policies dealing with this concern (SGSNBP, 2012). However 80% of Walkerton's catchment area is in the neighbouring Municipality of South Bruce and they were concerned about the possibility of increased regulatory pressure on their farmers. Although a town councillor objected, stating that they were elected to represent the water users of Walkerton, not farmers elsewhere (McPhee, J., 2012), the Mayor of Walkerton was sympathetic to his municipal counterparts. In the end, policies pertaining to nitrates were removed from the final SPP, providing a clear example of the potential for political influence to circumvent science. Furthermore, given the paramountcy of the CWA, this provides some basis for concerns over democratic accountability and a loss of municipal control over municipal affairs.

Cashore & Howlett (2007) have suggested that in certain instances rapid policy change may actually return to the original equilibrium, rather than establishing a new equilibrium. These cases would be more accurately be described as "faux-paradigmatic." In other words, instances where equilibrium conditions appear to have been disrupted,

[22] In Part Two, Chapter 4, of the Walkerton Inquiry O'Connor states: "One of the principal advantages of the new authorities, it was argued, would be a greater independence from municipalities. Without such independence, it is feared, improper political influence could adversely affect the process of promoting drinking water safety." (O'Connor, 2002b)

resulting in a paradigmatic policy shift, may be less dramatic than initially perceived in terms of actual effect. The broad adoption of SWP across Canada does appear to be consistent with a shift in policy paradigm. However in Ontario where substantial apparent change has shifted back to the status quo, the effective change is quite small. Broad deferral to existing instruments, as observed in RSN, is perhaps the clearest example. Contrasting policy images between the province and many SPC members, in conjunction with a protectionist attitude toward agricultural activities, may similarly curtail the overall impact of SWP in Ontario.

However to simply classify the outcomes of the CWA (as observed in our three case study SPR) as faux-paradigmatic is to discount the knowledge and capacity building that was achieved. Returning to the importance of shifting how we understand land-water connections and the challenges of sharing heavily populated landscapes, there was value in simply having an inclusive local process (Envi, Mun). The long-term benefit of this process in terms of shifting society's understanding of its interrelationship with the environment has yet to be determined. If the CWA has enabled a shift in the ideational understanding of water protection in such a way as to allow for future improvement, this could indicate the presence of a slower but progressively incremental change, even if the effect of the initial legislative change was not as substantial as intended.

4.6 Conclusions

Tracing the development of the CWA and subsequent implementation in three Ontario watersheds, I identified key actors and framed their actions within the social and historical context. In doing so I observed how their efforts shaped the CWA, and how this in turn

affected the decisions of others, ultimately shaping the source protection plans in three source protection regions. From these observations I was able to draw empirical and theoretical conclusions concerning SWP policy in Ontario, challenges experienced during development of source protection policies, and some of the challenges that lie ahead for implementation of local source protection plans. Some concluding thoughts on theoretical implications will follow a brief account what we have learned about SWP in Ontario, including some thoughts on challenges that may be encountered during implementation of the source protection plans.

Watershed-based SWP seemed like a policy idea whose time had come, with the institutionalisation of science-based identification and management of risks to municipal source water suggesting a paradigmatic shift in the provision of safe drinking water. However historical and political factors came together to exert tremendous pressure on subsequent implementation of this recommendation. Close examination through disaggregation of policy means- and ends-components raised a number of concerns. In particular, this approach isolated devolution (means-goal) to multi-stakeholder SPC (means-objective) as central factors for explaining the policy change observed during implementation of the CWA, and revealed the means by which a highly organised agricultural sector was able to assert its enduring dominance. In the context of significant political and public interest in securing the safety of municipal drinking water, the agricultural sector was first able to influence the terms by which it would be represented in the source protection policy process. Moreover, it was able to neutralise key opponents, to some extent. Once the CWA was in place, agricultural interests worked within the social subsystem (see Figure 1.2), to leverage political and economic influence, and align their

cause with strong social narratives, ultimately achieving a level of control over a policy process that was intended to address potential threats to the safety of drinking water sources, many of which are associated with agricultural practices.

As SWP in Ontario enters the next stage, and source protection policies written by the SPC come into force, there are two identifiable and interrelated concerns. First, there is the question of Conservation Authorities acting in the capacity of risk management officers during implementation. The argument in support of contracting Conservation Authorities to assess risk management plans and provide other risk management services is straightforward. They have the necessary scientific and organisational capacity, and resources allocated by the province to individual towns could be more effectively spent by leveraging this existing capacity. However they also have a statutory role to evaluate this same work, creating potential concerns around transparency and accountability. Equally concerning, some municipalities do not have the requisite expertise, yet they have still decided to use the implementation funds from the province to directly provide risk management services. This decision may be rooted in a general distrust of Conservation Authorities however these funds are time-limited (Ontario, 2013). Unless municipalities act in a timely manner, risk management work may not be completed prior to the termination of financial support. At that point the municipality would have to find other means to complete this work, and it is unclear how implementation of SWP policies would continue without trade-offs having to be made elsewhere in municipal budgets. This could only further degrade the bruised popularity of SWP, potentially leading to additional challenges with implementation, compliance and enforcement.

From a methodological perspective these findings underscore the importance of closely scrutinising how different variables embedded within a given policy change might affect the actual policy outcomes, either individually or in conjunction with historical factors, existing institutions, social narrative or political factors. An idealised SPC model was developed through political negotiations associated with the development of the CWA. In application however local decision-making authority and limited opportunities to appeal worked to institutionalise power asymmetries. Observed susceptibility to political influence raises additional concern around representativeness and the potential for capture.

The policy goals of those who designed the CWA were displaced as a result of the central position of the SPCs, combined with a contrasting policy image rooted in a sense of rural oppression that was held by many of their members. In spite of rigorous, science-based explanations, the durability of the narrative that the E.coli outbreak in Walkerton was a result of people simply not doing their jobs speaks to a fundamental unwillingness to accept that land-based activities were at fault. However it was also difficult for SPC members to legitimise SWP as a policy objective when it was widely understood by them to be a transfer of costs and blame to rural inhabitants by urban policy-makers and aligned experts.

While disaggregation of the CWA into means- and ends- components was helpful for understanding how various actors and institutional factors interacted within the confines of the specific policy process under examination, this methodological approach does have the effect of focussing the analysis, potentially at the exclusion of wider considerations. For instance, the study could have been focussed differently, perhaps emphasising the historical dimensions of the CWA and encompassing an account of the political decisions

that created the conditions within which the contamination of municipal drinking water in Walkerton could occur. Alternatively I could have prioritised the examination of institutional interactions with other policy areas and subsystems. For instance, the concurrent implementation of the *Green Energy and Green Economy Act* (2009), and ongoing urbanisation, were also aggravating the perception among residents of rural communities that their autonomy was being undermined to the benefit of large urban centres. These and other factors are not disconnected and likely affected how the CWA was perceived by local stakeholders, thereby shaping their response to its enactment. There are other data sources that could have been considered as well. Two in particular may be candidates for additional research, those being, Ministry of Environment (MOE) liaisons to the Source Protection Committees (SPC), and any available records of public consultation at the SPC level. MOE liaisons were uniquely positioned to observe interactions among SPC members, Source Protection Authority and Conservation Authority personnel, and agricultural actors. Moreover they would have direct insight into how the concerns of the provincial government evolved through the process. Examination of available records of public consultation would add valuable perspective on the degree to which the general public was engaged with the process, and the degree to which their concerns were reflected within SPC deliberations. Finally, this study deliberately focussed on implementation of the CWA in highly agrarian Source Protection Regions where there was an absence of large municipal actors. It would be interesting to broaden the analysis and include a more diverse set of SPR in order to determine the limitations of my findings, and to identify other causal explanations for observable institutional outcomes in terms of source protection policies.

In terms of contributions to the theory and practice of water governance, this study cautions against using watersheds as a policy-making unit. While watersheds lend themselves to managing water levels and flows, pollutant loads, and other bio-physical parameters, the fit is wrong for policy making (de Loe & Murray, 2013). The conceptual framework in Figure 1.2 is helpful for framing this argument in systems terms. As much as the watershed is a closed hydrological system (putting aside groundwater flows), it represents the ecological subsystem in this application of the conceptual framework. Social subsystem dynamics are connected to the ecological subsystem, but its component parts are not subsumed by the ecological subsystem. Ecosystem services flow into the social subsystem providing benefits and introducing associated incentives. However the actors that respond to these incentives need not be geographically located in the watershed. For instance, where political jurisdictions do not typically align with watershed boundaries it follows that governance processes will be similarly misaligned. Furthermore, some actors may not be beneficiaries of ecosystem services, but may instead be motivated to resist their provision. We see then that factors affecting actor strategies and water resources governance inevitably extend beyond the watershed boundaries. For example, orchestrated efforts by provincial-scale actors can easily overwhelm local representatives. The CWA put a significant amount of control over policy making in hands of the Ontario Federation of Agriculture, the result of which is that non-point source pollution management is now largely in hands of non-point source pollution producers, with limited opportunities for appeal.

Somewhat ironically, a bold policy move that was meant to empower local stakeholders, and distance decision makers in the provincial government from regulating

local land use, was perceived by local actors as an instrument intended to take power away from them. That said, where SWP policies were created by appointed committees under the influence of external agendas, and can only be appealed through a provincial mechanism, a loss of democratic accountability and substantial disempowerment was ultimately experienced by local stakeholders during implementation.

A good deal of scientific information and resource management capacity has been developed, and positive social externalities were observed in terms of a new appreciation and level of awareness around water issues. In the end however the CWA does appear to have concentrated and centralised power, though not in urban centres as was feared by local populations. New policies are in place to protect drinking water sources but the stringency of these policies was effectively contained by organised efforts from agricultural stakeholders, and an overlapping sense of rural marginalisation that was shared by many members of the three case study SPC. When combined with concerns over the funding of implementation, the overall effect appears to fall short of the paradigmatic change in water management that was originally envisioned.

Chapter 5 – Theoretical Implications for Ecosystem Services

To date the conceptual evolution of ecosystem services has been largely toward the development of typologies to identify and classify specific ecosystem services so that they may be valued in monetary terms. The primary objective is to ensure that nature "counts" in policy-making, either to address environmental externalities or to identify priceless natural assets for which there is no substitute. Rather than reducing ecological production processes into discrete and identifiable goods and services, or bundles thereof, my research returns to the earlier and more generalised conceptualisation, that human welfare is linked with nature. Benefits accrue to humans, either directly or indirectly, through goods and services provided by complex ecological systems. In this broader framing the systems perspective is more clearly in focus.

Ecosystem services arise out of a common ecology in a systems context that comprises two joint, complex systems – the environment and society – which together shape the production and use of ecosystem services. The "value" of one ecosystem service is not absolute but is in fact normative in its composition, with values varying across stakeholders relative to other ecosystem services and objectives of society. Social prioritisation of one ecosystem service, either through land use changes or production quotas, can change the availability of other ecosystem services. Meanwhile ecological conditions, whether natural or human-modified, work to define the set of available bundles of ecosystem services. These bundles (and the underlying ecological preconditions), then influence the direction of development and constitute a key factor in shaping social outcomes.

By using ecosystem services to link social and ecological subsystems, my research

developed new insights into causal mechanisms that underlie the central paradox that motivates this research; that despite increasing scientific knowledge and institutional safeguards, sustainable water management continues to be elusive. The use of ecosystem services in this analytical capacity has also resulted in new arguments that extend our theoretical understanding of ecosystem services as a concept. The three studies presented in this thesis have explored ecosystem services along three themes: (1) ecosystem services as a tool for system diagnostics, (2) ecosystem services as a values construct, and (3) ecosystem services as a locus of political contestation. This chapter will draw on the preceding three studies and discuss cross-cutting themes and findings that pertain to the evolution of ecosystem services as a concept. These conceptual arguments will then be taken up in Chapter 6 as I discuss the policy relevance of my findings and implications for innovative approaches to water governance.

5.1 Ecosystem Services, Values and Preferences

Although monetary valuation of ecosystem services has enjoyed substantial attention from scholars and practitioners (Absom et al, 2014), such approaches do not necessarily capture the broader dimensions of value (Vatn & Bromley, 1994; Farber et al., 2002; Jenerette et al., 2006; Fisher et al., 2008). From a policy perspective, decisions made can have implications for the relative availability of ecosystem services, whether or not the full range of affected ecosystem services are the substantive focus of a given policy process. As a result, water resources governance can directly or indirectly engage a plurality of interests, placing ecosystem services at the centre of political conflict between different policy actors. This section will discuss these other dimensions of value, and the embeddedness of

ecosystem services within policy discourse, with a particular focus on drawing linkages between the normative dimensions of ecosystems services provision, and the policy preferences of different actors as observed within the empirical case studies.

Among the standard categories of ecosystem services the valuation of cultural services, such as 'recreation' or 'sense of place', has been particularly perplexing. Challenges include the interconnected nature of benefits and the potential for double counting, incommensurability and inability to compare cultural and other values, and the plurality of values positions involved (Satz et al., 2013). Concerns around double counting are not unique to cultural services. A focus on final ecosystem goods and services offers one approach to circumvent these issues (Bateman et al., 2011; Nahlik, 2012). However the more salient questions where culture is concerned are those that pertain to values. "Values" in this sense are not the outcome of a utilitarian analysis concerning efficient allocation of resources, but instead equate with what is important to people – ethical aspects in addition to more pragmatic economic concerns. These in turn open up wider considerations for how we as a society, in the context of the social subsystem shown in Figure 1.2, ascertain the contributions of nature, and then account for these in policy making. From a values perspective, cultural services are inherently normative with valuation of cultural services defined more in terms of what *ought* to be. Generally we can expect a range of perspectives on what *should* be done, which will in turn politicise the valuation process. Moreover, it is apparent that other ecosystem services, in addition to those typically thought of as cultural services, can have important "values" (i.e. normative) dimensions. As observed with source water protection (SWP) in Ontario, values are also connected with external factors such as social narrative and media influence. This suggests

that individual and group preferences may be in a more reflexive relationship with governance context and the strategies of other actors. We know from behavioural economics and game theory that rational actors take into account the expected actions of others when forming their own strategies (Bowles, 2009; McCain, 2009), adding weight to the notion that preferences are not fixed but are endogenous to the process. Taken together the different variables that affect the preferences for, and hence value of, ecosystem services are many, and the associated normative dimensions of ecosystem services imply certain challenges for current approaches to ecosystem services valuation. Indeed, there are multiple dimensions of value at play: ecological, economic, social and cultural. Throughout this thesis we have seen how these different imperatives unfold within a socio-historical, political context where human action shapes both natural and social systems according to power structures such as political influence, property rights and current distribution of wealth.

In all three studies ecosystem services were not the explicit object of policy but were instead embedded and implicit in policy discussions, with a degree of competition taking place among involved actors over policy and associated ecosystem services outcomes. Examining these three studies through the lens of ecosystem services helped to define the policy context and how different actors influence governance processes. By highlighting the different values-orientations toward ecosystem services that are embedded within a decision-making scenario, I was able to develop a unique perspective on the motivation of involved actors. This helped structure an assessment of how different social and ecological subsystem elements within a governance scenario interacted, and led to unique empirical observations and theoretical perspectives.

In the Chelsea case study, municipal development hinged on water and wastewater infrastructure. However this development trajectory pitted rural living that valued and benefited from a specific set of ecosystem services, against denser development that would introduce social and ecological changes. Political contestation over different visions for the municipality had implications for the subsequent availability of ecosystem services.

