Valuing Ecological Services and Community Design - Implications for the Private Market and Local Government

by

Daniel Alexander Hegg
B.Comm., University of Victoria, 2006

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of the Requirements for the Degree of

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in the Faculty of Geography

Supervisory Committee

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Departmental Member

Dr. Monika I. Winn, (Faculty of Business)
Outside Member

Wm. Patrick Lucey, (Aqua-Tex Scientific Consulting Ltd.)
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Abstract

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Presently, conventional development does not adequately incorporate functional ecosystems into development design. Largely due to the intangible nature of most ecosystem services, functional ecosystems have not been directly identified as valuable and are, therefore, often ignored in economic decision frameworks. This has resulted in the degradation and loss of functional ecosystems and ecosystem services as the value and the associated costs of lost ecosystem services are not accounted for. The valuation of ecosystem services is a means by which ecological costs and values can be adequately represented in urban planning and decision-making processes. However, using current valuation methods, ecosystems are continuously being valued for their aggregated ecosystem service values and not for the value of their ability to resist/recover from disturbances and continue proving goods and services over time.

The Swan Lake watershed case study was utilized to show that the estimated ecosystem service values are not risk adjusted to reflect the functional condition of an ecosystem. Specifically, based upon the current valuation estimates alone and without reference to the functional condition, the estimated ecosystem service values for the Swan Lake study suggest that the watershed is in a good (proper) functional condition, when in-fact, the overall health of the watershed is in a poor condition of health and its resilience to disturbance is low. Furthermore, the estimated values do not reflect the loss of ecosystem services due to past urbanization and agricultural activities. Because the estimated values do not provide the critical information decision makers require, the valuation of the functional condition of ecosystems is recommended. Due to the complexity involved in valuing the functional condition of an ecosystem, the integration of ecosystem valuation methods and ecosystem evaluation assessments is proposed and explored.

In the context of post-urban planning and development, the proposed approach has immediate application as it would provide effective financial arguments for the preservation and restoration of ecosystems as well as facilitate more informed decisions in managing existing urban ecosystems for their function rather than ecosystem services. In a pre-development application, there exists a opportunity wherein an ecosystem's functional condition could be valued as part of an integrated development design and planning process (IDP).

The British Pacific Properties (BPP) Rodgers Creek development is used as a case study to describe how the proposed approach could be incorporated into the integrated design and planning (IDP) process. By clarifying the ecological tradeoffs between various land-use/development scenarios using a sieve analysis, the proposed approach could help a design team render more informed judgments regarding the functional condition of ecosystems and the value of the ecosystem services. The proposed approach also contributes to a much needed business case, which demonstrates that when urban developments are planned using an IDP process, where the landscape informs the design, there can be greater financial reward to the developer, community and municipality.

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Acknowledgments

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Chapter 1.

1.1 Introduction

"Ecosystems are the planet's life-support system" (WHO, 2005). Fundamental to human health and well-being, healthy (herein referred to as "functional") ecosystems provide valuable ecosystem services that contribute to social and economic well-being (MEA, 2003). However, due to the intangible nature of most ecosystem services, ecosystems have not been directly identified as valuable and are therefore, often ignored in national and corporate accounting frameworks, project appraisals, areas of economic policy and land-use decision frameworks (Hindmarch, Harris and Morris, 2006). Since they appear to have little or no economic value, many of the free direct and indirect life-supporting services provided by ecosystems are reduced or lost as functional ecosystems are impaired, degraded and destroyed as a result of economic activities; oftentimes, negatively affecting human well-being (MEA, 2003). Although the linkage between the continual loss, impairment, and degradation of functional ecosystems and human well-being has been extensively documented by the Millennium Ecosystem Assessment (2003), the value of ecosystem services remains overlooked in-lieu of short-term financial gain as the value of most ecosystem services are not accounted for in economic markets or internalized within economic decision frameworks (Balmford *et al.,* 2002; MEA, 2003).

Economics and the environment are not independent from one another; rather, they are dynamically interdependent (Gustavson, 1999; Limburg, O'Neill, Costanza and Farber, 2002; Straton, 2006). Ecological processes are impacted by economic activity (*e.g.,* extraction, pollution); similarly, ecological processes provide for and can constrain or limit economic activities. Although the linkage between the environment and economics appears clear and well defined, it is often difficult to measure. The few ecosystem services that are reflected in economic markets, and are internalized in economic decision frameworks, are those that directly contribute to economic welfare. However, the remaining ecosystem services - *i.e.,* the intangible ecosystem services (*e.g.,* cultural services, pollution abatement, clean air) - are not accounted for (King, 1997). Turner *et* *al.* (2003) state that the inability to value and internalize indirect ecosystem services is due to the complexity of nature itself and “that reliable estimates of all the services cannot be made” (p.7). The reason, according King (1997), is that the non-market valuation techniques employed are too expensive and there are too many ecosystem services to be valued. Therefore, only a sub-set of values is “captured” and “serve more to illustrate ecosystem values than to provide a comprehensive accounting of them” (King, 1997, p.7). Bingham *et al*. (1995), amongst others, suggest that there is limited information on predicting how socio-economic systems will affect ecosystem processes, functions, services and the overall functional condition of an ecosystem, thereby making it difficult, or impossible, to identify how ecosystem changes will impact socio-economic systems. As a consequence, economic markets or decision frameworks cannot properly or comprehensively determine how decisions may influence functional changes within ecosystems. In any case, as most decisions are based upon economic criteria, functional ecosystems are often not valued and are assigned an implicit value of zero in economic decision frameworks (Costanza *et al.,* 2007). Failing to recognize the value of functional ecosystems in economic decision frameworks, a market failure, often results in the loss, impairment, and degradation of functional ecosystems (MEA, 2003).

Market failures arise when there is no market for a good or service (*e.g.*, pollination); when goods or services display characteristics of public goods (*e.g.*, clean air); or when externalities are present (*e.g.*, increased crop production at the cost of polluted streams) (Fausold and Lilieholm, 1999; NRC, 2004; Great Britain H.M. Treasury, 2004; Schaeffer, 2008). In recognition that market failures are occurring, the ecosystem valuation discipline has endeavoured to improve existing ecosystem valuation methods or develop new ones in order to better estimate the total value of an ecosystem. The purpose of the valuation exercise is to provide information to help inform decisions on trade-offs (*i.e.*, rank priorities - such as the acquisition of land or environmental improvements), to inform decision makers on the value, to provide estimates on damages that have occurred or to measure the value of the assets to incorporate them into national income accounts (Costanza *et al.,* 1997; NRC, 2004; Hindmarch *at al.,* 2006). However, ecosystem valuation is a contentious topic as many people are opposed to the practice on ethical or moral grounds, asserting that ecosystems are valuable for their own sake and not merely for their usefulness to people (Ludwig, 2000; Dore and Burton, 2003). Valuation methods also suffer from problems of meaning and measurement as the definitions and concepts of ecosystems, ecosystem processes, ecosystem functions, and ecosystem services are rife with ambiguity and disagreement within the literature (O'Neill, 2001; de Groot and Hein, 2007; Fisher, Turner and Morling, 2009). The issue of semantics is often cited as a barrier to the improvement of ecosystem valuation methods and estimates (Limburg *et al.,* 2002).

Of particular importance and relevance to this thesis is the predominant theme of single service ecosystem valuation studies that occur in the literature, many of which are reductionistic (Kumar and Kumar, 2008). That is, most ecosystem valuation studies only focus on valuing the ecosystem services and do not take into account the factors that are necessary to maintain (or that influence) the functional condition of an ecosystem (*e.g.,* ecosystem processes, structures, regulation functions, and boundary conditions) (Figure 1). King (1997) explicitly comments on this issue stating, "ecosystem valuation methods attempt to assign values to ecosystem services, usually in absolute (dollar) terms, but usually without much regard for the specific ecosystem features or functions that generated them" (p.6). Therefore, the estimated ecosystem service values are not adjusted for risk or uncertainty (*i.e.,* the ecosystem is damaged, is near a threshold or at-risk of degradation). As a consequence, the estimated values do not reflect the critical information decision makers require - *e.g.,* the current functional condition, the direction or changes within ecosystems, the quality of ecosystem services and the urgency with which each may be occurring or changing as a result of man-made or natural perturbations (Toman, 1998; Straton, 2006). For instance, a riparian ecosystem in a functional condition will provide various regulating functions that result in valuable ecosystem services (Figure 2). Assuming that the fisheries value has been calculated without any consideration of the functional condition of the riparian ecosystem, the decision maker using the information may not be aware of upstream development activities (*e.g.,* the creation of a dam that would result in reduced flow of water), which would likely influence the functional condition of the riparian ecosystem, thereby effecting the regulating functions and the resulting ecosystem goods/services produced. Thus, the valuation has been done in isolation and is ineffective from a resource management or decision makers perspective as the decision maker is not provided information on how to manage the ecosystem.

This problem has not been overlooked as many ecosystem valuation researchers have made reference to the importance of the functional condition of an ecosystem (oftentimes referred to as ecosystem health, integrity, resilience and biodiversity) in their studies suggesting that future research efforts need be focused on these areas (*e.g.,* Bingham *et al.,* 1995; King, 1997; EPA, 2000; Folke *et al.,* 2002; Limburg *et al.,* 2002; MEA, 2003; de Groot and Hein, 2007; Mäler, Destouni and Li, 2007). However, the task of determining how to incorporate the necessary ecological factors into the valuation exercise continues to be elusive as many researchers simply provide lists of areas within both economics and ecology that need to be addressed, whereas others continue to refine current ecosystem valuation methods, neither of which is advancing ecosystem valuation knowledge forward (Norton and Noonan, 2007). Therefore, researchers must begin developing frameworks that adjust the estimated ecosystem value for the ecological factors necessary to maintain (or that influence) the functional condition of an ecosystem. Not accounting for the functional condition in valuation studies is counterintuitive since the continued production of ecosystem services is dependent upon continued ecosystem function. The intent of this thesis was to examine the link between ecosystem function and the economic values that are assigned to ecosystem services.

By exploring ecosystem valuation methods and the measurement of ecosystem function, this thesis contributes to the perceived deficiency in the ecosystem valuation discipline as noted above. Framed within the urban resource-planning context, this thesis argues that the values assigned to ecosystems or ecosystem services need to reflect the functional condition of an ecosystem. Addressing this issue is of particular importance as there is increasing recognition that the multi-functional use of an ecosystem (*e.g.,* stormwater control, flood abatement, carbon storage and sequestration, aesthetics) is economically more beneficial to cities and communities than if these systems are degraded (de Groot, 2006; Barraclough and Hegg, 2008a). Therefore, if resource managers and decision makers are going to undertake ecosystem valuation as a means to create a common language of which environmental and economic decisions can be compared (*e.g.,* development or preservation of lands), the functional condition of the ecosystem must be measured and valued. Valuing the functional condition of the ecosystem would provide resource managers and decision makers the critical information required to understand the ecological factors that influence the functional condition of an ecosystem (*e.g.,* the impacts of urbanization), as well as enable them to monitor and manage the functional condition of an ecosystem in such a way that they can maximize the value of the ecosystem services received by the surrounding community.

1.2 Ecosystems Functions and Services

Defined by the MEA (2003), ecosystems are a "dynamic complex of plant, animal, and microorganism communities and the nonliving environment interacting as a functional unit" (p.3). Existing at varying spatial (*e.g.,* a temporary pond, a forest, an ocean) and temporal scales, it is the linkages between the biotic and abiotic components that create these dynamic systems (MEA, 2003). Various types of ecosystems exist, such as deserts, prairies, forests, rivers, lakes, *etc*.

The ecosystem concept, according to O'Neill (2001), is a paradigm because it is a convenient way of labeling a complex system in order to understand the complexities involved.[1] It is a "relatively new research and management approach" (MEA, 2003, p.50). The term, ecosystem, was first coined in 1935, by Arthur Tansley who used the term to describe systems of interactive and integrated living and non-living things (Daily, 1997). However, many researchers point out that the underlying concept traces back to George Marsh (1864) who argued that natural resources were finite and noted that human

[1] As such, ecologists and those outside of the discipline have heavily criticized the concept; "at one extreme, ecosystem is a convenient term, relatively free of any assumptions, that indicates the interacting organisms and abiotic factors in an area. At the other extreme, the term, ecosystem is defined as a precisely defined object of a predictive model or theory" (O'Neill, 2001, pp. 3277).

actions tend to disturb natural systems resulting in the loss of what are now termed ecosystem services (Daily, 1997; O'Neill, 2001).

Over the last few decades, the ecosystem concept has evolved from describing ecosystems that are resilient, and when disturbed, systems that would return to an equilibrium or steady state to now describing ecosystems as having multiple equilibria (or an absence thereof) that are affected by both stabilizing and destabilizing forces (Holling and Meffe, 1996). In fact, "ecosystem equilibrium conditions are so rare, and that disturbance events are so common that most ecological systems never reach a dynamically stable climax stage" (Jelinski, 2005, p.281). Ecosystems are considered to be dynamic, heterogeneous, non-linear, open and scaled (Holling and Meffe, 1996; O'Neill, 2001; Jelinski, 2005). Ecosystems are complex and constantly changing due to natural succession or human-made perturbations.

Ecosystems are important to society in that they provide ecosystem services that are fundamental to human health and well-being. Defined, ecosystems services are the goods or services that society actively or passively derives from an ecosystem that contribute to human well-being (Costanza *et al.,* 1997 de Groot and Hein, 2007). The production of ecosystem services are based upon natural cycles, operating at various scales, that are "fuelled" by solar energy (Daily, 1997). These cycles are based upon the complex interactions between the boundary conditions, ecosystem processes and structures (MEA, 2003) (Figure 1).[2] For instance, the provision of clean water resulting from a healthy forested catchment will depend on: the ecosystem structures, such as the forest canopy, rooting patterns, soil types, *etc.*; the ecosystem processes (*e.g.,* photosynthesis, nutrient cycles, erosion, *etc.*); and, the landscape context, such as the type of ecosystem (*e.g.,* wetland *versus* stream), geology, climate, *etc.* (Vira and Adams, 2009).

[2] The boundary conditions are simply the landscape context - *i.e.,* the size, proximity to certain features of natural and human landscapes, slope, substrate geology, hydrology, precipitation, climate regulation - of which an ecosystem is situated (King, 1993). The boundary conditions influence ecosystem structure and processes (and *vice versa*). Ecosystem structures consists of the hydrological and geomorphic conditions as well as soils and fauna (Clouston, 2002). Ecosystem processes are the physical, biological or chemical changes or reactions that naturally occur within an ecosystem (Clouston, 2002). For example, photosynthesis or biomass production would both be processes that occur within an ecosystem (King, 1993).

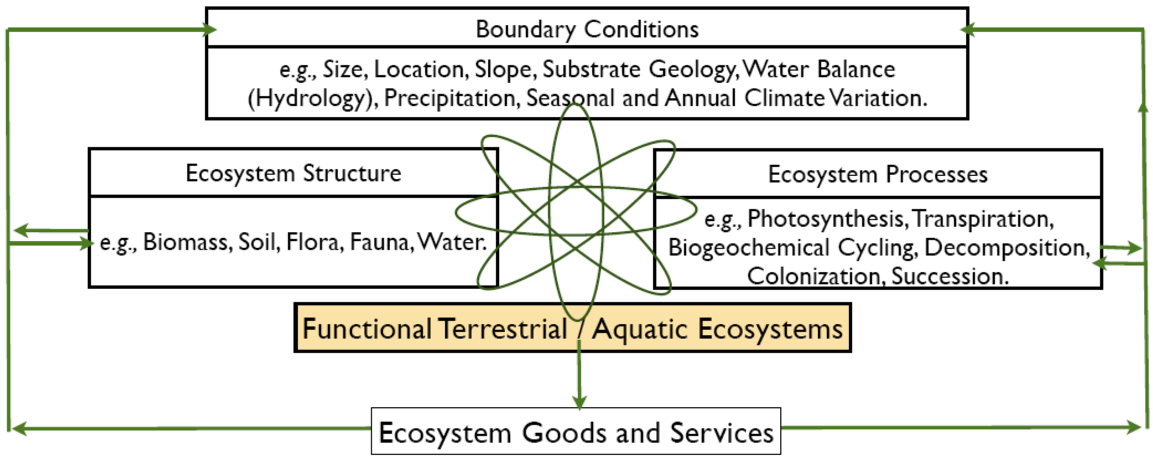

Figure 1. Interactions between the boundary conditions, ecosystem processes and structures that result in ecosystem services. *Note:* Based on Turner *et al.* (2003); NRC (2004); Farber *et al.* (2006); DEFRA (2007); de Groot and Hein (2007); and Fisher *et al.* (2008).

Ecosystem services arise from the regulating functions. Often referred to as supporting services or intermediate services, the regulating functions are necessary to the production of all ecosystem services (MEA, 2003). However, regulating functions differ from ecosystem services in that their influence or impact on society is indirect and tend to occur over a long period of time (MEA, 2003). For instance, society does not use the soil formation function directly; rather, society indirectly depends on the regulation function over time as it influences the food production ecosystem service in the short-term. Examples of other regulating functions can be seen in Figure 2. The distinction between ecosystem functions and ecosystem services is necessary as the terms are often used to refer to the same thing depending on the ecosystem typology classification applied which may ultimately result in the double counting of ecosystem functions and services (Fisher *et al.,* 2009).[3] Although, the direct and indirect contribution to human well-being is a common theme to defining what ecosystem services are, there is no clear consensus on what the final definitions between ecological functions and ecological services should be

[3] This is dependent upon how the ecosystem services are categorized. Three common ecosystem valuation typologies include: functional groupings, organizational groupings, and descriptive groupings (MEA, 2003). The challenge with defining a typology of ecosystem services is that few frameworks have consistently linked the ecological characteristics of ecosystems to their potential values (Kline, 2006; de Groot and Hein, 2007).

(Fisher *et al.,* 2009). Thus, for the purposes of this thesis, in order to promote consistency throughout, the above definitions for ecosystem services and functions have been taken from the Millennium Ecosystem Assessment (2003).

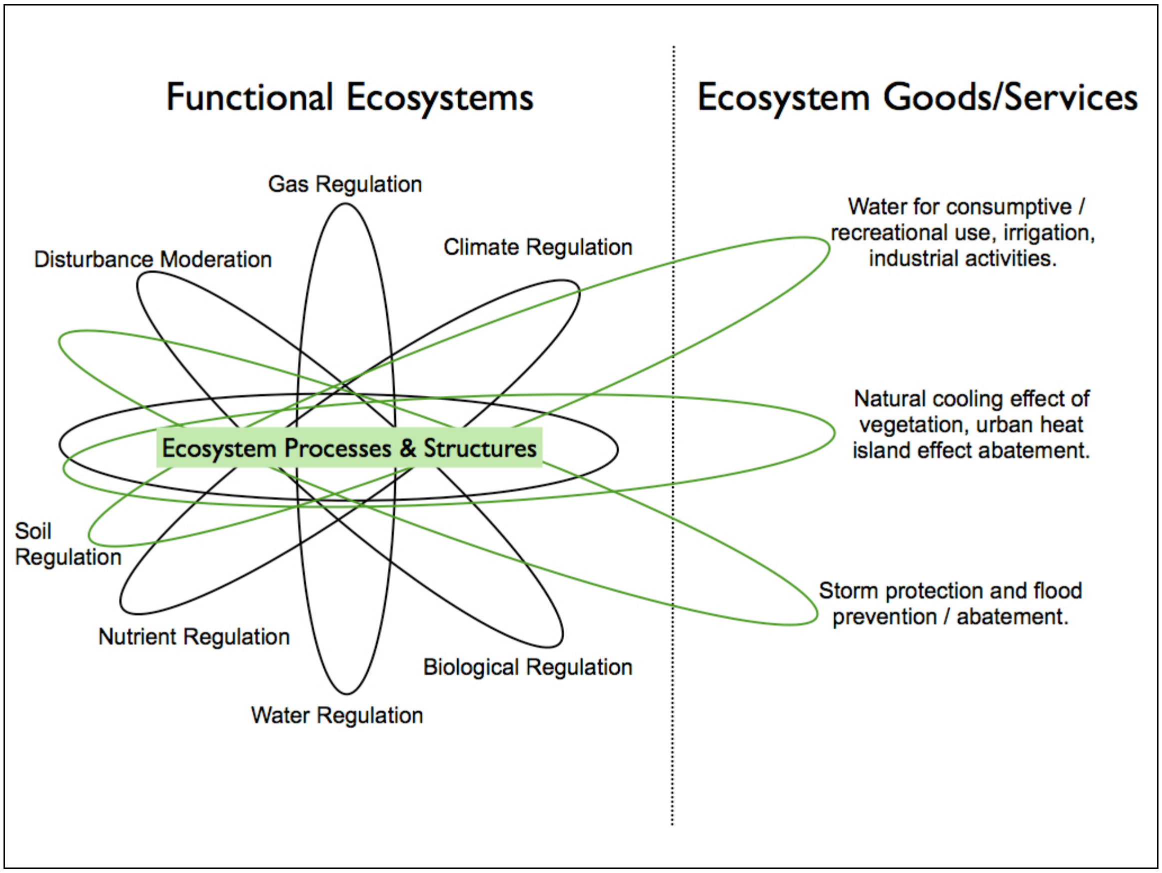

Figure 2. Conceptual relationship between functional ecosystems and ecosystem goods and services. *Note:* Adapted from "Defining and classifying ecosystem services for decision making" by Fisher *et al.,* 2009, Ecological Economics, 68, p.646. Copyright 2008 by Elsevier B.V. Adapted with permission of the author.

For society to benefit from the continued provision of ecosystem services, ecosystems must be healthy - *i.e.,* in a functional condition. The functional condition of an ecosystem generally refers to the continued operation of an ecosystem, "its integrated holistic dynamics, and not the role or job of an ecosystem" (King, 1993, p.20). Related to the regulating functions and services, when an ecosystem is in a functional condition, it has the required elements to "withstand disturbance and perform a variety of important ecosystem services" (Townsend, 2009, p.16). Because ecosystems are dynamic, heterogeneous and open, ecosystems can have a range of functional conditions within which they can operate (Prichard, 1998). For instance, ecosystems can be in a functional, functional-at-risk, non-functional or unknown condition (See Appendix C for the description of each). To what extent and when an ecosystem changes its functional condition depends on both the internal and external disturbances (disturbances can be either positive or negative) that influence ecosystem processes and structures (*e.g.,* changes in nutrient and energy inputs, soil removal, introduction of invasive species, changes in hydrology) (MEA, 2003).[4] The current functional condition of the ecosystem will tend to influence what regulation functions occur and the capacity (*e.g.,* more or less waste assimilation, more or less nutrient cycling) to which these will manifest as ecosystem services.[5]

Ecosystems have varying properties of resistance and resilience that influence the functional condition of an ecosystem (Edmonds, 2002; MEA, 2003). Ecosystem resistance is an ecosystem's ability to withstand disturbance and maintain its current physical state. Carpenter, Walker, Anderies and Abel (2001) define resistance as the amount of "pressure" that is required to cause a "given amount of disturbance" in an ecosystem (p. 766). It is the ability of the ecosystem to "spring back undamaged" (Edmonds, 2002, p. 24). The properties of ecosystem resistance and resilience is important to understanding how natural and anthropogenic pressures influence the ability of ecosystems to resist and recover from disturbances (MEA, 2003).[6] For instance, land changing activities, such as urban development, can have detrimental effects on ecosystems and increasing vulnerability to changes that previously could have been

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[4] For instance, natural forest fire cycles can be a positive disturbance as fire may be an integral component to seed germination (Holling and Gunderson, 2002).

[5] However, there is no one to one ratio of ecosystem function to the provision of ecosystem services. For instance, functional-at-risk and non-functional ecosystems could still provide ecosystem services; albeit, at a lower capacity and are not likely to be sustainable in the long-term.

[6] Recovery is the ability of a system to return to a state of ecosystem function (Palumbi, McLeod, and Grunbaumr, 2008). However, recovery can be incapacitated or impaired if an ecosystems structure, processes or boundary conditions are severely altered (*e.g.,* removal of soil, unnatural forest fires, increased frequency and volume of flows due to stormwater discharges, invasive species, human modifications to ecosystems) (MEA, 2003).

absorbed (Folke *et al.,* 2002). Reductions in resistance and resilience can occur by altering diversity, removal of vegetation, changing hydrologic regimes, releasing waste and pollutants, influencing climate patterns, *etc*. Such changes can cause ecosystems to flip, unpredictably, to irreversible states, resulting in ecosystems that are "more spatially uniform, less functionally diverse, and thereby more sensitive to disturbances that otherwise could have been absorbed" (Holling and Gunderson, 2002, p.60). At times, some ecosystems can be disturbed to such an extent that the ecosystem may not return to a functional condition (Prichard, 1998).

From a resource management or decision maker's perspective, ecosystems near unknown thresholds are the most challenging ecosystems to manage as they may change dramatically into a different physical state thereby influencing what ecosystem services are produced (Palumbi, McLeod and Grunbaumr, 2008).[7] Once an ecosystem has surpassed a threshold, it has likely entered into a new physical state that may exhibit a high degree of resilience or resistance (Holling and Gunderson, 2002). For instance, a clear lake receiving a constant stream of nutrient inputs due to anthropogenic activities may eventually surpass an unknown threshold and then progressively flip into an eutrophic stable state. From an ecological perspective, the lake has shifted to a new state. From an ecosystem service perspective, the lake no longer provides the same ecosystem services as before (*e.g.*, recreation, fishing, clean water), if any at all. From a decision-making or a resource management perspective, the lake has surpassed an unknown threshold into a eutrophic state which could be irreversible in the short term.[8]

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[7] The challenge to decision makers and resource managers is that gradual changes may not have an effect on the assessed functional state of an ecosystem (*e.g.,* eutrophication of lakes, fisheries management). As there is great difficulty and debate amongst researchers on what resilience is and how to measure the state of an ecosystem, decision makers are incorporating fail-safe rules such as the precautionary principle and the safe minimum standard to avoid surpassing unknown ecosystem thresholds (MEA, 2003). The precautionary principle states "when an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically" (Wingspread Conference on the Precautionary Principle, 1998). The safe minimum standard is simpler, yet follows the same principles as the precautionary principle, proposing that the overarching goal would be to preserve a renewable resource at a level that ensures that the ecosystem avoids extinction, unless the social costs were prohibitive or immoderate (MEA, 2003).

[8] See Folke *et al.,* 2004; Holling and Gunderson, 2002; MEA, 2003; Walker and Salt, 2006 for further discussion on ecosystem thresholds, resilience and resistance as it is not the purpose of this thesis to explore the measurement of these properties.

In the context of continuing to provide ecosystem services that are fundamental to human social and economic well-being, the ecological properties become very important as they enable ecosystems to resist or recover from disturbances thereby continuing to function as healthy ecosystems and provide the values society seeks (MEA, 2003; Folke *et al.,* 2004). Therefore, it is important for decision makers to be able to assess the trajectory and stability of an ecosystem if they are to ensure that existing functional ecosystems remain and that those ecosystems in a functional-at-risk condition are managed in such a way to avoid thresholds and promote recovery (Mooney, Pers. Comm.). By understanding that the current functional condition of an ecosystem influences the quality and quantity of ecosystem services over time, the rehabilitation of degraded ecosystems could become an integral, much needed, component to land-use management (Hobbs and Harris, 2001).[9]

Rehabilitation efforts would not necessarily attempt to restore the ecosystem to an original condition; rather, the goal in the short-term would be to stabilize the degradation process (to avoid continued losses) and allow for natural succession (Mooney, Pers. Comm.). For rehabilitation efforts to be successful, efforts must first focus on removing or altering the pressure causing the degradation (*e.g.,* land management, point and non-point source pollution, stormwater discharge, *etc.*) and then repairing the physical or chemical environment (Hobbs and Harris, 2001). In the long-term, the goal should shift to managing ecosystems for their functional condition in order to achieve valuable ecosystem services.[10] However, to succeed, rehabilitation efforts "need to not only be based on sound ecological principles and information, but also to be economically possible and practically achievable" (Hobbs and Harris, 2001, p.243).

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[9] Ecosystem rehabilitation defined as the "attempt to restore elements of a structure or functioning of an ecosystem without necessarily attempting complete restoration to a prior condition (MacMahon and Holl, 2001, p.247).

[10] The value to which the ecosystem services would be assigned would depend largely upon the context to which an ecosystem is situated. For instance, in the case of a water provisioning service, the value of this service may depend on the technology available (*e.g.,* water treatment, pipes, pumps) to transform the provisioning service (aka: regulating function) into a usable ecosystem service.

1.3 Research Aims and Organization of Study

Given the importance placed on economics in decision-making and cost-benefit analysis, not valuing ecosystems implies that they are worth nothing (Costanza *et al.,* 1997). However, estimating the economic value of ecosystem services without measuring or adjusting the estimated values for the functional condition of the ecosystem does not provide resource managers and decision makers the critical information required to understand the ecological factors (*e.g.,* ecosystem processes, structures, boundary conditions, properties of resistance, resilience and recovery) that influence the functional condition of an ecosystem. Thus, the main aim of this thesis is to examine the link between the functional condition of an ecosystem and the economic values that are assigned to ecosystem services. Framed within the urban resource-planning context, it is argued that the values assigned to ecosystems or ecosystem services need to reflect the functional condition of an ecosystem. Since the intent of this thesis is to explore how both the valuation of ecosystems and the measurement of ecosystem function could be utilized together in urban development planning, the question driving the research was: Is the estimation of ecosystem service values an accurate reflection of ecosystem function?

Chapter 2 begins to address the thesis question by first investigating if ecosystems should be valued. The intent of asking this question was to avoid the assumption that ecosystems should be valued in monetary terms without formally examining the arguments provided by those who oppose ecosystem valuation and those who do not. Of particular importance was to identify why there is a debate on the subject, since not valuing ecosystems is a choice - one that implies the ecosystem is worth zero. The review also provided critical information and insight into the strengths and weaknesses of the ecosystem valuation discipline.

Chapter 3 examines how ecosystems are valued by reviewing and evaluating the fundamental basics of ecosystem valuation. The chapter reviews fundamental ecosystem valuation concepts, including: Total Economic Value typology, use-values versus non-use values, temporal and spatial scales and ecosystem valuation methods. The importance of the review to the thesis topic is that it reveals that current ecosystem valuation methods do not explicitly require the measurement of the functional condition of an ecosystem.

Chapter 4 utilizes a local case study (Swan Lake watershed in Victoria, British Columbia) to illustrate the point that when the functional condition of an ecosystem is not assessed during the valuation exercise, the estimated ecosystem service values provides little, if not any, information on the functional condition of the ecosystem that generated them. Without this additional context when valuing ecosystems, in answering the thesis question, it is contended that decision-makers will not be able to adequately identify how ecosystem changes arising from man-made or natural perturbations will impact socio-economic systems.

Chapter 5 utilizes another local case study (British Pacific Properties Rodgers Creek development in West Vancouver, British Columbia) in order to assess whether or not the valuation of ecosystem services and the measurement of ecosystem function could be utilized as a planning technique, which results in the preservation and enhancement of functional ecosystems.

Chapter 2.