In the Mississippi Valley watershed, two things came into focus after modelling environment-economy linkages and examining patterns of demand for ecosystem services. First, this approach helped identify relevant policies which enabled scrutiny of these policies relative to the modelled system. Second, I was able to observe the influence of local politics on environmental governance, and how environmental externalities resulting from local decisions could influence social and economic outcomes elsewhere. Development in the upper watershed had consequences for the availability of ecosystem services in downstream census subdivisions (CSD). A subsequent review of relevant policies identified gaps in coverage where environmental externalities could result in the development of economic risk. In this case, policies that were focussed on a specific activity could affect a range of ecosystem services, either directly or indirectly through system effects. For instance, municipal official plans, and importantly variances to these plans, had implications for the provision of ecosystem services within and beyond municipal boundaries. The relative effectiveness of nutrient management practices carried out under the *Nutrient Management Act* (2002) provides another example. A large-scale breach in containment measures, for instance, involving multiple farms during a peak weather event, could be expected to impact water quality and associated ecosystem services in adjacent and downstream ecosystems.

Local politics were at work in the three source protection regions discussed in Chapter 4 as well. SWP as envisioned by the *Clean Water Act* (2006) was fundamentally a public health policy. Environmental protection, in terms of managing the risks from non-point source pollution, was a means to pursue safer drinking water. However the prospect of land-use management implicated a number of ecosystem services in the process. Clean source water as one ecosystem service was perceived to be in competition with agricultural and rural lifestyles and associated ecosystem services such as agricultural production, water and nutrient cycling, and pest regulation. Furthermore, beyond competing for ecosystem services and related property rights, clean water became a proxy battleground for competing visions of rural autonomy.

While this work is somewhat critical of monetary approaches to ecosystem services valuation, such approaches have been effective in this more utilitarian role. Indeed, monetary approaches pursue a complementary agenda in terms of seeking to make policy-making more accountable for environmental externalities. Valuing services in monetary terms has provided the foundations for payment for ecosystem services (PES) schemes that reimburse parties for actions taken to protect the supply of such services. The Economics of Ecosystems and Biodiversity (TEEB) supplies decision makers in all sectors with reports and methodologies to assist in the consideration of ecosystem services, and seeks to support national policy integration of ecosystem services. The World Bank's efforts on Green Accounting run in parallel with these recommendations. Built around the idea that wealth generation depends on a country's asset base, and that this includes natural capital, the World Bank underscores the importance of conservation, stating that a country's ability to maintain growth and wellbeing in the long term depends on the how well natural assets are

maintained.[23]

Instead this research presents a cautionary position. Monetary valuation has achieved practical ends, but it is important to understand the inherent risks associated with this approach. Employing a broader conceptualisation of ecosystem services, one that encompasses differing values perspectives and political contestation within an integrated systems frame, can help policy makers understand and identify sources of pressure on governance processes, and estimate the effects these pressures may have on resource management decisions and outcomes. Taking PES as an example, these schemes typically involve the identification of ecosystem services producers and beneficiaries, where the production of ecosystem services is a positive externality from specific behavioural choices on the part of the producers. To incentivise these beneficial choices the PES program establishes terms by which the beneficiary provides payment for the services received. This works to eliminate free-riding, incentivise further or ongoing production of ecosystem services, and potentially improve environmental conditions. Taking up this last point, assessing the degree to which ecosystems are preserved or restored is a somewhat normative position. For instance, PES programs may make ecosystems more productive by aligning incentives with selected uses. However, there are other ecosystem qualities beyond productivity that are important to humans, such as resilience. From a governance perspective, resilience scholars argue that approaches seeking to control the environment in order to maximise yield should be reconsidered, instead arguing for more open institutions that are flexible, involve significant dialogue and are capable of supporting

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23 http://web.worldbank.org/WBSITE/EXTERNAL/TOPICS/ENVIRONMENT/EXTEEI/0,,contentMDK:20487830~menuPK:1187769~pagePK:148956~piPK:216618~theSitePK:408050,00.html

significant social learning (Gunderson et al., 2006). Adding another consideration, Anderies et al. (2006) caution against leaning too far toward ecological resilience, suggesting that this could marginalise livelihoods and the rights of local people in decision making. Instead they suggest that trade-offs between social and environmental objectives should be political decisions rather than be left entirely to models and experts. While these are important considerations, my findings also suggest that there are inherent risks with this approach that would need to be accounted for. These include the possibility of capture by special interests (as was observed with the implementation of the CWA), or the marginalisation of local stakeholder concerns by political agendas (as observed in Chelsea).

Finally, the focus of ecosystem services research to date has largely been on supply of services, either individually or in bundles (Engel et al., 2008; Raudsepp-Hearne, 2010). This is equally true whether we look at monetary or non-monetary valuation (Martín-López, 2012), or provision of ecosystem services by a given ecological system (Burkhard et al., 2012). To better understand the competition over ecosystem services and underlying ecological resources my findings suggest a shift toward thinking in terms of demand is warranted. In the Mississippi Valley watershed case, using final ecosystem goods and services to model environment-economy linkages produced a geographical representation of demand, showing how patterns of demand on heterogeneous landscapes could lead to economic risk. In the case of the *Clean Water Act* (2006), it was demand and a concern over future ability to satisfy that demand that was at the root of the conflict. Viewing governance processes from the perspective of demand provides a space for examining political contestation and the resolution of conflicting values. This in turn will better enable

trade-offs to be made relative to key systems variables.

Policy making focussed on water resources (or other natural resources), inevitably intersects a range of ecosystem services. Consequently, these policy processes engage the normative and values perspectives of associated actors, who in turn work to bend policy making to privilege their respective interests. Economic and political leverage, as well as the relative institutional advantage that different actor groups enjoy, is used for this purpose. This underscores the importance of thinking about ecosystem services in terms of demand. An analysis of demand for ecosystem services, alongside underlying values and normative dimensions, can provide additional perspective into the motivation for how these assets are deployed. As such, the linkages between ecosystem services, values and policy preferences, underscore the analytical benefits of accounting for ecosystem service demand when seeking to understand the institutions and processes of water resources governance.

5.2 Boundary Objects and Institutional Interplay

A central proposition of my research was the use of ecosystem services as a boundary object, the premise being that boundary objects can be used to bridge overlapping social worlds, where these may also involve a degree of permeation between scientific and social accounts of the boundary object in question (Schröter et al., 2014). This follows closely on the preceding discussion of the plurality of values perspectives and associated ecosystem services preferences that are intrinsic in policy discourse. In some instances I have stretched the concept to also engage system elements in the natural and social worlds. While these elements can be seen as material features, they are also artefacts of their respective disciplines. For example, ecological components exist naturally, but the

ecological vocabulary we use to describe them is part of the social world of ecologists. They are used for communicating their priorities just as institutions are the medium through which social scientists work toward common ground, and the resolution of governance dilemmas that concern ecosystem services, either implicitly or explicitly. Where ecosystem services were used to examine technical features of natural or social subsystems I have tended to use the term 'analytical lens.' The intention was that it would frame the analysis as a more objective and linear exercise than would be entailed in any attempt to fully deconstruct the social dynamics of ecosystem services used as a boundary object. In this section we will look at ecosystem services in this latter role.

In attempting to disentangle preference formation, policy impact, and ultimate effect, in terms of a gain or loss of resilience, the mechanisms by which political actors mould the debate toward their individual objectives are of central interest. Political interplay in the form of deliberate linking of institutions are one possible means by which interests can be gathered and directed toward a given goal (Young, 2002a). This process of steering collective opinion can be assisted through the use of boundary objects that either have or can be infused with symbolic value, in this way embodying a political message and serving as a gathering point (Kimble et al., 2010). Evidently broadening support in this way can be a source of political power when contesting certain institutional outcomes. If contesting political actors observe different institutional frameworks, there may also be a degree of functional interplay occurring through the boundary object, between overlapping social worlds. In natural resources management this could involve claims and counterclaims about property rights across scale between higher levels of government and local actors. In an examination of issues of scale, Young (2002b) provides an example that

can be used to illustrate this theoretical point by contrasting land claims of national governments and those of indigenous groups. By various means governments come to view certain lands as public property over which they, as the public's representative, have authority. However indigenous people often maintain unique land-based social and cultural practices and thereby maintain their claim on the lands, which are generally understood in terms of common pool resources. Here we have two distinct overlapping worldviews on appropriate governance arrangements as pertain to certain lands, and these draw on a number of other institutional features including property rights as well as social practices and historical factors. In this example land tenure is a boundary object though which different institutional perspectives are interacting. Analysis of institutional interplay, in the context of overlapping and contrasting social (and institutional) perspectives, helps to expose the political dimensions of what is often a fairly contentious subject, while also linking institutional antecedents and outcomes with intervening political factors.

Similar theoretical arguments can be brought to bear on the case of source water protection (SWP) in Ontario. Positioning SWP as a boundary object reveals the institutional interplay that unfolds between two different institutional perspectives: those of the provincial government and those of highly agrarian, rural watershed communities. Differences in policy images between these two groups has been discussed in Chapter 4, and elsewhere (Ferreyra & Beard, 2007). However this classification can confine SWP to the role of policy-object for which there may be multiple, static images. It does not open up an analysis of the political use of existing institutions that takes place in contesting a dominant definition. The provincial architects of the CWA saw SWP primarily as a public health policy. However, in order to realise their policy goals, resource management

institutions needed to be engaged and the Department of the Environment was put in charge. This may have catalysed a certain perception that equated SWP primarily with environmental objectives. When agricultural organisations and media used terms such as "property rights" and "right to farm" (Romahn, 2005; Cumming, 2005; Van Dussen, 2014), these became powerful symbols that engaged deeply engrained social institutions within rural communities, such as autonomy and self-sufficiency (ABMV, RSN). In doing so they stuck a common chord that brought a heterogeneous stakeholder community together around shared institutions. In addition to contrasting policy images, institutional interplay unfolded in the context of overlapping but different worldviews on SWP (the boundary object), and the debate took on the features of a contest over public encroachment on private property. Where local stakeholders ultimately wrote the policies, there is a distinct asymmetry in decision-making power. This leads us to expect SWP policies that reinforce rural institutions, in particular those respecting the right to farm on private property, which is what I found.

While this is a theoretically interesting finding it also contributes to a recurrent theme in my work, where the interests of agriculture seem to varying degrees to be at odds with those of society. In the next chapter I discuss case selection and the effect this can have on research results and conclusions. However from the perspective of ecosystem services and systems-based analysis, it is important to highlight that agricultural lands produce a range of vitally important ecosystem services, and that many of these are not accounted for. From an ecological perspective this community of stakeholders are of central importance to improving social-ecological resilience. Their important contributions should be accounted for when seeking to improve water resources governance.

5.3 Ecosystem Services as a Diagnostic Tool for Policy Analysis

The degree and means by which different resource units (or ecosystem services), are valued is in part a matter of preferences. These in turn are partly contingent on socio-cultural values and norms, and are resolved within the social subsystem, with input and feedback from the ecological subsystem (either experienced or anticipated). In Mississippi Mills, landowners in the upper watershed are experiencing longer recreational seasons. In Chelsea, earlier inhabitants are worried that their environment and way of life will change. Farmers and rural inhabitants alike are worried that they are losing control over their land and right to self-determination. In all three studies ecological conditions and preferences are linked with values in a plurality of arrangements. These different perspectives interact relative to political and economic structures and forces, formal and informal institutions, and the actions and expected actions of other actors.

These findings underscore the importance of understanding ecosystem services in terms of demand. Policy outcomes are shaped by dynamic social processes that are in turn driven by demand for ecosystem services (or by resistance to the provision of specific ecosystem services). These policies then regulate access to the environment and in some cases affect the capacity and extent to which ecological systems can provide needed ecosystem services in the future. Political contestation over competing preferences for how to configure human-environment relationships underscores the fundamental importance of shifting our perspective to incorporate an analysis of demand for ecosystem services. Approaches to ecosystem services valuation that do not account for demand risk

overlooking key factors that determine actual use (which is a key variable for gauging relative scarcity, and therefore economic value). Furthermore, they will be limited in their capacity to anticipate how policy permits, privileges and prevents the use of different ecosystem services, thereby undermining more anticipatory approaches to resource management that account for ecosystem response in present day decision making.

Water resources governance processes will thus need to resolve a range of actors' preferences and political positions. Meanwhile, actors also link objectives from other resource and non-resource policy domains to water resources governance processes (see Section 6.1.1). My findings suggest that water governance could benefit from combining systems-based analysis with open and inclusive governance processes to stimulate social learning and values shifts relative to information on the range of inherent risks. By identifying relevant policies and institutions, as well as involved actors and an assessment of the magnitude and orientation of their interests, I was able to detail how the political economy of water, as a multi-use resource, plays out in a systems frame. This in turn has been instrumental for determining how certain ecological and economic risks might develop (see Chapter 2), or how social and cultural values can shape valuation in terms of preferred policy options (see Chapters 3 & 4), with relative risk perception exerting a shaping effect on these underlying values. Since policy decisions are a determinant of societal direction (Howlett & Ramesh, 2003), with resulting social values structuring the future political environment (Pal, 2010), extending a systems perspective in a temporal direction may help anticipate implications for future policy and political processes.

As Ostrom and others from her scholarly community have asserted, there is no panacea, no universal institutional form (Ostrom, 1990). Rather, what is appropriate for a

given situation will be a function of historical context, political and economic forces, cultural norms, and the idiosyncratic composition of local ecological conditions and constraints. Developing institutions and governance processes that embrace this complexity will serve to maintain the capacity of ecological systems to respond to the array of stressors, in turn increasing overall social-ecological resilience.

Chapter 6 – Implications for Policy, Governance and Resilience

The recognition that environmental externalities can have real economic costs is ancient in origin, dating back at least as far as Plato's reflections on the impacts of deforestation (Mooney & Ehrlich, 1997), with the emergence of environmental economics in the 1960s constituting a modern response (Pearce, 2002). Through a focussed application of the conceptual framework we have observed how economic activities in one location can impact the provision of ecosystem services elsewhere, revealing the embedded competition between different users in complex social-ecological systems, and providing more detail on the mechanisms through which damage to the environment can exert economic costs. The diagnostic value of this approach has broad policy relevance. It was used to identify gaps in the extant policy framework that could allow economic risks to develop as a result of ecological change. This approach also makes a more general case and provides a basis for improved policy approaches to managing upstream-downstream competition between users within watersheds, the importance of which is underscored by the observed and expected impacts of climate change. Moreover I have been able to make a number of general observations concerning the conceptual evolution of ecosystem services.

This chapter reviews themes and theoretical arguments that have been discussed so far to highlight the policy significance of my research findings. Each study has implications for water resources governance. These range from more general arguments to fairly specific recommendations. My findings also have theoretical implications that are relevant for policy studies. Those will be discussed later in this chapter. Section 6.3 provides a high-level overview of the research as a whole. Returning to the central paradox, I then discuss how the theory and methods developed and deployed in this research can be applied across

a wider range of water governance cases, with the ultimate objective of overcoming enduring challenges to the development of governance institutions that are better aligned with ecological resilience and sustainable water resources management objectives. This is achieved, in part, by clarifying the ecological dimensions of governance institutions and processes that are focussed on societal concerns. For example, infrastructure policy in Chelsea has implications for population density, community culture and the relative balance of power between different segments of the community. What is a fairly narrow policy focussed on municipal development has the potential to play out in a systems frame and redefine the manner in which Chelsea interrelates with its ecological surroundings. This innovative approach to water governance provides a more complete context for understanding actor-institution interactions and the impacts of associated institutional outcomes on social and ecological subsystems, either directly or as arise through feedback and systems effects. Limitations and future research will be touched upon before I close with some concluding remarks. First however I will discuss specific implications of this work for water resources governance.

6.1 Implications for Water Policy and Governance

Through analysis of governance processes, theoretical literature and empirical case studies, my research has reinforced some existing ideas and developed fresh perspective on a number of water resources governance challenges . This section will make some high-level observations on the relevance of my research findings for policy making processes before moving on to address three specific policy issues: watersheds as policy making units for water resources governance, implications for property rights, and the need for leadership on climate change.