2.1 Introduction

It is relatively easy to determine the monetary value of manufactured goods, corporations, buildings, land, *etc.,* because they can be measured, weighed or appraised using standardized assessment methods. However, the less tangible things that are more important to society, such as the value of human life or the value of human virtues cannot be assigned monetary value. Because we can measure and place a value on something does not make it any more real or significant than something we cannot adequately measure. For instance, functional ecosystems provide ecosystem services that are fundamental to human well-being and welfare; without these fundamental life-supporting ecosystem services, society would ultimately not survive (MEA, 2003). So, does assigning a monetary value to ecosystems make them any more important to society than they were prior to the valuation exercise? Arguably not, as placing a monetary value on an ecosystem has not improved the capacity of the ecosystem to provide services nor has it hindered the ecosystems capacity to do otherwise. Rather, the purpose of assigning values to ecosystems is that the valuation exercise provides information to help inform decisions regarding trade-offs, to provide estimates on damages that have occurred, to measure the value of the assets to incorporate them into national income accounts and to ensure that the full implications of decisions are considered. Those who argue against the valuation of ecosystems believe that the environment has value in and of itself, that is, the environment is intrinsically valuable and therefore, should not be valued.

Since the debate on whether ecosystems can be valued or not still rages within and amongst various disciplines a fundamental question must be addressed: Should we value ecosystems at all? The intent of asking this question is to avoid the assumption that ecosystems should be valued in monetary terms without formally examining the arguments of those who oppose ecosystem valuation and those who support it.

2.2 What Is Value?

There is a great deal of complexity that surrounds the question, should we value ecosystems, as the term value elicits both philosophical and ethical debate. In order to address this question, the term value must first be explored briefly as it can have different meanings both philosophically and semantically depending on the context in which it is applied.

The term value has philosophical underpinnings that originate from the schools of utilitarian and deontological philosophical thought and thus requires the distinction between intrinsic and instrumental values as well as anthropocentric and non-anthropocentric (biocentric) values (NRC, 2004). In the context of the natural environment, an ecosystem has instrumental value if it is useful in achieving a goal – *i.e.,* the ecosystem contributes towards achieving a means to an end other than itself (NRC, 2004; Straton, 2006). In contrast, an ecosystem is said to have intrinsic value if it has value in and of itself. In other words, the ecosystem has value independent of its contribution to humans or animals. For example, a fish population that provides nourishment for a population, whether it is human or animal, would be said to have an instrumental value because the fish population contributes to sustaining a population. This same fish population could also be said to have intrinsic value even if it did not contribute to human well-being. Anthropocentrism assumes that human beings possess a greater intrinsic value than non-human nature and, therefore, the value of non-human nature originates solely from its usefulness to humans. A non-anthropocentric approach proposes that the environment has value in and of itself, even if it is not considered to have value by humans. Non-anthropocentrism implies that humans are not more important than other living things (Straton, 2006). Both instrumental and intrinsic values can either be anthropocentric or non-anthropocentric and can be categorized in the following way: anthropocentric instrumental value, anthropocentric intrinsic value, non-anthropocentric instrumental value and non-anthropocentric intrinsic value (See Table 1) (Turner *et al.,* 2003). The distinctions between these values have arisen from the differing philosophical schools of utilitarian and deontological thought.

Table 1. Classification of environmental values

	Anthropocentric	**Non-Anthropocentric**
Instrumental	Total Economic Value (TEV). Consisting of use and non-use values (includes existence value as the continued existence of an entity generates value for humans).	The value of entities independent of humans - *e.g.,* the contributory value of ecosystem processes, structures, functions to ecosystem health, biodiversity, *etc*.
Intrinsic	Value is attributed to entities that are valuable in and of themselves (*e.g.,* cultural value). This is an anthropocentric value, as a human valuer has placed a value (monetary or non-monetary) on the entity.	Entities have inherent value. That is, they possess value independent of any valuer.

Note: Based on Turner *et al.* (2003); NRC (2004); and DEFRA (2007).

Utilitarianism is based upon the notion that "actions are right in proportion as they tend to promote happiness, wrong as they tend to promote the reverse of happiness" (Mill, 1969). It is a moral philosophy that operates on the principle that the derived happiness of individuals from a given object or service can be meaningfully measured and aggregated to reflect society's overall well-being (or happiness) (NRC, 2004). "In this sense, utilitarian values are instrumental and anthropocentric in that they are viewed as a means toward the end result of increased human welfare as defined by human preferences" (NRC, 2004, p.30). Although utilitarian values cannot be measured directly many analysts still use monetary valuation to express human preferences for ecosystem services (MEA, 2003; Kumar and Kumar, 2008). In contrast, the deontological approach, known as a "duty-generating" approach, implies a set of rights to exist (NRC, 2004).

The deontological philosophy takes an intrinsic approach to human beings - it implies that something with intrinsic value cannot be replaced, substituted or compensated by having more of something else (NRC, 2004). Under the deontological ethic, human beings are ends in themselves and thus there is simply no replacement for them - *i.e.,* there is no compensation for the impact on human health or the death of a human being (Booth, 1994). Although the original definition of intrinsic value was based upon the value of human life, the term now includes both non-humans and the environment (Booth, 1994; NRC, 2004). According to the MEA (2003), intrinsic value incorporates the more intangible side of the social equation such a culture, ethics and religion, and tends to focus more on the legislative and political aspects of a system, rather than on the economics. However, there is considerable debate on how instrumental and intrinsic values should influence decisions regarding the environment. For instance, Farber *et al.* (2006) suggest that intrinsic rights and moral obligations can be utilized to establish boundaries of which the utilitarian management decisions can operate within. In summary, the economic valuation of ecosystems is an anthropocentric approach based upon utilitarian principles (NRC, 2004).

From a semantics standpoint, the term value has different meanings. For instance, the Oxford English Dictionary (2008) provides three different meanings of the term: exchange value, the monetary worth of something; utility, the usefulness of something in regards to its purpose; or of moral importance, the principles/standards of ones behavior. According to de Groot, Stuip, Finlayson and Davidson (2006), these three definitions of value relate directly to the disciplines that are involved in ecosystem valuation: economics (exchange), ecology (utility) and sociology/philosophy (moral importance). The difficulty with the term value is that it does not consist of one concept, but of several related concepts derived from other disciplines. This complexity of the term value has resulted in various problems arising from these differing perspectives (Straton, 2006):

> [The] valuation of ecosystem goods and services is further confounded by the different perspectives of ecologists and economists […]. The ecologist's perspective lacks consideration of the social processes and human preferences that guide resource use; economists ignore the biophysical and ecological processes that sustain ecosystem goods and services. (p. 404)

Straton (2006) argues that a new economic framework is required; one that integrates the subjective elements of the term value between disciplines. Farber, Costanza and Wilson (2002) have discussed this in great length as well, stating that other valuation perspectives such as socio-cultural, ecological, biological, and biophysical value should be incorporated into the definition of economic value. Kline (2006) summarizes these perspectives quite succinctly:

> Socio-cultural value perspectives focus on the distribution of ecosystem services among members of society. An ecological value perspective measures value as the degree to which ecosystem services contribute to ecological objectives or conditions, such as healthy ecosystem function. Similarly, a biological value perspective measures the value of ecosystem services by their contributions to meeting biological objectives, such as the survival of individual species. A biophysical value perspective defines value in terms of direct and indirect inputs and outputs of mass and energy among ecosystem components. (p.12)

It is clear that the different perspectives on value provide an example of the varying objectives of each discipline and the inherent difficulties of defining and linking values between them. Given that the goal of this research was to examine the link between the functional condition of an ecosystem and the economic values that are assigned to ecosystem services, the modern definition of value provided by economists is inadequate. The economic discipline argues that value originates from individual preferences through their interaction with the market place (Straton, 2006). This definition of value does not allow for other perspectives, such as the ecological value perspective of ecosystem health defined by Kline (2006). Therefore, the MEA (2003) definition of value as “the contribution of an action or object to user-specified goals, objectives, or conditions” will be applied throughout this research as this definition attempts to include the perspectives of other disciplines (MEA, 2003, p.216). Specifically, this anthropocentric definition of value recognizes that the economic value of ecosystems is derived from the utility that human beings receive directly or indirectly from the ecosystem services that are provided by functional ecosystems. The quantification of the value of ecosystem services by economists is an anthropocentric instrumental approach and because of this, a long-standing debate has arisen on whether ecosystems should be valued or not.

2.3 Should We Value Ecosystems?

The valuation[11] of ecosystems has attracted considerable attention over the years from both economists and non-economists alike. For many non-economists (and some economists), the valuation of ecosystems or ecosystem services should not automatically imply that ecosystems or their services should be quantified in monetary terms (NRC, 2004). Those who believe that ecosystem's have intrinsic value argue that the valuation of ecosystems is unnecessary and inappropriate as ecosystems are invaluable or have infinite worth. Specifically, they argue that human welfare is dependant upon the stable and continuous provision of ecosystem services and any valuation of ecosystems is unethical as the need for protection / preservation of environmental assets is self-evident (Toman, 1998; Ludwig, 2000; Dore and Burton, 2003). It is further contended that economic values are less important than, and incompatible with, personal and social values; placing a dollar value on ecosystems is, by some accounts, unethical. For instance, Ludwig (2000) argues that economic values are of a "tertiary importance" when compared to personal values such as personal integrity and dignity, further using the example that "love and friendship cannot be bought" (p.32).

Costanza, Fisher, Mulder, Liu and Christopher (2007) rebut this argument, stating that they see "no logical conflict between identifying economic reasons for preserving natural systems and stating ethical reasons; in principle, these are mutually supportive rather than either/or justifications" (p.3). The authors affirm that the unethical argument stems from a false presumption that the valuation of ecosystems or ecosystem services implies that ecosystems can or should be traded in the marketplace, that is, valuation is just a process to get ecosystems into the marketplace (Costanza *et al.,* 2007). However, this is not the case as many ecosystem services are public goods. That is, they are non-rival (one individual's consumption does not prevent consumption by another) and non-excludable (others cannot be excluded from the use of a good) and the usage of private markets to manage ecosystem services would not maximize social welfare (Costanza *et al.,* 2007).

[11] Valuation defined as "the process of expressing a value for a particular good or service in a certain context (e.g., of decision making) usually in terms of something that can be counted, often money, but also through methods and measures from other disciplines (sociology, ecology, and so on)" (MEA, 2003, p.216).

Furthermore, the proclamation that the environment has infinite worth, implying "that nature is sacrosanct and beyond measurement", is an extreme approach and is not dissimilar to the business as usual (Milne, 1991, p.81).

Although, these approaches are on opposite ends of the spectrum, they are similar in that both leave a blank space for the environment in the cost-benefit or decision analysis. In either case, leaving the environmental space blank in the decision analysis implies that an ecosystem under question is worth nothing and as a consequence is likely to result in too little protection of the ecosystem (Costanza *et al.,* 1997; Balmford *et al.,* 2002; NRC, 2004). Furthermore, Costanza *et al.* (2007) assert that the process of making a choice is a value decision; one that implies that one alternative is more valuable than the other. For instance, the installation of a highway through a wetland implies a value decision as the choice indicates that the highway is worth more than the wetland. In other words, "to say that we should not do valuation of ecosystems is to simply deny the reality that we already do, always have and cannot avoid doing so in the future" (Costanza *et al.,*1998).

Those against ecosystem valuation have also argued that not all things that have a social value can be measured in monetary terms and valuation may actually provide society a disservice (Pearce, 1998; Dore and Burton, 2003). Ludwig (2000) argues that the application of ecosystem methods is limited at best and is "inappropriate and harmful" especially when ecosystem valuation methods are being applied to help determine public policy (p.31).[12] This can be true if ecosystem valuation is applied incorrectly or is taken out of context, which at times has resulted in "valuation backfires".[13] However, the intent of ecosystem valuation is to formally estimate the nonmarket values that the public already holds with respect to ecosystems (as well as identify users of ecosystem services), to help rank priorities and evaluate the effects of various development options, to provide estimates on damages that have occurred and to help explain why some areas should be

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[12] The author states three reasons for this: economic theories use simplified assumptions, market measures are inappropriate for addressing ecological questions (*e.g*., thresholds) and ecosystem valuation methods suffer from inherent flaws (*e.g.,* inter- and intra-generational problems) (Ludwig, 2000).

[13]King and Wainger (1999) provide examples of where ecosystem valuation studies were used to help inform a decision or policy making process, but had unintended effects.

preserved or rehabilitated (Costanza *et al.,* 1997; NRC, 2004; de Groot, 2006; Fisher *et al.,* 2009).

Including ecosystem values in a benefit-cost analysis has at times resulted in the inclusion and preservation of some ecosystems. For example, the New York Catskill/Delaware Watershed and the Charles River Basin, in Massachusetts, are common examples of how the application of ecosystem valuation to natural resource management can be beneficial. In 1990, New York City was facing cost estimates ranging from $6-8 billion for the installation of a traditional drinking water filtration facility. Using ecosystem valuation and deliberation / stakeholder engagement techniques, the city chose to pay landowners surrounding the area of its reservoirs to adopt alternative land management practices which only cost $2 billion; a minimum savings of $4 billion (Daily and Ellision, 2002). A similar example is the preservation of the Charles River Basin. The U.S. Army Corps of Engineers, the Commonwealth of Massachusetts and local governments acquired 8,500 acres of wetlands in the Charles River Basin to serve as a natural valley storage area for floodwaters. The cost of acquiring the wetlands was $10 million; the alternative approach (*i.e.,* constructing dams and levees) would have cost upwards of $100 million (Fausold and Lilieholm, 1999). In both the examples, the purpose of ecosystem valuation was not to determine a single number that describes the entire worth of an ecosystem; rather, the intent of the valuation exercise was to make sure that the social costs and benefits were represented in the decision making process (Pritchard, Folke and Gunderson, 2000).

Those opposed to the valuation of ecosystems also argue that valuation often suffers from intra- and inter-generational equity issues (Chavas, 2000; Ludwig, 2000; Dore and Burton, 2003). There is a perceived danger that ecosystem valuation may undervalue the needs and values of current and future generations (Milne, 1991). The concern is that policy and decision makers make uninformed decisions on what time scale and discount rate[14] to apply as well as determining which ecosystem services are valuable (Chavas,

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[14] When economists or decision makers need to evaluate benefits and costs over a specific time frame (more than one year), they must discount the values over time as a means to compare the differing cash or benefit flows. Known as the time value of money, decision makers must make allowances for the fact that an

2000). For example, an ecosystem service considered valuable today is likely to elicit the response that it is threatened and thereby the focus would be on a particular output or service (*i.e.* water quality), rather than taking a long-term focus on the opportunity or resilience costs of threshold changes. In the long term, the concern is that the management of ecosystem services rather than the health of an ecosystem can result in unintended, unforeseen and often irreversible changes in ecosystems that have associated economic costs (Holling and Meffe, 1996).[15] Discounting suffers from similar temporal setbacks.[16] The main concern with discounting of ecosystem service values is that the method often renders the long term damage of little importance to the present day for the net present value[17] of the cost of the damages is negligible and requires no preventative action (Ludwig, 2000).[18]

These arguments and concerns against ecosystem valuation have resulted in the suggestion that those within the discipline modify economic analyses to incorporate scientifically based rules such as the safe minimum standard (SMS) or the pre-cautionary principle (Chavas, 2000; MEA, 2003; Fisher *et al.,* 2008)[19]. The basic principle of the SMS is that it is based upon scientific knowledge and understanding and thus enables decision makers to place constraints and biophysical boundaries on the ecosystems in question, whilst still allowing decision makers to utilize ecosystem valuation for decision making (Milne, 1991). Furthermore, the recognition of the SMS principle is "practically

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individual's value time horizons differently than the present (a dollar today is worth more than a dollar tomorrow because the dollar can be invested today). For example, $500 received today at a 5% interest rate would be worth $638.14 five years from now. Whereas if $500 was received five years from now, its present value at a 5% interest rate today would be $391.76.

[15] Examples cited by the authors include the suppression of natural fires to protect homes, the use of dams to stabilize stream flows, channelization and "ditching" of streams and rivers to enable development within the floodplain. In the long term, many of these practices have backfired.

[16] There are many different theorems on discounting, such as consumption discounting, hyperbolic discounting (declining discount rates), gamma discounting (many constant discount rates), social discount rates, *etc*. (*e.g.,* Azar and Sterner, 1996; Newell, 2003; Sumaila and Walters, 2005; Hansen, 2006).

[17] Defined as "the interest rate used in determining the present value of future cash flows," the discount rate is an applied financial mechanism to determine the net present value of costs and revenues in the future (Pearce, 1993, p.54).

[18] As the future benefits and costs become quite low due to discounting there is a disincentive to collect information or to avoid causing damage as the discounted costs become so low (Bingham *et al.,* 1995).

[19] The process of applying the precautionary principle must be open, informed and must include potentially affected parties" (Wingspread Conference on the Precautionary Principle, 1998).

equivalent to socially recognizing their intrinsic value [ecosystems] and protecting them by law" before the ecosystem valuation, the anthropocentric instrumental approach, is undertaken (MEA, 2003, p.146).

In conclusion, ecosystem valuation is an important element to help reconcile competing values in the realm of public discourse and in policy-making (MEA, 2003). However, ecosystem valuation cannot and should not solely be used to determine what social actions are necessary as public discourse and policy-making requires an integrated and interdisciplinary approach (Toman, 1998; Pritchard *et al.,* 2000; MEA, 2003; de Groot*,* 2006).[20] Cognizant of inherent limitations, the ecosystem valuation literature continues to develop and refine methods to improve the reliability of estimated values and to provide a better informational base for the decision process and policy-making (Turner *et al.,* 2003; Fisher *et al.,* 2009). Consequently, many valuation studies have helped to heighten the level of importance and knowledge of Earth's ecosystems (Pagiola, 2008). For instance, Costanza *et al.* (1997) estimated values for 17 ecosystem services from 16 ecosystem types concluding that the value of global ecosystem services is estimated to be in the range of $16–54 trillion with an average of $33 trillion per year.

Although the work completed by Costanza *et al.* (1997) generated considerable controversy from a myriad of disciplines with researchers criticizing the authors on their methods and ethics surrounding the valuation (*e.g.,* Pearce, 1998; Dore and Burton, 2003), Costanza *et al.* (1997) expressed that they have achieved one of their many goals - to establish and "stimulate additional research and debate" on the valuation of ecosystems (Pearce, 1998, p. 27). On a more tangible basis, ecosystem valuation studies have been employed to demonstrate that underutilized (or what is perceived to be worthless land)f can provide valuable ecosystem services (Pagiola, 2008). For instance, there is increasing recognition that the multi-functional use of an ecosystem (*e.g.,* stormwater control, flood abatement, carbon storage and sequestration, aesthetics) is economically more beneficial to cities and communities than if these systems are degraded (de Groot*,* 2006;

[20] Ecosystem valuation can provide important information to the decision process, but valuation alone cannot determine what actions are appropriate as goals of equity, sustainability and fairness may trump the valuation (Toman, 1998; Schaeffer, 2008). See Costanza (2003) for further discussion.

Barraclough and Hegg, 2008a). Furthermore, valuation studies are useful in that they are the first step in establishing payment schemes, such as payment for ecosystem services (PES) in Costa Rica to wetland banking schemes in the United States, as a means to compensate landowners for preserving ecosystems (Pearce, 1998). Since economic markets play dominant roles in the environmental decisions humans undertake, the economic valuation of ecosystems can be a means to create a common language with which environmental and economic decisions can be compared (Pearce, 1998; MEA, 2003). The answer to the question of whether ecosystems should be valued is yes. There may be no right way to value ecosystems, "but there is a wrong way, and that is not to do it at all" (Pearce, 1998).

Chapter 3.

3.1 Introduction

Chapter 2 explored the question of whether or not ecosystems should be valued. The chapter concluded that ecosystems and ecosystems services should be valued as the valuation exercise can provide information to help inform decisions on trade-offs, to provide estimates on damages that have occurred or to measure the value of the assets to incorporate them into national income accounts (Costanza *et al.*, 1997; Pearce, 1993; NRC, 2004; Hindmarch *et al.,* 2006). Although the original intent of ecosystem valuation was to formally estimate the non-market values of ecosystems and ecosystem services, a theoretical approach, the ecosystem valuation exercise over time is becoming more applied (MEA, 2003; NRC, 2004). Specifically, ecosystem valuation is being utilized to establish various payment schemes and to demonstrate the value of land to provide multiple services to urban communities.

Since the intent of this thesis is to explore how both the valuation of ecosystems and the measurement of ecosystem function could be utilized together in urban development planning, the question that is addressed within this chapter is: how do we value ecosystems? Answering this question results in the review and evaluation of the fundamental basics of ecosystem valuation. The importance of the review to the thesis topic is that it reveals that current ecosystem valuation methods do not explicitly require the measurement of the functional condition of an ecosystem.

3.2 How Do We Value Ecosystems?

As mentioned in Chapter 2, the economic valuation of ecosystems is an anthropocentric instrumental approach. Developed by Randall and Stoll in 1983, a commonly applied framework developed to help categorize the value of ecosystems is the Total Economic Value (TEV) typology (Fromm, 2000). The TEV typology attempts to simplify the valuation exercise by distinguishing between various use and non-use values and can act as a checklist to help ecosystem valuers account for all the factors in order to achieve a complete analysis (Pearce, 1993; NRC, 2004).

The framework can be composed as follows (Pearce, 1993):

Total Economic Value (TEV) = Use Value [*Direct-use value* + *Indirect-use value* + *Option value*] + Non-use Value [*Existence value* + *Bequest value* + *Philanthropy value*].

Use values consist of direct-use, indirect-use, and option values. Direct-use values are simply the values that arise when humans use an ecosystem for consumptive or non-consumptive uses (MEA, 2003). Consumptive uses could include the harvesting of food products, hunting of animals, harvesting of timber for building or energy supplies or the installation of dams for hydropower, *etc*. Non-consumptive uses include recreational activities. Indirect use values stem from the indirect utilization of ecosystems services (*e.g.,* nutrient cycling processes in soil that contribute to agricultural production). Indirect use values tend to be the regulating functions of ecosystems and are often referred to as intermediate inputs or services as they contribute to the final production of ecosystem goods and services (MEA, 2003).[21] The direct and indirect-use values tend to be clear, easy to define values that are distinguished between use and non-use values and, as a result, these values are most often included in ecosystem studies. The more difficult use value to estimate is the option value of an ecosystem or ecosystem service. Option value is defined as the monetary value that people are willing to pay for an ecosystem to ensure that it is preserved for future use (Hein, van Koppen, de Groot and van Ierland, 2006). Option value is often misunderstood and confused with non-use values. Option value is a use value as it is the value that an individual would place to preserve an ecosystem or ecosystem services for *their* use at a future date (MEA, 2003). It is not the value that an individual places on an ecosystem or ecosystem to preserve it for others use, as this

[21] Fisher *et al.,* (2008) contend that the concept of ecosystem services has not been properly operationalized and thus ecosystem services should be qualified as intermediate or final ecosystem services. The authors provide an example stating that a final ecosystem service would the provision of food, whereas pollination would be an intermediate service. This classification is beneficial in that it recognizes the importance of intermediate services; however, careful attention to valuing these services as valuing both the intermediate and final services for food production would result in double counting. These intermediate inputs or services are referred to as the regulating functions of ecosystems (see Chapter 1 for discussion).

would be considered a non-use value. Option value is further complicated as it has a sub-set value associated with it, called quasi-option value. This value "represents the value of avoiding irreversible decisions until new information reveals whether certain ecosystems have values we are not currently aware of" (Hein *et al.,* 2006, p.213). For instance, the value of the carbon storage and sequestration service provided by vegetation is now considered to be an important ecosystem service; however, 30 years ago this was not the case. The challenge with measuring the quasi-option value is that it is very difficult to assess and estimate (Hein *et al.,* 2006). Overall, the main advantage of use values is that the estimated value is normally based upon current market prices. The disadvantage with use values is that the estimated value often understates the total value of ecosystem services (Milne, 1991). It is largely because of the underestimation of the total value of ecosystem services that non-use values and associated methods were developed (MEA, 2003).

Non-use values consist of existence, bequest and philanthropy values.[22] These non-use values can be either instrumental (*e.g.,* natural beauty) or intrinsic (*e.g.,* animal and plant species have a right to exist) and, thus, are difficult to measure and estimate (Wilson and Howarth, 2002; MEA, 2003; DEFRA, 2007). Non-use values are commonly distinguished from use values by the donation of sensitive land to land trusts or placing legal covenants on areas of land to ensure that the current condition of the natural environment will be passed on to future generations. Existence value, for instance, is the value derived from simply knowing an ecosystem exists. Depending on the individual, this value could include either instrumental or intrinsic values or both. Bequest value, sometimes referred to as integernational or intertemporal value, is the monetary value that people are willing to pay for an ecosystem to ensure that it is preserved for their heirs' future use. Consequently, it is quite similar to option value (Milne, 1991). Philanthropic value is also similar, but the value is derived directly from the value that an individual gains from knowing someone else benefits from the preservation of an ecosystem or an ecosystem service. The challenge with non-use values is that they are

[22] Philanthropy values are sometimes also called altruistic value (Hein *et al.,* 2006) or vicarious value (Milne, 1991).

often the most difficult and controversial values to estimate due to conceptual and empirical difficulties (MEA, 2003; Hein *et al.,* 2006).

A value often overlooked by the TEV model is the historical/cultural symbolic value as it is an intrinsic value (MEA, 2003). For instance, a particular mountain may be of great cultural significance to an aboriginal community. With regard to ecosystems and ecosystem services, some of these values may be intrinsic or instrumental in nature and, therefore, in the case of the latter, some economists suggest the use of participatory valuation assessments (often referred to as deliberative monetary valuation or group valuation). However, it is generally not feasible or acceptable to place economic values on the historical/cultural/symbolic element even if the values are of an instrumental nature (Fisher *et al.,* 2008). Therefore, rather than placing a monetary value on this type of value, decision makers can incorporate a process of public deliberation as a way to recognize this value in the decision-making process (MEA, 2003). Although not often recognized in the valuation literature, the process of public deliberation can be very important to the TEV framework as it is these stakeholders who hold the different values for an ecosystem or ecosystem service (Hein *et al.,* 2006). However, the identification of who the stakeholders are and what ecosystem services they use is a considerable challenge to ecosystem valuers. It is therefore important that ecosystem valuers assess the temporal and spatial scale by which ecosystem services occur and to whom they are valuable.

3.2.1 Ecosystem Scales

The scale to which ecosystem use and non-use values are captured is a very important element of valuing ecosystems both in terms of practicality and in equity (Turner *et al.,* 2003). For instance, local users may have a preference for short-term gain derived from the consumption or sale of the services derived from a forested ecosystem. But at a national or international scale over the long-term time, global users may express conservation-based preferences tied to the ecosystem functions (Bingham *et al.,* 1995; Hein *et al.,* 2006). De Groot and Hein (2007) state that in order to adequately analyze or model ecosystem services, three scales need to be considered: the ecosystem scale, the socio-economic system scale and the scale of landscape functions. The spatial analysis is important as it can help the ecosystem valuer identify the spatially overlapping ecosystem functions and services (de Groot and Hein, 2007).

According to the MEA (2003), ecosystems "can vary enormously in size" (p.3). They are complex and constantly changing due to natural succession or human-made perturbations.[23] The functional condition of ecosystems, and resulting provision of ecosystem services are based upon the interactions between the structures, processes and functions within the ecosystem and these occur at different spatial and temporal scales (King, 1997; de Groot and Hein, 2007). For instance, the properties of resistance, resilience and recovery and the subsequent functioning of an ecosystem can be influenced by microbes at the microscopic scale, insect outbreaks and fires at the site scale, or shifts in precipitation patterns at the global scale (MEA, 2003). According to Limburg *et al.* (2002), large-scale, long period changes set constraints on the smaller scale changes. However, the authors also note that the collective small-scales changes at the microscopic scale can influence and trigger large-scale changes. Thus, the challenge to ecosystem valuers is to determine which ecosystem scales are appropriate.[24] The challenge is that there is no ideal scale and therefore, the choice of scale will largely depend on the purpose of the analysis and available data (MEA, 2003).[25] As ecosystem services can be generated at a variety of ecological scales, the same ecosystem services can be received at a variety of different institutional scales.

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[23] Our understanding of how anthropocentric actions will influence an ecosystem's threshold and resilience levels are limited at best and therefore ecosystem valuation cannot be precise. For instance, ecosystems are metastable – *i.e.,* ecosystems may exist in a number of states, all of which may be equal; however, each state may provide different service, have different resilience and threshold levels and react differently once a threshold is reached (Holling and Gunderson, 2002).

[24] For instance, Straton (2006) comments that the quality of an individual grain of sand would not be relevant to understanding the functionality or value of the ecosystem of which it exists. Ecological value emerges when an ecosystem is functional; economic value, depending upon the scale of the institution, would be realized when ecosystem services are utilized (Straton, 2006).

[25] Despite the fact that determining the scale of an ecosystem can be rather subjective, the MEA (2003) suggests that the boundary should be meaningful. For instance, if evaluating the ecosystem services provided by a freshwater ecosystem, the boundary should at minimum be the watershed or the watershed in which the river is situated. Furthermore, the MEA (2003) recommends that if there is a mismatch between the processes, structures or functions that provide an ecosystem service and the chosen scale of the study, the service should be omitted from the assessment and re-evaluated at a different scale.

Distinguishing the socio-economic system scale is important because it enables the ecosystem valuer to establish a hierarchy of institutions (MEA, 2003). The higher institutional scales would include the international, multilateral, national, provincial and municipal, whereas the lower scales would include the village, family and individual (MEA, 2003). Establishing both the socio-economic system and the scale of landscape functions is important as all types of institutions are influenced by ecosystem services.[26] Determining which ecosystem services impact each institutional level may help reveal the interests of various groups or individuals within and therefore provide insight into the roles and the values held for ecosystems by each institution (Pritchard *et al.,* 2000; MEA, 2003; Hein *et al.,* 2006). For instance, an individual may express personal values for a particular ecosystem; however, the same person in an advisory or decision-making role may express different values for the same ecosystem. At a higher institutional scale, such as the provincial, the valuation is likely to be different as responsibilities change. Temporal scales can also be a challenge to ecosystem valuers as individuals' preferences and decision criteria tend to change when scales of time are involved.[27] For example, short-term values may shift to bequest values in a long-term situation (Bingham *et al.,* 1995).

In addition to accounting for changes in social preferences, ecosystem valuers are also restricted by insufficient ecological knowledge of ecosystems and cannot predict which ecosystem services will be valuable in the future until the new information, and future context) is known (Limburg *et al.,* 2002). For instance, ecosystems that sequester carbon are now being recognized for the value of carbon credits although the ecosystem service has always been present. The scale at which economic activities influence ecosystem properties and functions is an important factor to consider. For instance, a small farm may have a minimal impact on the local ecosystem, whereas in contrast, large-scale industrial agricultural activities can significantly influence and alter the function of

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[26] Turner, Odum, Costanza and Springer (1988) propose a classification scheme that would utilize scale qualifiers such as local omni-directional (*e.g.,* pollination) and regional directional (*e.g.,* flood protection).

[27] Temporal scales suffer from intra- and inter-generational equity issues (*e.g.,* discounting). The purpose of addressing future generations is to develop policies and to discount appropriately to ensure that future generations share fairly and proportionately in the net benefits that accrue when resource stocks are depleted or degraded. See Chapter 2 for discussion.

ecosystems. Since the temporal and spatial scales of ecosystems as well as socio-economic systems vary considerably, the relevant scales at which each applies will have to be established on a case-by-case basis (MEA, 2003; de Groot and Hein, 2007). As it applies to decision-making, identification of ecosystem boundaries, institutions, ecosystem function and services, as well as use and non-use values becomes extremely important as uninformed decisions may cause irreversible changes to ecosystems, ultimately impacting human well-being (Costanza, 2003; MEA, 2003; Turner *et al.,* 2003; NRC, 2004). Figure 3 summarizes the complexities and inherent challenges in valuing multiple uses.

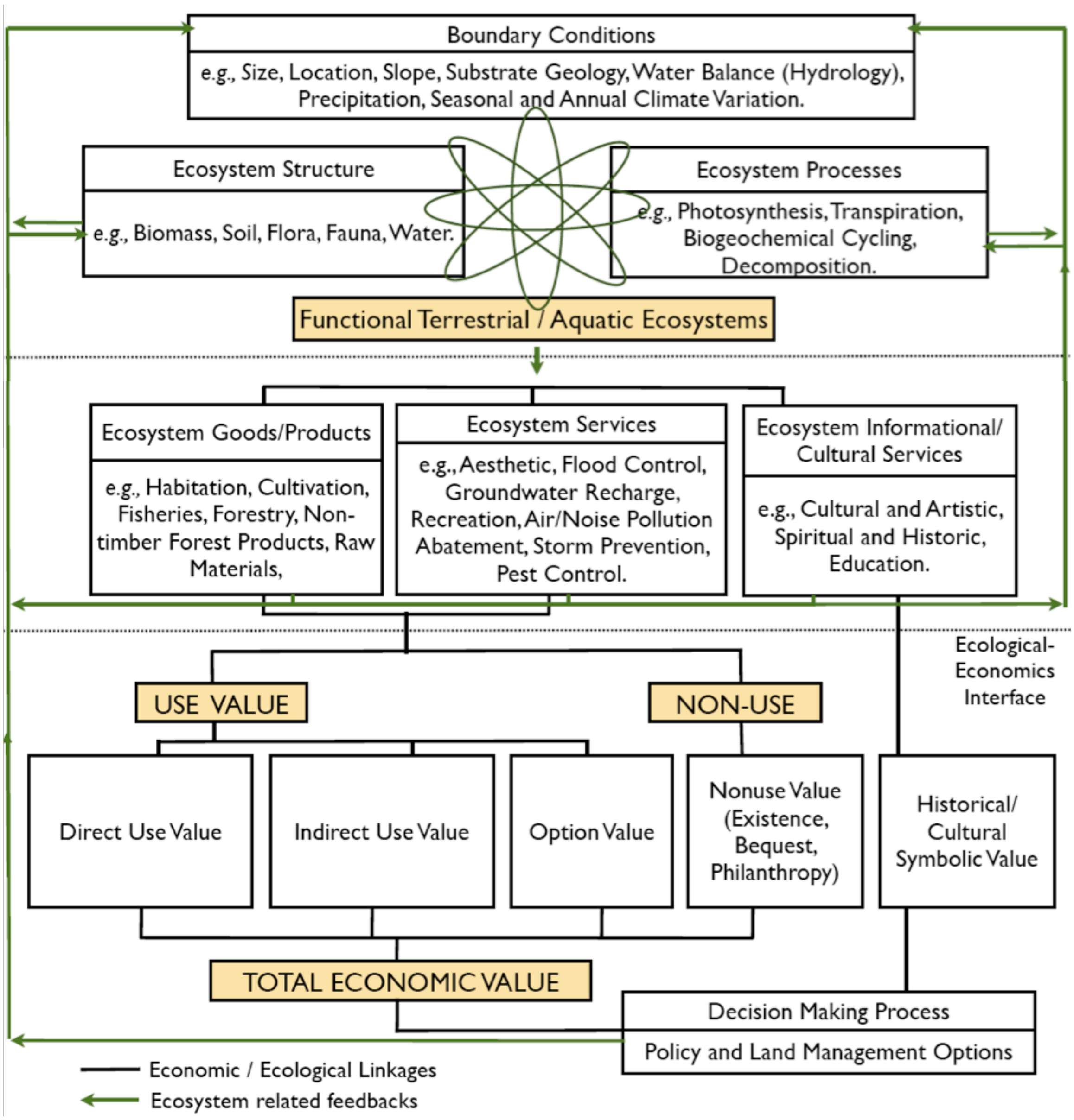

Figure 3. General framework for the analysis and valuation of ecosystem goods and services. *Note:* Based on Turner *et al.* (2003); NRC (2004); Farber *et al.* (2006); DEFRA (2007); de Groot and Hein (2007); and Fisher *et al.* (2008).

3.2.2 Ecosystem Valuation Methods

Use and non-use values are measured by a variety of valuation approaches (MEA, 2003). Specifically, there are two types of approaches to valuing ecosystems: physical and behavioral. Both the physical and behavioral approaches often undertake a dose-response approach to valuation of ecosystems. A dose-response approach typically attempts to identify relationships between physical changes within an ecosystem and the resulting change in value. For instance, using a dose-response approach, a researcher can estimate the cost of a policy change regarding fish licenses and fish populations. Valuation methods that utilize the physical valuation approach include the cost-based methods, the market price method, the productivity method, and the energetics (energy-based) method.

Behavioral methods are based on the assumption that an individual's preference for a particular ecosystem good or service can be observed through either direct interactions with markets (called the revealed preference method) or through the application of hypothetical markets (called the stated preference method). The revealed preference method can be further broken down into directly observed behavior (*e.g.,* number of visitations to a nature park) or indirectly observed behavior (*e.g.,* increased real-estate value due to greenspace views). For instance, a direct revealed preference method may calculate the value of an ecosystem service of a particular site using the number of visitors, distance travelled and time spent at the site. An indirect approach may use real estate prices to calculate the amenity value of a nearby ecosystem. Revealed preference methods utilize the following valuation techniques - travel cost, hedonic pricing and averting behavior methods. The stated preference method differs from the revealed preference method in that it depends upon hypothetical markets. This approach is hypothetical in nature as it measures what an individual might do, not what they may chose to do if the specific situation were to arise (MEA, 2003). For instance, a stated preference method would entail a survey asking respondents to state their willingness to pay (WTP) for an ecosystem service or their willingness to accept compensation (WTA) the loss of a service. Since the hypothetical market does not predict what would happen in real life, values are estimated using contingent, conjoint and group valuation methods. Each of the physical and behavioral methods is briefly summarized below.

Physical Ecosystem Valuation Method: Cost-based Approaches

The cost-based method originated from the opportunity cost concept developed in the late

1800s. Although utilized in cost-benefit analyses since the opportunity cost concept was developed, it is believed that the replacement and substitute cost methods began to be utilized in the environmental valuation discipline in the late 1960s when the discipline first emerged (Ropke, 2004). Cost based methods are based on estimating the cost (*i.e.,* a proxy) that would be necessary to replace an ecosystem service if it ceased to exist (*i.e.,* replacement cost) or the cost of providing substitute services (Bingham *et al.,* 1995). The value of the natural damage is based upon the restoration or replacement cost, other use values for which no market price exists and any other fees, charges, or payments that would have been collected by the government (King and Mazzotta, 2000). Since these approaches are based on actual market prices and estimates of cost, this method has become more popular over time.

The advantage of the cost-based approaches is that they estimate what it would actually cost to replace or substitute an ecosystem service using market prices. However, the drawback with the cost-based valuation method is that the methods rely on determining what attributes should be restored or replaced. This can be problematic for three reasons: first, it is technically impossible to restore an ecosystem to its original state[28]; second, the restoration effort may not be cost-effective; and third, it is assumed that human-made systems will provide the same quality and level of service to that which they are replacing (Bingham *et al.,* 1995; Chavas, 2000; Limburg *et al.,* 2002). Due to the complexity of the cost-based approaches, the valuation method tends to focus on a single service and overlook the functional condition of an ecosystem. According to the National Research Council (NRC) (2004), cost-based approaches should be considered the valuation methods of last resort and should only be employed when there are no reliable value estimates available.[29] However, if the cost-based method is to be employed, the following criteria are suggested: the alternative (or replacement system) provides the

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[28] Ecosystems cannot be restored in the same sense that furniture can be restored to an original pre-use condition. An ecosystem can be rehabilitated to a functional condition, but the physical state of the ecosystem will not be identical to the original condition.

[29] The National Research Council (2004) likens the replacement cost approach to the "cost of illness" approach in the health economics literature. These approaches are similar in that both add up the cost of treatment as the value of the benefit received; however, the main problem noted is that neither method accounts for individual or social preferences (NRC, 2004).

same service; the replacement/substitute system is the least-cost alternative; and that the replacement / substitution of an ecosystem service is demanded by stakeholders (NRC, 2004).

Physical Ecosystem Valuation Method: Market Price

The market price method is based upon Marshall's (1881) work of supply and demand equilibrium concepts. Often used to value changes in the quantity or quality of an ecosystem good/service, the market price method estimates the economic value of ecosystem services using consumer and producer surpluses of ecosystem good and services that are traded in commercial markets (King and Mazzotta, 2000). The strength of this method is that it uses price, quantity and cost data from established markets.

The main weakness of market price method is that the prices of the goods or services may not reflect the true economic value of the ecosystem good or service due to market failures. Market failures arise when there is no market for the ecosystem service provided. For example, property rights related to the ecosystem and associated services are not clearly defined; when ecosystem goods or services display characteristics of public or common goods; and when externalities are present (Fausold and Lilieholm, 1999; NRC, 2004; Great Britain H.M. Treasury, 2004; Schaeffer, 2008).[30] The market price method can be utilized to show how much an additional unit of an ecosystem good or service is worth, but the method is limited in that it cannot provide information on how much more or less of a resource users would receive if an ecosystem was in a functional condition or degraded.

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[30] According to Schaeffer (2008), market failures arise from incomplete property rights, which can occur when one or more of the following rights is missing: the right to exclude others from one's property, the right to transfer property either temporarily (*e.g.,* rent or lease) or permanently (*e.g.,* trade, sale, gift) and the right to garner all the benefits that accrue from the property. Incomplete property rights that lead to market failures tend to be associated with resources that exhibit characteristics of pure public or common goods (Milne, 1991). Pure public goods are non-rival (*i.e.,* the consumption of a resource by one individual does not prevent the consumption by another) and non-excludable (*i.e.,* the use of a resource by an individual does not preclude another's use) by nature. At the opposite extreme, pure private goods exhibit characteristics of excludability and rivalry. Common goods fall in between these two extremes (Milne, 1991). Externalities arise when there are no markets for the ecosystem goods/services – *i.e.,* there is no market to price the ecosystem goods/service – hence the development of non-market ecosystem valuation techniques (*e.g.,* contingent, conjoint, travel cost).

Physical Ecosystem Valuation Method: Productivity Method

First proposed by Wicksteed in 1894, then refined and termed the Cobb-Douglas production function in 1928, the productivity method is often utilized to estimate the value of ecosystem goods or services that contribute to the production of commercial goods (Garrod and Willis, 1999; NRC, 2004). The method is commonly used for scenarios where the resource in question is a perfect substitute for other inputs (*e.g.,* improved water quality results in lower treatment costs) and where producers of a marketable good benefit from changes in quantity or quality of the resource. Oftentimes, the focus of the studies that employ this method is on fish or clean water production (the intermediate inputs) that result in commercial fisheries values (the final output). For instance, the productivity method would attempt to value the clean water provisioning service of a wetland as it contributes to increased fisheries production downstream. Sometimes referred to as the net factor income approach, the derived value approach or the production function approach, the productivity method is advantageous as it can be easily applied and that it utilizes price, quantity and cost data from established markets (King and Mazzotta*,* 2000). The method is often used to value single services, such as fisheries linkages, flood control, storm protection, pollution abatement and water purification (NRC, 2004).

The method is limited in that it can only attribute value to goods or services that are used as inputs to the production of commercial goods/services (NRC, 2004). A major drawback to this method is that not all of the ecosystem regulation functions can be directly related to the service being valued (King and Mazzotta, 2000). Specifically, the focus of the method is only on one or more aspects of the production of an ecosystem service itself and not on the overall functional condition of an ecosystem. There is no explicit requirement that the functional condition of an ecosystem be measured or accounted for, rather the service in question is modeled under different scenarios with little regard to other ecosystem functions, services or functional condition. The reason for this is that the method is based upon the production function theorem of labour and capital. This is a theory “that describes the output as the product of a number of factors, some of which are principle unbounded” (Ludwig, 2000, p.32). This economic theorem holds that capital and labour can be interchanged without influencing the output. Applied to ecosystems, the theorem does not hold as it implies that ecosystem services can be interchanged or traded in the same linear fashion whilst still achieving the same output. For instance, applied to the water provisioning service of an ecosystem, this theory would assume that "any deficiency in water supply could be compensated by increasing solar input or the nutrient supply" (Ludwig, 2000, p.32). Such an assumption may not bode well for the function of an ecosystem as ecosystems are meta-stable and are subject to threshold changes. Therefore, managing for single ecosystem services can result in catastrophic changes to ecosystem function (Holling and Meffe, 1996). Consequently, this valuation method continues to receive considerable criticism from ecologists (Ludwig, 2000).

Physical Ecosystem Valuation Method: Energetics

Based on Lindeman's (1942) work on energy flows within ecosystems, Odum (1967) first assigned values to the energy flows that occur within ecosystems (Odum and Odum, 2000; Ropke, 2004). The fundamental premise behind Odum's work was that the production of ecosystem goods and services requires solar energy and therefore, the value should be based on the amount of energy required to produce them (Mcdonald and Patterson, 2007). Specifically, the energy value is expressed in units of embodied solar energy, which is then translated into a monetary value, which Odum (1983), called emergy. The advantage of this energetics method (sometimes referred to as the embodied energy approach) is that it concentrates on the relationships within natural systems and not on what human preferences would be for the system. Energy models have been used to value salt marshes, marine systems and aquatic systems.

The drawback with the energetics method is that it is extremely complicated, requiring considerable expertise to employ. Oftentimes, the challenge with these types of valuations is that they are not intuitive to the user and therefore are not accepted into the decision-making process (Fisher *et al.,* 2008; Schaeffer, 2008). Similarly, due to the nature of understanding required by both the valuer and the user, the method is often not applied. Finally, the emergy theory overlooks the varying roles and importance of regulating functions to maintain and sustain the function of an ecosystem. A similar method based upon the energetics methodology, called net primary production (NPP), has also emerged within the past two decades.

Ecological economists have begun to use net primary production (NPP) as a means to value ecosystems, specifically valuing the biodiversity[31] of an ecosystem (Costanza *et al.,* 2007). Net primary production (NPP) is the "accumulation of energy in plant biomass, or plant growth and reproduction" (Ricklefts, 1976, p.112). Those that utilize the NPP method argue that since these energy flows support both consumers and detritivores, "net primary production can be viewed as a flow that maintains the stock of natural capital that generates ecosystem services" (Richmond, Kaufmann and Myneni, 2007, p.455). To estimate the economic value, using conversion factors, the net primary productivity estimates are linked to carbon production or oil prices.

The strength of the NPP method is that it recognizes the importance of ecosystem function. Specifically, the model assumes that there is a positive relationship between both biodiversity and ecosystem functioning and biodiversity and net primary productivity (Costanza *et al.,* 2007). Many researchers argue that there is a positive correlation between the regulating functions of ecosystems and net primary production (Costanza *et al.,* 2007; Richmond *et al.,* 2007). However, most researchers measure the ecosystems structures or processes (*e.g.,* temperature, precipitation, soil moisture, species richness), but do not evaluate the functional condition of the ecosystem. Furthermore, most NPP analyses are performed using remote sensing techniques and aggregate data. Consequently, the measurement of ecosystem function is at best assumed as the link between biodiversity and ecosystem functioning and biodiversity and net primary productivity is based upon various in-house controlled experiments that result in correlations and not causal linkages (Costanza *et al.,* 2007; Vira and Adams, 2009). For instance, an ecosystem with a diverse amount of weeds may be considered to have a high

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[31] Biodiversity defined by Costanza *et al.* (2007) as "the variability among living organisms from all sources. This includes diversity within species, between species and of ecosystems" (p. 478).

level of biodiversity, but may not be in a functional condition. Finally, when using remote sensing, no on-site physical examination of ecosystem function occurs.[32]

Behavioral (Revealed) Ecosystem Valuation Method: Travel Cost

According to Garrod and Willis (1999), Harold Hotelling first conceptualized the travel cost method in 1947. However, it was not until the late 1950's that Trice and Wood (1958) and Clawson (1959) developed the methodology. Since then, travel cost methodologies have become extremely complex, utilizing statistical models. Travel cost studies attempt to value or infer the non-market value of an ecological service using the time/distance travelled and estimated trip costs an individual undertakes in order to visit a specific site (NRC, 2004). Travel cost studies have been used to value river fishing (Morey and Waldman, 1998); fishing and viewing wildlife in wetlands (Burt and Brewer, 1971; Creel and Loomis*,* 1992); swimming in lakes (Needleman and Kealy, 1995; Ward, Roach and Henderson, 1996); beach use (Haab and Hicks, 1997); boating on lakes (Siderelis, Brothers and Rea, 1995); and camping (Boxall, McFarlane and Gartrell, 1996). There are three types of travel cost methods available to researchers: zonal, individual and random utility models (RUM).

The zonal travel cost method is the cheapest and easiest to employ out of all the travel cost methods. The method estimates an aggregated value for an ecosystem using the number of visitors to the site, the distance traveled based on geographic zones and demographic data. The drawback with the approach is that it cannot easily value a change in the quality of the recreation of a site and the method would only be able to value a drastic change in ecosystem services, not a marginal change in ecosystem services (King and Mazzotta, 2000). The difference between the zonal and the travel cost method is that the individual travel cost method relies on survey data from individual visitors to the site and thus requires considerably far more preparation and data collection. However, the individual travel cost method would have more precise estimates than the zonal method (King and Mazzotta, 2000). A more intensive and subsequently expensive travel cost

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[32] Using remote sensing can be a challenge as an ecosystem may actually be a monoculture (*e.g.,* agriculture) and is managed using fertilizers and other inputs. Such as system would have a high NPP, but it is artificial.

study is the random utility travel cost model (RUM). Random utility models are empirical models that assume people know their preferences, but note that there are characteristics or preferences that are not accessible to the observer (NRC, 2004). Random utility models have an advantage over the travel and zonal cost calculation methods in that they can estimate the qualitative differences between various recreational sites (King and Mazzotta, 2000). The major drawback of all three travel cost methods is that none of these methods can assign values to features and functions that users of the site do not find valuable or are not aware of.

Behavioral (Revealed) Ecosystem Valuation Method: Hedonic Pricing

The hedonic model was based upon consumer theory developed by Lancaster in 1966; Rosgen (1974) provided the theoretical model on which the hedonic model is based (Milne, 1991; Garrod and Willis, 1999). Commonly used in ecosystem valuation studies, hedonic models compare how different characteristics influence the price of similar marketed goods (most commonly used is real-estate markets). The hedonic approach can either use a single-stage or a two-stage estimation process (Steinnes, 1992). The first stage model uses physical changes in the biological resource or ecological service to estimate the change in economic value. A second-stage model estimates the total economic cost of a total change in ecological services based on a first-stage model results.

The advantage of the hedonic model is that it utilizes data from actual market transactions (King and Mazzotta, 2000). However, hedonic models are limited in that they may not be effective in valuing an ecosystem change or the function condition of an ecosystem if the those who purchase the asset cannot physically see or are not aware of the functional condition of an ecosystem (Milne, 1991). Furthermore, the asset prices may be influenced by other variables the researcher is not aware of or has not controlled for. Other limitations of the model include: speculation error (researcher determines which variables should be included in the price function) and the missing variable bias (data is not available or does not exist) (Kulshreshtha and Gillies, 1993).[33] With regard to decision-making, the hedonic model is limited in that it can only provide a before and after analysis; the model cannot predict future environmental improvements.

Hedonic valuation models have been widely applied to determine the value of other ecosystem services, such as: proximity effects of aquatic systems (Mahan, Polasky and Adams, 2000; Paterson and Boyle, 2002; Wu, Adams and Plantinga, 2004; Netusil, 2005; Colby, 2002); view of a river/lake (Kulshreshtha and Gillies, 1993; Lansford and Jones, 1995); fragmentation of land uses and buffer width sizes around residential properties (Geoghegan, Wainger and Bockstael, 1997; Acharya and Bennett, 2001; Cho, Poudyal and Roberts, 2008); varying classes of wetlands (Doss and Taff, 1996; Anderson and Cordell, 1988; Mooney and Eisgruber, 2001); size and type of forests (Garrod and Willis, 1992; Mansfield *et al.,* 2005); and quality (Jaksch, 1972; Chattopadhyay, 1999; Zabel and Kiel, 2000).

Behavioral (Revealed) Ecosystem Valuation Method: Averting Behavior

The averting behavior method was first conceptualized in Zeckhauser and Fisher's (1976) unpublished paper, "*Averting behavior and external diseconomies"* (Smith and Desvousges, 1986). The averting behavior valuation method, referred to as the damage cost avoidance or the defensive expenditure method, has been less commonly used in the past, but is increasingly being used to evaluate the willingness to pay for improved health or the cost of avoiding undesirable health consequences (NRC, 2004). The averting behavior method estimates the value of ecosystem services by observing the costs people are paying to avoid future negative effects. Oftentimes, the method is used to value drinking water contamination (Harrington, Krupnick and Spofford, 1989; Abdalla, Roach and Epp, 1992).

This model is based upon the presumption that individuals will change their behavior to avoid an undesirable health outcome or threat and thus analyzes the changes in behavior

[33] An example of speculation error is Paterson and Boyle's (2002) study, who point out that "previous studies that simply measure proximity to and extent of environmental conditions may have produced different statistical results if visibility variables were included in the estimation" (p.424).

and the subsequent cost of substitution to infer the value of a service. The model is limited in that it can only provide *ex-post* analysis. Furthermore, for this method to be effective participants must - be aware of the problem; believe that the problem will adversely affect the health of themselves or a household member; recognize activities that a participant can undertake to avoid or reduce exposure to the problem; and, be able to make expenditures that result in adequate protection (NRC, 2004, p.89). Since the method measures individuals' responses to a problem, the measurement of ecosystem function is not required. In addition, due to the complexities and requirements involved, many of these studies are not commonly undertaken. Consequently, this method is quite limited in terms of which ecosystem services can be valued (NRC, 2004).

Behavioral (Stated) Ecosystem Valuation Method: Contingent Valuation

Ciriacy-Wantrup (1947) first proposed the contingent valuation method. According to Portney (1994), Ciriacy-Wantrup noted that some ecosystem services are public goods and hypothesized that one way to obtain information on the value of such services would be to ask individuals directly. It was not until 1963, however, that Robert Davis designed and implemented a contingent valuation survey (Portney, 1994). Contingent valuation is a stated preference method, as it is based on asking respondents questions regarding ecosystem services in order to determine, hypothetically, the respondents' willingness to pay (WTP) on a monetary basis to preserve (or restore) an ecosystem service, or determine what the respondents are willing to accept (WTA) as monetary compensation for the forgoing (or loss) of a service. Contingent valuation is often used to measure the value of an ecosystem service by asking respondents to state their value directly or to describe a range in which the value resides. It is a popular ecosystem valuation method as it can be used to value both use and non-use values.

Although extremely popular, there are many inherent challenges and limitations with the contingent valuation method. For instance, these studies are open to hypothetical bias where respondents are unfamiliar and inexperienced with identifying and determining dollar values for ecosystem services, strategic bias where respondents lie about their preferences, warm glow where respondents receive a warm-glow from expressing support for good causes, but do not feel the ecosystem good/service is important to them and zero-bid or protest response (where respondents are protesting some aspect of the study, or feel that the environment should not be valued (Milne, 1991; King and Mazzotta, 2000). Furthermore, individuals may express a reaction to an event, rather than evaluating the state of the ecosystem or ecosystem service (Boyle, Poe and Bergstrom, 1994; Diamond and Hausman, 1994). Often times, respondents do not take into account personal income restraints when providing an estimate of willingness to pay (Milne, 1991).[34] Finally, WTP and WTA studies are hypothetical in nature as they examine what a respondent might do and not what they actually do (Diamond and Hausman, 1994). Contingent valuation studies have been used to value wastewater, erosion control, natural purification of water, habitat for fish and wildlife (Loomis, 2000); wetland values (Woodward and Wui, 2001); restoration of marshes (Morrison, Bennett and Blamey, 1999); riparian restoration (Holmes, 2004); and watershed quality (Farber and Griner, 2000).

Behavioral (Stated) Ecosystem Valuation Method: Conjoint Valuation

Conjoint valuation is quite similar to contingent valuation in that is a stated preference method that utilizes hypothetical markets to determine what respondents are willing to pay (or accept for the loss) of a given ecosystem service. First conceptualized by Luce and Tukey (1964), conjoint analysis gained popularity in business and marketing (Green, Krieger and Wind, 2001). Its first known ecosystem valuation application was by Rae (1983) who valued air quality (NRC, 2004).

Conjoint valuation (also referred to as implicit valuation) differs from the contingent valuation method in its approach to ecosystem valuation. Specifically, conjoint valuation is based on asking respondents to choose or rank bundled attributes. However, the method differs from contingent valuation as it does not directly ask people to estimate their willingness to pay, but this is inferred from the chosen or ranked scenario by the respondent (King and Mazzotta, 2000; NRC, 2004). There are three different formats to

[34] WTP is technically limited by an individual's income and therefore they cannot be willing to pay more for an ecosystem service than they earn on a life-time income basis, even if they might wish to pay a greater amount.

which a conjoint valuation study can be carried out - contingent ranking (compare and rank alternatives), discrete choice (identification of most preferred choice) and paired rating (compare and rate alternatives in terms of value and preference) (NRC, 2004). The advantage of the method is that it can be used to value or prioritize a policy or action (albeit hypothetically) and due to its trade-offs structure, the method avoids some of the inherent problems with contingent valuation such as estimating values (King and Mazzotta, 2000). The method differs from the other behavioral valuation methods in that it can estimate the value of a functional ecosystem. However, the estimated value would be theoretical and would not be grounded in empirical data, as it is based upon respondents' preferences/choices in a hypothetical market. Furthermore, trade-offs between hypothetical situations increase decision-making complexity on the part of the respondent, which may result in frustration and loss of interest with the study they are participating in (NRC, 2004).

Behavioral (Stated) Ecosystem Valuation Method: Group Valuation

Within the past two to three decades, group valuation has been receiving attention from ecological economists since the method is based upon the principle that "public decision making should result, not from the aggregation of separately measured individual preferences, but from open public debate" (de Groot, Wilson and Boumans, 2002, p.404). The method is an attempt to address the social equity question of how ecosystem goods and services can be valued in such a fashion as to ensure fair treatment to competing social groups (Wilson and Howarth, 2002). The group valuation exercise utilizes contingent and conjoint valuation methods to determine the deliberated value. Group valuation differs from these methods in that the values are derived through group discussion and consensus building, not from interviewing and aggregating individual preferences. The method has been utilized primarily in land use planning (Wilson and Howarth, 2002).

The advantage of the group valuation method is that it can resolve conflict and build trust between participants and institutions, incorporate public knowledge and values into an environmental decision making process, improve understanding of environmental problems and act as a way to identify and engage fringe stakeholders (NRC, 2004; de Groot and Hein, 2007; Norton and Noonan, 2007). Pritchard *et al.* (2000) argue the process of dialog and deliberation makes a valuation exercise more objective. The group valuation approach is limited in that it can be difficult and costly to organize, it is subject to the same strengths and weaknesses of the valuation method undertaken as part of the approach and participants may be difficult to engage as they may have preconceived notions or wish to protect certain decision processes (NRC, 2004). Finally, groups are likely to have strong incentives to manage an ecosystem to maximize one of a few ecosystem services and pay little or no attention to how managing for a bundle of services will impact the functional condition of an ecosystem (Pagiola, 2008).

Ecosystem Valuation Method: Benefit Transfer

Benefit transfer is not considered a true ecosystem valuation method because it takes the estimated ecosystem values determined in other valuation studies and applies these values to the site under study, rather than developing estimates itself (MEA, 2003).[35]

The benefit transfer method has the advantage of being quicker and cheaper than undertaking any primary economic valuation research and is now commonly used to value ecosystems (Costanza *et al.,* 1997; EPA, 2000; MEA, 2003; NRC, 2004; DEFRA, 2007). As a result, the US Environmental Protection Agency has developed the only peer-reviewed guidelines for these types of analyses (EPA, 2000; NRC, 2004). Although benefit transfer is useful in the initial stages of planning and has been widely used in appraisal/compensation/liability cases in order to help estimate the magnitude of the value (or loss), it has been the subject of considerable controversy because it has been used inappropriately (MEA, 2005; DEFRA, 2007). If used correctly, benefit transfer can provide reliable and valid value estimates of services (MEA, 2003).

There are two ways to perform a benefit transfer - the transfer of mean site benefit measures or the transfer of the demand functions/WTP equations - both of which have

[35] It is not clear when or by whom the methodology was developed. Garrod and Willis (1999) comment that an early application (1970-80's) of the benefit transfer method was by the United States Forest Service.

their inherent drawbacks (Rosenberger and Loomis, 2001). For instance, Kirchhoff, Colby and LaFrance (1997) developed a methodology to evaluate the performance of direct benefit transfer and benefit function transfer, concluding that if benefit transfer is to occur, then benefit transfer function is a far better tool, as the former has an error around 40% even across seemingly similar ecosystem services. Desvousges, Naughton and Parsons (1992) noted some problems with implementing a benefit transfer stating that the data from existing studies - is often not available; have broad categories; do not establish between site characteristics and the value of improvements in a meaningful way; do not provide prices of substitutable items in a useable way; and, differ between user and non-user values. Loomis (1992) examined the error rates between transferring the average value of the benefit and transferring the function equation from a fishing recreation study site in Oregon with one in Washington. The authors concluded that transferring the demand equation resulted in an error of 5-15%, whereas transferring the mean site value resulted in errors of 5-40%. The EPA (2000) suggests that if a benefit transfer study is to be undertaken that a researcher should adhere to the following criteria - the studies that are transferred are based on adequate data, correct empirical technique, and sound economic method; the study contains information on statistical data between sites and the socioeconomic characteristics of the affected population; and that there is an adequate number of studies in order to enable the researcher to develop credible statistical inferences concerning the applicability of the transferred value.

3.3 Valuation Challenges

In summary, there is a variety of ecosystem valuation methods that can be applied to measure the varying use and or non-use economic value of ecosystem services (Table 2 summarizes each of the valuation methods). When calculated, use and non-use values can be aggregated to estimate the total economic value (TEV) of an ecosystem. However, it is important to note that the TEV typology does not capture the total value of an ecosystem since it does not value the overall function of the ecosystem. Rather, the typology values the "interdependent elements of ecosystem services" (Turner *et al.,* 2003, p.2). Thus, the aggregation of the values within the TEV typology results in the undervaluation of an ecosystem since the continued functioning of an ecosystem would be of more value than that of its aggregated services (Turner *et al.,* 2003; Kline, 2006). Consequently, even the most comprehensive valuation assessment of an ecosystem using current valuation methods cannot be presumed to have calculated the functional value of an ecosystem. At best, the assessment has only calculated the lower value bounds at a single point in time (Costanza *et al.,* 1997; NRC, 2004; Fisher *et al.,* 2008). Other researchers and reports have commented and criticized the ecosystem valuation discipline for this fundamental flaw as well (*e.g.,* Bingham *et al.,* 1995; King, 1997; EPA, 2000; Folke *et al.,* 2002; Limburg *et al.,* 2002; MEA, 2003; de Groot and Hein, 2007; and Mäler *et al.,* 2007).