6.1.1 Water-Centricity in Governance

Watersheds have long been considered an ideal unit for balancing the quality and quantity of water supply with varying demands and pressures on the resource (Bandaragoda, 2010). The hydrologic argument for watersheds as a basis for water management is reasonably straightforward. With the exception of groundwater flows, evapo-transpiration, and exports of water in bulk or as an embedded material (for instance in agricultural products), water that falls as rain or snow follows hydrological pathways through the watershed to its outlet. The effects of any human activity that intervenes in this process can readily be measured and managed (Biswas, 2004). However, numerous fault lines have been revealed in this conceptualisation and much has been written about the challenges and failures of integrated watershed management (Medema et al., 2008; Jeffrey and Gearey, 2006; Saravanan et al., 2009). These governance challenges stem from the tension between a conceptualisation of water management that is grounded in natural processes, and the implementation of this conception where human processes are the dominant variable. The suitability of watersheds as a unit for crafting policy that can mediate our interactions with water resources is a subject that has come up several times in this work, and is an open debate (Cohen & Davidson, 2011). My research has shown how contestation between interests can challenge watershed-based governance processes, in particular when actor networks or objectives are to some extent external to the watershed.

A good first step is to distinguish between water management and water governance. In a basic sense, governance involves influencing and making policy decisions while the focus of management is on implementing the outcomes of these decisions. The manner in which actors may influence decision making depends on a number of factors. In

Chelsea interested individuals were able to participate in a workshop that sought to identify and weight different concerns around the development of water and wastewater infrastructure. Other modes of influence included local media or addressing political representatives directly. There were numerous opportunities and venues through which societal actors could influence the development of the CWA, including the O'Connor Inquiry, subsequent Advisory Committee activity, online consultation, media, and engaging government representatives. Many of these venues were not equally accessible to all stakeholders, with organisational actors generally having greater access. For instance, the online Environmental Registry provided an open opportunity for feedback however findings from expert interviews indicated that most decisions were largely formed through interactions with a core group of 300 to 400 stakeholders prior to being posted for online comment (Gov). Governance took on a new form during implementation of the Act. Now private citizens were no longer operating at the periphery, with little influence on decision making, but instead representative citizens were the decision makers.

Collaborative approaches that involve small group discussion often point to mutual learning as a primary output (Wilson & Horwath, 2002). SPC members (RSN) and stakeholder groups (Mun) echoed these findings. While not wholly convinced that source water protection (SWP) was necessary to ensure safer drinking water, or that the expense was justified, there was a shared sentiment that the process resulted in significant learning at the local level and that this may have long-term value. This could include increased capacity of local governments to defend local interests in the face of pressure from higher levels of government (Ivey et al., 2006). Participation can also increase local buy-in, backstopping voluntary compliance while also potentially increasing adaptive capacity

(Engle, 2010). However the degree to which common understanding arrived at within SPC deliberations reached the wider community is uncertain. While there was some media coverage, and extensive efforts by SPC to reach out, testimony from SPC members is mixed but generally suggests a relatively low level of engagement (ABMV, SGSNBP, RSN). Accordingly, it is questionable whether a general values shift occurred, thereby resulting in a widespread appreciation of the impact of individual action on the safety of drinking water sources, or in an awareness of the constraints and concerns of other watershed residents.

Respecting that communities can be sites of conflict as well as collaboration, social learning can be a positive development when it helps address information asymmetries and exposes individuals to other viewpoints. While these findings have also raised a number of concerns, empirical evidence from the SPCs reveals high levels of collaboration, even in the face of opposing interests (for instance between land owners and municipalities). Arising from a sense of rural unity in the face of urban advance, we observed a broad coalition of support that to some degree recast the findings of the O'Connor Inquiry in defence of agricultural land use. The addition of a decision-making function to a small group process, where values were shifting in part due to external forces, created the potential for capture and an asymmetrical distribution of power. This is illustrated by the degree to which the Ontario Federation of Agriculture had influence over policy making, both directly through the SPC and source protection plans, but also through the efforts of farm media. The main outcome of concern being that non-point source pollution management is now largely in the hands of non-point source pollution producers.

Based on concerns over legitimacy and accountability, source water protection

practitioners argue that decision making should reside with higher levels of government (de Loë & Murray, 2013). Considering the potential for capture at the local level, my research reinforces these cautions against full devolution of policy making authority. Instead local participation has a key role in supporting more informed decisions at provincial level. Legitimacy, conflict resolution, and social learning can all benefit from collaboration, but when results are non-binding this can result in a failure to follow-through (de Loe & Murray, 2013). A binding commitment to act on recommendations could enhance the integration of these different components of water governance. This is consistent with Principle 5 of the modified *ideal deliberative procedure*, as discussed in Chapter 3. Moreover, if participatory governance processes are informed about how socio-cultural, economic and environmental risks may arise from system effects, this could lead to a shift in values that support enhanced social-ecological resilience, and more stable and certain progress toward a range of social, economic, ecological and cultural goals.

So while there are good arguments to be made for local involvement, and for the usefulness of watersheds for managing hydraulic parameters, the "fit" is wrong for decision making. The factors that affect governance and actor strategies extend beyond the watershed boundaries, and orchestrated intervention by provincial-scale actors can overwhelm local representatives. Furthermore "water problems" are not just about water. Indeed, the primary objectives of actors engaging in water governance may be quite separate from the objectives of water resources governance, irrespective of the question of whether watersheds are an appropriate jurisdiction for decision making (de Loe & Patterson, 2017). For instance, oil sands development has a significant impact on water governance however the main objectives are energy development. Similarly, municipal

development in Chelsea and Lanark Highlands has clear impacts for water resources locally and downstream, but the focus was on municipal development and revenue generation. Moreover, water resources governance affects and intersects a wide range of interests that may be in opposition to one another to varying degrees. Political contestation among actors of varying levels of power and influence can be highly determinant of observed outcomes, including perceived governance failures (Johns, 2008a). Finally, political jurisdictions do not generally align with watershed boundaries, and relevant legislation is routinely created at higher levels of government where it is subject to imperatives that can differ from local level (e.g., watershed) concerns (Davidson & de Loe, 2014). Since the drivers of change in the water sector may originate within other sectors and respond to other priorities of society (Jaspers and Gupta, 2014), it is therefore essential to cast any analysis of water governance within its wider social context, while also ensuring that local concerns are not lost in the process.

6.1.2 Property Rights and Incentive Structures

To this point, institutions have been discussed in various empirical contexts but are generally referred to in the abstract, as collections of rules and practices applying to some substantive matter of common concern. Taken together these institutions define the options and incentives that shape actor strategies and decisions, with property rights being a key institutional element among these rules and practices. In many instances property rights help mediate our interactions with nature, and a functioning property rights system, including adequate monitoring and enforcement, is an important institutional feature when resources are either scarce or only renewable to a limited extent. However property rights

can also be quite rigid. Where systems effects result in quasi-open access conditions, property rights may structure persistent overuse and sub-optimal outcomes. In this section I will briefly summarise findings from environment-economy modelling in the Mississippi Valley watershed. These will be extended to an analysis of water rights in Canada before touching on wider implications, as indicated by my research.

Consistent with ideas of interacting resource regimes (Ostrom, 2009), my systems-based analysis of ecosystem services has shown how property rights can interact with one another at an ecological level such that exercising property rights in one location can affect the ability of others elsewhere to do the same. Whether or not different uses are rivalrous depends on the assemblage of ecological linkages, patterns of land use and policy coverage; in other words the systems context. Two examples from the Mississippi watershed case study are the linkages between upstream and downstream users, and those between farmers and users of adjacent or downstream water systems and wetlands. Each case raises questions over duty of care and the contingency of property rights. These factors will be now discussed in more detail leading into an analysis of implications for water rights in Canada.

Riparian rights are commonly used to manage upstream-downstream interactions between rights holders. The focus of these rights systems is on maintaining the quantity and quality of water available to downstream water users. My central argument is that water needs to be conceptualised more broadly, and that downstream rights holders can be more accurately identified by examining the range of water-based ecosystem services that are being used. Framing water in a systems-context provides a more nuanced characterisation of how different actors, and associated rights entitlements, are interrelated. The same core

arguments can be made when the interactions in question are between agricultural actors holding private property rights, and others in adjacent or downstream ecosystem types. In this case, activities on agricultural lands do not necessarily affect water through direct use, but may nevertheless affect water quality, quantity and the availability of ecosystem services elsewhere. In both of these situations, a systems-based analysis can identify interactions at an ecological level that may result in an implicit duty of care. These core concepts, that water-use is not limited to extractive or consumptive uses, and that these wider uses introduce new considerations in terms of duty of care, will now be extrapolated to the case of water rights in Canada.

Water allocation in the southern provinces of Canada (except Quebec), is generally governed under two systems: riparian rights and prior allocation (Christensen & Lintner, 2007). In a riparian rights system the right to use water is contingent on the ownership of land that is adjacent to the water body from which the water will be extracted. There is also an obligation to downstream riparian rights holders such that one person's use does not affect the quality or quantity of water available to others. As Canada expanded westward, settlers brought this system of water rights with them. However it was not appropriate for the more resource-constrained environment of western Canada. Development of the resource would necessarily affect downstream conditions. As a result, users could not meet the duty of care embedded in riparian rights. A new system of prior allocation was developed. This is the second main approach to governing water allocations in Canada. Also known as first-in-time first-in-right, irrigation districts and cooperatives have the right to use a fixed volume of water annually, regardless of environmental conditions and other users. This is subject to a ranking system based on seniority, and the fraction of total

permitted withdrawals that is actually used determines what is left in the system for ecological and other uses.

There is an unavoidable tension between acknowledging the ecological linkages between rights holders (as embedded within riparian rights), and the practical need to impose some level of downstream impact in order to develop the resource, and support the economic wellbeing of society. However the priority system also privileges those actors, primarily agriculturalists, who were involved in its creation. New uses such as urban development and ecological flows are disadvantaged within the priority system. In 1999, the Alberta Water Act established water markets. These markets removed conditions of appurtenancy, where rights are legally tied to land ownership, and allowed for the transfer of rights between holders. However all pre-existing rights were grandfathered in, and this has meant that they are essentially untouchable. These generally belong to irrigation districts and cooperatives and may only be transferred following a successful plebiscite. Technically, the state actually owns the water (Glenn, 2010), meaning that water allocations involve rights of use, not of ownership. However water rights are highly political, with agriculturalists holding significant political clout. Taken together, this amounts to a politically entrenched system of property rights that crowds out new uses, which from a systems perspective may be of higher priority (if they are needed to sustain the ecological resource), or higher value (for instance urban or industrial development). Moreover, unlike other jurisdictions, such as Australia, where all rights are pooled and rights holders get a fixed percentage of available water resources in any given year, the allocations in western Canada are volumetric and fixed. Although there is a ranking system that adjusts total allocations during times of shortage, giving priority to the earliest rights

holders, given the strength of political interests, seasonal restrictions would be expected to encounter a certain amount of resistance, undermining the overall responsiveness of the system. With climate change introducing greater seasonal fluctuations, this system of rights could result in significant additional pressure being placed on a strained resource that experiences periods of increased scarcity. Applying an ecosystem services systems-based perspective suggests that this system of property rights may not only structure sub-optimal outcomes in terms of disadvantaging higher value uses, it may also negatively impact the resource base itself, undermining social-ecological resilience and introducing a range of risk factors. More fluid property rights could help address resilience concerns, and would be consistent with a shift toward making institutions more responsive to feedback from the environment. However existing rights holders could be harmed by such an approach. Some manner of compensation would be needed to reach an equitable solution and to overcome political resistance.

However property rights are only one part of the institutional incentive structure that guides decision making. Institutions are in part defined by practice and norms. Recalling concerns around the appropriateness of aggregating individual preferences to value public goods where groups of people have a shared interest (Farber et al., 2002), engaging rights holders in dialogue with other users to better understand how system elements, including rights entitlements, can interact, may help address information asymmetries among involved parties and open up rigid institutional structures in order to engage with system challenges. Municipal representatives suggested that a key outcome of the source protection process was an awareness of the concerns and constraints of others. How this translates into progressive policy is another matter. As mentioned, water in

Canada ultimately belongs to the state (Glenn, 2010), where ownership can be defined as possession of the rights to access, manage, exclude and alienate (Schlager & Ostrom, 1992). However, the manner in which the rights of ownership are deployed is a function of political circumstances. These in turn are subject to attitudinal shifts among the electorate. Movement on water rights in Canada could benefit from working toward a shared understanding of social-ecological resilience, and in particular a discussion of economic and other risk factors that are embedded in current rights regimes.

In addition to ensuring that compensation and fair treatment are part of any changes that follow political action on water rights, it is worth noting that changing rights regimes could also result in wider cultural changes. In Chelsea, the property rights of longer-term inhabitants would include expectations and felt entitlements to a certain sort of environment, with these surroundings being interlinked with their chosen lifestyle. Changes that would result in higher-density urban development would also change the demographic makeup of the municipal population. Newcomers with different values would become increasingly influential, leading to concerns that community culture, largely reflecting that of prior inhabitants, would be increasingly subject to change. For western provinces any movement away from the current system of prior allocation would increasingly mobilise new users and displace old users, mainly agriculture. Given the relative scale and presence of agricultural land use, it is unlikely that the overall cultural landscape would change dramatically, at least not in the short term, but it would be expected to change. If perceived threats to existing cultural institutions are not anticipated and accounted for in the analysis, they could become a source of concern and political resistance.

From an instrumental perspective, systems-based modelling may be helpful for identifying where risk factors stemming from property rights are relatively more intense, and where intervention could enhance social-ecological resilience. The form and processes by which the details of any intervention are determined should have technical inputs but should not be technocratic. Affected stakeholders need to be engaged while taking precautions to avoid capture. The prospect of more fluid property rights that are responsive to ecological feedback is fraught with concerns. Some level of compensation would be required or else the economic benefits that are associated with functioning property rights systems could begin to erode. What constitutes an appropriate level of compensation, who gets compensated, and how are these decisions made, are all difficult questions and outside the scope of this research. However if social-ecological resilience is to be taken seriously this is a vitally important discussion to undertake.

6.1.3 Climate Change

The implications of climate change for water resources governance are many. Increasing temperatures lead to higher rates of evapo-transpiration. These in turn result in more water vapour circulating in an atmosphere that itself has trapped more energy and is subject to greater levels of meteorological activity. In much of Canada this means a higher frequency of more intense precipitation events with longer dry spells in between (Lemmen et al., 2008). The expected hydrologic conditions upon which water management policy and protocols have been established are changing dramatically, and our governance processes are struggling to keep up.