The fundamental issue with the ecosystem valuation discipline is that there is a lack of current economic methods capable of generating precise value estimates because the valuation methods themselves do not require the measurement of ecosystem function (NRC, 2004). A review of each of the ecosystem valuation methods in Table 2 reveals that no method explicitly requires the measurement of ecosystem function, since all valuation "schemes" or methods make an implicit assumption about the functional condition of the ecosystem being valued (Limburg *et al.,* 2002; Fisher *et al.,* 2008). Thus, an ecosystem valuer, even if there is accessible ecological information, is not likely to account for the factors necessary to maintain ecosystem health or the inherent uncertainties such as thresholds (King, 1997; Toman, 2000; King and Wainger, 1999). Winkler (2006) comments on this issue explicitly:

> The strength of the economic valuation methods is that their concept of value incorporates the relationship between humankind and ecosystem products. However, the economic valuation methods also face severe difficulties. Often they do not adequately take account of the internal structure of ecosystems. Hence, they neglect the ecological interdependencies of different ecosystem entities. (p.84)

The United States EPA (2000) completed a comprehensive review of ecosystem valuation methods and concluded that the ecosystem valuation methods force a researcher to "value ecological service flows separately and then sum these estimates rather than constructing prices for changes in the structure and function of entire ecosystems" (EPA, 2000, p.111). Instead of focusing on the intercomplexities within ecosystems such as structures, processes, functions, resistance, resilience, recovery that produce the ecosystem services themselves, current economic valuation methodologies have focused simply on the services, which continue to result in simplified value conclusions that do not capture the "expanse, nuances and intricacies of the many ecosystem services" (Kumar and Kumar, 2008, p.808). Many researchers that have made reference to the importance of ecosystem function (oftentimes referred to as health, integrity, resilience and biodiversity) in their studies and have suggested that future research efforts need be focused on these areas. However, the task of determining how to incorporate the necessary ecological factors into the valuation exercise seems to exceed the scope of many papers. Many simply provide lists of areas within both economics and ecology that need to be addressed. Only a few researchers have attempted to address ecosystem function and valuation together; those who did have proposed and developed applications that utilize indicators (*e.g.,* King, 1997; King and Wainger, 1999; Hein *et al.,* 2006; de Groot and Hein, 2007) or complex mathematical models (*e.g.,* Mäler *et al.,* 2007).

It is clear from the preceding discussion that current ecosystem valuation methods do not properly or comprehensively account for the functional state of an ecosystem under study, because there is no explicit requirement to do so. Current valuation methods simply provide an estimate of an ecosystem service at a point in time. That is, the estimated ecosystem service values do not provide the critical information that decision makers may require. For instance, simply calculating a value for an ecosystem for conservation purposes, without any context to the function of an ecosystem and landscape context, is likely to warrant the same level of conservation amongst various other ecosystems being managed. However, some ecosystems may be at risk and require immediate attention and greater resources than others if functional changes (thresholds) are to be avoided. Without assessing the functional condition of an ecosystem, the past changes in the functional condition, its current state and trajectory, the quality of the ecosystem services and the urgency to which these conditions need to be addressed all remain unknown (Toman, 1998; Straton, 2006). Because current ecosystem valuation methods implicitly assume the functional condition of an ecosystem, the estimated values have not been adjusted for the risk that the ecosystem is damaged or at-risk.

Turner *et al.* (2003) explain that the inability to incorporate ecosystem function into valuation studies is due to the complexity of nature itself and "that reliable estimates of all the services cannot be made" (p.7). The reason, according to researchers, is that there is insufficient ecological knowledge on ecosystems and on how socio-economic systems will affect ecosystem function (Bingham *et al.,* 1995; MEA, 2003; NRC, 2004; de Groot and Hein, 2007). There is also a fundamental uncertainty regarding the minimum level of ecosystem function (Fisher *et al.,* 2008), thus making it difficult or impossible to identify how ecosystem changes will impact socio-economic systems and *vice versa* (Bingham *et al.,* 1995; Toman, 2000).[36] The cost of undertaking ecosystem studies, as well as issues of double counting have been cited as an impediment to effective valuations (King, 1997; NRC, 2004; Fisher *et al.,* 2008). The issue of semantics is also a barrier to improving ecosystem valuation methods and estimates (Limburg *et al.,* 2002). For instance, the terms "ecosystem services", "intermediate services", "ecosystem functions", "ecosystem values", "ecosystem benefits" often times relate to the same thing. However, these words have different meanings depending on the context, the discipline and the perspective of the researcher involved and may result in varying value estimates for the same ecosystem depending upon the terminology being used (Norton and Noonan 2007). Therefore, the reliability and accuracy of the ecosystem value estimates are unknown. Finally,

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[36] Some researchers are now arguing that ecosystems "at-risk" cannot be effectively valued using conventional economic methods (*e.g.,* Prichard *et al.,* 2000; Limburg *et al.,* 2002). Examination of the reasoning behind this argument uncovers some implicit assumptions within the ecosystem valuation literature. For instance, Limburg *et al.* (2002) propose a continuum of ecological/economic conditions. At one end of the continuum is the marginal regime (*e.g.,* a small change in structure or function does not cause an ecosystem to change states) where there is a high degree of certainty that the ecosystem is stable, at the other extreme is the non-marginal regime where an ecosystem, under the right conditions, may change ecosystem states (Fisher *et al.,* 2008). The authors contend that the former can be economically valued using the valuation methods described previously; however, the non-marginal regime cannot be "easily" valued as the "assumptions about stability, substitutability, stable equilibrium, *etc*." no longer hold and marginal changes are not easy to define in ecosystems (Limburg *et al.,* 2002, p.417). For those ecosystems that exist in the non-marginal regime or spectrum, the development or application of an index that measures the various functional states and trajectories of ecosystems is recommended.

perspectives of ecologists and economists are often times cited as a limitation to effective valuations. Norton and Noonan (2007), for instance, are rather blunt on the issue stating that "ecologists still think like ecologists and economists still think like economists" (p.664). The authors argue that the ecologist's perspective lacks the human component, whilst economists continue to ignore the biophysical processes and structures that when in balance with the landscape result in functional ecosystems and provide ecosystem goods and services. The presence of thresholds within ecosystems should suggest to both economists and ecologists that greater collaboration is required.

Based upon the aforementioned reasons, according to Norton and Noonan (2007), a new valuation method is required. It must be one that builds on the existing strength of ecosystem valuation methods, that incorporates knowledge on thresholds and various properties of ecosystems and integrates the ecological understanding that functional ecosystems provide ecosystem goods and services that contribute to economic value. The challenge is that there has been considerable inaction in the development of new valuation methods; rather, the focus of the valuation discipline has been on improving existing methods (Norton and Noonan, 2007). It is clear from the preceding discussion that current ecosystem valuation methods do not properly or comprehensively account for the functional state of an ecosystem under study and thus, a new question is raised: is the estimation of ecosystem service values an accurate reflection of ecosystem function?

Table 2. Summary of ecosystem valuation methods

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Cost-Based Approaches: Replacement /Substitute[1]	Physical (1)	- Methods may provide a rough indicator of economic value. - It is easier to measure the cost of producing benefits (*e.g.,* estimating the cost of building a water filtration facility) than estimating non-use or indirect values.	- Method may undervalue or omit important ecosystem functions. - There tends to be a focus on the valuation of a single or few ecosystem services at a macro scale (*e.g.,* province/state or national scale) and overlooks the function of an ecosystem - Substitute method undervalues the ecosystem as the capital replacement (*e.g.,* drinking water treatment facility) only replaces one ecosystem service. Substitute goods are unlikely to provide the same level of quality and type of good. - Costs are not an accurate measure of benefits (*e.g.,* methods do not consider individual or social preferences) and as a result are likely underestimates of value. - It is technically impossible to restore an ecosystem to its original state and that the restoration cost may not be cost-effective.	1. Assess the ecosystem service(s) being provided (*e.g.,* how they are being provided, to what level) and to whom. 2. Identify the least costly alternative means of providing the service(s). 3. Calculate the cost of the substitute or replacement service(s). 4. Gather evidence that the public would be willing to accept the substitute or replacement service(s) in place of the ecosystem service(s).	No
Market Price Method[1]	Physical (1,2,3)	- Price, quantity and cost data are available due to established markets (*e.g.,* fish, water, timber). - The method uses	- Method may undervalue or omit important ecosystem functions. - Market data is only available for a limited number of ecosystem goods and services. - The true economic value of an	1. Use market data to estimate the market demand function 2. Estimate consumer surplus pre- and post- impact/decision. Calculate change in consumer surplus.	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Market Price Method (con't)		standard, accepted economic techniques.	ecosystem good/service may not be accurately represented in the markets due to externalities, embedded inputs, policy failures and seasonal variations. - The method is limited in that it cannot provide information on how much more or less of a resource users would receive if an ecosystem was functional or degraded.	3. Estimate producer surplus pre- and post- impact/decision. Calculate change in producer surplus. 4. Calculate total economic change – *i.e.,* the sum of the change in consumer and producer surplus.	
Productivity Method[1,2]	Physical (2)	- Price, quantity and cost data are available due to established markets. - Method is relatively straightforward to apply. - Time frame to complete study: 2-4 Months.	- Method may undervalue or omit important ecosystem regulation functions (as well as ecosystem function). - Method is limited to valuing the resources that can be used as inputs to the production of marketed goods and services. - Not all ecosystem functions are directly related to the services that are related to the production of marketed goods. - Requires advanced knowledge of production theory and econometric methods; working knowledge of renewable resource or engineering models. - Cost $30-50,000.	1. Determine the physical effects of the environmental change and the identification of external factors using empirical estimates on known production effects, research-based field trials or laboratory experiments. 2. Develop a production-function – *i.e.,* a functional relationship between the inputs and the outputs. 3. Estimate the costs of the inputs given a change in the change of an ecosystem service (*e.g.,* cost of purification to reduced water quality). 4. Estimate the benefits of alternate options (*e.g.,* protecting the resource). 5. Compare the benefits against the costs.	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Energetics: Emergy /Net Primary Productivity[5]	Physical (1,2)	- Method concentrates on the relationships within natural systems rather than depending on what human preferences may be.	- Emergy method overlooks the roles and importance of regulating functions to maintain and sustain the function of an ecosystem - NPP method is designed to value biodiversity using correlations between the regulating functions of ecosystems and net primary production. However, most studies are reductionistic in that only a few ecosystem processes or structures are used to correlate between biodiversity and NPP. There is no evaluation of the function of an ecosystem as a whole. - Methods do not reflect the value of human preferences. - Methods are extremely complicated (expertise in energy modeling and ecological principles) and can be difficult to employ and as a result are commonly used. - Emergy valuations are often not intuitive to users and therefore may not be accepted into the decision-making process.	The following is a description of how to complete an emergy analysis (performing a net primary productivity analysis would be based upon the same principles as the emergy method, but usually utilizes remote sensing): 1. Define system and temporal boundary. 2. Identify all major energy sources and material resources flowing into and stored within the system boundary. 3. Diagram these systems using energy systems language. 4. Estimate energy sources and material resources by direct measurement, or deriving estimates from production/financial records and other relevant sources (*e.g.,* weather data). 4. Convert energy quantities into energy units (Joules), mass units (grams) or monetary units (US Dollars).	No
Travel Cost: Zonal /Individual[1,2]	Behavioral: Revealed (1)	- Method is one of the most widely used for valuing site-specific recreation benefits.	- Method may undervalue or omit important ecosystem functions. - Method relies on a subjective decision by the researcher to include or exclude specific elements (*e.g.,* study	**Zonal Travel Cost Approach:** 1. Identify site and collect data from visitors relating to their points of origin and the number of visits to a sire within a specified time	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Travel Cost: Zonal /Individual (con't)			difficulty in estimating the value of time - *i.e.,* opportunity cost of time *versus* leisure time). - Method cannot value quality improvements or ecosystem function at a site, nor can it differentiate the effects of substitute sites. - The availability of substitute sites will influence the estimated values. - Method cannot assign values to features and functions that users of the site do not find valuable. - Method has difficulty establishing value when users visit multiple sites or when multiple day trips are involved. - Values estimated by the method are fixed for any given site at a specific point in time and cannot be identified statistically. - Cost range $50-150,000 (lower end). - Time frame to complete study 0.5-2 years.	period. 2. Define zones of origin and allocate visitors to the appropriate zone. 3. Calculate zonal visits per household to the site and average travel cost from each zone to the site. 4. Use census data to derive variables relating to zonal socio-economic characteristics. 5. Estimate trip generating function. 6. Derive demand curve and obtain zonal household consumer surplus estimates. 7. Calculate aggregated zonal consumer surplus. 8. Aggregate zonal consumer surplus estimates to obtain an estimate of total consumer surplus. **Individual Travel Cost Approach:** 1. Identify site and use a questionnaire survey to collect data from visitors relating to cost of travel to the site, the number of visits to the site, recreational preferences, socio-economic characteristics, *etc*.	

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Travel Cost: Zonal /Individual (con't)				2. Identify travel cost model. 3. Derive demand curve and obtain household consumer surplus estimates. 4. Calculate aggregate consumer surplus for the site.	
Travel Cost: - Random Utility Model[1,2]	Behavioral: Revealed (1,2)	- Method enables researcher to determine how the qualitative difference of ecological attributes between different sites influence a users decision (and value) to select a particular site.	- Method may undervalue or omit important ecosystem functions. - Method relies on a subjective decision by the researcher to include or exclude specific elements (*e.g.,* difficulty in estimating the value of time - *i.e.,* opportunity cost of time versus leisure time). - Method has trouble differentiating the influence of substitute sites and to the degree of which ecosystem services are correlated (*e.g.,* fish versus scenery). - Lack of consistent data between site attributes (*e.g.,* comparative problems). - Method runs into difficulty when users visit multiple sites or when multiple day trips are involved. - Method is more expensive than the traditional travel cost method as it has greater data and expertise requirements - *e.g.,* advanced knowledge of demand theory, statistics and econometrics, survey design and sampling	1. Identify all possible sites that a visitor might select, their quality characteristics, and the travel costs to each site and use a questionnaire survey to collect data from visitors relating to cost of travel to the site, the number of visits to the site, recreational preferences, socio-economic characteristics, ecosystem services, etc. 2. Use econometric models to derive demand curve for site-specific characteristics/attributes. 3. Calculate aggregate consumer surplus for site-specific characteristics/attributes.	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Travel Cost: - Random Utility Model (con't)			procedures. - Cost range $50-150,000 (higher end). - Time frame to complete study 1-2 years.		
Hedonic Pricing[1]	Behavioral: Revealed (1,2)	- Widely used method. - Method can be used to estimate ecosystem values based upon actual choices. - Data is normally quite reliable and timely (property markets are relatively efficient in responding to new information and are updated yearly). - Time frame to complete study: 4-6 Months.	- Method may undervalue or omit important ecosystem functions. - Method requires researcher to control for all other available attributes (researcher may be limited by data availability). - Method assumes that the people are aware of the environmental difference or attribute (*e.g.,* ecosystem function) being measured and have access to the same information. - Method only evaluates before and after situations (*i.e.,* it cannot predict future values if ecosystem function were to change). - Asset markets are influenced by external factors that may not be accounted for (*e.g.,* taxes, interest rates, economic cycles). - Requires specialized skills in statistics (*e.g.,* advanced knowledge of demand theory, statistics and econometrics; skilled data manager) and can be costly ($30-50,000). - Method requires the collection and processing of large amounts of data.	1. Identify the good or service to be investigated. 2. Collect data on residential property sales within a designated area of study and within a specific time period. Data should include: selling prices and locations of residential properties, property characteristics (*e.g.,* lot size, number and size of rooms, and number of bathrooms, garage), neighbourhood characteristics (*e.g.,* property taxes, crime rates, and quality of schools), accessibility characteristics (*e.g.,* distances to work and shopping centers, and availability of public transportation) and environmental characteristics (*e.g.,* distance to green space, type of ecosystem, air pollution, *etc*.). 3. Statistically estimate a function that relates property values to the property characteristics. 4. Analyze data using regression analysis. 5. Assess the validity and reliability	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Hedonic Pricing (con't)				of the exercise.	
Averting Behavior[1]	Behavioral: Revealed (1,2)	- A common approach to valuing drinking water.	- Method omits important ecosystem functions - there is no requirement to measure the function of the ecosystem since the approach values a behavioral response to a situation. - To be effective participants must be educated and aware of the problem (and associated risk) as well as have available options to make different choices. - Method tends to only value a single ecosystem service and cannot value many ecosystem services (commonly used to value drinking water quality). - Method can be extremely expensive and is subject to data limitations.	1. Assess the ecosystem service(s) being provided and to whom. 2. Use a questionnaire survey to collect data from those impacted by the damage. 3. Estimate potential physical damage to property over a period of time. 4. Calculate actual dollar amount of damage or the amount people spent to avoid damages.	No
Contingent Valuation[1]	Behavioral: Stated (1,2,3,4)	- Most widely applied method due to its versatility (*e.g.,* the method can be used to measure both use and non-use values. - Researchers continue to refine and improve the method to improve results (*e.g.,* validity and	- Method may undervalue or omit important ecosystem functions. - Method is prone to hypothetical bias, strategic bias, warm glow and zero-bid/protest response. - There is considerable controversy with the method as there is no actual transaction. - Method relies on consumer preferences. - Payment questions can influence the	1. Set up the contingent valuation or hypothetical market. 2. Obtain WTP or WTA amounts. 3. Estimate mean and median WTP and/or WTA amounts. 4. Aggregate the WTP or WTA amounts. 5. Assess the validity of the CV exercise.	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Contingent Valuation (con't)		reliability). - Method is not difficult to employ and is not difficult to analyze and describe.	value (*e.g.,* What are you willing to pay for a service? *vs.* What are you willing to accept for the loss of the service?). - Method is subject to the embedding effect (*i.e.,* respondents provide similar value estimates for a single service and for a suite of assets). - Estimates of non-use values are difficult to validate. - Method is expensive ($50-100,000) and time-consuming (6-12 months) due to pre-testing, survey work and analysis. - Value estimates are often not grounded in scientific or market-based data. - Requires advanced skills in survey design, sampling procedures, data management, statistics and econometrics.		
Conjoint Valuation[1]	Behavioral: Stated (1,2,3,4)	- Method can be used to value the outcomes of an action or policy decision. - Method enables respondents to think in terms of trade-offs rather than estimating dollar values.	- Method may undervalue or omit important ecosystem functions. - Respondents may find some of the tradeoffs difficult to evaluate resulting in frustration or loss of interest due to unfamiliarity with ecosystem goods and services and if the choices (*i.e.,* bundles) are too complex. - There is considerable controversy	1. Identify the good or service to be investigated. 2. Develop a suitable experimental design for bid amounts. 3. Design questionnaire survey and incorporate referendum CVM questions. 4. Validate bids. 5. Develop an appropriate model of	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Conjoint Valuation (con't)		- Method can determine both values and priorities of respondents in regards to the environment (*e.g.,* rank scenarios) - Method minimizes response biases that can occur in Contingent valuation methods.	surrounding the method, as it does not represent an actual transaction. - Surveys may be too limited and force respondents to make choices that they would not normally choose. - The validity and reliability of the method for valuing non-market ecosystem goods/services is largely untested. - Method is expensive ($50-100,000) and time-consuming (6-12 months) due to pre-testing, survey work and analysis. - Estimates are often not grounded in scientific or market-based data. - Requires advanced skills in survey design, sampling procedures, and data management; advanced knowledge of demand theory, statistics and econometrics.	willingness to pay. 6. Estimate welfare measures based on willingness to pay.	
Group Valuation[3]	Behavioral: Stated (1,2,3,4)	- Method can better incorporate public knowledge and values into an environmental decision making process. - Method can resolve conflict and build trust between participants and	- Method may undervalue or omit important ecosystem functions. - Group valuation can be difficult to organize. - Participants may be difficult to engage especially when they have preconceived expectations or when they wish to protect certain decision processes. - Group valuation is subject to the	The focus of the method is on the social values for ecosystem services given a specific scenario arising from policy or land-use change. The value of an ecosystem service or bundle of services is not determined from an aggregation of individual preferences; rather, the value is derived from the deliberation process. However, the	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Group Valuation (con't)		institutions. - Method can improve participant understanding of environmental problems. - Method can be used to help identity and engage fringe stakeholders.	same strengths and weaknesses of the valuation method undertaken (*e.g.,* contingent or conjoint). - Method can suffer from a form of group think - *i.e.,* the group only focuses on a subset of benefits the ecosystem provides. This is likely to result in mismanagement of the ecosystem under study.	application of contingent or conjoint valuation does not change.	
Benefit Transfer[1,4]	N/A (1,2,3,4)	- Does not take much time to complete (1-3 months) and can be a cost-effective ($10-20,000) approach. - Method can be used a screening tool to determine if a more in-depth valuation study is required.	- Method may undervalue or omit important ecosystem functions. - Method is controversial as it is often not considered a valuation method and has been misused. - The data surrounding the estimates from the study site may be inadequate or inaccessible. This limits the ability of the researcher to make the needed adjustments to transfer the values from the study site to the policy site. - Most study sites do not establish between site characteristics and the value of improvements in a meaningful way. - Study sites do not provide prices of substitutable items in a useable way. - Studies differ between user and non-user values. - Adequacy of existing studies (study	1. Describe the policy case so that its characteristics and consequences are understood. 2. Identify existing studies or values that can be used for the transfer. 3. Determine whether the existing values are transferable – *i.e.,* evaluate values from study sites against a criteria (*e.g.,* the studies that are transferred are based on adequate data, correct empirical technique, and sound economic method; the study contains information on statistical data between sites and the socioeconomic characteristics of the affected population, and; there is an adequate number of studies in order to enable the researcher to	No

Ecosystem Valuation Method	Physical / Behavioral Approach	Benefits of Approach	Drawbacks with approach	Valuation Procedure	Measure Ecosystem Function?
Benefit Transfer (con't)			sites) may be difficult to assess. - Benefit transfers can only be as accurate as the initial value estimate completed at the study site. - Study site estimates can become dated quite quickly.	develop credible statistical inferences concerning the applicability of the transferred value). 4. Adjust the study site values to reflect the policy site characteristics.	

Legend: 1-Direct-use Value; 2-Indirect-use Value; 3-Option Value; 4-Nonuse Value.

Note: [1] Valuation procedure description based upon Pearce (1993); King and Mazzotta (2000); EPA (2000); MEA (2003); NRC (2004); and DEFRA (2007). [2] Valuation procedure description from Garrod and Willis (1999). [3] Valuation procedure description based upon Wilson and Howarth (2002); and Splash (2007). [4] Valuation procedure from Rosenberger and Loomis (2001). [5] Valuation procedure description based upon Odum and Odum (2000); Costanza *et al.* (2007); and Richmond *et al.* (2007). The remainder of this table is based upon the works of: Milne (1991); Pearce (1993); Bingham *et al.* (1995); Apogee Research (1996); Turner *et al.* (1988); Garrod and Willis (1999); EPA (2000); King and Mazzotta (2000); Rosenberger and Loomis (2001); MEA (2003); NRC (2004); Winkler (2006); de Groot and Hein (2007); and Splash (2007).

Chapter 4.

4.1 Introduction

Chapter 3 reviewed and evaluated the fundamental basics of ecosystem valuation. It revealed that current ecosystem valuation methods do not explicitly require the measurement of the functional condition of an ecosystem. As a result, most ecosystem valuers continue to neglect accounting for the functional condition of an ecosystem - a critical issue that ensures the continued provision of ecosystem services. Because current ecosystem valuation methods do not properly or comprehensively account for the functional condition of an ecosystem under study, a new question was raised: Is the estimation of ecosystem service values an accurate reflection of ecosystem function?

In order to address this question, an ecosystem valuation study completed by D. Hegg and L. Townsend for the Swan Lake watershed is utilized. The intent of using the case study is to illustrate the point that when the functional condition of an ecosystem is not assessed during the valuation exercise, the estimated ecosystem service values provides little, if any, information on the functional condition of the ecosystem that generated them. It is also contended that ecological evaluations which assess the ecological factors, tend to overlook and do not assess the value of the ecosystem services provided. Thus, in the context of urban planning, an ecosystem valuation or an ecosystem evaluation assessment, applied alone can only provide decision makers with only half of the necessary information. That is, the estimated ecosystem service values do not provide the critical information that decision makers may requires. The past changes in the functional condition of an ecosystem, its current state and trajectory, the quality of the ecosystem services and the urgency to which these conditions need to be addressed all remain unknown. Using the Swan Lake case study, it is argued that the ecosystem valuation discipline needs to begin assessing the functional value of an ecosystem, not simply the value of the ecosystem services. Due to the inherent challenges in achieving such a recommendation, the integration of ecosystem valuation and ecosystem evaluation assessments is recommended.

4.2 Description of Swan Lake Watershed

Located on southern Vancouver Island, British Columbia, the Swan Lake watershed is a watershed of the 46,000 hectare (ha) Colquitz River Watershed (Figure 4). The Swan Lake watershed drains into the Colquitz River (Figure 5). The 1,200 ha watershed consists of a variety of land-uses including both residential and commercial sectors, agricultural land and contains a 48.06 ha Swan Lake/Christmas Hill Nature Sanctuary, that is widely recognized for its habitat, recreational and educational value (Buchanan *et al.,* 2007). The Nature Sanctuary consists of protected land consisting of a mix of riparian and upland species surrounding a 9.2 ha eutrophic lake (Townsend, 2009) (Figure 6).

4.2.1 Functional Condition of the Swan Lake Watershed

Over the past 200 years, the Swan Lake watershed has undergone many changes in land use (Figure 7). Specifically, the watershed has been hydrologically modified by agricultural activities, including 40 plus years of point source pollution by two wineries, sewage discharges, urbanization, and the construction of a channel connecting Swan Lake and Blenkinsop Lake (Edmonds, 2002; Buchanan *et al.,* 2007). The outcome of these past activities has been a loss of functional condition and/or degradation of both terrestrial and freshwater ecosystems which has resulted in a loss of valuable ecosystem services (Edmonds, 2002; Malmkvist, 2002).

According to Townsend (2009), who assessed the health and resilience of the Swan Lake watershed, the watershed is in poor health and its resilience to disturbance is low. Because land-use change alters the natural hydrology of a watershed, the functional condition of the streams within a watershed can decline quite quickly. For instance, the Center for Watershed Protection (2003) states that most stream quality indicators begin to decline when watershed imperviousness reaches 10% with severe degradation occurring when watershed imperviousness extends beyond 25%. In the case of the Swan Lake watershed, 25% of the watershed consists of impervious surfaces (Townsend, 2009). At a finer scale, Buchanan *et al.* (2007) undertook field assessments using the Proper Functioning Condition (PFC) assessment of all of the streams concluding that approximately 35% of Swan Creek was rated to be in a properly functioning condition (PFC), 15% as functional at risk and 50% as non-functional (see Appendix C for description of the PFC assessment). Furthermore, 23% of Blenkinsop Creek was rated to be in a functional condition, 11% was rated as functional-at-risk and 66% was rated as non-functional. Both Blenkinsop and Swan Creek suffer from poor water quality due to continued non-point source pollution arising from agricultural activities and stormwater discharge (Barraclough and Hegg, 2008b). As a consequence of historical point source and ongoing non-point source pollution, both Blenkinsop and Swan Lake are eutrophic.[37] Reasons cited for the poor functional ratings of the aquatic ecosystems was that many of the creeks within the watershed have been subject to enclosure in pipes, channel excavation and straightening, encroached riparian zones, introduction and proliferation of invasive species and increased volume and frequency of flows due to stormwater discharges and increased impervious area within the watershed (Buchanan *et al.*, 2007).

4.3 Objective of Swan Lake Watershed Valuation Study

Swan Lake watershed was selected as a candidate for a valuation study for three reasons. First, it is representative of the effects that urbanization and land-use activities can have on ecosystems over time; specifically, the degradation of fresh water ecosystems. Second, there is readily available historic and current information on the functional condition of the ecosystems within the Swan Lake Watershed.[38] And third, within the past decade, various stream rehabilitation projects were undertaken within the watershed (*e.g.,* Malmkvist, 2002; Edmonds, 2002), many of which have been shown to have significant

[37] Both lakes were rated to be in PFC. However, the PFC system does not assess the "subtle processes such as shifts in nutrient regime and biological characteristics, which are factors of concern in this system" (Buchanan *et al.,* 2007, p. 70). The lakes were largely rated to be in PFC based upon the fact that the integrated assessment team felt that both lakes had the capacity to withstand wind and wave energy. However, Buchanan *et al.* (2007) state that although both lakes are in PFC, they are far from their potential and desired future condition.

[38] Lack of ecological information (either by access or lack of funds) is a commonly cited barrier to effective ecosystem valuations. The freshwater ecosystems within the Colquitz Watershed have been assessed by Aqua-Tex Scientific Consulting Ltd. in 2007 using the Proper Functioning Condition Assessment in which the author participated as a member of the integrated team (Buchanan *et al.,* 2007). Townsend (2009) has also undertaken various vegetative and hydraulic studies of the Swan Lake Watershed as a part of her thesis to assess the overall resilience of the sub-watershed.

financial benefit to the developer, community and municipality when stream restoration objectives are tied into urban planning and development (Barraclough and Hegg, 2008a).

The objective of the Swan Lake watershed valuation study was to assess and highlight the value of the ecosystem services provided by urban vegetation. Due to the size of the watershed itself and the limited resources and time available, the study was undertaken at two different scales. The first was a broad analysis of the watershed using GIS tools (ArcGIS and CITYgreen) and the application of a market-based ecosystem valuation method. This provided a high level, less rigorous estimate of the value of the energy dissipation and the value of carbon storage and sequestration. A subsequent, more fine-grained analysis was undertaken on the Swan Lake/Christmas Hill Nature Sanctuary. The Nature Sanctuary analysis involved field work utilizing the Urban Forest Effects Model (UFORE) and application of three ecosystem valuation methods: market-based, cost-based and the benefit transfer method. The ecosystem services valued in the Swan Lake/Christmas Hill Nature Sanctuary analysis included energy dissipation, carbon storage and sequestration, air pollution abatement, flood mitigation and general recreation. Because ecosystems provide a myriad of services, only a few ecosystem services were selected and valued. Selection of the ecosystem services to be valued was based upon identifying which ecosystem services would be of interest to urban planners/decision-makers well as identifying the services that a community would consider valuable. The valuation study completed by Townsend and Hegg (in progress) concluded that ecosystem valuation could be a means to facilitate more informed decisions in land-use planning and be used to provide financial arguments for the preservation and restoration ecosystems.

4.3.1 Methods

In the Swan Lake watershed valuation study, field investigations using GIS remote sensing tools, an urban forestry planning model and data collected by Townsend were used to estimate the value of the ecosystem services of the Swan Lake watershed and the Christmas Hill/Swan Lake Nature Sanctuary. Both the UFORE and CITYgreen models were identified as the most commonly applied and available urban forestry planning tools that identify the economic value of vegetation in urban areas. Market price and cost-based ecosystem valuation methods were employed to estimate the value of energy dissipation and flood abatement, respectively. Benefit transfer, a commonly applied ecosystem valuation method, was also employed to estimate the value of general recreation. Description of the UFORE and CITYgreen models, field work and mapping activities as well as the application of the three ecosystem valuation methods are fully described in Appendices A and B.

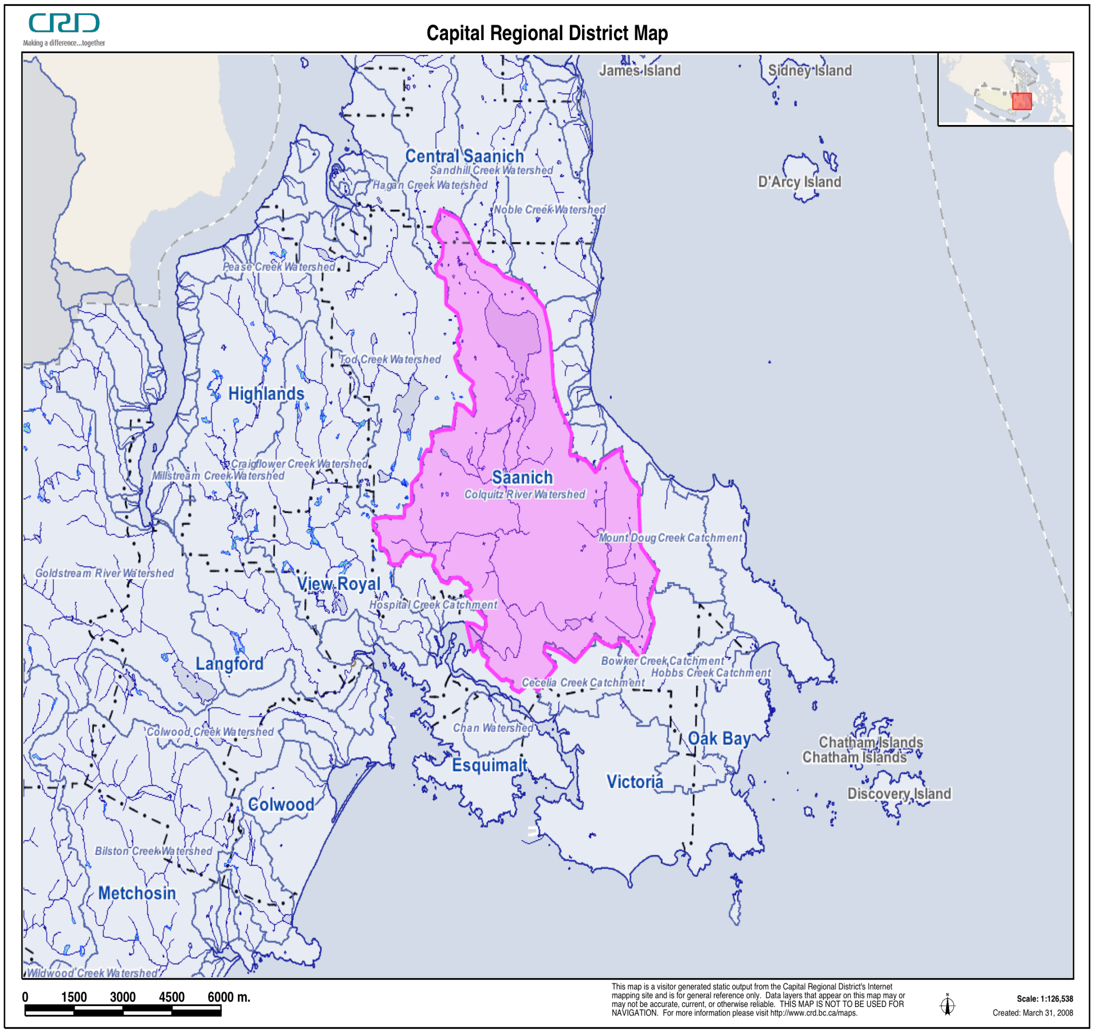

Figure 4. Location of Colquitz River Watershed (Pink Polygon) in the Municipality of Saanich, Southern Vancouver Island, British Columbia. *Note:* From *Colquitz River Watershed Proper Functioning Condition Watershed Assessment* (p.2) by Buchanan *et al.,* 2007, Victoria, B.C: Aqua-Tex Scientific Consulting Ltd. Copyright 2007 by Aqua-Tex Scientific. Reprinted with permission of author.

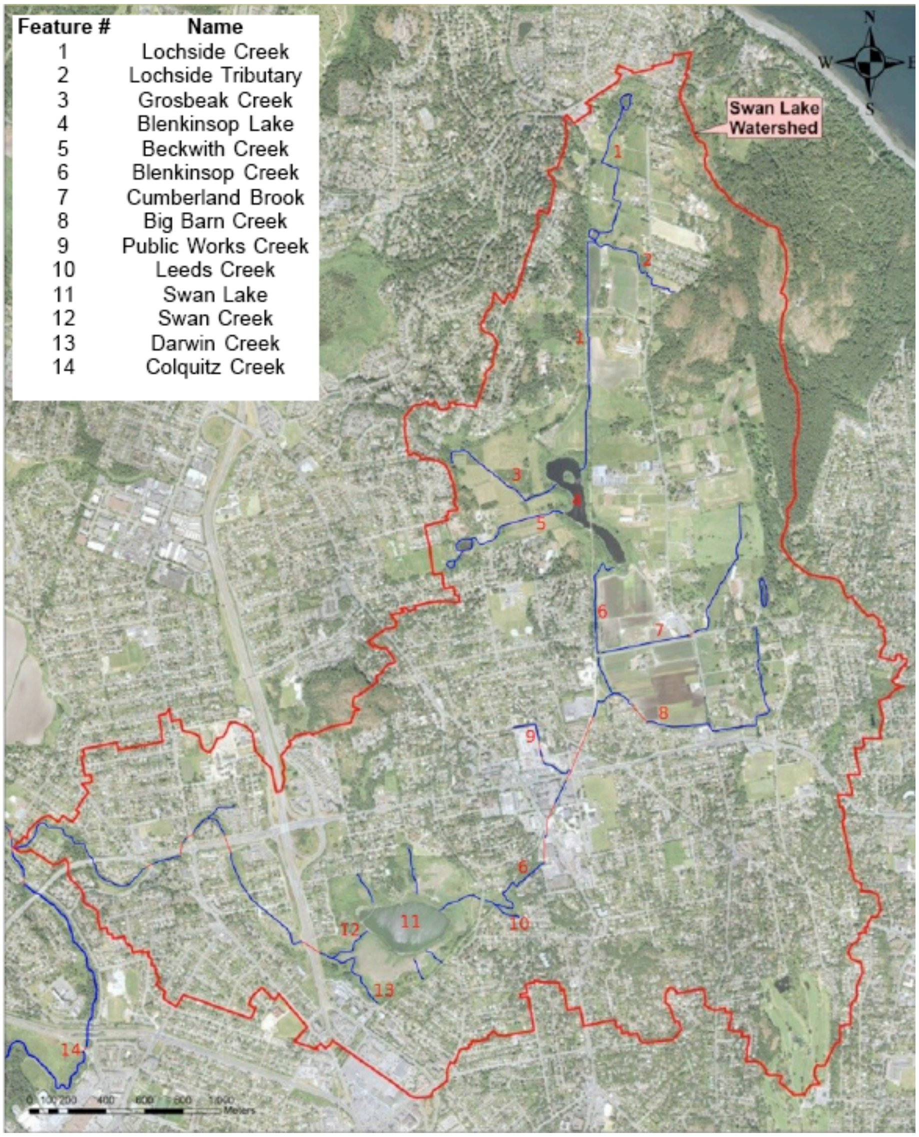

Figure 5. 2007 Ortho-photo of Swan Lake Watershed. *Note:* The watershed is delineated in the thick red outer line. Open channels are shown in blue; culverts are pink in color. From “Urban watershed health and resilience, evaluated through land use history and eco-hydrology in Swan Lake watershed (Saanich, B.C.)” by Townsend, 2009, University of Victoria, p.8. Copyright 2009 by Townsend. Reprinted with permission of author.

Figure 6. 2007 Ortho-photo of the Swan Lake/Christmas Hill Nature Sanctuary. From CRD, 2007.

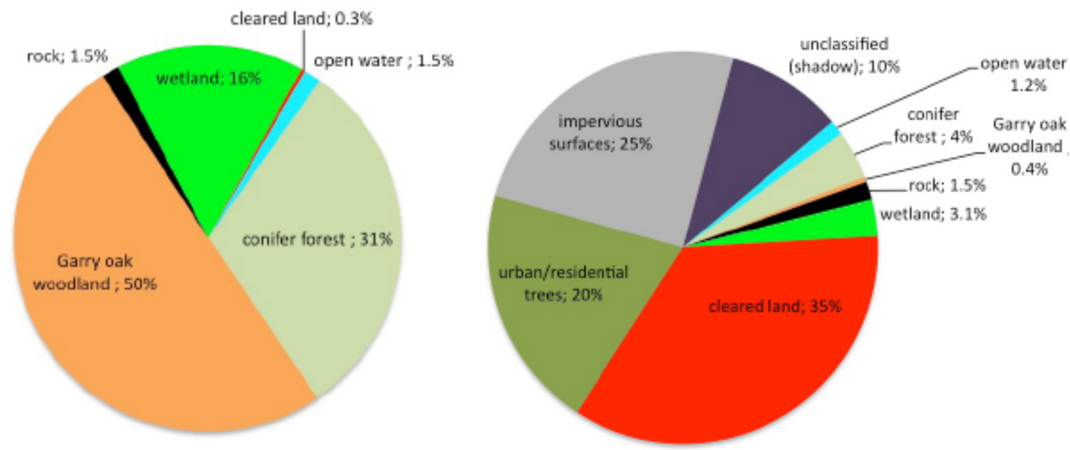

Figure 7. Comparison of land use changes in the Swan Lake Watershed for 1858 and 2007. *Note:* From "Urban watershed health and resilience, evaluated through land use history and eco-hydrology in Swan Lake watershed (Saanich, B.C.)" by Townsend, 2009, University of Victoria, p.91. Copyright 2009 by Townsend. Reprinted with permission of author.

4.3.2 Valuation Findings

Patterns of human settlement – *i.e.,* urbanization - have significantly altered natural hydrological and nutrient cycles by changing the nature of the landscape (Kravcík, Pokorný, Kohutiar, Kovác, and Tóth, 2007). Specifically, the removal and replacement of natural vegetation with impervious areas consisting of buildings, roads, sidewalks, *etc.* has resulted in increased surface temperatures within cities, oftentimes referred to as the urban heat island effect (Alberti, 2005). The urban heat island effect is an inadvertent modification of the local climate that can result in increased air pollution and energy costs arising from the greater usage of air conditioners in buildings (Akbari, Pomerantz and Taha, 2001). Rather than storing solar energy and re-radiating the energy as heat back into the atmosphere as buildings and other paved surfaces do, vegetation has a "significant cooling effect and air-conditioning capability" by protecting the ground from overheating and drying out by regulating humidity and temperature regimes through shading and the process of evapotranspiration (Kravcík *et al.,* 2007, p.15).[39] For example,

[39] The air conditioning effect is largely due to two properties of water. First, water has a high specific heat capacity - *i.e.,* the ability to absorb energy without changing temperature. For instance, a small amount of heat applied to a specific volume of air will change the temperature of the air rapidly as air has a low

Kravcík *et al.* (2007) estimate that a large tree with a surface area of $80m^2$ could intercept about 450kWh of solar energy per day. From the intercepted amount of solar energy, the tree would use 280kWh to transpire 400 litres of water per day; the remainder of energy that is not absorbed is reflected, used for photosynthesis and some reflected/absorbed by the ground. Therefore, in the course of an entire day, such a tree would utilize 20-30kW of power, an amount of energy comparable to what 10 or more air-conditioning units would require (Kravcík *et al.,* 2007). This is not to say that such a tree has the equivalent cooling effect of 10 air conditions, rather it is a way to demonstrate how much solar radiation vegetation can absorb and therefore keep an area from warming up; hence, the cooling effect.

In the Swan Lake valuation study, the energy dissipation estimate was undertaken at both the watershed and Nature Sanctuary scale. Rather than using the number of air conditioners, the amount of energy absorbed by the vegetation was converted into a coal tonne equivalent and then valued at the May 2009 short price for coal, which resulted in an estimated annual value of $35,205,562 for the watershed (Table 3) and $2,069,332 for the Nature Sanctuary (Table 4).[40] The purpose of the valuation exercise was to demonstrate the amount of solar energy that vegetation absorbs over the period of a year. The value estimate provides a basis for the argument that preserving, enhancing and/or restoring vegetation in urban areas could help buffer temperature extremes in cities.[41] For instance, vegetation is now being applied to buildings (*e.g.,* green roofs, green walls) to reduce building energy costs (Takebayashi and Moriyama, 2007; EPA, 2008)

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specific heat capacity. If the same amount of heat were applied to the same volume of water, the temperature of the water would not change much (Ricklefts, 1976). Second, water has a high thermal conductance (*i.e.,* “the rate of diffusion of heat through water: approximately 30 times that of air”), which results in a “thermally constant and uniform environment” (Ricklefts, 1976, p.29). Through the process of evapotranspiration, vegetation uses water to absorb solar energy which results in the regulation of humidity and a uniform temperature; hence, the cooling effect (Ricklefts, 1976).

[40] Coal was selected as a proxy as a means to recognize an often-overlooked ecosystem service – *i.e.,* the conversion of biomass into fossil fuels (Hodas, 2007). Over millions of years, ecosystems have collected and stored solar energy as biomass, which over time is converted into fossil fuels such as coal, natural gas and petroleum.

[41] Akbari *et al.* (2001) estimate that the mitigation of urban heat islands using vegetation could reduce US air conditioning energy use by 20%, saving over $10 Billion annually.

Furthermore, enhancement of urban forests within cities is now being viewed as a climate adaptation strategy (Snover *et al.,* 2007).

The remaining solar energy that is not absorbed/reflected by the ground/vegetation, or used to transpire water is converted into energy by the photosynthetic process (Bormann and Likens, 1979). Photosynthesis is a process in which vegetation converts solar energy as light into chemical energy so that vegetation can grow and maintain many of the regulating functions that result in ecosystem services, such as carbon storage and sequestration (Ricklefts, 1976). Photosynthesis occurs when plants take in solar energy, carbon dioxide and water to produce a glucose and oxygen (Ricklefts, 1976). Plants use the glucose for energy or as a building block (in conjunction with various minerals) to construct complex organic compounds, such as fats, oils and cellulose, which are used in plant production (Ricklefts, 1976). Plant production is divided into plant respiration (*i.e.,* regulation and maintenance of plant health) and net production (*i.e.,* plant growth and reproduction) (Ricklefts, 1976). It is the net production that is of interest to those valuing the carbon storage and sequestration service as net production refers to the process by which carbon dioxide is transformed into above- and below-ground biomass and stored as carbon (C).

The carbon storage and sequestration service was assessed on two different scales: the Swan Lake watershed scale and the Swan Lake/Christmas Hill Nature Sanctuary scale. Each scale was valued using two different methods (CITYgreen and UFORE respectively) to assess the amount of carbon stored and sequestered by vegetation. To assess the Swan Lake watershed, the CITYgreen GIS-based program was employed. Using a 2005 land cover dataset, the CITYgreen program assessed the total carbon stored and sequestered by vegetation within the Swan Lake watershed to be 32,590 tonnes and 254 tonnes per year, respectively (Appendices A and B). Using a value of $25 per tonne, the monetary value estimate of the total carbon stored and annually sequestered for the area in 2009 is estimated to be $840,122 and $6,343 respectively.[42]

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[42] $25 per tonne of carbon is the price selected by the British Columbia Provincial government for every tonne of Greenhouse Gas (GHG) (Office of the Premier, 2007). The 2005 sequestration rate was used to estimate the total amount of carbon stored for 2009. See Appendix A for discussion.

Table 3. Estimated ecosystem service values for the Swan Lake Watershed

Ecosystem Services	**Estimated Value**
Carbon Stored	$840,122
Carbon Sequestered	$6,343 Per Year
Energy Dissipation	$35,205,562 Per Year

To assess the carbon storage and sequestration value for the Swan Lake/Christmas Hill Nature Sanctuary, a more detailed assessment using the UFORE model was undertaken. Using field measurements that included species identification, diameter breast height, crown height and width, crown condition, light availability, die-back, leaf area and shrub understory, the carbon stored and sequestered for the Swan Lake/Christmas Hill Nature Sanctuary is estimated to be 1873 tonnes and 81 tonnes/year, respectively. The total estimated value in 2009 based on a value of $25 per tonne would be $46,836; annually, the carbon sequestration value is estimated to be $2,029 (Table 4).

Concerns over the impact that the continued release of greenhouse gases (GHG) has on global climate change has led to an increased interest in reducing atmospheric carbon dioxide (CO2) concentrations. As a result, carbon markets, as well as policies to reduce carbon emissions, have emerged. Managing urban forests to store carbon dioxide, in particular, is becoming a popular climate change mitigation strategy in order to reduce atmospheric carbon dioxide (CO2) concentrations (Conway and Urbani, 2007; McHale, McPherson and Burke, 2007). For instance, The California Climate Action Registry (2009) recently released the *"Urban Forest Project Verification Protocol"* which is a guide for verifying urban forestry projects in order to achieve carbon credits. In addition to carbon storage and sequestration, urban forests can also provide a myriad of other benefits such as the reduction in building energy costs when used to shade or block wind (McPherson, Nowak, Rowntree and Rowan, 1994), reduction in stress and improved health (Ulrich, 1984), increased property values (Dwyer, McPherson, Schroeder and Rowntree, 1992), rainwater interception (Xiao and McPherson, 2002) and improved air quality (Scott, McPherson and Simpson, 1998; McPherson, Simpson, Peper, Maco and Xiao, 2005; Nowak, Crane and Stevens, 2006).

Detrimental air quality in cities has become a multibillion-dollar problem that continues to negatively affect human health, damage vegetation and reduce visibility (McPherson *et al.,* 1994). In most cases, industrial/factory emissions and vehicle emissions are often cited as the major sources of air pollution (Research Group, 2002). In particular, urban forests are being recognized in their ability to improve air quality. For instance, Nowak *et al.* (2006) contend that managing urban forest canopy cover would improve air quality and help meet air quality standards as urban forests can remove air pollutants such as carbon monoxide (CO), nitrogen dioxide (NO_2), ozone (O_3), particulate matter < 10 microns (PM10) and sulphur dioxide (SO_2). Furthermore, McPherson *et al.* (1994) estimate that large trees remove 60-70 times more air pollution than smaller trees due to the characteristics associated with larger trees.

Air pollution abatement by vegetation differs for various pollutants. For instance, removal of O_3, SO_2 and NO_2 by vegetation will depend on leaf-level uptake and photosynthesis/transpiration processes, whereas CO and PM10 are removed via physical deposition (Scott *et al.,* 1998; Nowak *et al*., 2006). However, the removal of the pollutants is influenced by length of leaf-on season, species type, climate patterns, boundary conditions and pollutant concentrations (Nowak *et al*., 2006). To estimate the air pollution abatement value provided by the vegetation within the Swan Lake/Christmas Hill Nature Sanctuary, the UFORE model was employed. Using local air pollution values, meteorological data and field measurements, the air pollution abatement service was estimated to be $314,904 per year for the nature sanctuary. By influencing the urban heat island effect, vegetation not only moderates local temperatures but can also influence local climate, thus affecting both air pollution and precipitation patterns (Scott *et al.,* 1998; Kravcík *et al.,* 2007). As noted earlier, vegetation can also intercept rain fall and, as a consequence, reduce the rate and volume of stormwater runoff, lowering flood risk as well as reducing the cost of stormwater infrastructure and water quality treatment (Xiao *et al.,* 2002; CWP, 2003; Asadian and Weiler, 2008). Using L. Townsend's fieldwork and a cost-based ecosystem valuation method, a conservative flood control value of $83,098 was estimated for the Swan Lake Nature Sanctuary.

Table 4. Net annual value of the ecosystem services for the Swan Lake Sanctuary

Ecosystem Services / Management Costs	Estimated Value / Cost
Carbon Sequestered	$2,029
Energy Dissipation	$2,069,332
Air Pollution Abatement	$314,904
Flood Benefit	$83,098
General Recreation	$1,234,304
Operating and Maintenance Expenses	$(425,000)
Volunteer Time Expense	$(125,000)
Total Annual Net Value	**$3,153,667**

As demonstrated above, ecosystems can provide a myriad of valuable services. Ecosystems also provide less tangible social benefits such as general recreation, beautification, education, scientific research, privacy, wildlife habitat, cultural value, sense of place and well-being (MEA, 2003). Various ecosystem valuation methods, such as those described in Chapter 3, have been devised, and are continually being improved, in order to better estimate and quantify the economic value of these intangibles. For instance, using the ecosystem valuation benefit transfer method, the annual lower bound value estimate for general recreation in the Swan Lake/Christmas Hill Nature Sanctuary is $1,234,304. Although the methods for estimating the values for the flooding and general recreation may not be considered to have much scientific rigor, compared to those values estimated by the peer-reviewed UFORE model, the values themselves provide an argument for the preservation and rehabilitation of ecosystems. In the case of the Swan Lake/Christmas Hill Nature Sanctuary, the total estimated value of the five ecosystem services ($3,701,638) exceeds the annual cost of $550,000 to manage the area (Morrison, Pers. Comm.).

It is clear from the results of the Swan Lake/Christmas Hill Nature Sanctuary valuation study that the ecosystems provide valuable services that on an annualized basis exceed the cost to maintain the area. In addition, the large dollar value estimate derived from only a few ecosystem services provides a financial argument for the preservation and rehabilitation of ecosystems in urban areas. However, when the values at either scale are compared to the ecological information on the areas, the estimated values do not suggest that there are ongoing water quality problems, eutrophic lakes or kilometers of degraded stream and aquatic habitat within. Rather, an annual value of $35,211,905 per year for the Swan Lake Watershed is likely to suggest to decision makers that the watershed is in a good functional condition, when in-fact, the overall health of the watershed is poor (Townsend, 2009).[43] Nor does the net annual value of $3,153,667 suggest that there are various water quality problems within Swan Lake/Christmas Hill Nature Sanctuary. Furthermore, when the current ecosystem service annual value of the watershed is compared to a historical annual value estimate for the energy dissipation service alone, $49,518,927 (a difference of $14,313,365), it becomes increasingly clear that ecosystem degradation and loss has occurred over time (Table 5).

Table 5. Difference in the annual value of energy dissipation for the Swan Lake Watershed between 1858 and 2009

	MJ	**Coal (tonnes)**	**Value**
Energy dissipated by latent heat (ET) in 1858, throughout growing season (MJ)	15341509719	611118	$49,518,927
Energy dissipated by latent heat (ET) in 2009, throughout growing season (MJ)	10907071500	434475	$35,205,562
Value of "Lost Service"	**4434438219**	**176643**	**$14,313,365**

A possible reason for the inaccurate assumption of good health is likely due to the expectation that degraded ecosystems are less capable in providing the ecosystem service values deemed useful by society (Malmkvist, 2002). In addition, the estimated values do not reflect the loss of ecosystem services - the loss of a historically supportive salmon system, loss of forested areas to invasive species and other ecosystem services, such as: fishing, swimming and other recreational activities, cultural/amenity values, carbon storage and sequestration - due to degradation arising from urbanization and agricultural activities (Edmonds, 2002; Malmkvist, 2002). Had the lost ecosystem service values been

[43] The value of $35,211,905 is derived from the annual energy dissipation value ($35,205,562) and the carbon sequestration value ($6,343).

estimated, the value of the ecosystem services for the Swan Lake Watershed would have certainly been reduced to a lower, if not negative, value. Overall, the estimated ecosystem service values derived in the Swan Lake watershed valuation study do not intuitively reflect the watersheds poor condition of health and, thus, begs the question, is the estimation of ecosystem service values an accurate reflection of ecosystem function?

4.4 Is the Estimation of Ecosystem Service Values an Accurate Reflection of Ecosystem Function?

In order to address this question, an analogy using natural and human-made capital is presented. Ecosystems are often referred to as natural capital, meaning the stock of ecosystems that provides a yield of valuable ecosystem goods and services indefinitely (Costanza *et al.,* 1997). In contrast, human-made capital is generated though economic activity arising from human ingenuity and technological development (Hawken, Lovins, and Lovins, 1999). Both types of capital are similar in that both require functional systems (*e.g.,* corporations, markets, ecosystems) to produce the valuable goods and services. They also differ in many ways; specifically, in that human-made capital is traded in the economic markets whereas natural capital is, for the most part, excluded from the marketplace. In very simplified terms, the value of any corporation at any given time reflects all known information about the entity – *i.e.,* its functional condition. This includes the current financial condition, market/firm uncertainty goods and services produced, industry information and management skills. Thus, the value of a corporation is not simply generated by the total value of the goods and services produced (and *vice versa*). Rather, the value is determined by the interaction of the components such as management, resilience, adaptability and competitive advantage that enable a corporation to produce the goods/services indefinitely that results in profit (King, 1997). In summary, the value of a corporation (human-made capital) is based upon information on the current functional condition and anticipated trajectory (ability to continue producing goods and services to earn revenue) of the entity as a whole. Ecosystems are not valued in this way.

As described in earlier chapters, ecosystem valuation methods only focus on valuing select goods and services produced by ecosystems. Using the Total Economic Value (TEV) framework, these values are summed and purportedly reported to be representative of the total value of an ecosystem. However, unlike human-made capital where the total value is based upon all available information and indicators that provide information on less known information such as volatility, diversity and vulnerability, the estimated total value of an ecosystem is simply based on the aggregated value of all the ecosystem services assessed. Current attempts to value or at valuing ecosystems are akin to valuing a corporation based upon the current production of goods and services at a given moment in time. This is a limited approach as the value does not represent other future earnings good/service lines being offered, the total amount of assets owned or what will be acquired in the near future, nor does it account for the less tangible characteristics such as talent, goodwill, resistance, vulnerability, *etc*. Essentially, ecosystems are not being properly assessed and valued for their functional condition. The reason, as noted in Chapter 3, can be largely attributed to the fact that ecosystem valuation methods do not take into account the factors necessary to maintain or that influence the functional condition of an ecosystem. Thus, the valuation exercise cannot determine if an ecosystem is in a *functional*, *at-risk condition*, a *non-functional condition* or near one of these thresholds. As a consequence, ecosystems are not being properly assessed for their ability to resist/recover from disturbances and continue providing goods and services over time that are fundamental to human health and well-being. In essence, the value of estimated ecosystem services, although they contribute in part to the total value, are not an accurate reflection of ecosystem function, *because the values do not account for the factors necessary to maintain function, nor do the values account for the current functional condition or the trajectory of which the assessed ecosystem is headed.* An example of a ditched stream can help put this argument into context.

Streams, like many of those in the Swan Lake watershed, are often ditched to drain water off the landscape as quickly as possible in order to support agricultural production or reduce flooding; both valuable ecosystem services to farmers and municipalities, respectively. Using ecosystem valuation techniques, a ditch could be assessed to determine the value of the increased agricultural production using the productivity method or could be assessed for its flood value using cost-based methods. The value of either ecosystem service, particularly the latter, is likely to have a very large economic value even though the functional condition of the stream has been severely degraded (if not lost entirely) and many other ecosystem services have been lost as a result.[44] The purpose of the example is to demonstrate that the estimated value of the ecosystem services is not reflective of the functional condition of the stream as a ditch, nor does the value reflect the loss of other ecosystem services. Because there is no requirement to evaluate the functional condition of an ecosystem, in the context of urban centers, a ditch is likely to be perceived to have more value than a functional stream.

In summary, based upon the ecosystem service value estimates alone, decision makers cannot gauge and compare the historical condition to the present, nor can decision makers assess how the landscape context is influencing the functional trajectory of the ecosystem of study. Information on the trajectory of the ecosystem is extremely important to decision makers as it influences the urgency with which the functional condition of an ecosystem may be changing as a result of anthropogenic or natural perturbations. Finally, without additional information decision makers are likely to presume that the ecosystem is in a healthy condition.

Thus, in answering the question - Is the estimation of ecosystem service values an accurate reflection of ecosystem function? - based upon the work completed in Chapter 3, the Swan Lake Watershed valuation study and the prior discussion on valuing human-made capital, the answer is no and a more holistic analysis of ecosystems is required. Specifically, the ecosystem valuation discipline needs to begin assessing the functional value of an ecosystem, not simply the value of the ecosystem services, in order to avoid many of the noted problems with ecosystem valuation. However, such a recommendation is impractical in the short-term as many of the existing barriers to effective ecosystem

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[44] In the short-term, there is a perceived financial gain to the individual or land-owner. However, in the long-term, there are economic costs associated with the lost ecosystem services that often require the installation of human-made structures, such as dams, levees, stormwater pipes, *etc*. Such human-made structures tend to deteriorate over time requiring future maintenance and replacement. These costs are borne collectively by society, not by the individual or land-owner who originally caused the degradation.

valuations have not yet been overcome. These include the considerable expense to research and monitor ecosystems, lack of known thresholds, inability to value and internalize indirect ecosystem services due to the complexity of ecosystems, impracticality of collecting all necessary information and semantics. Furthermore, valuing the functional condition of an ecosystem would require additional and extensive scientific research in collaboration with valuation specialists and experts from other disciplines. Various questions would also have to be addressed, including (but not limited to):

- *Does a scientifically peer-reviewed standard exist that measures the functional condition of all various types of ecosystems on a standardized basis?*
- *What are the functional boundaries, both spatially and temporally, of an ecosystem? How are these defined?*
- *At what level of function does an ecosystem provide goods/services? How would ecosystem threshold changes be accounted for?*
- *Do the goods and services contribute to the functional value of an ecosystem, or should the functional value of the ecosystem be isolated from the value of the goods and services?*
- *How do you value the functional trajectory (risk) of an ecosystem?*
- *What is a non-functioning ecosystem (absence of function) worth?*
- *What is the bare minimum level of ecosystem function required to achieve a net positive financial value?*
- *Because management and landscape context would influence the functional condition of an ecosystem, how are these (and other factors) accounted for in the functional valuation assessment?*
- *Does the physical scarcity of functional ecosystems increase the functional value? How would scarcity be assessed locally? regionally? globally?*

Rather than choosing to wait for scientists and valuation specialists to develop the perfect methodology to asses the functional value of ecosystems, economic markets for human-made capital can be sought to help develop more practical approaches to valuing ecosystems. Returning back to the discussion on human-made capital, in economic markets the value of a corporation is not simply based upon what goods and services are produced. Rather, specialists and institutions determine the value by collecting, analyzing and disseminating information and utilizing indicators. Although much of this information is available, markets are also subject to various inefficiencies such as incomplete and asymmetric information such as adverse selection, moral hazard and the principle agent problem. To navigate around these issues, those who participate in the markets rely on indicators that address scales and nested hierarchies in order to better understand and gauge trends within local and global economic and political markets in order to assess the context and functional condition as well as the trajectory of the entity (King, 1997). Many of these indicators assess trends of resilience, diversity, vulnerability, and volatility.

Similar to economic markets that utilize indicators and rating systems to determine trends and assess uncertainty (risk) respectively, scientists use indicators in ecological evaluations to assess and draw conclusions about the health of an ecosystem (King, 1997; Andreasen, O'Neill, Noss and Slosser, 2001). Thus, taking the lead from economic markets - that utilize indicators and rating systems to determine trends and uncertainty of valued capital - what is proposed is the integration of ecosystem valuation and ecosystem evaluation assessments to facilitate the inclusion of natural capital in economic decision frameworks. The following discussion provides the reasoning behind this suggestion.