From a review of the relevant policy framework in the Mississippi Valley

watershed it is apparent that key provincial instruments, such as the *Nutrient Management Act* (2002), *Drainage Act* (1990) and *Pesticides Act* (1990), have not explicitly engaged with climate change. This has two implications. First, building capacity to adapt to the impacts of climate change is not being mainstreamed within relevant policy. Moreover, the lack of provincial leadership may excuse a lack of action at a municipal level. From a systems perspective these conditions exacerbate political factors that underlie ecologically derived economic risks. Prior to the contamination of a municipal water supply in Walkerton, the focus of water pollution management in Ontario was on point source emitters. No specific policy instruments were in place at the time to address non-point source pollution, and the clearest policy response since has been Ontario's *Clean Water Act* (2006). However this legislation is focussed on protecting source water for municipal utilities rather than water quality throughout the watershed (Johns, 2008b). Broader management of non-point source pollution falls largely to the *Nutrient Management Act* (2002), *Pesticides Act* (1990) and *Drainage Act* (1990), none of which make explicit provisions for addressing the impacts of climate change. Furthermore, while the Provincial Policy Statement (PPS) states that, "Planning authorities shall consider the potential impacts of climate change that may increase the risk associated with natural hazards" (MMAH, 2014), the degree to which official plans follow the PPS varies from municipality to municipality. The Official Plan for Lanark Highlands makes no reference to climate change (MMAH, 2016), while Mississippi Mills' Official Plan acknowledges climate change, but is primarily focussed on mitigation of greenhouse gases (MMAH, 2006). Given the significant impact climate change is expected to have on the watershed, the lack of policy protections increases the likelihood that the ecologically-derived economic risks, as

discussed in Chapter 2, will materialise. While both Official Plans have policies in place to safeguard ecosystem integrity and natural heritage, municipal sources indicate that by-law variances are relatively easy to obtain (Mun), and this has the potential to undermine such provisions, especially where there is a permissive municipal council. Taken together these factors demonstrate how competing interests can undermine climate change adaptation. Furthermore, these findings suggest that the social-ecological and political conditions in the watershed could allow identified economic risks to develop.

While municipalities have a role to play, as outlined in the PPS, system effects involve factors that are outside municipal boundaries, and are in that sense beyond their jurisdiction. It is therefore appropriate to look to higher levels of government for leadership on adaptation. While there is some indication that provincial governments are in the process of responding, it will take time to adjust and get all parties onside (WS, Mun). Moreover, reflecting on the political challenges associated with source water protection (SWP), we could expect the province to move carefully on mainstreaming climate change within provincial policy. Nevertheless, the Mississippi watershed may be an exemplar of a much wider problem where upstream-downstream competition, ecological linkages between rights holders in neighbouring ecosystem types, and gaps in the policy framework can result in pronounced ecological and economic risks, and a loss of social-ecological resilience. These factors and potential downside risks would likely be more pronounced in more densely populated watersheds.

Considering political disincentives at a provincial level, there are strategic opportunities for federal action that do not overstep constitutional boundaries. If federal tax data on active businesses, currently available through the Canadian Business Patterns

survey, were georeferenced, this would enable more accurate modelling of joint social-ecological systems. Rather than working on a census subdivision-scale (CSD), georeferenced data would locate active economic entities within the CSD, increasing the resolution of the model and the accuracy and effectiveness of any attempted policy interventions. Also, leadership in policy areas where there is a federal presence, such as fisheries and navigable waters, could create needed policy protections and open up political space for provincial action. Recalling that the Mississippi River Water Management Plan uses historical data for lack of a better option, there is an apparent need for additional research to provide the scientific basis for adaptation measures. Moreover, the actual implementation of climate change adaptation needs to be better understood. Ultimately many decisions will have to be made locally so any top-down action would benefit from a collaborative approach that accounts for the concerns and values of local stakeholders.

6.1.4 Institutions, Politics and Policy Processes

If ecosystem services are not the explicit focus of the policy process, their relative value may nevertheless be determined by decisions made and the balance struck between the different policy objectives of concerned stakeholders and governance actors. In other words, ecosystem services are embedded within many policy scenarios and, depending on decisions made, there can be a range of implications for ecosystem services. So it matters who determines policy goals and by what process. Alongside and connected with institutional and economic imperatives, political factors further shape policy outcomes for water resource governance. Small group deliberation has the potential to form and transform values. Through deliberation, participants are exposed to the ideas and concerns

of others and this can stimulate a level of values convergence. In the Chelsea case study the outcomes of the valuation process were turned over to decision makers. Source protection committees (SPC) were also observed to converge on a set of shared values, not only within groups but also across the SPC. In part this was driven by social narrative and a common concern about urban control over rural communities. This narrative was in turn mobilised and strengthened by media activities, particularly those of farm media outlets. It was also embedded within a longer historical series of events and institutional developments. The concurrent implementation of the *Green Energy and Green Economy Act* (2009), led to the widespread installation of large windfarms. These changed rural landscapes and raised the spectre of additional changes to come; changes that were perceived to largely benefit urban areas, once again at rural expense. This was a source of considerable frustration for many rural communities (Stokes, 2013). As such, the formation of values within the small group SPCs was shaped not only from within but also by factors external to the deliberative process. Another key difference when compared to the Chelsea case was that the outcomes of these small group processes were not handed over to policy makers. Instead, the SPC were the decisions makers, with resulting policies directly affected by any values shift among committee members.

These cases demonstrate how deliberation can help integrate widely held social values within policy making and any associated valuation of ecosystem services. However, if decisions are made external to the process and without accountability to it, the impact of this integration of values may be diminished, as was observed in the Chelsea case. In the limit, the legitimacy of the process may suffer, potentially undermining policy implementation and future policy participation. Alternatively, if decision making occurs

within the small group, the deliberative process could be captured by a subset of involved interests. This would challenge the ability of policy makers to realise design goals. In the case of the *Clean Water Act* (2006) (CWA), social narrative exerted a shaping effect on the manner in which many SPC members understood the policy objectives, and the context for their work. The result was that a dramatic difference in policy images emerged. The policy intention of the provincial government was to provide safer drinking water. However local stakeholders that were charged with implementing the policy perceived it quite differently, largely through a lens of urban indifference for rural areas.

Returning to issues of institutional fit, interplay and scale, producers and consumers of ecosystem services in the Mississippi Valley watershed are often not co-located geographically. The spatial separation could be as close as neighbouring ecosystem types, or in the case of upstream-downstream linkages, producers and consumers can be removed by several census subdivisions (CSD). The Mississippi in turn is one of many subwatersheds within the larger Ottawa River watershed, and may therefore produce ecosystem services that accrue to stakeholders outside as well as inside of the Mississippi watershed. Moreover, with a number of CSDs spanning watershed boundaries, the decision-making processes at the local level will in part respond to interests and concerns that are external to the watershed. Taken together we observe a high degree of misfit, as well as horizontal interplay between municipal-level institutions that affect ecosystem services provision, for instance in the Mississippi Mills CSD. These institutions also interact vertically with the watershed-level and the Mississippi River Water Management Plan (2006), as well as various provincial instruments and statutes including the Ontario Wetland Evaluation System (OWES), the *Nutrient Management Act* (2002), the *Drainage*

Act (2002), and the *Pesticides Act* (2002). Indeed, even some federal level provisions, such as regulations on fish protection and navigation, are applicable. Natural resource management institutions are therefore under a range of pressures from other institutions and actors from all levels. For instance, wetland management in Lanark Highlands, and the provision of wetland and downstream ecosystem services, is affected by provincial regulatory requirements and enforcement of those requirements. We can assume that whether or not farmers install and maintain filter strips will depend, in part, on their expectation of enforcement. The level of enforcement in turn is a function of resources and perceived benefit. Ideally these two calculations would be made relative to the expected impacts of climate change, in terms of occasional spikes in hydraulic loading on these strips, and the likelihood of compromised ecological systems downstream (for instance due to low flow conditions in receiving waters or drying wetlands). However climate change is either not acknowledged, identified but not acted upon, or acted upon but not for the purposes of local adaptation (see Section 6.1.4), within applicable policies. While these factors can lead to downstream economic impacts through systems effects, is it unlikely that these impacts figure into the incentive structures of either farmers or provincial enforcement officials.

Water management can also involve processes of institutional change, where discretion in interpretation or enforcement may lead to changes over time (Mahoney & Thelen, 2010). For example, wetlands designated through OWES benefit from additional protection to the degree they are incorporated within municipal plans. In Lanark Highlands a development-friendly town council was found to have a permissive stance on the enforcement of limitations embedded in its Official Plan. By-law variances were approved

without consideration of ecological linkages and system effects, thereby undermining those protections. The results from environment-economy modelling show how this could reverberate through the social-ecological system to introduce risks locally and downstream. From an institutionalist perspective, fiscal pressures at the municipal level provided an incentive toward additional residential and recreational property development. This led to the use of discretion in implementing local governance processes, and resulted in institutional drift. Existing rules were either neglected or enacted in ways that did not confront other objectives, such as development (Mahoney & Thelen, 2010). The net result is that institutional safeguards may not achieve the intended level of environmental protection.

The institutional dynamics underlying the development and implementation of the CWA raise similar concerns. Numerous umbrella organisations including the Ontario Federation of Agriculture, Conservation Ontario, and the Association of Ontario Municipalities, were engaged at the provincial level during consultation on the CWA. These groups testified at the O'Connor Inquiry, they were members of the Advisory Committee on Watershed-Based Source Protection Planning, and they engaged with government during the development and regulatory roll-out of the CWA, both directly and online through the Environmental Registry. These three main stakeholder groups also had significant influence at the watershed level during the implementation of the CWA by the SPC. Meanwhile, provincial policy designers had relatively little influence at that level beyond the inclusion of a non-voting MOE SPC liaison. This mismatch of resources and influence is in keeping with a decentralisation of decision-making; however it also created the opportunity for larger agendas to take hold at the local level. The efforts of the

agricultural sector were particularly coherent across scale. The presence of a broader agricultural agenda in a dominant position at the watershed level calls into question the fit between source water protection (SWP) as a policy objective and watershed-based policy making. While the technical aspects of SWP fit well within the watershed, the primary instrument through which SWP would be delivered was only watershed-based in form, not in function. These concerns are deepened by the presence of tacit support from an informal coalition that formed to defend rural livelihoods and communities.

Previous studies have commented on how the CWA could affect existing institutional arrangements, alongside concerns around democratic accountability. This research has drawn similar conclusions, in particular around the options for appeal, and the manner in which the CWA has structured the governance of non-point source pollution.

The case of the Raisin-South Nation SPR highlights additional considerations. Unlike the other two source protection regions (SPR) in the analysis, many of the source water protection policies in the Raisin-South Nation SPR that address agricultural pollutants default to the provisions of the *Nutrient Management Act* (2002) (NMA), or other existing regulation. The CWA and the NMA are fundamentally different instruments. While the CWA is ostensibly driven by science-based risk management, we have seen that institutional outcomes are also highly contingent on political forces and external agendas. Meanwhile the NMA lays out a set of prescriptive regulations, specifying setbacks and other requirements based on certain thresholds. Where applicable, these regulatory requirements constitute the source protection policies in the Raisin-South Nation SPR. In the Saugeen Grey Sauble Northern Bruce Peninsula and Ausable Bayfield Maitland Valley SPR new nutrient management policies were layered over top of existing NMA regulations,

effectively changing the institutional arrangements pertaining to nutrient management. The CWA states that where multiple nutrient management policies are applicable the instrument that offers the greatest protection for drinking water shall prevail. Over time however layering can result in displacement of earlier institutions (Mahoney & Thelen, 2010). These effects are more likely where ambiguity exists in the nutrient management regime (for instance around grazing and pasturing activities), and could be exacerbated by uncertainty about the extent of paramountcy the CWA has over other acts, alongside any perception that the CWA's science-base and focus on safe drinking water results in a more stringent set of safeguards. As practices continue to develop on the ground, the rules in use may begin to favour SWP risk reduction policies on issues where there is less ambiguity, whether or not they actually provide a higher level of protection for drinking water. These instruments also offer more local control in terms of discretion in implementation, further incentivising their use.

The manner in which decisions were made in the Raisin-South Nation SPR is also of interest. Ultimately the policy decision to defer to existing policy instruments came from the SPC however this decision was consistent with opinions expressed by Source Protection Authority personnel well in advance of SPC deliberations (Thompson, 2006a, 2006b). As the legal force of the CWA came into scope the only way to achieve these policy preferences, and to ensure that NMA policies applied, was to embed them in the Source Protection Plan. Recalling that Conservation Authorities were non-voting and intended only to be a source of technical information relevant to establishing the science-base for policy making, they had limited official capacity to defend the status quo. However they were in a position of influence. In this case, policy advice was offered in the media

and possibly during SPC deliberations. This resulted in a displacement of science-based risk management approaches called for by the CWA, and a retrenching of NMA regulatory requirements within a Source Protection Plan produced under the CWA. As a result, the net policy change was limited whereas source water protection was intended to result in measurable improvements and risk reduction. Nevertheless, the net effect in terms of protecting drinking water sources may be the best possible outcome considering the alternative: a politicised policy making process under heavy influence of the agricultural sector with limited avenues for appeal. The situation presents a dilemma. The central policy objective was to reduce risk through science-based source water protection. However, following on O'Connor's recommendations the process was also to be watershed-based. The importance of voluntary compliance when addressing non-point source pollution further increased the need for local input in order to seek and maintain legitimacy. To maximise local control, and distance itself from source protection, the provincial government decided to go beyond consultation and devolve actual decision-making to the local level. So there is a highly developed scientific framework on the one hand, but this feeds into a politicised policy making environment on the other, where individual actors have significant capacity to interpret science advice relative to other concerns, raising some doubt over the degree to which the original intentions around risk reduction have been realised.

Similar overlap between political factors and institutions was observed across the empirical case studies. Indeed, institutions were vulnerable to intervention by political forces during processes of institutional formation and implementation. Furthermore, cultural perspectives can have a normative effect on the policy process, while policy

decisions can potentially impact community culture. In Chelsea the town council had a vision for growth that was at odds with the interests of many of its citizens, as expressed in the Vision Chelsea report. Increased development would result in an influx of new residents. This in turn could bring about a shift in political power as a growing population within the town core shifts the focus of municipal politics, affecting the level of influence and representation experienced by residents living outside the town core. Previously the majority of residents lived on large treed lots with ample space between neighbours and ready access to forest ecosystem services. Any shift of political attention toward the interests of town dwellers could have implications for the relative provision of different ecosystem services. In the Mississippi Valley a thriving and growing recreational industry provides a strong incentive for developers and municipal politicians to increase the numbers of cottage lots on lakes in the upper watershed. This process is already underway, as seen around White Lake. It is experiencing problems with blue-green algae (WS) but continues to be developed (Mun), despite concerns over ecological carrying capacity. In the process, political representatives have demonstrated a willingness to grant variances to municipal by-laws, with choices in one CSD resulting in repercussions elsewhere. This underscores the larger issue where interests of one jurisdiction may be different from neighbouring jurisdictions, or other levels of government. As with the Chelsea case, it also raises questions around whose preferences count – existing or future users – and to what extent. At a minimum these policies should do no harm however there may also be benefits from development that accrue to existing users, for instance through increased municipal revenues, and these would need to be taken into account.

Some degree of reflexivity between political concerns and water governance, or

policy-making that has implications for water governance, is appropriate in a democratic society. Nevertheless, it is important to balance stakeholder concerns with ecological considerations, and avoid capture of governance processes by powerful interests. In looking at the institutionalisation of the CWA, situating the locus of power at a watershed level exposed governance processes to a host of political pressures that were ultimately determinant of institutional outcomes. In-depth study of the political and institutional dynamics of river basin organisations describes a dilemma related to decentralisation. More centralised and closed decision-making processes were found to be better able to develop technically sound and rational policy. However this was achieved at the expense of legitimacy. Meanwhile, more decentralised processes have higher levels of legitimacy but lead to sub-optimal outcomes from a technical perspective (Meijerink & Huitema, 2014). Capacity at a watershed level to represent and communicate local interests to higher levels of government has been identified as a key ingredient for success in source water protection planning (de Loe & Murray, 2013). Nevertheless, politics loomed large around the issue of source water protection in Ontario.

6.2 Policy Relevance for Policy Studies

In this section I will discuss the implications of my research for the theory of policy studies, including how I measured policy change and how this led to particular findings. I will also discuss a possible extension of Young's work on institutional fit. The implications of case selection for the generalisability of findings from the Chelsea case, and my institutionalist analysis of source water protection policy outcomes in Ontario, will also be discussed. On both counts it is difficult to make conclusive statements that are capable of filling any particular theoretical gap. Rather, the results of my research are better suited to either

support or detract from existing theory, while also resulting in new hypotheses, as well as some causal templates that may be useful in other similar settings.

When approaching a complex research scenario it is important to identify the most relevant variables among the many possible candidates. This helps to avoid overloading the analysis such that it becomes difficult to make any conclusive statements. Analysis of policy change presents one such challenge. Many factors have some connection to the case at hand but only some are of core relevance, and among those some are deterministic of observed outcomes while others are dependent on the process of change itself. Cashore & Howlett's means-ends framework (Cashore & Howlett, 2007; Howlett & Cashore, 2009) provides a useful conceptual expansion on Hall's levels of policy change (1996). Application of this framework to the analysis of drinking water policy in Ontario helped isolate source protection committees (SPC) as being of central importance to explaining policy change observed during implementation of the *Clean Water Act (2006)* (CWA). While the existence of a connection between the SPC and the policies that were created at the watershed-level is obvious and implicit, the disaggregation of drinking water policy over two historical time-steps revealed a pattern of engagement over time. Through this analysis it was apparent that devolution of decision making to the watershed level was the critical policy variable, with the strategies of the three main stakeholder groups responding to one another as they pulled in different directions to shape the specific terms upon which this devolution of decision making would take place.

Disaggregation of the CWA confirmed what other scholars have concluded about the appropriateness of the watershed as a decision-making unit. Interestingly, the misfit between SWP as a policy objective, and the watershed as a unit of governance to pursue

that objective, developed largely as a result of the devolution of decision making. Had decision-making authority remained with the province it is possible that the observed policy failure would not have developed. Accordingly, in addition to spatial, functional and temporal dimensions of institutional fit, this suggests that there is a degree of contingency between institutional fit and broader governance configurations. This can only be construed as a working hypothesis however it does underscore the value of disaggregating policy change into component parts, while potentially extending Young's work on the environmental implications of institutional design.

Environment-economy modelling in the Mississippi watershed raised analogous concerns. Here a systems-based analysis revealed how geographical separation between producers and consumers of water-based ecosystem services could lead to a range of risks. However it was in part relative to the governance context that these risks could develop. The political economy of decision making at the census subdivision level resulted in a degree of institutional drift such that existing instruments, which did fit the substantive management challenge of land-use planning and wetland protection, were bent to accommodate local priorities, potentially at the expense of neighbouring jurisdictions. Meanwhile, incentives affecting the enforcement and effectiveness of provincial instruments were similarly disconnected from the potential outcomes of systems effects. So the issue was not simply that producers and consumers of ecosystem services were not co-located, or that policies of an appropriate geographic scale were not in place. The issue of fit once again arises from institutional outcomes of relevant governance mechanisms, in this case reflected by an inability to encompass and manage the impacts of systems effects. My analysis of the joint social-ecological system identified affected parties who are

currently excluded from relevant governance processes. Further exploration of the working hypothesis that governance configurations, as determined by system linkages and the political economy, are a determining factor for issues of fit, could be expected to benefit from systems modelling of competing demand for ecosystem services, and from disaggregation of policy change into component parts.

Case selection can introduce elements of bias, for instance, if cases are chosen based on the dependent variable this can result in selection bias (Geddes, 1990). This further supports the disaggregation of policy change in order to better understand the interrelationships between relevant variables. It also prompts some discussion of cases chosen for this research and the potential for bias within the research results. Chelsea is a town with unique circumstances. It is located in Quebec only a short drive from Ottawa, Ontario. This is significant in two ways. Due to provincial tax laws there are strong financial disincentives to living in Chelsea. These are exacerbated by differentials in health care provision between the Quebec and Ontario sides of the Ottawa River. Perhaps as a result, Chelsea has not experienced the level of development pressure one would expect, given its abundant natural amenities and close proximity to Ottawa. Geological conditions have also limited the potential for development. As discussed in Chapter 3, the addition of municipal services opens up new development options that will result in higher density, (and likely) lower cost, housing units. The municipality and developers alike anticipate these will be attractive with the end result being a dramatic spike in municipal population. While many rural towns and peri-urban areas can expect to undergo development and intensification as the trend toward urbanisation continues to expand nearby cities, the unique circumstances of Chelsea may amplify this situation. So while we may expect a

difference in values between longstanding residents and newcomers, the extent of any impacts on culture and relative disempowerment of current residents may be unique to Chelsea and not as dramatic in other locations.

In my analysis of institutional change following the enactment of the CWA, the selection of source protection committees was based on the presence of a strong agricultural sector and the absence of large municipalities. These criteria introduced an element of bias, and while the efforts of agricultural interests through the Ontario Federation of Agriculture and farm media can be expected to have broad effect, additional research would be required to measure the impact on the membership, discourse and source protection policies of other SPC. The informal coalition that formed around a shared sense of rural marginalisation may still be found in other SPC, but the presence of larger municipalities, who are clear beneficiaries of SWP and have less of a stake in agriculture, could reduce its influence.

Primarily based on qualitative case studies, this research did not attempt to produce statistically significant results that are generalisable to a population level. Indeed, case selection has introduced a degree of bias, as has been discussed. Taking this more qualitative and nuanced approach has however enabled the consideration of a wider set of variables. Rather than offering new theory my research had added empirical weight to existing approaches. For instance, deploying Cashore and Howlett's (2007) framework within a broad institutionalist analysis of the *Clean Water Act* (2006) underscored the critical importance of differentiating between dependent and independent variables. This in turn led to the working hypothesis that governance context, determined in part by social-ecological systems linkages, can be a key consideration for institutional design. This

hypothesis was further defined through reflection on the Mississippi Valley Watershed case study.

6.3 Research Review and a Return to the Central Paradox and the Pursuit of Resilience

My research set out to examine the linkages between natural and social worlds, and the extent to which ecosystem services could be used as a boundary object in order to disentangle the many technical, socio-economic and institutional elements, including actor effects, and provide fresh insight into the factors underlying my central research paradox: why in spite of increasing knowledge and institutional safeguards do we still struggle to manage water resources sustainably. Examining water governance within a systems context led to several interesting theoretical and empirical findings.

Accounting for culture and values (Retallack & Schott, 2014), broader socio-economic interests, and patterns of demand for ecosystem services across shared landscapes (Brauman et al., 2007; Vrebos et al., 2015), proved useful for tracing how factors within the political economy affect institutional development and implementation in the context of water resources governance. This added weight to challenges around conventional notions about the appropriateness of watersheds as a basis for water resources governance (de Loe & Patterson, 2017). My doctoral research confirmed that policy making at the watershed level is heavily exposed to, and shaped by, external factors. Economic interests, media, and the priorities and policy processes of other levels of government all affect whether decision making at the watershed level can realise its expected benefits for water governance.

While a common hydrological system underpins the provision of these numerous

resources and services, decision making is often more narrowly focussed. Narrow approaches can undermine system resilience, in part because they prioritise certain ecological characteristics at the expense of overall ecological integrity (Walker & Salt, 2006; Tallis & Polasky, 2009). Moreover, there is a high degree of interplay between institutional regimes governing the different resources present on shared landscapes (Young, 2002a), making resource-specific institutions of interest to a wider range of stakeholders than initially perceived, and vice versa. Beyond their interest in the resources and services provided by the environment, actors pursue other inputs and respond to market demand, either domestic or international. It follows that there may be complementary policy objectives that are important to these actors in the pursuit of their varied interests. Decisions made in policy domains which may appear to be at arm's length from environmental management may nevertheless have an influence on the institutional outcomes of environmental governance (Adger et al., 2005). As a result, political and economic actors from a broad range of policy areas may be determinant of environmental outcomes (Levy & Newell, 2005), underscoring the importance of shedding water-centric views in order to more fully comprehend the drivers, actors and institutions that shape water governance.

Substantial progress has been made using various economic techniques to monetise ecosystem services for integration within decision making, and current trends show an increased interest in non-monetary approaches to valuation (Abson et al., 2014). Although this is an encouraging trend, it is largely an extension of the original project of economic valuation, undertaken in order to ensure that ecosystem services are accounted for in decision making. This "add-on" approach will not fully engage the social and political

dynamics that occur when values interact with one another in complex governance scenarios. The issue of cultural services provides a salient example.

Cultural services comprise a category of ecosystem services whose definition continues to shift (Satz et al., 2013). However moral "values" are increasingly understood to comprise a core component of defining the relative importance these services. Engaging values has relevance for shaping individual preferences, and depending on governance processes used, it may affect the direction of group preferences. Moreover, consideration of values and cultural perspectives is an important determinant of what *will be*, in addition to what *ought to be*, since policy choices help determine the path forward for society, and this has implications for culture. Accordingly, the debate around how best to account for cultural ecosystem services should encompass the importance of maintaining culture itself, as well as providing cultural services.

Conflicting value-bases and competing rationales can politicise decision making, the outcomes of which have implications for the provision of ecosystem services, and potentially social-ecological resilience. These dynamics were observed in all three studies, with different visions for societal direction having varied ecological implications, and hence affecting the relative provision of different ecosystem services. Indeed, politicised policy processes with embedded ecosystem services dimensions may develop policy goals that are not necessarily aligned with the continued production of the ecosystem services themselves. This calls for a more nuanced approach to understanding and crafting policy to manage the interrelationships between social and ecological systems, and the ecosystem services on which we depend. Assessing values can provide insight into actor motivation and why political-economic resources are being applied toward various ends. Thus, simply

confining the role of values to that of a non-monetary input into the valuation of ecosystem services could greatly overlook their relevance to important aspects of decision making and implications for society.

Where values and normative perspectives on policy priorities are of central importance to understanding actors-institutions interactions within the social subsystem, this is a subject that lends itself to examination through use of a boundary object, in order to reveal divergence and alignment among different perspectives. While scholars argue that ecosystem services have the potential to be used as a boundary object, current approaches lack adequate breadth and need to engage with literature on governance, among other areas (Abson et al., 2014). Moreover, an absence of collaborative frameworks is preventing ecologists from engaging with decision-making processes at a normative "purposive" level (Reyers et al., 2010). Ecosystem services' role and value as a boundary object depends on its usefulness by stakeholders in governance processes (Schröter et al., 2014). However the grounding premise that "nature provides" may situate nature as a thing apart from society, marginalising traditions that do not understand nature to be a separate category. As a result, an emphasis on nature may result in ecosystem services being more divisive than integrative (Hodgson et al. 2007). Meanwhile, resilience scholars argue that the capacity of social and ecological systems to adapt and endure is co-dependent (Berkes et al., 2008), and that governance approaches are needed that encompass both social and ecological factors, enable learning through open institutional arrangements within the joint system, and have linkages to relevant factors outside the system (Gunderson et al., 2006).

A good deal of policy work implicitly involves ecosystem services when making trade-offs between social objectives. Taking source water protection as an example, it is a

policy objective that involves at least six ecosystem services.[24] These services benefit other policy areas as well however SWP, as a response to Walkerton, was at its core a public health policy. In developing the source protection plans, local stakeholders drew on values that were partly shaped by external narratives as well as economic interests to structure a public health policy that had substantial influence on a number of ecosystem services. As a result these decisions have a wider policy effect through the social-environmental linkages that connect these services to users in other policy domains. In the Mississippi Valley case, modelling linkages between ecological and social subsystems called attention to the environmental-ecosystem services dimensions of existing policy arenas and processes. Equally important, a systems-based analysis showed promise as a means to delineate the economic dimensions of environmental issues with a useful degree of specificity.

Highlighting the utility of environment by translating ecosystem services into monetary terms was perhaps a necessary first step, demonstrating the value of considering environmental factors to the widest possible audience. However, as it becomes an increasingly common thing to understand our wellbeing as interlinked with environmental health, there may be room to move forward and value ecosystem services in a more nuanced way. For instance, as we think and talk about cultural values we can be exposed to the cultural and values-perspectives of others. If mutual understanding makes an impression and shapes our individual and collective decisions, we may start to become what Schumacher (1973) called, "our better selves," thus underscoring the potential value

[24] Relevant ecosystem services include: water cycling (supporting service); nutrient cycling (supporting service); water flow regulation (regulating service); erosion regulation (regulating service) disease regulation (regulating service); water purification and waste treatment (regulating service).

of ecosystem services as a boundary object for comprehending highly interwoven, normatively charged, social-ecological systems.

Given the close relationship between societal values, politics and policy making, working toward a shared set of values that are consistent with the core principles of intra- and inter-generational equity is an important step for transitioning toward more sustainable models of development. By drawing on a systems-based analysis of ecosystem services, and associated risks and benefits, small group processes could be used to induce such a values shift. While individuals are not solely motivated by self-interest, the literature on the co-evolution of institutions and individual behaviours does suggest that evolutionary mechanisms, in terms of individual fitness, underlie the convergence on norms that outwardly appear to primarily benefit the group, sometimes at the expense of individual wellbeing (Gintis, 2003). These norms are stable when the costs to the individual of group-beneficial behaviour are not excessive (Bowles et al., 2003), and when legal and cultural institutions reinforce them (Bowles, 2004). The perception of individual benefit from improved societal outcomes may lead participants in small group processes to support decision making that results in increased social-ecological system resilience.

Returning to the central paradox, as the intensity of water use continues to rise it is reasonable to suggest that the competition over ecosystem services and underlying resources that is embedded in policy and governance processes will similarly rise, perpetuating observable challenges with sustainable water resource management. Reorienting toward governance approaches that are consistent with social-ecological resilience is one over-arching policy objective. However such approaches require responsive institutions that adapt to feedback from the environment. Considering the

politically charged nature of these policy challenges, such a reorientation would need significant public support, which in turn requires knowledge building and a broadly held sustainability ethos. The instrumentality of ecosystem services as a pedagogical concept could support a broadening of how nature is understood and valued, potentially making space for a reconsideration of efficiency as the dominant economic principle. At its core this proposition is about identifying and managing the ecological determinants of socio-cultural, economic and political risk. To do this effectively, the social factors that induce these underlying ecological conditions need to be understood as well. A systems-based approach, using ecosystem services as a boundary object to examine and debate the interrelationships between the natural and social world, can be instrumental in that regard. Importantly, such an approach could highlight the dynamics of demand between different users within the social world and how these are expressed through the political economy, exerting substantial pressure on governance processes and institutional outcomes. Moreover, from a systems perspective, institutions may be ill-equipped to manage and represent marginalised interests in the face of political forces within a complex governance network that enables the orchestration and concentration of power. While systems approaches alone will not necessarily address these factors, they can be used to disaggregate complex systems, identify key actors and contested elements, and inform a reassessment of governance processes to ensure fairer outcomes. If integrated with deliberative processes such that social-ecological trade-offs and risk factors are discussed as part of a values transformation that informs decision making, these approaches can reduce information asymmetries within stakeholder communities, and between these communities and governmental actors. This could result in governance processes that are

capable of developing values-based policies that respect social and ecological constraints, including interacting resource regimes and institutional variables. Ensuring sufficient transparency between decision makers and stakeholders, and maintaining the flexibility to adapt to changing system realities would further improve resilience outcomes.

My doctoral work has developed tools and approaches that can be helpful in the pursuit of institutional outcomes that are both fairer and better aligned with social-ecological resilience. For example, the reformulated FEGS-CS (Tables 2.1 & 2.2), could be used or adapted by local and regional-level stakeholders to define relevant ecosystem elements and associated economic interests for inclusion within an application of the culturally sensitised, values-based environmental decision-making framework discussed in Chapter 3 (see Figure 6.1, below).