4.4.1 Discussion

As contended by King (1997), ecosystem valuation and ecosystem evaluation assessments are situated at two differing ends of the spectrum. The former attempts to assign values to ecosystem services, but rarely gives consideration "to the specific features or functions that generated them" (King 1997, p.6). The latter, the ecosystem evaluation assessments (such as the Index of Biological Integrity (IBI), The Shannon Index, *etc.)* provide information on the form and structure of the ecosystem, but do not address the ecosystem service values that may be provided. Essentially, the information from ecosystem evaluations is not translated into an economic value on which most land-use decisions are based. Thus, each ecosystem valuation or evaluation assessment applied alone can only provide decision makers with half of the necessary information. Simply put, either approach utilized in isolation from the other makes it difficult or impossible to identify how ecosystem changes will affect socio-economic systems and *vice versa.* Therefore, the integration of ecosystem valuation and ecosystem evaluation assessments could provide decision makers with the necessary ecological information as well as the economic value of the ecosystem services.

Valuing ecosystems without the context of the functional condition is likely to result in mismanagement of ecosystems because without any context, understanding or measurement of the functional condition of the ecosystem, ecosystems will simply be managed for what is perceived to be the highest and best use or left unmanaged. For instance, in the context of urban planning, many municipalities continue to manage aquatic ecosystems as aqueducts in order to transport waste products and stormwater (Malmkvist, 2002). As a watershed is developed over time, the functional condition degrades and, as a result, other ecosystem services are progressively lost. Because the functional condition of the aquatic ecosystems is considerably reduced, there are unintended consequences as well, including: lowered water quality, lowered water table, increased stream erosion and sediment transport, increased pollutant concentration and loading, flashier flow regimes and reduced flood control, many of which have economic costs (Malmkvist, 2002).[45] Rather than managing ecosystems simply to achieve an ecosystem service, an integrated ecosystem valuation/evaluation approach would provide decision makers with the necessary ecological information to manage ecosystems for their functional condition, rather than for an ecosystem service, resulting in the sustainable production of multiple ecosystem services.

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[45] These are characteristics of the "urban stream syndrome" (CWP, 2003; Grimm *et al.,* 2008). Oftentimes, the unintended consequences affect downstream developments (flooding, sediment deposition, *etc*.) and have economic costs associated with them as some situations require the installation of expensive grey infrastructure which provides fewer and less reliable services than what the original aquatic ecosystem originally provided. Human-made capital, such as infrastructure, is also attributed with having higher upfront capital and long-term economic costs (depreciation, replacement, maintenance) when compared to the cost to rehabilitate or maintain existing functional ecosystems (Barraclough and Hegg, 2008a).

Because there is no requirement to assess the functional condition of an ecosystem in ecosystem valuation methods, ecosystem service value estimates tend to focus on a single point in time and, thus, do not reflect the critical information decision makers require, including the current functional condition, the direction or changes within ecosystems, the quality of ecosystem services and the urgency to which each may be occurring or changing as a result of human-made or natural perturbations (Toman, 1998; Straton, 2006). However, an integrated ecosystem valuation/evaluation approach would force valuers to assess the historic functional condition for the purposes of developing a benchmark by which the current functional condition can be assessed. Assessing both an historic and current, value would enable decision makers to contrast and highlight the value that has been lost (or gained) over time. The analysis could also be used to identify areas for rehabilitation and plan for future ecosystem services given the current political, social and economical limitations of the landscape within which an ecosystem is situated.

Ecosystem conservation and rehabilitation efforts based on ecosystem valuation estimates, performed in isolation, are more likely to fail, as the estimated ecosystem service values cannot identity key underlying mechanisms that link urban patterns to changes in ecosystem function (Alberti, 2005). Because functional measurement is necessary to understand the system (Pickett *et al.,* 2001; Edmonds, 2002), simply calculating the value of an ecosystem for conservation purposes, without any context to the functional condition of an ecosystem and landscape context, is likely to warrant the same level of conservation amongst various other ecosystems being managed. However, some ecosystems may be at risk and require immediate attention and greater resources than others. Thus, the information provided by an integrated ecosystem valuation/evaluation approach would better assist decision makers in determining which ecosystems are at risk and have the most value, thereby enabling decision makers to rank and address ecosystem priorities more effectively. The integration of ecosystem valuation and ecosystem evaluation assessments could begin to address some of the ecological-economic questions that cannot be addressed by either approach alone:

- *"Is the landscape stable [functional] in the context of human [socio-economic] needs and expectations? Or is it losing its essential characteristics, becoming* *progressively biotically impoverished and contributing in the process to the destabilization of other ecosystems?" (Woodwell, 2002, p.21).*

- *What is the value of this change? And is this the highest and best use of the ecosystem (e.g., preservation, agriculture, urban development)?*

Integrating ecosystem valuation and ecosystem evaluation assessments would capitalize on the strength of ecosystem valuation methods, which attempt to incorporate the relationship between society and ecosystem goods/services, whilst taking account of the ecological-economic interface. Compelling both ecologists and economists to work together in a meaningful way, this valuation approach could help researchers focus more on the intercomplexities within ecosystems (*e.g.,* structures, processes, functions, resistance, resilience, recovery) that result in the sustainable production of ecosystem services, and thus develop new methods to overcome many of the current barriers to achieve reliable and accurate ecosystem valuation estimates.

4.5 Future Research

Future research must focus on the development of a new ecosystem valuation framework that evaluates both ecological and economic impacts of the undesirable effects of anthropogenic or natural alterations to ecosystems (Norton and Noonan, 2007). The proposal of integrating both ecosystem valuation and ecosystem evaluation assessments is a positive step in this direction.

Due to the complexity of the task, only a few researchers have developed conceptual models that attempt to link the landscape context to which an ecosystem is situated to their respective ecosystem service values (see King, 1997; Mäler *et al.,* 2007). These models are important as they attempt to increase the accuracy of the estimated values (a much needed step if ecosystem service values are to ever enter into accounting frameworks). However, the drawback with many of these models is that they are based upon production-function models designed to adjust the economic value of the ecosystem service using a weighted index that is based upon the health of the ecosystem. The problem with simply adjusting an ecosystem service value up or down is that the resulting economic value provides little, if any, information on an ecosystems functional condition and trajectory, even if the valuation estimate is more accurate. Thus, the task should be focused on how to develop a risk-rating system that adjusts the estimated ecosystem service value for the functional condition, but also provides a risk-rating assessment to provide the necessary ecological information to the decision maker.

Such a system could be akin to bonds rating assessments utilized in economic markets that help advise investors of the level of risk associated with a specific type of capital. Specifically, bond-rating systems classify bonds ranging from high quality/low risk bonds as “Aaa” to high risk/low quality bonds as “C” or “junk bonds” (see Table 6). In the context of ecosystem valuation, ecosystem valuers would able to continue valuing single or a few ecosystem services using existing methods, but would also assess the functional condition and trajectory of the ecosystem in order to classify and rate the level of inherent risk/uncertainty based on the landscape context. A functional risk/uncertainty rating associated with an ecosystem service value estimate would provide decision makers more information on the functional level of the ecosystem as well as its trajectory.

Table 6. Example of bond rating system related to investment risk and grade

Rating	**Risk**	**Grade**
Aaa	Highest Quality	Investment
Aa	High Quality	Investment
A	Strong	Investment
Baa	Medium Grade	Investment
Ba, B	Speculative	Junk
Caa/Ca/C	Highly Speculative	Junk
C	In Default	Junk

Note: Table adapted from Investopedia (2009).

However, in order to develop the proposed a risk rating system that integrates ecosystem valuation methods and ecosystem evaluation assessments for the purposes of land-use planning and development, the following challenges will have to be addressed.

4.5.1 Future Research Challenges

A major challenge to valuing ecosystems is that there is no standardized ecosystem valuation typology due to issues of meaning and measurement. For instance, the globally recognized Millennium Ecosystem Assessment (2003) uses the term ecosystem services to describe both the goods and services provided by ecosystems as well as the ecosystem functions that provide the services and, thus, leaves much room for interpretation between the terms function and services (Kline, 2006; de Groot and Hein, 2007). However, these terms must be identified separately as a comprehensive valuation of an ecosystem may result in double counting if the terms are not properly differentiated from one another (de Groot and Hein, 2007). More importantly, placing values on ecosystem services may be problematic if decision-makers begin trying to manage ecosystems for their services rather than the functional condition, which may result in unforeseen consequences.

With regards to the functional assessment of ecosystems, the challenge is that most functional assessments focus on aquatic ecosystems and attempt to describe the functional condition of the ecosystem by assessing biodiversity using indicator species as a numbered index. The drawback with this type of approach is that the assessments tend to be based on the assumption that the diversity of species or structural integrity is related to the functional integrity of the environment, which ultimately results in inventories of species, not functional assessments based upon the ecological criteria (Edmonds, 2002; Malmkvist, 2002; Woodwell, 2002; Mattson and Angermeier, 2007; Barraclough, Pers. Comm.). Thus, the identification of trajectory or what is causing the degradation in a system cannot be identified (Woodwell, 2002). Number indices can also be problematic as they may not effectively communicate a problem to a decision maker and thus result in false positives and negatives.[46] The challenge is that assessments, like the Proper Functioning Condition (PFC) assessment, which measure the functional condition of an ecosystem by assessing "both abiotic and biotic factors as they relate to physical function" (Buchanan *et al.,* 2007, p.ii), do no have an equivalent index to assess terrestrial ecosystems; nor does a combined terrestrial/aquatic index exist (Townsend, 2009; Barraclough, Pers. Comm.). An additional challenge with many current ecosystem evaluation assessments, including the PFC assessment, is that they have not been developed to assess the functional condition of urban ecosystems (Edmonds, 2002).[47]

Describing and measuring ecosystem services that are linked to urban policy and management actions also needs to be further researched (Kline, 2006). Recognizing that ecosystem degradation continues to occur from urban development activities, decision makers have often relied on urbanization indicators such as the percentage of impervious area within the watershed, buffer ordinances as well as sustainable development strategies (*e.g.,* green development rating schemes) to guide planning and development activities. However, these approaches are limited in their ability to protect the functional condition of ecosystems (Alberti, 2005, Grimm *et al.,* 2008). According to Pickett *et al.* (2001), the problem is that few studies have examined how urban patterns affect ecosystem function. Rather, considerable research has been focused on the correlation of ecosystem indicator changes to aggregated measures of urbanization (Alberti, 2005; Verburg, Van de Steeg, Veldkamp and Willemen, 2009).[48] Although these studies can be informative from a post-urban development standpoint to assess current conditions, the indicators are problematic as they cannot easily identify or isolate where the degradation

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[46] A false positive occurs when an indicator suggests a loss of biodiversity due to human-made activities, when in fact the change was natural. False positives are considered benign, as they tend to result in the conservation of an area. However, false negatives are more alarming as they tend to result in ecosystem degradation and tend to result when a negative change in the ecosystem has occurred, but the index indicates no significant change has taken place. The result is either no protection of the area or an activity is allowed to proceed within the area, which tends to worsen the existing condition (Andreason *et al.,* 2001).

[47] The PFC has been modified by Aqua-Tex Scientific Ltd. to assess the function of urban aquatic ecosystems, but it still remains a challenge.

[48] For instance, The Center for Watershed Protection (2003) concluded that when the percentage of impervious surfaces within a watershed exceeds 25%, the aquatic ecosystems are likely to exhibit characteristics of excessive degradation. This indicator does not express to what extent the degradation will be or how new development will affect ecosystems (Burton and Pitt, 2000; May, 2003).

is coming from, nor can the indicators always predict how ecosystems will respond to new development or changes in climate or both (Alberti, 2005).

These challenges considered, what that remains now is to identify ecosystem evaluation assessments that measure the functional condition of both aquatic and terrestrial ecosystems, and develop standardized procedures to which the selected ecosystem evaluation assessments can be integrated with applicable ecosystem valuation assessments, in the context of urban planning.[49] While this task exceeds the scope of this thesis, recommendations to address the challenges can be made.

4.5.2 Recommendations For Future Research

To be effective in urban planning and development, a terrestrial ecosystem evaluation assessment is required.[50] Akin to the PFC assessment for aquatic ecosystems, the terrestrial evaluation assessment should be able to assess the functional condition of terrestrial ecosystems as well as identify areas of where degradation is occurring and why. Like the PFC approach (Appendix C), the terrestrial assessment should be able to assess historic, current condition/trend as well as future conditions in order to develop benchmarks to which future evaluations (audits) can be assessed against so that an decision makers can determine if management activities are improving or reducing the functional condition of an ecosystem. The assessment should be able to communicate relevant information to decision makers in a form of an index that is intuitive and understandable to users. Furthermore, the terrestrial assessment should integrate with the PFC (or similar type) assessment(s), so that the functional condition of a landscape can be adequately assessed. To undertake this recommendation, the creators of the PFC model (U.S. Bureau of Land Management, the U.S. Forest Service and Natural Resources Conservation Service) should first be sought for advice and a model (if one exists) acquired and modified accordingly, so that a similar assessment can be devised.

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[49] For instance, a standardized procedure could simply be a process to which an ecosystem condition assessment is first undertaken and a valuation assessment following suit.

[50] See Andreasen *et al.* (2001) for a comprehensive description of the necessary characteristics for a terrestrial index of ecological integrity.

Following the PFC development model will require collaboration between university researchers, public/private sector scientists, landowners and community groups so that the newly developed method is peer-reviewed, is field tested and is user friendly. In order to develop standardized procedures to which terrestrial/aquatic ecosystem assessments can be integrated with ecosystem valuation methods, a comprehensives framework for urban planners will need to be identified and modified accordingly or created.

Until the uncertainties surrounding ecosystem valuation estimates are better identified and managed, the value estimates will keep them from being accepted into accounting frameworks (Gret-Reganey, Walz and Bebi, 2008). What is therefore required is a standardized multiple criteria decision analysis framework, that relates the provision of ecosystem functions and the value of the services to the level of the risk associated given the current and future landscape context.[51] The framework must also be flexible enough to define and evaluate ecosystems at varied ecological and institutional scales in order to enable long range valuation planning. For instance, the process should be able to help users differentiate between ecosystem functions and services as well as identify what services are valuable to whom at different scales. The framework should be able to incorporate the characterization of both aquatic and terrestrial ecosystems by identifying processes that lead to the degradation and loss of these systems as well as identify and provide implementable strategies such as policies and complimentary regulations that enforce high density low impact development, no net run-off of stormwater, green developments, buffer zones, *etc*. for areas that can be rehabilitated, improved or preserved.

The multiple criteria decision analysis framework should be a multi-disciplinary and multi-stakeholder interface that enables urban planners and community stakeholders to work in collaboration with experts in order to comprehend and understand the importance and economic value of functional urban ecosystems and how to manage these systems. Since much of the technical experience regarding urban ecosystems, and the valuation

[51] To address this requirement, the relationship between land-use, ecosystem function and the provision of ecosystem goods and services needs to be better understood (Alberti, 2005; de Groot and Hein, 2007).

thereof, remains isolated from decision makers/urban planners/community groups, the usage of both terrestrial and aquatic functional assessment systems in conjunction with ecosystem valuation must be adequately supported with scientific and technical experience. Otherwise, placing untrained people that are isolated from the technical/scientific experience in charge of complex ecosystems will be a "recipe for disaster" (Cullen, 1990, p.208). Universities can help address this issue by creating multi-disciplinary degrees that focus on both theoretical and applied learning using relevant examples and case studies such as field trips, field work and co-op positions.

4.6 Conclusion

Historically, urban development has been allowed to expand with very little restriction, often "[ignoring] all but the most glaring changes in local ecosystems" until a crisis occurs and forces the issue into the political agenda (McGranahan and Marcotullio*,* 2005, p.807). The reason can be attributed to the fact that the economic value of most ecosystem services are not internalized within land-use economic decision frameworks. In the case of the Swan Lake watershed, not incorporating the economic value of the ecosystem services or the functional condition of ecosystems, in urban development planning, has contributed to the expansion of the urban fringe in such a way that it has resulted in a watershed that is poor in health as many of the aquatic ecosystems are an at-risk or non-functional condition.

In recognition that market failures are occurring, the ecosystem valuation discipline has endeavoured to develop new and improved existing ecosystem valuation methods in order to better estimate the total value of an ecosystem. However, these methods do not incorporate the account the factors necessary to maintain (or that influence) the functional condition of an ecosystem, having a direct impact on the ecosystem services produced as was noted in the Swan Lake Watershed study. On the other hand, ecological evaluations assess the factors necessary, but tend to overlook, and do not assess, the value of the ecosystem services provided. And in the context of urban planning, each assessment applied alone can only provide decision makers with half of the necessary information.

Therefore, the combined measurement of the functional condition of ecosystems and the valuation of ecosystem services in urban development planning is deemed necessary.

Until there are better ways to value the functional condition of ecosystems, the integration of ecosystem valuation methods and ecosystem evaluation assessments is a necessary incremental step in order to achieve such a goal. In the context of post-urban planning and development, the approach has immediate application, as it would provide effective financial arguments for the preservation and regeneration of ecosystems as well as facilitate more informed decisions in managing existing urban ecosystems for their function rather than ecosystem services. Specifically, the approach could be used to communicate the extent to which the value of the ecosystem services are currently being provided, which ecosystem services are declining and help identify what management actions are necessary to stabilize the ecosystem and stop the decline. This information would be vital as decision makers could also use the information to assess the impact of future alternative land-use or ecosystem management scenarios based upon social, environmental and economic criteria: a form of triple bottom line evaluation.

In the long-term, an anticipated challenge with valuing the functional condition of ecosystems post-development is that such systems tend to be degraded or destroyed due to various constraints and modifications and, thus, the full-value of an ecosystems functional condition may not (or ever) be adequately captured. However, from a pre-development standpoint, in which many of the ecosystems are in a functional condition, there exists an opportunity to value an ecosystem's functional condition as part of the development design process in order to preserve (or create) the functional value. Specifically, the proposed approach when utilized as a component of the IDP process could be one such incremental step in order to achieving the long-term goal of valuing the functional condition of ecosystems. Although the process is not likely to describe the full encompassing functional value of an ecosystem, it may be the best method because the IDP process attempts to incorporate and adequately represent all interested parties and/or stakeholders values and concerns regarding the environment as part of the design process. Using the British Pacific Properties development in West Vancouver as a case study, Chapter 5 explores the application of the proposed approach to integrated design by asking the following question: can an integrated ecosystem valuation/evaluation approach be utilized in urban development planning and design?

Chapter 5

5.1 Introduction

Chapter 4 utilized a valuation case study to support the argument that without the proper context of the functional condition of an ecosystem, the estimated ecosystem service values provides little information on the functional condition of the ecosystem that generated them. The chapter demonstrated that the ecosystem valuation discipline needs to begin assessing the functional value of an ecosystem, not simply the value of the ecosystem services. Due to the inherent challenges in achieving such a recommendation, the integration of ecosystem valuation methods and ecosystem evaluation assessments was proposed.

Using the British Pacific Properties (BPP) development in West Vancouver as a case study, Chapter 5 explores the following question: can an integrated ecosystem valuation/evaluation approach be utilized in urban development planning and design? The chapter is structured as follows: a brief discussion on how conventional urban developments are designed. To begin addressing the question posed, the integrated design and planning (IDP) process that British Pacific Properties (BPP) designed and employed for the 215-acre Rogers Creek Development is explained and contrasted against the traditionally designed communities in the context of building sustainable communities. Using the BPP development as a case study, the question is answered, while characterizing barriers and providing recommendations.

5.2 The Impacts of Development Design

5.2.1 Conventional Development Design

Local governments have historically focused on the built components of cities as a result of economic, social and political priorities (McGranahan and Marcotullio, 2005). As a consequence, under current urban development practices, development responds to legislation, economy, safety, community, beauty and ecology in a decreasing level of importance (Van Bohemen, 2002). Treated as a valueless commodity, the functional importance and value of ecosystems remains overlooked in the traditional or conventional development design.[52] The open spaces that remain are left in their original condition because they cannot be developed either due to the high costs to prepare the area for development or because of legislative requirements (Arendt, 1996).[53] In application, this design process results in urban sprawl, which is the outward expansion of the urban containment boundary through the conversion of natural landscapes into scattered and fragmented urban developments.

Because nature is considered last in the conventional design process, urban developments tend to eradicate or severely impact functioning ecosystems and exaggerate other associated problems that have associated economic costs. These costs include reliance on overburdened infrastructure, community disconnectedness, light and noise pollution, urban heat island effect, air and water pollution and a reduction in individual health and safety (Frenkel and Ashkenazi, 2008). However, because of market failures, the economic costs that arise from the loss of ecosystems and ecosystem services are not included in fiscal impact analyses, as the analysis only quantifies the direct and current impacts of a proposed development and not the spillover effects on to the environment (Fausold and Lilieholm, 1999).[54] Although there was no conscious goal to achieve such an irrevocable modification to the landscape that has future economic costs associated,

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[52] Established as a bylaw, zoning is a means to guide land-use and land-cover change by directing how and where development can occur by controlling lot sizes, permitted use, density, setbacks, conservation zoning. The problem with zoning is that it is not fine-grained enough to account for site-specific ecosystem function; nor does it account for the value of the services provided (Environmental Law Clinic *et al.,* 2007). Other legislative requirements, like the Riparian Area Regulation (RAR), that establish buffer zones tend to be ineffective as well, because ecosystem degradation is not avoided; rather, it continues indirectly (CWP, 2003; May, 2003).

[53] Legislators have typically enforced a top-down approach to ecosystems that result in laws, incentives, penalties and agreements to preserve and control ecosystems (Holling and Meffe, 1996; Hindmarch *et al.,* 2006). Focused simply on the protection of ecosystems, many policy instruments have neglected the importance of ecosystem function, overlooking the underlying issues that cause the degradation of functional ecosystems in the first place (*e.g.,* non-point source pollution, agriculture, inadequate buffer zone widths, aggregation of many small hydrology changes in a watershed).

[54] The fiscal impact analysis only assess the additional economic costs of providing social and structural services and compares these cost to the revenues earned from a development if it was allowed to proceed. The analysis is quite limited, as it does not consider the longer-term impacts and costs such as future residential growth spawned by the proposed development or the future replacement of infrastructure as well.

Arendt (1996) contends that urban sprawl and its associated costs have come about because "minimal thought has been given to the basic development design standards contained in local ordinances which ask little, if anything, with respect to [the value of] conserving open space" (p.6).

This current imbalance is due to the fact the functional importance and value of the ecosystem services are not internalized in land-use planning frameworks. What is therefore required is an alternative design process that recognizes the importance and value of functional ecosystems as an integral component in development design. Because the alternative design process would require both the evaluation of ecosystem function as well as an assessment of the economic value of the ecosystem services, the proposed approach described in Chapter 4 could be utilized in an integrated design and planning (IDP) process.

5.2.2 Integrated Development Design: BPP as a Case Study

Ignoring the aggregated economic costs or the spill-over effects that arise because of the way conventional developments are designed and built places entire communities at risk as these communities may not be able to accommodate changes arising due to climate change with minimum disruption or cost (Snover *et al.,* 2007). Rather, the cost and disruption of the changes arising from climate change could be immense.[55]

Given that it is the responsibility of local governments to guarantee the provision of services and infrastructure and improve the quality of life by ensuring the safety, health and welfare of their communities now and into the future (Costa, 2008), it is becoming increasingly apparent that governments must begin to "anticipate trends and changes that

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[55] Human induced climate change - *i.e.,* the significant and long-lasting changes to current weather patterns - is anticipated to manifest in environmental changes including: increased climate variability, increased frequency and duration of extreme events, increased biodiversity loss, increased species extinction rates and catastrophic changes to ecosystem function (IPCC, 2007). From an economic and social standpoint, changes in both the climate and the environment are anticipated to result in impacts to municipal water resources/supplies, shortfalls in food production, greater competition for hydropower and natural resource supplies, increased pollution, infrastructure failure/overload, changes in demographics, economic instability and increased rural poverty (Snover *et al.,* 2007; IUCN, 2009).

could affect their environment, economy, and community wellbeing" (Snover *et al.,* 2007, p.27). Because conventionally designed developments have a significant influence and oftentimes negative impact on the natural, economic and social environment, many local communities are demanding that developments and local governments incorporate goals of sustainability into their design process and regulations in order to reduce climate change risk (MEA, 2003; Costa, 2008).

Such a case occurred with the BPP development in West Vancouver, B.C., when a conventional area development plan for the 215-acre parcel of land was presented to council in 1995, 2000 and 2004 and was refused on all three accounts by council and the community (Boyle, Pers. Comm). The proposals were denied approval on the basis that the proposed conventional development would result in an unsustainable development. After the third refusal and the release of the Official Community Plan (OCP) in 2004, which mandated that the BPP Area Development Plan (ADP) achieve a leading edge sustainable community, BPP could no longer undertake the conventional design process (Boyle, Pers. Comm.). Rather, BPP had to undertake an integrated approach to development design that adhered to four community-building principles described by the OCP (BPP, 2008):

- Create a strong community;
- Establish a sensitivity and connection to the natural environment and mountain qualities;
- Encourage a diverse community;
- Focus on environmental and economic sustainability.

It is these four community-building principles that have guided the development of the key organizing principles for the 2008 Rodgers Creek Area Development Plan (Appendix D). According to Boyle (Pers. Comm.), the BPP integrated development design process was split into three phases: 1.) technical analysis; 2.) working group; and, 3.) bylaw development.

Phase one of the design process was to collect the technical information of the landscape to which the development would be situated. Rather than ignoring the features and values of the landscape and allowing zoning ordinances to guide the development design as conventional developments do, the BPP process utilized Ian McHarg's method of ecological planning by letting the landscape inform the design (McHarg, 1972). McHarg's method was to separately map individual variables of the landscape to "distinguish the capacity of each area for development, the susceptibility to despoliation and the restraints and opportunities inherent in the landscape" (McHarg, 1972, p. 73).[56] In the case of BPP, this involved the collection of technical background data on cultural values, as well as environmental (vegetation communities), recreational (trails) and topographic features (slope, soils, *etc.*) of the landscape (Appendix D). This included the assessment of the functional condition of aquatic ecosystems. Using the Proper Functioning Condition (PFC) assessment, which characterizes aquatic ecosystems based upon their level of functional condition (*i.e.,* Proper Functioning Condition (PFC), Functional-at-Risk Condition (FAR; with an upward or downward trend) or a Non-Functional (NF) condition), all aquatic ecosystems within the Rodgers Creek development area were assessed by an integrated team including an aquatic ecologist, hydrologist, vegetation ecologist, fish biologist and geotechnical engineer and an economist (see Appendix C for a description of PFC).[57] Once all the features were separately mapped, they were then combined to reveal the complexities of the landscape (areas that had the highest and lowest ecological value) in order to identify where pockets of development could be located - called a sieve analysis (Appendix D).

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[56] The theory behind this mapping exercise is to develop a visual tool to display spatial information in a quick and concise manner for the public.

[57] The PFC assessment takes place at two scales: the hydraulic reach scale in order to identify and assess the aquatic ecosystem under evaluation and the watershed scale as most land alteration activities affect the hydrological response of watersheds (Alberti, 2005). The PFC assessment is based upon identifying the capability and potential of the system, based upon the historical condition of the landscape prior to human activities. Potential is defined as the potential natural community an aquatic ecosystem could achieve "given no political, social, or economical constraints" (Prichard, 1998, p. 6). Capability is defined as what could be achieved given current the political, social, or economical constraints. The PFC assessment enables users to identify and prioritize areas of restoration and can operate as design criteria as well (Malmkvist, 2002). However, the PFC approach is limited in that it can only assess aquatic ecosystems.

Embarking on a citizen involvement approach, Phase 2 of the development process involved creating an interdisciplinary team that consisted of district staff, council, community groups, volunteers, landowners and technical experts (Boyle, Pers. Comm.). The purpose of the group was to engage in a deliberative process in order to envisage "a future community for the Rodgers Creek area and establish detailed principles for the area development plan" (Boyle, Pers. Comm., p.4). The purpose of the IDP process was to ensure that all groups could provide input into the development design process to ensure that concerns were recognized and adequately addressed prior to presenting the development plan (conventional developments take a more combative approach). Within a time frame of 14 months and under the guidance of the four community building principles, the group established key organizing principles for the area development plan, undertook two open houses to solicit community input on the direction of the plan, reviewed and modified the plan as technical issues arose and met with council twice (Boyle, Pers. Comm.). By the end of the fourteen months, the proposed area development plan was presented to council that described six distinct development areas (pods) with two associated housing mix options.[58] Along with an amenity package, a fiscal impact analysis for each housing option was also presented.[59] Once the Area Development Plan (ADP) was approved[60], phase 3 of the development process consisted of the drafting of three bylaws, which were adopted after third reading, to set out the development permit guidelines based upon the ADP for the Rodgers Creek area.[61]

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[58] Two housing mix options were presented. Option A was based on standard zoning ordinances for the area (2.5 units per acre), which would result in 536 units with an average apartment size of 2,255 sq. ft. Option B is based upon determining the maximum total square footage for floor area (1,875,000sq ft) and from this value, a more socially diverse housing mix of 736 units was calculated. Option B was recommended by the integrated team and was also approved by council. It is estimated that the Option A would yield a revenue of $462 million, whereas option B's revenues are estimated to be $378 million (BPP, 2008; Vancouver Real Estate Direct, 2008). However, the value of the lost or preserved ecosystems is not included in these calculations.

[59] Although the integrated design team implicitly recognized the importance of the functional ecosystems within the development area, the economic values of the ecosystem services were not incorporated into the fiscal impact analysis. BPP's amenity package includes the integration of green building standards, provision of a service site of 0.8 hectare for District use, a $7.94 million cash contribution, the preservation of certain mountain biking routes and the extension of a collector road (Boyle, Pers. Comm.).

[60] The ADP achieved 4th reading without any opposition (Lucey, Pers. Comm.).

[61] The by-laws support the rezoning of the 215 acres to allow for 736 units. Under the Local Government Act, the zoning exempts the Rodgers Creek Development from any zoning changes for 10 years to secure the amenity package provided by BPP. And finally, the zoning amendments created one zone for the entire parcel of land to provide "density certainty with measured flexibility" (Boyle, Pers. Comm., p.5).

As a result of the integrated “science and community” based design process, the proposed BPP development preserves 55% of the land that has the highest ecological and social value to the community (Croll, Pers. Comm.). Rather than another constructed hillside of homes where ecological and social values are secondary to the economic criteria, the Rodgers Creek development is proposed to be a network of forested multi-use neighbourhoods that recognize the importance of preserving ecosystems in order to incorporate the ecological and social values into the development. Recognizing the importance of functional ecosystems in development design, the area development plan states that creeks and sensitive areas will be protected, creek crossings minimized and low impact trails built. In addition, the six development pods and the transportation network connecting these pods are designed to have the smallest footprint possible as well as incorporate design principles that reduce the indirect impact that urban developments tend to have on the landscape (*e.g.,* usage of green infrastructure in design to limit the impact on the hydrology of the landscape, integrated stormwater management planning, avoidance of fragmentation, preservation of animal corridors and habitat). Although not quantified in development planning or the fiscal impact analysis, the preservation of 55% of the landscape ensures that the untouched areas will result in the continued provision of economically beneficial ecosystem services such as the storage and sequestering of carbon, downstream flood avoidance, reduction in air pollution and the regulation of local climate over time.[62]

Undertaking an integrated and collaborative design process that incorporated principles of sustainability fundamentally changed the design of the traditional BPP development to one of a sustainable community.[63] From a developers’ perspective, the IDP process is perceived to be costly and time consuming up front; however, engaging with the community and government using an integrated design process oftentimes results in faster approval times and avoided construction interest costs, marketing advantages and

[62] Although the economic value of the ecosystem services was not quantified for the Rodgers Creek development area, direct-use values (*e.g.,* mountain biking, recreation) and non-use (*e.g.,* preservation) values were expressed by community members during open houses and information sessions.