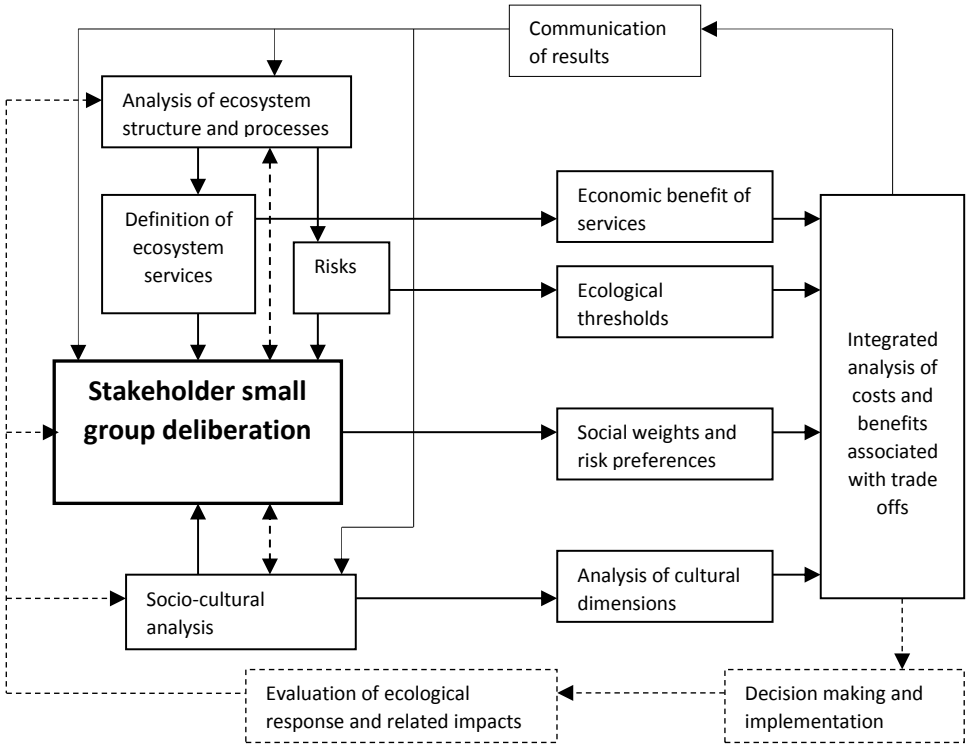

Figure 6.1 – Conceptual framework for culturally sensitised, values-based environmental decision making

Using the reformulated FEGS-CS in this capacity would only provide an approximate account of linkages between complex ecological and social subsystems. As such it should not be considered an equal substitute for engaging ecological (and cultural) expertise to interpret these critical factors. That said, the framework in Figure 6.1 was tested at the municipal level to assess a policy development process in Chelsea, Quebec. Any future application of the framework at the local level would likely encounter resource constraints (as discussed in Chapter 2), presenting a substantial challenge to the development of exhaustive ecological and socio-cultural analyses. Accordingly, this may be a reasonable compromise and second-best approach, fit to task and expected constraints.

The three case study Source Protection Regions from Chapter 4 provide some additional perspective on the framework. Structured around Source Protection Committees, source water protection policies were made endogenously within a small group process. As a result, the degree to which social weights and preferences (of the group), were balanced against ecological and other considerations is uncertain, but my research did conclude that self-interest generally prevailed. Locating final decision-making authority outside of the small group process, as indicated in Figure 6.1, could help in this regard. However the legitimacy of the process requires some level of accountability on behalf of decision makers for any divergence between policy recommendations coming from stakeholder deliberations, and decisions made by an external authority. Challenges around communication back to stakeholder communities also raised concerns about the representativeness of the source water protection policy-making process. Nevertheless, the presence and influence (if unchecked) of cultural concerns was clear, and these cases illustrate the potential for values-shifts within small group processes at a regional level.

Ensuring that appropriate checks and balances are in place, such as making sure decision-making is external to but also accountable to participants, small group processes can help balance stakeholder interests with ecological concerns, when the social, economic and cultural implications of ecological change is made explicit and integrated within a dynamic valuation and decision-making process. As appreciation of systems effects grows, it should fuel a shift in the public ethos toward ecological sustainability (if largely driven by concerns for societal well-being). Governance institutions and processes will become increasingly hardened against the influence of powerful interests if and where they are broadly understood to be at odds with (highly valued) resilient social-ecological systems.

It is too early to say definitively that these mechanisms will work in all instances. However, the Chelsea and *Clean Water Act* (2006) case studies, along with observations from the Mississippi Valley watershed, suggest that values are central to institutional outcomes, and that these values are not fixed but are rather plastic, subject to forces internal and external to governance processes. Ecosystem services have been useful for delineating systems linkages that in turn help clarify actor interests, and the potential for ecological change to result in a range of social, economic, political and cultural risks. Accepting that political discourse is complex and subject to distortion, if people understand and accept a systems-based approach to determining these risks, it could be expected to dramatically affect values positions, and by extension the institutional outcomes of water resources governance processes. Ultimately, resilient social-ecological systems are needed to ensure inclusive sustainable societies. Fostering knowledge and engaging values is essential for developing needed institutions that are responsive to ecological feedback. The model and framework developed in Chapters 2 and 3, respectively, can be instrumental in this regard.

6.4 Limitations and Future Research

In this section I will discuss some of the specific limitations of my research. In doing so I will be able to identify ways in which future research could be improved. Moreover, in some instances these limitations lead directly to future research opportunities. We will however begin with some lessons learned from the three research studies.

One of the founding premises of this work was the need to engage with ecological complexity. The implications of this assertion became evident as the research progressed, most dramatically with the environment-economy modelling in the Mississippi Valley watershed. While the final ecosystem goods and services approach was fruitful, it shifted the focus away from underlying ecological structure and functions (such as regulating and supporting services), toward the final goods and services that are the product of these ecological features. A more comprehensive ecological characterisation of risk was beyond my ability. Ecological expertise would be helpful for translating specific ecological and geographical factors into a measure of relative risk. Risk estimates would be further improved by geolocating and better specifying the nature of the various businesses and other users. Ecological (and socio-cultural) expertise would have been equally helpful for more fully testing the valuation framework discussed in Chapter 3.

While I was able to secure interviews with ten high-level stakeholders who were involved in key roles during the creation and implementation of the CWA, the uptake of the survey by SPC members was relatively modest. There was less turnover of SPC

membership than was intended by the regulations. As a result, the overall pool was smaller than originally expected (nominally members were expected to rotate out on a three-year cycle (CWA, 2015b)). In total, 15% of SPC members in the three source protection regions participated in the survey[25]. This response rate could have been improved if I had attended one or more meetings with each committee to present my project and answer questions about the survey and my objectives. Unfortunately, this was not possible due to the timing of SPC meetings and limited financial resources. Nevertheless, the importance of in-person communication is something to be mindful of in future research.

In addition to the limitations and concerns about the generalisability of results from case study research (see Section 6.2), my research was framed within a very broad conceptual framework. This was difficult to avoid given the interdisciplinary nature of the work, and the overarching proposition that social and ecological systems are linked by politically contested ecosystem services, and that better understanding of these linkages can lead to innovation in water resource governance. The three focussed and specific research studies do provide detailed insight into aspects of this larger proposition. Furthermore, the conclusions drawn from these studies have been cast within existing literature to add weight to current thinking, for instance around the appropriateness of watersheds as a unit of governance, or the need to account for a plurality of values when managing natural resources. In other cases my research has resulted in working hypotheses on how the literature might be extended, with the limitations of my work also opening up an interesting research agenda.

[25] There were 9 respondents out of 69 total possible participants, as determined by reviewing all minutes for the three SPC.

The manner in which ecosystem services can function as a boundary object is a case in point. Recalling that boundary objects are found where social worlds overlap, and that they can be used to link different disciplines together in a common purpose, I have deployed these general criteria in two ways. First, I used ecosystem services as an analytical lens to trace linkages between social and ecological systems. The second application examined the connection between political contestation over policy outcomes and the preferences of different policy actors for particular ecosystem services. In this case different social priorities, in terms of preferred ecosystem services, overlap in a shared policy space, for instance source water protection planning in Ontario or municipal development in Chelsea. These two applications of the boundary object concept can be taken up elsewhere to further test the working hypothesis that governance configurations and political economy are a determining factor for issues of institutional fit (see Section 6.2).

As a specific example, the nature of institutional interactions between indigenous peoples and national or sub-national governments in Canada, specifically as relate to resource extraction, is an interesting area for further study. Between these parties there are a plurality of legal and cultural institutions involved, as well as some overlap in terms of perceived institutional constraints. The institutional arrangements within which actors formulate their respective strategies are therefore different but connected, through the boundary object itself – the resource in question – but also through a shared acceptance of specific institutions. Nevertheless, their participation in governance processes could be complicated by institutional differences. Examining these challenges in terms of actor effects and institutional interplay would provide new perspective on the fit of resource

governance institutions relative to the varied institutional perspectives of involved actors. Institutions that exhibit reasonable fit from one perspective could be wholly inappropriate from another, potentially resulting in negative environmental (and social) outcomes. Furthermore, this approach would provide insight into the relative influence of governance configurations and political economy on institutional outcomes. Finally, traditional institutional arrangements typically draw on traditional knowledge. As we seek to improve the integration of traditional knowledge, a better understanding of related institutional approaches could lead to better policy response.

The binational Great Lakes Water Quality Agreement (GLWQA) presents an interesting opportunity to use ecosystem services as an analytical lens to examine actor-institution interactions, and the impact on institutional outcomes. The Great Lakes are a resource system that currently supports the social and economic wellbeing of over 13 million Ontarians who live in the basin (Ontario, 2014). The GLWQA was first signed in 1972 to address mounting environmental damage to the Great Lakes. It was most recently renegotiated in 2012. Canada currently seeks to meet its obligations under the GLWQA through the Canada-Ontario Agreement (COA). This agreement organises federal and provincial responsibilities under the Constitution relative to a series of identified goals. These pertain to a wide range of environmental concerns from climate change, wetlands and habitat, to pollution control. As a result of this broad mandate a large number of interests are affected by the agreement, and the governments of Canada and Ontario consult widely during COA renegotiation. A systems-based approach could be used to examine co-existing resource regimes to describe spatial and temporal variation in property rights, and associated governance mechanisms, by identifying and tracing the internal and external

sources of institutional change. This would involve examination of the COA consultation processes during renegotiation to assess how different interests engaged with the process and how their respective property rights were affected over time. Combined with a comprehensive survey documenting persistent and emerging environmental challenges, and tracking progress relative to politically negotiated goals and commitments made in successive agreements, this would provide a high-level characterisation of how factors within the political economy are correlated with observed institutional and environmental outcomes.

6.5 Closing Statement

While there are limitations to the research presented in this thesis, the conceptual framework and research findings are coherent with ongoing discourse in the broader literature on ecosystem services, water resources governance, institutionalist approaches to policy analysis, and social-ecological resilience. Persistent water resources management challenges, combined with the impacts of a changing climate, speak to the need for more effective and adaptive institutions. These will necessarily require innovative approaches to resource governance, in turn benefiting from more nuanced understanding of how interacting institutional perspectives are resolved within the political economy of resource governance. This research concludes that systems-based approaches, using ecosystem services as an analytical lens and boundary object to delineate system linkages and coordinate action, even where there may be a lack of disciplinary consensus, have the potential to develop new and important theoretical perspective on these issues.

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Appendices

Appendix A – Typology of Ecosystem Services

The typology presented in Table A.1 comes from the framework document that was developed by the World Resources Institute for the Millennium Ecosystem Assessment (WRI, 2003). While numerous efforts have been made to define and organise ecosystem goods and services, the four main categories below (provisioning services, regulating services, cultural services and supporting services), have been widely used and are arguably the current standard.

Provisioning Services	Regulating Services	Cultural Services
Products obtained from ecosystems	*Benefits obtained from regulation of ecosystem processes*	*Nonmaterial benefits obtained from ecosystems*
■ Food	■ Climate regulation	■ Spiritual and religious
■ Fresh water	■ Disease regulation	■ Recreation and ecotourism
■ Fuelwood	■ Water regulation	■ Aesthetic
■ Fiber	■ Water purification	■ Inspirational
■ Biochemicals	■ Pollination	■ Educational
■ Genetic resources		■ Sense of place
		■ Cultural heritage

Supporting Services

Services necessary for the production of all other ecosystem services

■ Soil formation ■ Nutrient cycling ■ Primary production

Table A.1 – Millennium Ecosystem Assessment typology of ecosystem services

Appendix B – FEGS Model Parameters and Data

This section provides addition data that was used by the model presented in Chapter 2. Economic data drawn from the Canadian Business Patterns Survey (Statistics Canada, 2008), and the agricultural survey (Statistics Canada, 2011), is presented in Table B.1. Farm data was distilled from 42 different farm types into three representative groups: (animal) grazers, concentrated animal feeding operations (CAFO), and (crop) farmers. The figures are provided according to census subdivision (CSD), and have been scaled according to the percentage of each CSD that lies within the Mississippi watershed (CSD_{WS}).

	Addington	North Frontenac	Central Frontenac	Greater Madawaska	Lanark Highlands
Grazers	15	3	57	21	78
CAFO	9	3	35	23	33
Farmers	12	28	99	36	128
Foresters	14	13	8	6	24
Fur/hide trappers and hunters					
Industrial Dischargers	4	5	6	5	15
Industrial processors	3	4	4	5	12
Utilities					
Residential property owners	2	7	19	9	10
Tourism	5	40	19	9	15

	Tay Valley	Drummond N. Elmsley	Mississippi Mills	Carleton Place	Beckwith
Grazers	70	112	133	0	43
CAFO	30	47	63	0	18
Farmers	148	164	234	0	67
Foresters	7	6	18	3	1
Fur/hide trappers and hunters	1				
Industrial Dischargers	7	7	40	40	3
Industrial processors	6	5	34	38	2
Utilities			1	1	
Residential property owners	19	11	49	64	6
Tourism	9	3	41	61	3

Table B.1 – Sub-beneficiaries by census subdivision

Landcover by ecosystem type is found below in Table B.2. Land Information Ontario feature codes are given for each of seven different ecosystem types used in the reformulated Final Ecosystem Goods and Services Classification System. These codes are attached to individual, georeferenced polygons and were used to generate maps in Chapter 2 (Figures 2.3 & 2.4). Where there are sub-categories of a given type of landcover (e.g. different types and mixes of deciduous and coniferous forests), multiple codes have been combined into one overarching category (e.g. forests or wetlands). The areas of landcover shown are for the watershed portion of the CSD.

Tables B.3 – B.12 bring together coded and non-coded economic data with landcover information on a CSD_{WS} basis. Using the reformulated FEGS-CS framework (Tables 2.1 and 2.2), economic data was used to determine total number of users of a given FEGS by ecosystem type. The number of beneficiaries in the top left corner of each table is the total number for the CSD as a whole. The number of unique beneficiaries was determined (i.e. a given beneficiary is only counted once, even if drawing more than one FEGS in a given ecosystem type), and then scaled using the ratio of CSD_{WS}:CSD to derive the values listed in Table 2.4. As noted, there are a number of FEGS that would be expected to benefit resource dependent businesses (Table 2.3). However not all sub-beneficiary categories were correlated with the economic data. This limited the ability to quantify the total number of users of these FEGS. Areas of landcover are also shown in brief. These values are used with the figures on total beneficiaries to determine the intensity of use by FEGS and landcover type.