[63] Robinson *et al.* (2008) characterize a sustainable community as a dense, mixed land-use community that incorporates sustainable water and transportation systems, a net zero energy use system and a diverse local economy.

value appreciation, as was the case with the BPP development (Lucey, Pers. Comm.). Because sustainability was incorporated into the design principles of the development itself, a high-density, clustered type mixed-use style development that incorporates an extensive network of trails as well as a system for pedestrians, cyclists and transit in order to reduce the dependency on vehicular transportation has been proposed. Green buildings, which are designed to be energy efficient, have also been incorporated into the development design, thereby reducing energy dependence and waste. Finally, the functional condition of the ecosystems has been preserved. Undertaking an IDP process is proactive and is "more effective and less costly than responding reactively to climate change impacts" as it reduces future risks associated with conventional development design and a changing climate (Snover *et al.,* 2007; Costa, 2008).

5.3 Discussion

Because the goal of the IDP process is to engage in a deliberative process that enables an integrated design team to make consensus based decisions regarding the design of the sustainable communities, the integrated approach proposed in Chapter 4 can apply quite well to the process. For instance, ecosystem valuation approaches like the group valuation approach (see Chapter 3), incorporates public input (though stakeholder deliberation) as an attempt to address the social equity question of how ecosystem goods and services can be valued in such a way to ensure fair treatment among competing stakeholders (Wilson and Howarth, 2002). The ecosystem valuation exercise helps identify and reveal the interests of various groups or individuals as well as provide insight into the role and the values held for functional ecosystems. Thus, when applied within an IDP process, the proposed valuation approach could be used to derive meaningful estimates that stakeholders have for a particular ecosystem by assessing the economic value of the ecosystems to both developers and stakeholders.[64] By assessing the economic value of ecosystem services in development planning, the developer can

[64] Although such a process is not likely to result in "a complete convergence of values", Wilson and Howarth (2002) contend that the process of deliberation can achieve a form of compromise to which the best interests of society are achieved, rather than simply an individual's preference (p.436). Wilson and Howarth (2002) describe the philosophy behind the deliberative process, how it relates to social equity in detail as well as the process and rules. Therefore, it will not be explored further in this thesis.

identify and preserve ecosystems that add financial value to the development (*e.g.,* amenity value, reduced infrastructure costs).

The approach would also help the integrated design team clarify the ecological tradeoffs between various land-use/development scenarios. For instance, using GIS, the proposed approach could be used to inform an additional layer in the sieve analysis by identifying areas of high economic value in terms of ecosystem services (Figure 8) and the subsequent the areas of high ecological risk (Figure 9). The result would be a more informed development (similar to BPP's proposed development), that is designed in such a fashion to preserve or enhance the function of ecosystems, whilst capitalizing on the highest and best use values that can be derived from a functional ecosystem, which translates into increased economic value to the developer.[65] In addition, the mapping exercise could also be used to identify ecosystems currently at high risk and high value, thus, enabling local government to prioritize rehabilitation efforts accordingly.[66] This is of increasing importance as local governments are looking to utilize ecosystems as ecological infrastructure as a means to build adaptive capacity in communities (Snover *et al.,* 2007).

In summary, the proposed approach could help decision makers "balance the impacts of different planning options on the economic accounting of a region, and guide them in selecting sustainable and economically feasible development strategies" (Gret-Reganey *et al.,* 2008, p.157). However, to be able to choose among the economically feasible development strategies in order to design sustainable communities it is necessary to understand how the provision of valuable ecosystem goods and services is affected and therefore the evaluation of the functional condition of an ecosystem must also be a part of

[65] Quantifying the economic value of the ecosystem services preserved within a landscape could also be used to the developers benefit. For instance, the outcome of the proposed ecosystem valuation/evaluation approach provides a quantified value of the ecosystem services being preserved. The developer could use this information to negotiate for additional Transfer Development Rights (TDR's) or density bonusing as well as reduce the cost of the amenity package paid to the community (Croll, Pers. Comm.).

[66] For instance, the current functional condition and value could be assessed against a desired future functional condition and value. Based upon the current trajectory, decision makers could prioritize efforts based on the ecological information as well as the economic cost of lost ecosystem services and thus prioritize rehabilitation efforts accordingly. Such a process could be a means to plan for climate change.

the analysis and design. Hence, the recommended integration of both ecosystem valuation and evaluation assessments as the proposed approach would provide decision makers with the necessary ecological information as well as the economic value of the ecosystem services. Thus, in answering the question, the proposed ecosystem valuation/evaluation approach could be utilized in the IDP process as the approach acts as a means to create a common language to which environmental and economic decisions can be compared. However, although quite applicable to integrated design for the purposes of designing sustainable communities, the application may be limited as many barriers to the development of sustainable communities remain.

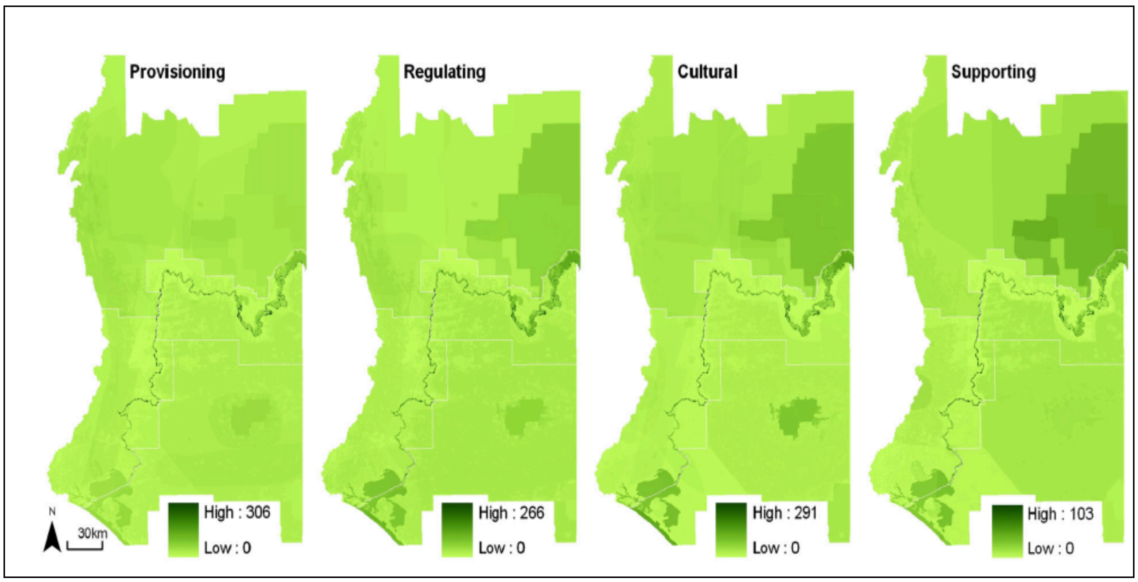

Figure 8. Example of mapping estimated ecosystem service values. *Note:* The authors used the Millennium Ecosystem Assessment's ecosystem typology to describe and organize the ecosystem services as seen above in each of the pictures. The purpose of this example is to show how the valuation of ecosystems could be mapped and utilized as a layer in the sieve analysis. From "Mapping community values for natural capital and ecosystem services" by Raymond *et al.,* 2009, Ecological Economics, p.1310. Copyright 2008 by Elsevier B.V. Reprinted with permission of author.

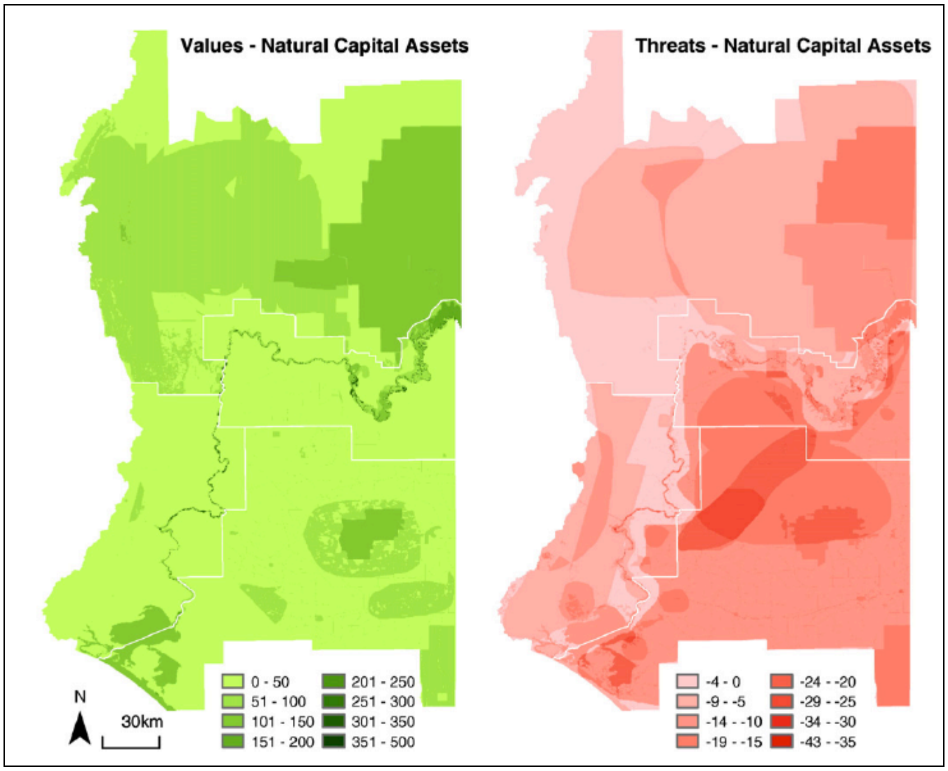

Figure 9. Example of mapping ecosystem service values and ecological risk. *Note:* In the above example, utilizing a public forum, the authors asked the public to identify areas of high risk (*i.e.,* the threats). The purpose of this example is to demonstrate how areas of ecological areas could be assessed and mapped for risk (on the right) using both aquatic and terrestrial risk assessments, which can be then compared to the mapped ecosystem values (on the left). From "Mapping community values for natural capital and ecosystem services" by Raymond *et al.,* 2009, Ecological Economics, p.1311. Copyright 2008 by Elsevier B.V. Reprinted with permission of author.

5.3.1 Barriers

Although ecosystem valuation is a useful means to help decision-makers identify and quantify stakeholder's values for ecosystems or ecosystem services, the debate on whether or not to value ecosystems still rages on. Bringing ecosystem valuation into what is normally a combative situation between the developer and stakeholders may only delay the process further, thus increasing costs, as the topic of valuing ecosystems remains a contentious subject. In addition, if a valuation exercise was applied, some developers are likely to select and value the ecosystem services with the highest value, as this value could perceivably be used as a negotiation strategy to offset the value of the amenity package paid to the community.[67] However, this may not be such a concern, as the total value of ecosystems is oftentimes so high it not believed to be an accurate estimate.

There are considerable barriers within local government that inhibit the implementation of sustainable communities. As noted by Robinson *et al.* (2008), the organizational culture within local and provincial governments tend to be structured around single-discipline (silo) thinking which tends to discourage collaboration across disciplines and groups both internally and externally. Other notable problems include: a lack of leadership within the governmental organization; authentic power of sustainability departments (either they do not exist or, if they do, they have little authority and decision making power); and there is limited knowledge or experience due to limited resources available (Robinson *et al.,* 2008). Finally, there is oftentimes no feedback mechanism, which allows local and provincial government to monitor the effectiveness (or ineffectiveness) of policies developed by the local or provincial government regarding urban development. These are oftentimes outdated and are not flexible enough to allow for innovation. Because these barriers exist, urban developers are more inclined to continue applying the conventional design process to urban development, as there is less financial risk associated.

5.3.2 Recommendations

Considerable barriers to the development of sustainable communities that lie within local government need to be overcome. To reverse this trend, innovative and forward thinking policies and complimentary regulations will have to be developed with the input of both the private and public sector if the goal is to develop more sustainable communities. Until local and provincial governments mitigate the risks to urban developers by changing

[67] BPP's amenity package includes the integration of green building standards, provision of a service site of 0.8 hectare for District of West Vancouver's use, a $7.94 million cash contribution, the preservation of certain mountain biking routes and the extension of a collector road (Boyle, Pers. Comm.). In total, the amenity package is approximately $20 million in value (Vancouver Real Estate Direct, 2008).

policy/regulations and creating organizational cultures that are innovative and embrace sustainability, conventional developments will continue to be built. A shift in culture that nurtures innovation and collaboration amongst departments so that sustainability initiatives can be adequately supported and implemented is needed. Such strategies to achieve this could include a change in job descriptions from single disciplines to those that require interdisciplinary knowledge and understanding of sustainability and climate change whilst designating a central and authoritative group that "institutionalizes strong inter-divisional collaboration" (Robinson *et al.,* 2008, p.12).

Because many local governments are limited by funding, the private sector can be engaged to play an greater role in stimulating demand for new and innovative sustainability orientated communities. However, in order to leverage the private sector and create meaningful opportunities, policy and regulations must be complimentary, locally relevant and practical. For instance, establishing the following policies would help achieve the goal to create sustainable communities (Barraclough and Hegg, 2008a):

- Use the development review process to consider the use of variances and density bonusing to secure or restore public amenities (*i.e.,* open space, riparian areas, landmarks, or cultural features) that provide valuable ecosystem services.
- Utilize development control bylaws to achieve a more appropriate development in terms of streetscape, pedestrian environment, view protection, overall site design, and compatibility with the function of the landscape.
- Support the understanding of growth management and sustainable development best management practices (*i.e.,* low impact development, changing policies to focus on maintaining and enhancing ecological function, rather than determining buffer strips, allow the future proofing of buildings), through public events and online and printed information.
- Encourage accessibility through the incorporation of building support systems as design features and where appropriate, make them visible to the public (*i.e.,* green roofs, white roofs, energy retrofits and water use monitoring).

- Establish protocols that require the evaluation, and valuation, of the impact that new developments will have on ecosystems (*e.g.,* loss or gain of valuable ecosystem services) using a fiscal impact analysis.

For policy and regulations to be effective and maintain relevance, measuring how local and provincial governments inhibit the acceleration towards building more sustainable communities, using a feedback-loop that ties into policy decisions that measure the effectiveness of policies and regulations regarding sustainability and climate change is needed (Robinson *et al*., 2008).[68]

5.4 Conclusion

In planning and designing urban developments for a changing environment, sustainable development strategies, like the IDP process, are critical to creating behavioral change both in the private and public sector. The BPP study is a case in point as the IDP process engaged the developer, district staff, council, community groups, volunteers, landowners, technical experts to work together collaboratively, rather than confrontationally in order to achieve a desired objective - a leading-edge sustainable community. However, a major challenge in urban planning and design is in communicating the social importance, and value of, functional ecosystems to urban developers and decision makers in such a way that the goal shifts from simply developing the landscape to one that allows the landscape to inform the design via a manner that enhances triple bottom line values.

The economic valuation of functional ecosystems could be one way to help address this challenge as the incorporation of the proposed ecosystem valuation/evaluation approach into integrated design could provide the means to which the fair comparison of various land-use options can commence on both economic and ecological criteria. Measuring the function of an ecosystem and then applying financial values to the services provided would be an extremely useful tool to enhance developers' and/or the municipalities'

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[68] This would first require an integrated team to identify criteria for measurement, develop a baseline, and to define what a successful policy would look like prior to deployment. Periodic assessments would be required.

ability to evaluate trade-offs between ecosystem management regimens and social actions that may alter the function of an ecosystem. Furthermore, the proposed approach when utilized in integrated design could contribute to building the much needed business case to plan urban developments (like BPP) in such a way that preserves / enhances the functional condition of ecosystems and financially benefit from tangible and less tangible ecosystem services. The proposed approach could also act as an audit tool thereby providing the financial and municipal sectors with a system to evaluate, and manage risk in, development applications and investment, while optimizing for the functionality of the ecosystem, thereby influencing policy and the private sector to develop ecologically sensitive developments. However, for the approach to make a legitimate contribution to the advance of sustainable communities, it must be structured and utilized in an IDP framework.

Although considerable barriers to sustainable communities exist, society must begin to recognize that "man and nature are indivisible, and survival and health are contingent upon an understanding of nature and her processes" (McHarg, 1972, pp. 73). Therefore, the best way to preserve functional ecosystems is to manage human development in a holistic/systems thinking approach that is based upon social, environmental and economic criteria, not simply based upon economics alone.

Chapter 6

6.1 Conclusions

The following key research questions of this thesis are revisited and discussed in light of the findings of the previous chapters.

Research Question 1: Should we value ecosystems?

The valuation of ecosystems continues to remain a contentious topic as many researchers are opposed to the exercise on moral grounds, asserting that human well-being is dependant upon the stable and continued provision of ecosystem services. Others contend that any form of ecosystem valuation may actually provide society a disservice. Many of the arguments against the valuation of ecosystems may actually be moot as society and economic markets, for the most part, do not explicitly value ecosystems or measure the impact that decisions have on ecosystems. Ignoring the value of ecosystems has resulted in the degradation and loss of ecosystems and ecosystem services as the costs of degradation are not accounted for in most economic decision frameworks.

Although ecosystem valuation methods are imperfect, the valuation exercise can be beneficial as a means by which ecological costs and values can be adequately represented in decision making processes. Thus, in the context of urban development where the building of homes and cities is "not unity with nature, but conquest… and that is built upon economics" (McHarg, 1972, p.24), ecosystem valuation can help decision makers to evaluate the long-term consequences of ecosystem degradation/loss as a result of economic objectives. Therefore, the answer is yes - ecosystems should be valued. However, this answer has a qualifier as the value of ecosystem services cannot solely be used to determine what social actions are necessary, as that requires an integrated and multidisciplinary approach. Ecosystem valuation is a means to help achieve decision making goals regarding ecosystems, but it is not an end in itself.

Research Question 2: How do we value ecosystems?

The review of the ecosystem valuation methods and procedures of "how to" implement a valuation study revealed that there is no explicit requirement to the measure the functional condition of an ecosystem. Rather, the functional condition is assumed.

The problem with not accounting for the functional condition of an ecosystem in a valuation study is that the estimated ecosystem service values are not adjusted for the risk of uncertainty that the ecosystem is damaged or at-risk of degradation and may fail to provide the same amount of ecosystem services in the future. More importantly, the estimated values do not provide the critical information decision makers require, including the current functional condition, the direction or changes within ecosystems, the quality of ecosystem services and the urgency with which each may be occurring or changing as a result of human-made or natural perturbations. In the context of urban development and planning, if ecosystem valuation estimates are to be used to rank priorities, to evaluate the effects of various development options, to provide estimates of damages that have occurred or to help explain why areas should be preserved or rehabilitated, the functional condition of the ecosystem being valued must be assessed as part of the valuation exercise. Without this vital information, the valuation exercise has resulted in a simplified value conclusion that provides little information on how to manage the ecosystem or the uncertainties in its long-term health. Therefore, a new valuation method is required, one that builds on the existing strength of ecosystem valuation methods, that incorporates knowledge on thresholds and various properties of ecosystems as well as integrates the ecological understanding that functional ecosystems provide ecosystem goods and services that contribute to economic value.

Research Question 3: Is the estimation of ecosystem service values an accurate reflection of ecosystem function?

Using work completed by D. Hegg and L. Townsend, a valuation of the Swan Lake Watershed and the Swan Lake/Christmas Hill Nature Sanctuary was undertaken and then compared to the functional condition of the watershed. The two valuation analyses undertaken at a watershed ($35,211,905 per year) and the Nature Sanctuary ($3,153,667), when compared to the ecological information on the areas, do not suggest that there are ongoing water quality problems, eutrophic lakes or kilometers of degraded stream and aquatic habitat. When the watersheds annual estimated ecosystem service value for 1858 is compared to the current annual value estimate for 2009, it becomes increasingly clear that ecosystem degradation and loss has occurred over time. Thus, the estimated values of the watershed and the Nature Sanctuary do not reflect the loss of ecosystem services due to past urbanization and agricultural activities. Had the lost values been estimated, the value of the ecosystem services for the Swan Lake Watershed and Nature Sanctuary would have been reduced to a lower, if not negative, value. However, based upon the current valuation estimates alone and without reference to the functional condition, the values are likely to suggest to decision-makers that the watershed is in a good functional condition, when in-fact, the overall health of the watershed is in a poor condition of health and its resilience to disturbance is low (Townsend, 2009).

Using current valuation methods, ecosystems are continuously being valued for their aggregated ecosystem service values and not for their functional value. Thus, the answer to the above question is no. Recognizing the challenges and barriers associated with undertaking a more holistic valuation of ecosystems – *i.e.,* valuing the functional condition of an ecosystem – an integrated approach that measures both the functional condition of an ecosystems as well as value the ecosystem services is proposed. In application, the assessment of an ecosystems' functional condition would be completed when an ecosystem valuation study is undertaken and would provide the ecological context that is inherently missing from current economic estimates of ecosystem services. Applied to post-development areas, such an approach would enable decision makers to manage ecosystems in order to achieve or maintain a functional condition to attain sustainable production of desired ecosystem services within the landscape context, rather than setting priorities to manage ecosystems for single ecosystem services without regard to the functional condition. Thus, the information provided by the approach would better assist decision makers in determining which ecosystems are at risk and have the most value, thereby enabling decision makers to rank and address ecosystem priorities (including rehabilitation) more effectively.

Future research efforts will need to focus on developing an integrated valuation/evaluation approach that adjusts the estimated ecosystem service value for the functional condition, but also provides a risk-rating assessment to provide the necessary ecological information to a decision maker. This will require the development of a standardized multiple criteria decision analysis framework that relates the provision of ecosystem functions and the value of the services to the level of the risk associated given the current and future landscape context. The framework should also be a multi-disciplinary and multi-stakeholder interface that enables various groups to work in collaboration with experts.

Research Question 4: Can an integrated ecosystem valuation/evaluation approach be utilized in urban development planning and design?

Because the traditional design process continues to focus on the built and economic/social components of development, nature is often treated as a valueless commodity. The result is urban sprawl that has long-term economic, social and environmental costs that increase the vulnerability of communities to the risks associated with an unprecedented and changing climate.

In order to address the aforementioned question, it must first be recognized that the traditional design process is a linear process that can be characterized by various tradeoffs, which tends to oversimplify causal factors. In contrast, integrative thinking involves a multidirectional and systems thinking approach in order to identify creative solutions to a problem or process (Martin, 2007). Simply put, the conventional design process views nature as a liability; whereas, the integrated design and planning (IDP) process, views the environment as an asset. Related to this approach is that the IDP process recognizes the importance of preserving, enhancing and maintaining healthy/functional ecosystems as a part of development design as these systems are characteristically resistant and resilient to external shocks. The IDP process also implicitly recognizes the value of the ecosystem services that are provided by functional ecosystems. Because the IDP process requires the measurement of ecosystem function, as well as recognizes the value of the services provided, the proposed approach would apply quite well to the IDP process.

Specifically, the proposed approach could be incorporated into the IDP process to help the design team render informed judgements regarding the functional condition of ecosystems and the value of the ecosystem services held by the group and, thus, clarifies the ecological tradeoffs between various land-use/development scenarios using a sieve analysis. The proposed approach would also contribute to a needed business case method which proves that when urban developments are planned using an integrated approach that preserves / enhances the functional condition of ecosystems, there is the opportunity to financially benefit from tangible and less-tangible ecosystem services. Simply put, it is more profitable to the developer (*e.g.,* Barraclough and Hegg, 2008a). Because of the IDP process, the local municipality and surrounding community also benefit as local ecosystems are preserved / enhanced and, thus, avoid the long-term economic costs associated with ecosystem degradation arising from land-use and a changing climate.

6.2 Discussion

6.2.1 A Changing Climate

Recognizing the implications of a changing climate, many local governments are developing policies that incorporate adaptation and mitigation strategies in order to increase the adaptive capacity as a risk reduction strategy to ensure continued economic and social progress of communities (Snover *et al.,* 2007). In general, climate change adaptation strategies are those that "prepare the current landscape and its inhabitants for the new climate, whereas mitigation strategies are those that attempt to slow down the process of climate change" (Mazza, 2008, p.1). These strategies are being implemented to reduce the risk and, thus, improve the resilience of a communities and ecosystems.

Akin to ecosystem valuation and evaluation approaches, both climate change mitigation and adaptation strategies compliment one another and neither are likely to be as effective if employed alone (Mazza, 2008). "For cities to effectively respond to global climate change, both mitigation and adaptation strategies – and economic markets for them – will be required" (Grimm *et al.*, 2008, p.758). An adaptation strategy, such as an ecosystem evaluation assessment, applied alone might describe a level of risk, but without mitigation strategies like ecosystem valuation and available economic markets, the risk information does not translate into financial criteria in land-use decision frameworks. In the context of the proposed integrated ecosystem valuation/evaluation approach, land-use decisions to convert a landscape from a natural setting into residential / industrial areas are based upon economic criteria, not ecological, due to market failures. Therefore, an adaptation strategy designed to preserve ecosystems without a complimentary mitigation strategy and supporting economic markets are more likely to fail, as there is no financial incentive to preserve ecosystems.

In planning and designing urban developments for a changing climate, mitigative and adaptive strategies are "critical to the arenas of governance, decision making and behavioral change" that result in sustainable communities (Robinson *et al.,* 2008, p.3). The BPP case study is a case in point as the IDP process engaged district staff, council, community groups, volunteers, landowners, technical experts, as well as the developer to change their previous behaviours and work together collaboratively, rather than confrontationally to achieve a desired objective - a sustainable community. Therefore, local governments must begin undertaking an integrated and collaborative approach that deviates from the conventional design process to a design with nature approach that recognizes the importance and value that functional ecosystems have within an urban development. However, to transition from conventional development design to integrated development design - that results in sustainable communities - there must be a transition from a "resource-based economy to [an] ecosystem-based economy that recognizes and values environmental goods and services" (Harford, Vanderwill and Church, 2008, p.3). To support this shift in thinking, local government and associated policies/regulation must change if the goal is to create more sustainable communities. Until local and provincial governments mitigate the risks to urban developers by changing policy and creating cultures that are innovative and embrace sustainability, conventional developments will continue to be built.

6.2.2 Future Applications

Integrated Resource Management (IRM)

Combining whole systems thinking and integrative design, the Integrated Resource Management (IRM) approach is an urban resource management philosophy that takes a city-wide approach to "integrating the design of [urban developments] in accordance with ecological principles" (O'Riordan, Lucey, Barraclough and Corps, 2008, p.1). The approach itself is the next step in integrated urban planning and design as it is governed by the same principles to those of BPP; however, in application, IRM differs in it's approach to development design. Specifically, using the IRM philosophy, the design and planning of an urban development is no longer guided by accounting for the costs of managing urban wastes as most developments do (including BPP's Rodgers Creek Development). Rather, the IRM approach recognizes the value in using wastes as resources by removing heat from sewage to heat buildings, converting liquid and solid wastes into fuel and to utilize functional ecosystems to act as both green and ecological infrastructure.[69] Thus, in regards to the proposed integrated ecosystem valuation/evaluation approach, the IRM philosophy recognizes, measures and capitalizes on the social importance and economic value of sustained ecosystem services provided by functional ecosystems.

In application, the IRM approach proposes to reduce the production of Green House Gases (GHG), reduce landfill costs, create opportunities to restore depleted groundwater supplies, reduce potable water consumption, lower energy use, reduce land costs associated with solid and liquid waste facilities and rehabilitate urban ecosystems whilst completely changing the design of urban landscapes. The IRM approach can also be employed to retrofit existing cities and guide new developments and thus can be used as a climate change strategy to increase the adaptive capacity of communities. However, for

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69 The term ecological infrastructure refers to both functional aquatic and terrestrial ecosystems (*e.g.,* building engineered wetlands for storm water retention and water quality) that provide valuable ecosystem services to the municipality and surrounding community (Postel, 2008). Green infrastructure is a "structured by a hybrid hydrological/drainage network, complementing and linking relict green areas with built infrastructure that provides ecological [services]" (Ahern, 2007, p.267). The following are examples of green infrastructure (*e.g.,* Project Urban Rain Garden – see Hegg, 2008) and ecological infrastructure projects (*e.g.,* Willowbrook/Glandford Subdivision, Baxter Pond – see Barraclough and Hegg, 2008a).

IRM to be effectively implemented at either a urban development or city scale, complimentary policy/regulations will need to be changed and economic incentives, such as carbon markets, required (O'Riordan *et al.* 2008).

6.3 Concluding Remark

The goal of this thesis was to examine the link between ecosystem function and an estimation of ecosystem service values. Because of the way in which the total value of an ecosystem is calculated, it is not made apparent that the economic value of ecosystem services is dependent upon the ecological function of an ecosystem. In the context of urban planning and development, not only is the value information misleading, users of the information cannot determine how the various functional conditions of ecosystems will be influenced by development, nor can they assess how the values will change. It is, therefore, hoped that ecosystem valuation researchers will begin to shift their focus from simply improving and refining existing methods to developing new methods that attempt to value the functional condition of an ecosystem. Such an understanding of both the function - and value - of an ecosystem would enable decision makers in any discipline to be able to make more informed decisions on sustaining ecosystems in an unprecedented and changing economic, social and environmental climate.

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Appendix A. Methods

A.1 Urban Forest Effects Model (UFORE)

Developed in the 1990's by researchers at the United States Department of Agriculture (USDA) and Forest Service, UFORE is a science-based, peer-reviewed computer model designed to calculate urban forest ecosystem service values based on field data inputs and available data sets from external sources (*e.g.,* weather and pollution data sets).

The UFORE process involves collecting quantitative and qualitative field measurements (*i.e.,* identifying species, diameter at breast height, crown height and width, crown condition, light availability, die-back, leaf area and shrub understory) from randomly or subjectively selected plots using quadrant sampling. The data is entered into an excel spreadsheet, and completed spreadsheets are then uploaded into a database, or sent directly to the UFORE researchers (US Forest Service), for processing. The UFORE model uses local air pollution values, meteorological data and structural characteristics in order to quantify and monetize the ecosystem services provided by urban forests for a given area. The results are then statistically extrapolated upward to estimate totals for a study area. The UFORE model has been used to date in approximately 50 urban forest studies (Nowak *et al.*, 2008).