Ecosystem types	Land Information Ontario Feature Code	Addington km^2	North Frontenac km^2	Central Frontenac km^2	Greater Madawaska km^2	Lanark Highlands km^2
Freshwater	20	27.5979	152.7049	40.0338	1.6343	48.4888
Wetlands	80, 81, 82, 83	5.7508	25.3809	20.1401	0.7936	52.7289
Forests	210, 211, 212, 220, 221, 222, 230, 231, 232, 233	282.3202	724.5838	246.5089	34.1903	771.6536
Agroecosystems	121, 122	1.7135	7.7050	10.9810	0.0373	59.5138
Grassland	110	0.0000	0.0696	0.3201	0.0000	1.5455
Barren/Rock & sand	30, 32, 33	2.8932	5.6738	9.4862	1.8524	3.0466
Grassland & herb	100, 110	2.4118	8.4837	9.7341	0.0000	38.6150

Ecosystem types	Land Information Ontario Feature Code	Tay Valley km^2	Drummond N. Elmsley km^2	Mississippi Mills km^2	Carleton Place km^2	Beckwith km^2
Freshwater	20	13.2125	16.5504	7.0301	0.2672	13.7846
Wetlands	80, 81, 82, 83	7.4869	6.9656	11.1782	0.3010	4.7187
Forests	210, 211, 212, 220, 221, 222, 230, 231, 232, 233	138.3621	62.5997	244.6053	1.0068	31.8686
Agroecosystems	121, 122	32.0209	47.7788	139.7569	0.9956	28.1001
Grassland	110	0.6316	0.6224	1.5995	0.0239	0.5798
Barren/Rock & sand	30, 32, 33	2.9836	0.0823	3.9873	0.0000	0.0489
Grassland & herb	100, 110	11.9500	7.6185	47.4720	0.1628	1.5736

Table B.2 – Landcover by census subdivision and ecosystem type

Grazers	15	Rivers and Streams	27.60	Percent CSD in WS	0.227598463
CAFO	9	Lakes and Ponds		Population	2532
Farmers	12	Wetlands	5.75	Population in WS	576
Foresters	14	Forests	282.32		
Fur/hide trappers and hunters	0	Agroecosystems	1.71		
Industrial Dischargers	3	Grassland/herb	2.41		
Industrial processors	3	Barren/Rock & sand	2.89		
Utilities	0	Shrubland			
Residential property owners	2				
Tourism	5				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	30	18					
flora *	15	15	N/A	N/A	15	N/A	15
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		27	14	27	27	9	24
soil		12	14	27	12		
presence of environment *	10	10	7	7	7	7	7
pollinators		12		12	12		
(de)predators		12		12	12		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	8.42	8.42	4.78	7.74	7.74	3.64	7.06
CSD(WS) TOTAL across FEGS, Intensity	0.31	1.46	0.02	4.52	3.21	1.26	

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	0.247	0.712					
flora *	0.124	0.594	N/A	N/A	1.416	N/A	
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		1.069	0.011	3.586	2.548	0.708	
soil		0.475	0.011	3.586	1.132		
presence of environment *	0.082	0.396	0.006	0.930	0.661	0.551	
pollinators		0.475		1.594	1.132		
(de)predators		0.475		1.594	1.132		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.3 – Addington Highlands, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	3	Rivers and Streams	152.70	Percent CSD in WS	0.702126612
CAFO	3	Lakes and Ponds		Population	1842
Farmers	28	Wetlands	25.38	Population in WS	1293
Foresters	13	Forests	724.58		
Fur/hide trappers and hunters	0	Agroecosystems	7.70		
Industrial Dischargers	4	Grassland/herb	8.48		
Industrial processors	4	Barren/Rock & sand	5.67		
Utilities	0	Shrubland			
Residential property owners	7				
Tourism	45				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	14	7					
flora *	3	3	N/A	N/A	3	N/A	3
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		31	13	31	31	3	6
soil		28	13	31	28		
presence of environment *	56	56	52	52	52	52	52
pollinators		28		28	28		
(de)predators		28		28	28		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	46.34	61.09	45.64	58.28	58.28	38.62	40.72
CSD(WS) TOTAL across FEGS, Intensity	0.30	2.41	0.06	7.56	6.87	6.81	

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	0.064	0.194					
flora *	0.014	0.083	N/A	N/A	0.248	N/A	
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		0.858	0.013	2.825	2.566	0.371	
soil		0.775	0.013	2.825	2.317		
presence of environment *	0.257	1.549	0.050	4.739	4.304	6.435	
pollinators		0.775		2.552	2.317		
(de)predators		0.775		2.552	2.317		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.4 – North Frontenac, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	57	Rivers and Streams	40.03	Percent CSD in WS	0.305029754
CAFO	35	Lakes and Ponds		Population	4556
Farmers	99	Wetlands	20.14	Population in WS	1390
Foresters	8	Forests	246.51		
Fur/hide trappers and hunters	0	Agroecosystems	10.98		
Industrial Dischargers	6	Grassland/herb	9.73		
Industrial processors	4	Barren/Rock & sand	9.49		
Utilities	0	Shrubland			
Residential property owners	19				
Tourism	19				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	102	63					
flora *	57	57	N/A	N/A	57	N/A	57
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		156	8	156	156	35	92
soil		99	8	156	99		
presence of environment *	44	44	38	38	38	38	38
pollinators		99		99	99		
(de)predators		99		99	99		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	42.70	61.01	14.03	59.18	59.18	22.27	39.65
CSD(WS) TOTAL across FEGS, Intensity	1.07	3.03	0.06	5.39	6.08	2.35	

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	0.777	0.954					
flora *	0.434	0.863	N/A	N/A	1.787	N/A	
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		2.363	0.010	4.333	4.891	1.125	
soil		1.499	0.010	4.333	3.104		
presence of environment *	0.335	0.666	0.047	1.056	1.191	1.222	
pollinators		1.499		2.750	3.104		
(de)predators		1.499		2.750	3.104		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.5 – Central Frontenac, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	21	Rivers and Streams	1.63	Percent CSD in WS	0.033335087
CAFO	23	Lakes and Ponds		Population	2485
Farmers	36	Wetlands	0.79	Population in WS	83
Foresters	6	Forests	34.19		
Fur/hide trappers and hunters	0	Agroecosystems	0.04		
Industrial Dischargers	5	Grassland/herb	0.00		
Industrial processors	5	Barren/Rock & sand	1.85		
Utilities	0	Shrubland			
Residential property owners	9				
Tourism	59				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	54	26					
flora *	21	21	N/A	N/A	21	N/A	21
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		57	6	57	57	23	44
soil		36	6	57	36		
presence of environment *	73	73	68	68	68	68	68
pollinators		36		36	36		
(de)predators		36		36	36		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	4.07	4.33	2.47	4.17	4.17	3.03	3.73
CSD(WS) TOTAL across FEGS, Intensity	2.49	5.46	0.07	111.72	#DIV/0!	1.64	

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	1.101	1.092					
flora *	0.428	0.882	N/A	N/A	#DIV/0!	N/A	
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		2.394	0.006	50.945	#DIV/0!	0.414	
soil		1.512	0.006	50.945	#DIV/0!		
presence of environment *	1.489	3.066	0.066	60.777	#DIV/0!	1.224	
pollinators		1.512		32.176	#DIV/0!		
(de)predators		1.512		32.176	#DIV/0!		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.6 – Greater Madawaska, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	78	Rivers and Streams	48.49	Percent CSD in WS	0.889353593
CAFO	33	Lakes and Ponds		Population	5128
Farmers	128	Wetlands	52.73	Population in WS	4561
Foresters	24	Forests	771.65		
Fur/hide trappers and hunters	0	Agroecosystems	59.51		
Industrial Dischargers	15	Grassland/herb	38.62		
Industrial processors	12	Barren/Rock & sand	3.05		
Utilities	1	Shrubland			
Residential property owners	10				
Tourism	15				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	139	93					
flora *	78	78	N/A	N/A	78	N/A	78
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		206	24	206	206	33	111
soil		128	24	206	128		
presence of environment *	40	40	25	26	26	26	25
pollinators		128		128	128		
(de)predators		128		128	128		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	144.96	218.78	43.58	205.44	205.44	51.58	120.95
CSD(WS) TOTAL across FEGS, Intensity	2.99	4.15	0.06	3.45	5.32	16.93	

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	2.549	1.569					
flora *	1.431	1.316	N/A	N/A	1.796	N/A	
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		3.475	0.028	3.078	4.744	9.633	
soil		2.159	0.028	3.078	2.948		
presence of environment *	0.734	0.675	0.029	0.389	0.599	7.590	
pollinators		2.159		1.913	2.948		
(de)predators		2.159		1.913	2.948		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.7 – Lanark Highlands, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	70	Rivers and Streams	13.21	Percent CSD in WS	0.345132168
CAFO	30	Lakes and Ponds		Population	5571
Farmers	148	Wetlands	7.49	Population in WS	1923
Foresters	7	Forests	138.36		
Fur/hide trappers and hunters	1	Agroecosystems	32.02		
Industrial Dischargers	7	Grassland/herb	11.95		
Industrial processors	6	Barren/Rock & sand	2.98		
Utilities	0	Shrubland			
Residential property owners	19				
Tourism	9				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	113	77					
flora *	70	70	N/A	N/A	70	N/A	70
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		218	7	218	218	30	100
soil		148	7	218	148		
presence of environment *	35	35	28	28	28	28	28
pollinators		148		148	148		
(de)predators		148		148	148		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	48.66	87.32	12.08	84.90	84.90	20.02	44.18
CSD(WS) TOTAL across FEGS, Intensity	3.68	11.66	0.09	2.65	7.10	6.71	

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	2.952	3.550					
flora *	1.829	3.227	N/A	N/A	2.022	N/A	
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		10.049	0.017	2.350	6.296	3.470	
soil		6.823	0.017	2.350	4.274		
presence of environment *	0.914	1.613	0.070	0.302	0.809	3.239	
pollinators		6.823		1.595	4.274		
(de)predators		6.823		1.595	4.274		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.8 – Tay Valley, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	112	Rivers and Streams	16.55	Percent CSD in WS	0.370896112
CAFO	47	Lakes and Ponds		Population	7487
Farmers	164	Wetlands	6.97	Population in WS	2777
Foresters	6	Forests	62.60		
Fur/hide trappers and hunters	0	Agroecosystems	47.78		
Industrial Dischargers	7	Grassland/herb	7.62		
Industrial processors	5	Barren/Rock & sand	0.08		
Utilities	0	Shrubland	4.88		
Residential property owners	11				
Tourism	3				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	171	119					
flora *	112	112	N/A	N/A	112	N/A	112
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		276	6	276	276	47	159
soil		164	6	276	164		
presence of environment *	21	21	14	14	14	14	14
pollinators		164		164	164		
(de)predators		164		164	164		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	68.62	110.16	7.42	107.56	107.56	22.62	64.17
CSD(WS) TOTAL across FEGS, Intensity	4.15	15.81	0.12	2.25	14.12	274.90	13.15

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	3.832	6.336					
flora *	2.510	5.964	N/A	N/A	5.453	N/A	8.5124
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		14.696	0.036	2.143	13.437	211.812	12.0845
soil		8.732	0.036	2.143	7.984		
presence of environment *	0.471	1.118	0.083	0.109	0.682	63.093	1.0640
pollinators		8.732		1.273	7.984		
(de)predators		8.732		1.273	7.984		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.9 – Drummond/North Elmsley, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	133	Rivers and Streams	7.03	Percent CSD in WS	0.890302801
CAFO	63	Lakes and Ponds		Population	12385
Farmers	234	Wetlands	11.18	Population in WS	11026
Foresters	18	Forests	244.61		
Fur/hide trappers and hunters		Agroecosystems	139.76		
Industrial Dischargers	75	Grassland/herb	47.47		
Industrial processors	68	Barren/Rock & sand	3.99		
Utilities		Shrubland	7.02		
Residential property owners	54				
Resource dependent businesses	81				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	339	208					
flora *	133	133	N/A	N/A	133	N/A	133
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		367	18	367	367	63	196
soil		234	18	367	234		
presence of environment *	210	210	135	135	135	135	135
pollinators		234		234	234		
(de)predators		234		234	234		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	422.00	513.70	136.22	446.93	446.93	176.28	294.69
CSD(WS) TOTAL across FEGS, Intensity	60.03	45.96	0.56	3.20	9.41	44.21	41.98

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	42.932	16.566					
flora *	16.843	10.593	N/A	N/A	2.494	N/A	16.868
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	
fish *	N/A	N/A					
open space		29.230	0.066	2.338	6.883	14.067	24.857
soil		18.637	0.066	2.338	4.388		
presence of environment *	26.595	16.726	0.491	0.860	2.532	30.144	17.121
pollinators		18.637		1.491	4.388		
(de)predators		18.637		1.491	4.388		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.10 – Mississippi Mills, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	0	Rivers and Streams	0.27	Percent CSD in WS	1
CAFO	0	Lakes and Ponds		Population	9809
Farmers	0	Wetlands	0.30	Population in WS	9809
Foresters	3	Forests	1.01		
Fur/hide trappers and hunters	0	Agroecosystems	1.00		
Industrial Dischargers	150	Grassland/herb	0.16		
Industrial processors	145	Barren/Rock & sand	0.00		
Utilities	1	Shrubland	0.64		
Residential property owners	61				
Tourism	58				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	296	150					
flora *	0	0	N/A	N/A	0	N/A	0
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		0	3	0	0	0	0
soil		0	3	0	0		
presence of environment *	269	269	119	120	120	120	119
pollinators		0		0	0		
(de)predators		0		0	0		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	414.00	269.00	122.00	119.00	119.00	119.00	119.00
CSD(WS) TOTAL across FEGS, Intensity	1549.32	893.56	121.17	119.52	731.03		185.94

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	1107.726	498.269					
flora *	0.000	0.000	N/A	N/A	0.000	N/A	0
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		0.000	2.980	0.000	0.000		0
soil		0.000	2.980	0.000	0.000		
presence of environment *	1006.684	893.563	118.195	120.526	737.169		185.938
pollinators		0.000		0.000	0.000		
(de)predators		0.000		0.000	0.000		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.11 – Carleton Place, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Grazers	43	Rivers and Streams	13.78	Percent CSD in WS	0.389758773
CAFO	18	Lakes and Ponds		Population	6986
Farmers	67	Wetlands	4.72	Population in WS	2723
Foresters	1	Forests	31.87		
Fur/hide trappers and hunters	0	Agroecosystems	28.10		
Industrial Dischargers	2	Grassland/herb	1.57		
Industrial processors	2	Barren/Rock & sand	0.05		
Utilities	0	Shrubland	11.15		
Residential property owners	6				
Tourism	3				

	Freshwater	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	65	45					
flora *	43	43	N/A	N/A	43	N/A	43
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		110	1	110	110	18	61
soil		67	1	110	67		
presence of environment *	11	11	9	9	9	9	9
pollinators		67		67	67		
(de)predators		67		67	67		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

CSD(WS) TOTAL across FEGS, Absolute	28.84	47.16	3.90	46.38	46.38	10.52	27.28
CSD(WS) TOTAL across FEGS, Intensity	2.09	9.99	0.12	1.65	29.47	215.09	2.45

	Rivers & Streams	Wetlands	Forests	Agroecosystems	Grassland/herb	Barren/ Rock & Sand	Shrubland
water *	1.838	3.717					
flora *	1.216	3.552	N/A	N/A	10.650	N/A	1.503
fauna *	N/A	N/A	N/A	N/A	N/A	N/A	N/A
fish *	N/A	N/A					
open space		9.086	0.012	1.526	27.245	143.393	2.132
soil		5.534	0.012	1.526	16.595		
presence of environment *	0.311	0.909	0.110	0.125	2.229	71.697	0.315
pollinators		5.534		0.929	16.595		
(de)predators		5.534		0.929	16.595		

* This FEGS does benefit resource dependent businesses, but the extent of this benefit is indeterminant due a lack of data on sub-beneficiary populations

Table B.12 – Beckwith, sub-beneficiaries by FEGS and ecosystem type (total and intensity)

Table B.13 presents the systems dimensions for a number of risk scenarios in the Mississippi Mills and Lanark Highlands CSD. These are organised by FEGS with information given on the impacted beneficiary, competing beneficiary (or beneficiaries) and the implicated ecosystem type(s). Additional information is given on climate change dimensions of the risk scenario. For instance, how the expected impacts of climate change could exacerbate or give rise to the ecologically-derived economic risk.

Final Ecosystem Good or Service	Beneficiary at risk	Ecosystem Types Implicated	Competing Beneficiary	Climate Change Effects
Mississippi Mills				
Water (quality)	Industrial Processors	Freshwater	Industrial dischargers	Less dilution during low flows
Presence of environment	Recreational users	Freshwater	Industrial dischargers	Less dilution during low flows
Water (quality)	Industrial Processors	Freshwater, agroecosystems	CAFO, grazers, farmers (fertiliser, pesticides, pathogens)	More runoff during peak events, less dilution
Presence of environment	Recreational users	Freshwater, agroecosystems	CAFO, grazers, farmers (fertiliser, pesticides, pathogens)	More runoff during peak events, less dilution
Pollinators, (de)predators	Farmers	Wetlands, agroecosystems, grassland/herb	Grazers (landuse affecting habitat)	Species effects such as migration and competition
Lanark Highlands				
Presence of environment	Recreational users	Wetlands, agroecosystems, grassland/herb	CAFO, grazers, farmers	More runoff during peak events
Pollinators, (de)predators	Farmers	Wetlands, agroecosystems, grassland/herb	CAFO, grazers, farmers (capturing water onsite)	Lower soil moisture affecting habitat
Pollinators, (de)predators	Farmers	Wetlands, agroecosystems, grassland/herb	CAFO, grazers, farmers (fertiliser, pesticides, pathogens)	More runoff during peak events

Table B.13 –Summary of systems-based risk scenarios in Mississippi Mills and Lanark Highlands

Appendix C – Source Protection Region Farm Data and Policy Comparison

Data on total number of farms and land coverage is provided in Table C.1 for the three case study source protection regions (SPR). Farm type information was aggregated into two main categories: livestock and crop farming (OMAFRA, 2016a, 2016b). It is also worth noting that watershed-level data was not available from the Ontario Ministry of Agriculture and Rural Affairs. Accordingly the information below is county-specific. The counties listed for each SPR are those that encompass the majority of the area of the watershed with a minimum of spillover into adjacent watersheds (e.g. other SPR).

	Livestock		Crops		TOTAL	
	Farms	Hectares	Farms	Hectares	Farms	Hectares
Ausable Bayfield Maitland Valley SPR						
Huron County	1,141	14,530	1,326	245,189		
TOTAL	1,141	14,530	1,326	245,189	2,467	259,719
Saugeen Grey Sauble Northern Bruce Peninsula SPR						
Bruce County	1,175	46,412	836	154,653		
Grey County	1,332	41,658	916	113,514		
TOTAL	2,507	88,070	1,752	268,167	4,259	356,237
Raisin-South Nation SPR						
United Counties of Stormont, Dundas and Glengarry	732	15,627	845	144,780		
United Counties of Prescott and Russel	519	7,909	468	98,818		
TOTAL	1,251	23,536	1,313	243,598	2,564	267,134

Table C.1 - Number of farms and land under production by farm type

Further information on the threat specific policies for the three case study SPR is available in Table C.2 below. This information is taken from their respective source protection plans (ABMV, 2015a; ABMV, 2015b; RSN, 2014; SGSNBP, 2016), and these policies are applicable where the activity is considered a significant threat to drinking water. Circumstances where activity could be a significant threat depends on Wellhead Protection Area (WHPA) or Intake Protection Zone (IPZ), grade and other factors (see the Raisin South Nation Source Protection Plan, Appendix E). Activity that is beyond the defined threshold and associated with a WHPA/IPZ vulnerability score of 10 will typically amount to a significant threat.

Threat	Policy Type	Ausable-Bayfield Maitland Valley	Saugeen, Grey Sauble, Northern Bruce Peninsula	Raisin-South Nation
Road salt: Application	RMP	Existing and future activity where 80% impervious surface	Where existing and future activity are a significant threat to drinking water	
Road salt: Handling and storage	Prohibition		Existing and future activity over 500 tonnes (exposed)	Future storage where it would be a significant drinking water threat
	RMP	Existing and future activity over 5000 tonnes		Existing storage where it would be a threat to drinking water
Outdoor Confinement	Prohibition	Existing and future in WHPA-A Future activity in WHPA-B	Future activity including expansion in WHPA-A	
	RMP	Existing activity in WHPA-B	All vulnerable areas other than future activity in WHPA-A	Activities not subject to requirements of the *Nutrient Management Act* (2002) that are or could be a significant threat to drinking water
Grazing and Pasturing	Prohibition	Existing and future activity in WHPA-A above 1 nutrient unit per acre		

	RMP	Existing and future activity in WHPA-B Existing and future activity in WHPA-A below 1 nutrient unit per acre	In all vulnerable areas where existing and future activity are a significant threat to drinking water	Activities not subject to requirements of the *Nutrient Management Act* (2002) that are or could be a significant threat to drinking water
Commercial fertilizer: Application	Prohibition		Future activity in WHPA-A	No application in WHPA-A
	RMP	Existing or future uses where livestock density is greater than 1 nutrient unit per acre, or greater than 80% managed land	Existing and future use in all vulnerable areas, except future use in WHPA-A	Activities not subject to requirements of the *Nutrient Management Act* (2002) that are or could be a significant threat to drinking water
Commercial fertilizer: Storage	Prohibition	Future activity > 2500 tonnes		
	RMP	Existing activity > 2500 tonnes	All vulnerable areas, existing and future use, plus facilities specifications for liquid and solid fertilizers	Activities not subject to requirements of the *Nutrient Management Act* (2002) that are or could be a significant threat to drinking water
Pesticide: Application	RMP	Existing or future activity > 1 hectare	Existing and future application not regulated under the *Pesticides Act*	Existing and future application not regulated under the *Pesticides Act* (1990) that are or could be a threat to drinking water
Pesticide: Storage	Prohibition	Future activity where storage for retail sale > 250kg, or storage for manufacturing or wholesale > 2500kg	Future activity in WHPA-A	Future activity at commercial outlets where this could be a significant threat to drinking water
	RMP	Existing activity where storage for retail sale > 250kg, or storage for manufacturing or wholesale > 2500kg	All activity except future activity in WHPA-A	Existing and future application not regulated under the *Pesticides Act* (1990) that are or could be a threat to drinking water

Fuel handling and storage	Prohibition	Future storage of 250 litres at least partially below grade or 2500 litres above grade	Future activity > 2500 litres, or bulk plant manufacturing or refining fuel > 250 litres; Future activity of > 3000 litres in Kinkardine, or > 2000 litres in Meaford	Future storage of liquid fuels at facilities other than private outlets and farms
	RMP	Existing storage of 250 litres at least partially below grade, or 2500 litres above grade	Expansion of existing facility >2500 litres; existing activity > 2500 litres, or bulk plant >250 litres but <2500 litres; existing and future facilities between 250 and 2500 litres at least partly below grade; Existing activity of > 5000 litres in Kinkardine, numerous other specific provisions for other municipalities hosting marina facilities (see below)	Existing and future storage of fuel oil, liquid fuels, existing storage at facilities other than private outlets and farms

Table C.2 - Comparison of selected threat policies from three source protection plans

The following additional information regarding fuel handling and storage policies in Saugeen Grey Sauble and Northern Bruce Peninsula specifies thresholds that are specific to individual drinking water systems. These policies applies where the storage of fuel is a significant drinking water threat (existing and future activity).

1. for Events-based Area for the Kincardine Drinking Water System (as shown on Map 5.1.K.K.1) where fuel is stored in a quantity of 5,000 L or more (EBA-5000), and 10,000 L or more (EBA-10000);

2. for Events-based Area for the Meaford Drinking Water System (as shown on Map 5.2.M.M.1) where fuel is stored in a quantity of 5,000 L or more (EBA-5000), and 12,000 L or more (EBA-12000);

3. for Events-based Area for the Wiarton Drinking Water System (as shown on Map 5.2.SBP.W.1) where fuel is stored in a quantity of 5,000 L or more (EBA-5000), and 8,000 L or more (EBA-8000);

4. for Events-based Area for the Lion's Head Drinking Water System (as shown on Map 5.3.NBP.LH.1) where fuel is stored in a quantity of 5,000 L or more (EBA-5000), 7,500 L or more (EBA-7500), and 22,500 L or more (EBA-22500);

5. for Events-based Area for the Owen Sound Drinking Water System (as shown on Map 5.2.OS.RN.1) where fuel is stored in a quantity of 15,000 L or more (EBA-15000), 25,000 L or more (EBA-25000), and 50,000 L or more (EBA-50000);

6. for Events-based Area for the Southampton Drinking Water System (as shown on Map 5.1.SS.S.1) where fuel is stored in a quantity of 13,000 L or more (EBA-13000), and 22,500 L or more (EBA-22500);

7. for Events-based Area for the Thornbury Drinking Water System (as shown on Map 5.2.BM.T.1) where fuel is stored in a quantity of 50,000 L or more (EBA-50000), and 100,000 L or more (EBA-100000)

Appendix D – Interview and Survey Protocols for Fieldwork

The following two subsections provide the interview and survey protocols used in thesis fieldwork for Chapters 2 & 4. Subsection D.1 provides the protocol used to interview natural resource managers and other stakeholders in the Mississippi Valley watershed in order to better understand and contextualise environment-economy modelling. Subsection D.2 provides the interview protocol used for engaging subsystem experts who were involved in the formulation of the *Clean Water Act* (CWA) (2006), as well as key stakeholders at the SPR-level who were involved with the implementation and development of source protection plans. The survey protocol circulated to SPC members is also provided.

D.1 Assessing Environment-Economy Linkages in the Mississippi River Watershed

<u>Interview Protocol, Part A: For representatives of stakeholder organisations involved in water governance in the Mississippi Valley watershed</u>

Preamble: Thank you for providing me with your time. The questions below are designed to help us understand how economic risk stemming from environmental factors is understood, specifically in the context of climate change, and how these risks are managed in the Mississippi River watershed.

Background [*Purpose of this section is understand how long the respondent has been in their position and the extent to which they work on the issues being discussed.*]

1. How long have you been with [*name of organisation/group*] and how long have you been involved with climate change planning here or elsewhere?

2. Do you/have you engaged with other governmental or non-governmental agencies/organisations on climate change adaptation planning and policy?

Economic Risk [*These questions are meant to understand how the origins of economic risk are understood by different groups/agencies within the watershed*] To begin with I would like to ask you a few questions about how economic activities are seen to depend on and affect the natural environment – or not – by different groups, agencies or individuals active within the watershed.

1. In terms of linkages between economic sectors and environment, do people think of the environment as a source of economic risk? Can you give any examples?

2. Are there some examples of how one activity may alter the environment and thereby affect the opportunity or conditions for another activity to take place?

3. Are there any locations within the watershed where there exists a greater potential for economic risk as a result of ecological change due to, (a) other economic activity, or, (b) the projected impacts of climate change?

Climate Change [*These questions are meant to understand the degree to which climate change adaptation is being taken up within the watershed.*]

1. Which stakeholders are actively implementing adaptation measures?

2. Are there any instances where stakeholders are not implementing adaptation measures in spite of observed changes and expected future impacts to themselves or others?

3. Do you have a sense of what motivates different stakeholders to undertake adaptation measures, or not?

Policy Framework [*These questions are intended to verify or counter the results of my modelling work and analysis.*]

1. What are the most important policy instruments in place for managing the impacts of climate change on water quality and quantity, and the ecological function of the watershed broadly speaking?

2. What are most important policy instruments for managing economic threats posed by climate change?

3. Where are there gaps in the policy framework in terms of managing the impacts of climate change?

Present a summary of my modeling analysis for discussion.

D.2 Investigating Actor-Institutions Interactions and the *Clean Water Act* (2006)

Interview Protocol: For Conservation Authority personnel, SPC Chairs, government officials and members of the policy subsystem

Preamble: Thank you for your time. The questions below are designed to help us understand the way the socio-political context within which the CWA was developed, which players were involved (or conspicuous by their absence), and the nature of their actions. Not all the questions will be applicable to your expertise. We will therefore only focus on those questions you are comfortable answering given your area of work.

Background [*Purpose of this section is to ask a few background questions to understand what position the interviewee occupied at the time and how that positioned them to interact with the development of the Act, whether they have prior positions that also inform their answers, and whether their current work might inform or affect their answers.*]

1. Could you please provide a little background on your own career? How long were you with [*name of organisation/department*] prior to the initiation of development of the Clean Water Act (CWA)? Where are you working now?

Economy [*This section will seek to understand how different stakeholders engaged the creation of the Act*] There is a potential contradiction between policies that seek environmental outcomes but are socially derived, in large part relative to economic criteria. However it is also essential that local economies not be unnecessarily disrupted by policy change.

1. Were any economic sectors of particular concern when developing the terms of the CWA, and if so why?
2. Do the source protection committees (SPC) give equal voice to all parties, and if not why would one group be privileged or excluded?

Politics [*These questions are meant to understand the political climate at the time the Act was developed.*] There are a number of analyses that paint a politicized picture of various factors that lead up the events in Walkerton, in particular around concerns about the capacity of the Ministry of the Environment's to effectively manage these matters.

1. How did political scene at the time affect the scope, ambition and direction of the CWA?

The Act [*These questions get to the heart of what motivated the act, which reflect on how the goal of source water protection is weighed against other priorities.*] Acknowledging that a broad societal concern over the need to protect drinking water gave rise to the CWA, the shape such provisions would eventually take are contingent on the terms of the Act itself.

1. Which stakeholder groups were most actively engaged with formulation of the Act and what were their main objectives?
2. Where did the idea of devolving governance to SPC come from and how did various stakeholders react?
3. What were the main concerns of policy-makers in terms of trade-offs with other policy objectives and associated risks related to devolution to local level?
4. Has the CWA been successful in terms of effectively protecting source water or has it largely been a paper exercise? What factors have contributed to these outcomes?

Source protection committees [*These questions focus in on the SPC and how well they functioned.*] Although they received support and guidance from the Province and from the lead Conservation Authority, the source protection committees ultimately held the pen when it came to creating the Source Protection Plans.

1. Which stakeholders have done comparatively well and which have ended up at the margins?
2. Why have the plans taken longer to develop than originally hoped?
3. How has power been distributed at the local level through this instrument?
4. What are the different social-values perspectives/priorities that come to bear on the work of the SPC through membership, media, guidance, etc?

Survey Protocol: For SPC members

Background [*These questions seek to identify which stakeholder group the respondent represented and at what point they participated on the SPC*]

1. Which stakeholder group did you represent on [*name of SPC*]?
2. When did you start and how long were in that role?
3. What is your occupation now and is this the same as before your time on the SPC?

Source protection committee function [*These questions are meant to gather perspective on how the SPC actually function, to provide some perspective on design versus function as well as the actions of different stakeholder groups.*]

1. Did any single group seem to have more influence over the development of the plan than others? Why?
2. Were there any mechanisms in place to prevent undue influence?
3. Why did it take nine years following enactment of the CWA to approve SPP?
4. Did any actors seek to hasten or delay the development of the SPP, and if so why?

Values, symbols and culture [*These questions will get at issues of values and fairness, seeking to tease out socio-cultural factors that could help disaggregate a range of goals being pursued through the SPC.*]

1. Was there any discussion of source water protection in local media and if so what was being said?
2. In addition to "Walkerton" what other symbolic references were used in deliberations, and in what context (for instance "independence" or "family")?
3. In addition to source water protection what other individual and societal priorities were discussed by the SPC members?
4. Did the process produce a fair outcome?