UFORE calculates the following (USDA Forest Service, no date):

- *UFORE-A: Anatomy of the Urban Forest. The model quantifies urban forest structure (e.g., species composition, tree density, tree health, leaf area, leaf and tree biomass) based on field data. Field data are collected by two means; an inventory of all the trees is undertaken or a representative sample of the population is selected. Data collection also includes identifying land use (e.g., street tree) and distance and direction to nearby buildings (if applicable).*

- *UFORE-B: Biogenic Volatile Organic Compound (VOC) Emissions. The model quantifies: hourly urban forest VOC emissions (isoprene, monoterpenes, and other VOC emissions that contribute to O3 formation) based on field, specie* *specific characteristics (e.g., leaf-on season) and meteorological data (e.g., local air temperature, weather and leaf-on season), and O3 and CO formation based on VOC emissions.*

- *UFORE-C: Carbon Storage and Sequestration. The model calculates total stored C, and gross and net C sequestered annually by the urban forest based on field data. The monetary values of carbon storage and sequestration are based upon the social cost of carbon.*

- *UFORE-D: Dry Deposition of Air Pollution. The model quantifies the hourly amount of pollution removed by the urban forest and associated percent improvement in air quality throughout a year. Pollution removal is calculated for O3, SO2, NO2, CO, and PM10 based on field, pollution concentration, and meteorological data. Values were based on median per ton of pollution externality values that are used in energy-decision-making studies. These values, in dollars per metric ton (t) are: NO2 ($6752/tonne), PM10 ($4508/tonne), SO2 ($1653/tonne), CO ($959/tonne). The value for O3 was set to equal the value for NO2 (Nowak et al., 2006). These monetary values of pollution removal are based on avoided externalities – i.e., the costs borne by society for rising health care due to air pollution.*

- *UFORE-E: Energy Conservation. The model estimates the effects of trees on building energy use and consequent emissions of carbon from power plants.*

The UFORE model and assumptions are explained in detail in Nowak and Crane (2000), Nowak, Crane, Stevens, and Ibarra (2000), Nowak *et al.* (2006) and Nowak *et al.* (2008). The assumptions are summarized in Townsend and Hegg (in progress). A brief summary of the strengths and limitations of the model follows.

A.1.1 UFORE Summary

UFORE is a detailed model of urban forest structure, based on extensive research and tailored to the local species and climate. The main strengths of the UFORE model is that it is based on actual field data specific to the study area, it is a publicly available and peer-reviewed model and there is considerable technical support. The model is also subject to its limitations. For instance, an issue related directly to this thesis is that the model does not assess the functional condition of an ecosystem; rather, the model assesses tree health (susceptibility to disease and insect outbreaks) and not the context of the landscape. In addition, many of the estimated parameters (*e.g.,* dieback, light coverage) are difficult to measure in the field. Furthermore, the vegetation calculations are based upon algorithms developed in other studies – some of which are quite dated and may not properly model urban tree growth (Nowak *et al.,* 2008). Nevertheless, the estimates are based upon scientifically and statistically peer-reviewed methods (Townsend and Hegg, in progress).

A.2 CITYgreen

CITYgreen is a proprietary software application program developed by American Forests to analyze the economic benefits of tree canopy and other green space (American Forests, 2008). It is promoted as a tool for influencing public policy regarding urban trees and has been used by several US and Canadian cities for educational and planning purposes (American Forests, 2008). An extension of ArcGIS software, the CITYgreen program is a GIS-based system that analyzes the economic benefits of tree canopy.

A CITYgreen analysis is based on a land cover dataset (*e.g.,* ortho-photo) that has been previously prepared. The land cover features are digitized directly by tracing polygons (superimposing coloured shapes over tree canopies on an ortho-photo or by digitizing groups of trees as a single polygon using the editing tools in ArcGIS). The CITYgreen software incorporates basic research conducted predominantly by the U.S. Forest Service and a storm-water model (TR-55) developed by NRCS (Natural Resource Conservation Service, a division of USDA). Because CITYgreen is based upon research conducted within the US, users can only select from a list of 55 US cities in order to perform a CITYgreen analysis. If the city is not in the list, a city that best represents the area of study is selected.

CITYgreen is designed to calculate the following:

- Air pollution abatement (calculated for carbon monoxide, nitrogen dioxide, ozone, particulate matter (< 10 microns), and sulfur dioxide);
- Total carbon stored and net annual carbon sequestration by the urban forest;
- The volume of storm-water runoff coming from the land cover based on a 2-year, 24 hour rain event; and,
- Difference in water pollutant loading between the test case and a reference scenario.

The detailed workings of the model are not explicit in the documentation provided with the software, or have any detailed workings of the model been published; however, Townsend and Hegg (in progress) examined and documented the assumptions of the model. A brief summary of the strengths and limitations of the model follows.

A.2.1 CITYgreen Summary

CITYgreen is a GIS-based model designed to estimate the economic value of urban forests. It is a useful program to help estimate the benefits of green infrastructure as it highlights the importance of urban forestry and the economic consequences of lost green space. Combined with other GIS tools, the strength of the model lies in its ease of use and ability to quickly estimate the value of urban forests. However, the model also suffers from limitations as well. First, the model does not account for understory vegetation and only estimates the carbon values of areas delineated as forest in the program, which is likely to result in an inaccurate estimate of carbon storage and sequestration. Second, the carbon values are based upon a very general characterization of the urban forest structure in terms of age distribution. Third, as no physical measurements are taken, the model does not take into account variations in tree species, canopy height, canopy diameter, canopy density, leaf area, shrub understory, wet/dry/chemical pollution deposition, boundary layer, *etc.* and, therefore, pollution abatement calculations are not likely to be an accurate representation of the pollution abatement value that the urban forest under study actually provides.[70] Forth, the model simplifies the drainage characteristics of the study area by allowing the user to adjust only 4 parameters (the percent slope, hydrological soil group, rainfall distribution and rainfall intensity), which are used to calculate a curve number (CN) for the entire study area.[71] As the curve number applies to the entire study area, it does not allow for any variability that would influence the hydrology within sub-watersheds (*e.g.,* varying slopes, soils, water features, culverts). Finally, the model estimates the amount of non-point source contaminant that would arise from a replacement land cover scenario (*i.e.,* replacing treed area with a chosen land cover type) based upon a linear regression table of pollutant loadings and curve numbers. However, regressing curve numbers and pollutant loading based on land-cover over-simplifies a complex relationship resulting in "gross errors" in the estimation of pollutant loadings (Schuett, Pers. Comm.).

A.3 Data Collection

The following describes the data collection activities that was undertaken to collect the data required to value the ecosystem services for both the Swan Lake Watershed and the Swan Lake/Christmas Hill Nature Sanctuary discussed in Chapter 4.

A.3.1 GIS Mapping of the Swan Lake Watershed

Background and historical information and maps of the Swan Lake watershed was collected and provided by L. Townsend. The Victoria Official Map of 1858 was converted into digital format and then georeferenced by the Municipality of Saanich GIS department. Using ArcGIS 9.2 software, a shape file outlining the 2005 Swan Lake watershed boundary was created and added as a layer to the digital 1858 map. Using the

[70] For instance, Longcore, Li and Wilson (2004) compared the Ozone removal per acre value derived by a CITYgreen model to that of the results derived in a UFORE model for the City of Los Angeles and concluded that the CITYgreen values are approximately 50% lower than the UFORE values.

[71] A curve number (CN) indicates how much rainfall becomes direct run-off (NRCS, 1986).

boundary and historical information provided by L. Townsend and features on the 1858 map itself, areas of water, riparian vegetation, cleared areas, Garry Oak and conifer forest were delineated as polygons called shape files by D. Hegg on the digital map in order to calculate the total amount of area for each land-class (Table 7).

Table 7. 1858 Land cover classes within the Swan Lake Watershed

Land Cover Type	**Area (ha)**
Open water (Swan & Blenkinsop Lake)	17.4
Old growth conifer forest	391
Garry oak woodland	605
Rock	17.5
Wetland (Swan & Blenkinsop Lake)	154
Cleared land	3.4
TOTAL	**1188**

The CITYgreen analysis was based on a 2005 ortho-photo of the Swan Lake watershed provided by the Capital Regional District (CRD), to Caslys Consulting (a local GIS analysis firm), for a larger-scale study of the Capital Region. Caslys Consulting was contracted to run CITYgreen for Swan Lake watershed, using funds provided to L. Townsend through a Sarah Spencer Research award (Townsend, 2009). In order to use CITYgreen, shape files need to be classified as one of the land cover types listed in the program menu. D. Hegg and L. Townsend advised Caslys Consulting on what land cover classifications to assign to the 2005 ortho-photo for the custom Swan Lake watershed analysis (Table 8). As a Canadian version of this software was in development when this study was undertaken (the only working version had information for the Ottawa area), the city of Seattle was used to estimate the air pollution values.

Other inputs to the CITYgreen model include the following:

- A precipitation value of 50.4mm was used. This value is the 24-hour, 2-year storm event as per the Municipality of Saanich's rainfall intensity curve (Townsend and Hegg, in progress);
- A digital elevation model was acquired from BC Government Terrain Resource Information Management (TRIM);
- The city of Seattle was used as the reference city for air pollution;

- Rainfall was adjusted to Type IA (west coast rainfall patterns); and
- Hydrologic soil group Type B – Somewhat Pervious was selected. This selection was based on L. Townsend's research on general soil characteristics of the Swan Lake Watershed area.

Table 8. 2005 Land cover types expressed in terms for the CITYgreen analysis

Land Cover Type	**CITYgreen Land Cover Class**
Open Water	Open Water
Agriculture (cleared land within the Agricultural Land Reserve)	Row Crops
Exposed soil	Fallow, bare soil and row crops
Grass (cleared land outside of the ALR)	Open space, grass/scattered trees, >75% ground cover in grass
Gravel	Impervious, unpaved, gravel
Impervious (roads, roofs, parking areas)	Paved
Marsh (incorrectly identified, mostly in residential areas)	Trees with grass/turf understory (50-75% cover)
Shadow	Open space/grass
Shrub	Shrub, ground cover < 50%
Trees	Trees, forest litter understory

Note: From Townsend and Hegg (in progress).

A.3.2 Estimation of Carbon Storage and Sequestration

Carbon Storage and Sequestration for the Swan Lake Watershed

Because the Swan Lake watershed is composed of a greater proportion of conifer forest (Table 9) and conifers retain more carbon in leaf biomass, estimates for carbon storage and sequestration are expected to be quite conservative as the CITYGreen model uses a weighted average (or Type 4 distribution). Specifically, the carbon calculation in the model is based upon the allometric equations for two species (sugar maple and white pine) and the categorization of a forest into 4 different classes based on diameter breast height width (DBH): Type 1 – a young population; Type 2 – a older population (20 to 50 years of age); Type 3 – a balanced distribution; and, Type 4 – a weighted average of the three distributions. The CITYgreen model appears to use the Type 4 distribution (although this is not explicitly stated in the manual and no options could be found in the program to alter the urban forest to correspond to another distribution Type - reverse-calculations of the results corresponded with average values). CITYgreen carbon storage and sequestration values are reported in short tons and thus were converted into tonnes using the following conversion factor: short ton = 0.90718474 tonnes. Because the carbon storage and sequestration values are anticipated to result in a conservative (under-) estimate for the watershed, the values from the CITYgreen model were used for this study and the 2005 values were extrapolated to 2009. However, due to concerns with how the other services (air pollution, stormwater and pollutant loadings) are derived in the model, the values are not reported, but can be found in Appendix B.

Carbon Storage and Sequestration for the Swan Lake/Christmas Hill Nature Sanctuary

Vegetation communities within the Swan Lake/Christmas Hill Nature Sanctuary were delineated using ortho-photos and L. Townsend's knowledge of the site based on ground-truthing. The total area in hectares (ha) of each of the vegetation communities for 2007 was calculated using the Capital Regional District's public GIS viewer called the Natural Areas Atlas (CRD, 2007) (Table 9). Once each vegetation community was delineated, fieldwork and literature values were employed to estimate the amount of carbon stored and sequestered. Table 10 displays the results of the carbon storage and sequestration for each of the vegetation communities within the Swan Lake/Christmas Hill Nature Sanctuary.

Table 9. 2007 Land cover classes within the Swan Lake Watershed

Land Cover Type	**Area (ha)**
Open water (Swan & Blenkinsop Lake)	15.8
Conifer forest (Mount Doug)	46.6
Garry oak woodland (Christmas Hill)	4.93
Rock	17.5
Wetland (Swan & Blenkinsop Lake)	49
Agriculture / private open space	191
Commercial / light Industrial	54.2
Residential	809
TOTAL	**1188**

Mixed Deciduous/Coniferous, Garry Oak and Mature Cottonwood Communities

Based on the UFORE methods, a circular plot with a radius of 11.4m was surveyed in each of these tree communities, in order to estimate the amount of carbon stored and sequestered. L. Townsend subjectively selected plot locations, as most vegetation communities were small in comparison to the dominant vegetation types. These plots were classified as the following:

- **Mixed Deciduous/Coniferous Forest**: A mixed conifer-deciduous regenerating forest composed primarily of *ca.* 30 year-old Douglas-fir (*Pseudotsuga menziesii*) (with one veteran specimen), and an understory of cherry (*Prunus* sp.), dense shrub and herbaceous cover;
- **Cottonwood:** A mature cottonwood (*Populus balsamifera* ssp. *trichocarpa*) dominated site, with an understory of red-osier dogwood and alder; and
- **Garry Oak**: A mature Garry oak (*Quercus garryana*) ecosystem that also included Douglas-fir with a grass-dominated understory.

Table 10. 2007 Land cover types within the Swan Lake/Christmas Hill Nature Sanctuary

Land Cover Type	Area (ha)	Carbon Stored (tonnes)	Carbon Sequestered (tonnes/year)
Open water	9.24		
Aquatic vegetation	0.84	8.06	
Wetland grass (majority is reed canary grass)	13.67	153.68	12.71
Terrestrial grass	4		
Tall shrub (willow/red osier dogwood)	9.74	1056.79	64.77
Mixed conifer/deciduous (upland species)	0.781	59.04	0.81
Garry oak & Douglas fir	0.955	146.50	-2.60
Mature cottonwood/alder/poplar	0.854	291.81	4.87
Hardhack	1.63	17.93	
Low shrubs (largely consisting of snowberry, rose and Himalayan blackberry)	3.32	27.56	
English hawthorn	0.28	2.86	0.06
Other (miscellaneous deciduous trees)	2.75	28.05	0.55
TOTAL	**48.06**	**1792.28**	**81.17**

Note: Carbon storage and sequestration estimates for land-cover types that could not be calculated have been left blank in the table.

Once Townsend and Hegg delineated each of the plots using flags, following the specified methods outlined in the UFORE manual, vegetation measurements were undertaken and documented in a field book. Once, all measurements were undertaken for each plot, the values were transferred into an Excel spreadsheet and emailed (support@ufore.org) directly to UFORE researchers for processing. Upon receipt of the processed data, the carbon storage and sequestration estimates for each of the twenty three plots was converted into tonnes/hectare. This enabled extrapolation of estimates based upon the size of the community in hectares. The processed data from the three UFORE plots was also used to estimate the carbon storage and sequestration values of the *English Hawthorn* and the *Miscellaneous Deciduous communities*.

English Hawthorn Community

Carbon storage and sequestration was estimated based on the single cherry (*Prunus* sp.) values surveyed in the UFORE mixed deciduous/conifer plot. Because no carbon storage and sequestration literature values could be found for the English Hawthorn, it was reasoned that since this species grows to a similar size as cherry, the carbon storage and sequestration estimates would be similar (Townsend, Pers. Comm.). The carbon storage and sequestration estimate was converted into tonnes/hectare.

Miscellaneous Deciduous Community

This community entailed the remainder of deciduous trees originally mapped, minus the deciduous communities listed above (Townsend and Hegg, in progress). Carbon storage and sequestration was estimated based on the single cherry (*Prunus* sp.) values surveyed in the UFORE plot in the mixed deciduous/conifer plot. The carbon storage and sequestration estimate was converted into tones/hectare.

Tall Shrub Community

This community is dominated by willow (*Salix* spp.) and red osier dogwood (*Cornus stolonifera*), with a sparse herbaceous understory (Townsend and Hegg, in progress). Using sample methods described in Peet, Wentworth and White (1998), L. Townsend stratified and established two 0.1-hectare plots in the tall shrub community fringing Swan

Lake. Each large plot consisted of ten 10m by 10m subplots aligned in a single row, for a total of twenty plots. L. Townsend collected field data (species present, percent cover, number and diameter breast height measurements). Because the purpose of the plots was to initially characterize the shrub community of the Swan Lake Watershed (a component of L. Townsend's thesis), additional information was estimated after the fact in order to apply the UFORE model, as described below. Once, measurements and estimates were completed for each plot, the values were transferred into an excel spreadsheet and emailed directly to UFORE researchers, for processing. The information estimated by Townsend (Pers. Comm.) is as follows:

- The tree height (woody specimens > 2.5 cm DBH) was estimated based on LiDAR data acquired from Terra Remote Sensing Inc (Townsend, Pers. Comm.).
- The crown attributes (height to crown base, and diameter) were estimated based on recollections and was roughly correlated with diameter breast height (DBH).
- A tree condition value of "Good" was assigned to all trees, based on general observations and recollections of limited tree mortality or dieback.
- All woody plants less than 2.5 cm DBH were designated as shrubs, and as per UFORE methods were assigned an area of coverage and an average height. 1.5 m was used as a standard height for all shrub species.

Wetland Grass Community

This community largely consists of reed canary grass (*Phalaris arundinacea)* with a small amount of giant mannagrass (*Glyceria maxima)* (Townsend and Hegg, in progress). Carbon sequestration of this community was estimated using the average value calculated by Jelinski (2007) for *Phalaris arundinacea* in a Wisconsin wetland. Carbon storage was estimated based on a small sample of biomass measurements by L. Townsend of above-ground material, combined with a calculation of estimated below-ground biomass using a formula from Jelinski (2007).

Aquatic Vegetation Community

Carbon storage for this community was estimated using the calculated value for *Typha latifolia*, from Rothman and Bouchard (2007) in Lake Erie. No carbon sequestration rate was calculated for this community since references in the literature could not be found.

Hardhack (Spirea douglasii, var. *douglasii) Community*

Carbon storage was estimated based on formulas in Smith and Brand (1983); average diameter and density (stems per m2) were calculated based on measurements of small hardhack thickets within the Tall Shrub community plots. Carbon sequestration was not estimated.

Low Shrubs Community

This community largely consists of snowberry (*Symphoricarpos albus*) rose (*Rosa nutkana*). Carbon storage was estimated based on formulas in Smith and Brand (1983); average diameter and density (stems per m2) was estimated to be 1.5 cm and 10 cm, respectively (Townsend and Hegg, in progress). Carbon sequestration was not estimated.

Because all carbon values were based on data collected in 2008, the estimated yearly sequestration value was added to the stored carbon value in order to estimate a carbon storage value for 2009.

A.3.3 Estimation of Air Pollution

Air pollution values for 2008 were generated from the UFORE analysis performed on the twenty-three plots discussed above: Mixed Deciduous/Coniferous, Garry Oak, Mature Cottonwood and Tall Shrub. Air pollution values were extrapolated to the Swan Lake/Christmas Hill Nature Sanctuary based upon the size of the vegetation community, but was not extrapolated to the year 2009, as this would require extensive recalculation and additional fieldwork. Unlike the carbon estimates, no air pollution values were determined for the *English Hawthorn*, *Aquatic Vegetation, Hardhack (Spirea douglasii), Low Shrub and Miscellaneous Deciduous communities* as the air pollution values are quite complicated and any error in the calculations on the part of the researcher would result in a material error (18.82% of the total area was not assessed for an air pollution value). Thus, the air pollution abatement value estimate ($314,904) for 2009 is a conservative underestimate for the area.

Due to the size of the *Wetland Grass Community* an air pollution estimate was derived. A custom UFORE dataset was generated for the large grass community in the Swan Lake/Christmas Hill Nature Sanctuary in order to measure the carbon storage value of the grass (Hoehn, Pers. Comm.). Using the standard methods for determining pollution flux, under the guidance of the UFORE team, two modifications to the air pollution model were made: first, an estimated leaf area index (LAI) of 5.95 was used; the LAI value is based on a similar site in Wisconsin (Jelinksi, 2007). Second, assuming that moisture availability is plentiful, a two-sided layer of stomata was modeled (Townsend, Per. Comm.).

A.4 Ecosystem Valuation

A.4.1 Market Price Method: Energy Dissipation

To estimate the amount of energy dissipation by the type of land cover, the following information was acquired (Townsend, 2009):

- Land cover data for the watershed in 1858 and 2007 (Tables 7 and 9)[72];
- Monthly global solar energy values (MJ/m^2) collected on an hourly basis at a station in Vancouver, B.C were acquired from Environment Canada;
- Estimates of the latent heat dissipation of solar radiation energy for each type of land-cover were taken from the literature by L. Townsend (Table 11);

The growing season was defined as May 1st to September 30th. Hourly radiation values were averaged over each day; these daily average values were then averaged for each month to give a monthly total radiation value in Mega Joules (MJ).[73] Based on the land-

[72] Land cover classes were identified using 2007 ortho-photos as these photos were readily available at the time of the study. Since 2005/2007, according to Townsend (Pers. Comm.) the land-cover classes have not considerably changed in the Swan Lake watershed as the area is already highly developed.

[73] It should be noted that the monthly averaged values were not adjusted for the weather patterns of the regions and is likely to result in an underestimate of the value of the ecosystem service as the data is from a weather

cover type, the radiation values were adjusted for the amount of solar radiation energy that was dissipated for each month. These values were then summed and divided by the amount of energy (MJ) in a tonne of coal (25,104 MJ).[74] Using a May 2009 spot price value of $81.03 ($68.95 US converted into Canadian currency) per tonne of coal, the total energy value in coal was determined for 2009. The spot price was taken from the Energy Information Administration (2009), which is a group that reports on official energy statistics from the US Government.

Table 11. Summary of energy references

Land Cover Type	**Proportion of Solar Energy Dissipated**	**Reference**	**Notes**
Open Water	0.375	Burba *et al.,* 1999	Reported range of 20-55%
Conifer Forest	0.42	Humphreys *et al.,* 2003	50-Year-old regenerating Douglas-fir forest on Southern Vancouver Island, B.C.
Deciduous Forest	0.42	Bormann and Likens, 1979	Regenerating Eastern hardwood Forest
Rock	0		Assumed to have a value of zero (no vegetation)
Wetland	0.62	LaFleur, 2008	Mean of values for 5 studies of marshes
Agriculture / Open Space	0.33	Barnes *et al.,* 2000	Calculated from a Bowen Ratio (Latent:Sensible) for "Grassland and Cropland" of 0.67
Commercial / industrial	0.13	Grimmond and Oke, 1999	From a light industrial area in Vancouver, B.C. (44% impervious)
Residential	0.33	Grimmond and Oke, 1999	From measurements in a residential area in Vancouver, B.C.

Note: Adapted from Townsend and Hegg (in progress).

station in Vancouver (Victoria tends to receive more sun than Vancouver). A more detailed study would require the deployment of numerous micrometeorological stations in both urban and vegetated sites over a couple of years (Townsend, 2009).

[74] According to the World Coal Institute (2007), Bituminous coal will have an energy value of 6000 kcal/kg. Converting the value into kcal/tonnes and using a conversion factor of 1 kcal = 0.004184 MJ, a tonne of Bituminous coal has an energy value of 25,104 MJ (World Coal Institute, 2007).

A.4.2 Cost-Based Method: Flood Mitigation

Wetlands are commonly recognized for their ability to reduce the magnitude or prevent downstream flooding (Barbier, Acreman and Knowler, 1997). When not filled to capacity, wetlands can reduce the magnitude of flood peaks by holding back floodwaters and slowly releasing the waters downstream (Ming, Xian-guo, Lin-shu, Li-juan and Shouzheng, 2007). By retaining water in the surface water of lakes, marshes, *etc.*, or by storing water in the soil, wetlands can act as “natural reservoirs”, thereby reducing the need for costly human-made infrastructure (Barbier *et al.,* 1997). Thus, “wetlands can be compared to a substitute proxy such as human-made reservoirs” (Ming *et al.,* 2007, p.221). As 37 hectares of wetlands surround the 9.2-hectare lake, the Swan Lake/Christmas Hill Nature Sanctuary has a natural flood control system (Townsend, 2009).

Using the same method and proxy applied by Ming *et al.* (2007) to value the flood mitigation benefits of wetland soils within the Momoge National Nature Reserve, Jilin Province, the People's Republic of China, a flood value of $83,098 was determined for the Swan Lake/Christmas Hill Nature Sanctuary. Employing the alternative/substitute cost method, Ming *et al.* (2007) calculated the flood benefit provided by the wetlands by calculating the volume of surface water stored by the wetlands and multiplying the value by the estimate for the capital cost of reservoir construction ($0.08/m3). To estimate the flood control value for the Swan Lake/Christmas Hill Nature Sanctuary, L. Townsend (Pers. Comm.) calculated the volume of water contained in the basin of the lake and on adjacent floodplains for varied types of storm events (Table 12) (Townsend, Pers. Comm.). Using the $1.16/m3 value ($0.08/m3 adjusted for time and currency - Table 13) from Ming *et al.* (2007) and the 1/1-year storm event, the flood value for the Swan Lake/Christmas Hill Nature Sanctuary was determined. The 1/1-year storm event was selected for the valuation analysis, as this type of event is likely to happen on a yearly basis. As a result, the flood value is quite conservative, whereas actual downstream flooding costs (if they were to occur) would be conservatively estimated to be in the millions.

Table 12. Swan lake flood storage volumes

Event	**Max Stage (m)**	**Base Storage (m3)**	**Additional Basin Storage (m3)**	**Floodplain Storage (m3)**	**Total Volume (m3)**
Low Water	11.74	237530	N/A	N/A	237530
Filled to Rim	12.1	237530	11621	N/A	249151
1/1 Year Flood	12.6	237530	11621	71358	320509
1/7 Year Flood	13	237530	11621	162252	411403
1/100 Year Flood	14	237530	11621	445833	694984

Note: Adapted from Townsend (Pers. Comm.).

Table 13. Conversion table for flood abatement ecosystem service

Ecosystem Service	**Original Value**	**US CPI 2004**	**US CPI 2008**	**US$ to Can$**	**Adj. Value**	**Reference**
Flood Abatement	$0.88	188.9	212.709	1.1752	$1.16	Ming *et al.*, 2007

Note: US CPI values are from: United States Department Of Labor, 2009. Exchange rate is from: Yahoo Finance, 2009.

A.4.3 Benefit Transfer Method: Recreational Value

The Swan Lake/Christmas Hill Nature Sanctuary is a protected 48-hectare parcel of parkland that acts as both a sanctuary to local flora and fauna as well as a place of retreat, cultural heritage and recreation for the local community (Swan Lake Christmas Hill Nature Sanctuary, 2006). The benefit transfer method was employed as a means to estimate the lower bounds of the recreation value of the Swan Lake/Christmas Hill Nature Sanctuary.

Because a single value transfer study site with similar characteristics to the Swan Lake/Christmas Hill Nature Sanctuary could not be identified (although such a study may exist), an average value was used to identify what the lower limit recreational value for the site could be. A mean value of $22.77 per person was used to estimate the recreational value of the study site. The value was taken from Rosenberger and Loomis

(2001) who performed the most recent in-depth meta-analysis on 163 valuation studies that covered all type of recreation activities for the U.S. Department of Agriculture and Forest Service. Rosenberger and Loomis (2001) analysis is quite detailed splitting the values from the studies into different geographical location and into 22 different recreational activities. The person/day value of $45.84 ($22.77 adjusted for time and currency - Table 14) was then applied to the lower estimate of the total number of visitors to the Swan Lake Nature Sanctuary (80,775 visitors) (T. Morrison, Pers. Comm.).[75] Since the value applied was derived from an average of studies with unknown characteristics, many of which are based on hypothetical markets, the $45.84 value was adjusted downward by a factor of 3 to reflect List and Gallet (2001) empirical findings that on average participants in hypothetical markets overstate their preferences by a factor of about 3. Thus, a conservative general recreational value for the Swan Lake/Christmas Hill Nature Sanctuary is $1,234,304.

Table 14. Conversion table for general recreation ecosystem service

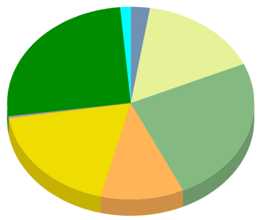
Ecosystem Service	**Original Value**	**US CPI 1989**	**US CPI 2009**	**US$ to Can$**	**Adj. Value**	**Reference**
General Recreation	$29.74	124	212.709	1.1752	$45.84	Rosenberger and Loomis (1989)

Note: US CPI values are from: United States Department Of Labor, 2009. Exchange rate is from: Yahoo Finance, 2009.

[75] T. Morrision (Pers. Comm.), manager of the Swan Lake/Christmas Hill Nature Sanctuary, estimated that for the year 2007: 8,600 individuals participated in guided community programs, 6,750 students participated in guided school programs, 425 people took part in guided bird walks around the lake and 65,000 to 70,000 visitors used the trail systems and facilities on their own. Thus, for 2007 it is estimated that between 80,775 and 85,775 individuals visited the site in the year 2007.

Appendix B. CITYgreen Results

Analysis Report
for
Swan Lake Watershed



Land cover areas are in acres.

Cropland: Fallow: Bare Soil	70.5	2.4%
Cropland: Row Crops	464.0	15.8%
Impervious Surfaces: Paved	728.2	24.8%
Open Space - Grass/Scattered Trees	319.8	10.9%
Open Space - Grass/Scattered Trees: Grass cover > 75%	549.3	18.7%
Shrub: Ground cover < 50%	7.4	0.3%
Trees: Forest litter understory	757.4	25.8%
Water Area	39.0	1.3%
Total:	2,935.6	100.0%

Total Tree Canopy: 757.4 acres (25.8%)

Air Pollution Removal

Nearest Air Quality Reference City: **Seattle**

	Lbs. Removed/yr	Dollar Value
Carbon Monoxide:	4,051	$1,729
Ozone:	22,279	$68,445
Nitrogen Dioxide:	10,127	$31,111
Particulate Matter:	20,928	$42,928
Sulfur Dioxide:	10,127	$7,600
Totals:	**67,511**	**$151,812**

Carbon Storage and Sequestration

Total Tons Stored: **32,589.99**

Total Tons Sequestered (Annually): **253.72**

Stormwater

Water Quantity (Runoff)

2-yr, 24-hr Rainfall: 2.13 in.

Curve Number reflecting existing conditions: 74
Curve Number using default replacement landcover: 84

Additional Storage volume needed: 4,528,216 cu. ft.

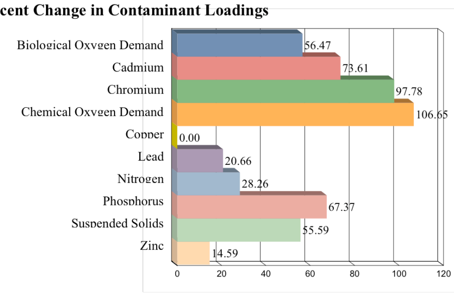
Construction cost per cu. ft.: $0.93

Total Stormwater Savings: **$4,211,241**

Annual costs based on payments over 20 years at 6% interest: **$367,155** **per year**

Water Quality (Contaminant Loading)

Percent Change in Contaminant Loadings



Appendix C. Proper Functioning Condition (PFC) Assessment

C.1 Summary Description of PFC

Jointly developed by the U.S. Bureau of Land Management, the U.S. Forest Service and NRCS, Proper Functioning Condition (PFC) is a qualitative assessment tool based upon quantitative science that measures the state and health of riparian-wetland areas. The PFC assessment is not based upon values (*e.g.,* whether the stream is visually pleasing, suitable habitat for fish), but rather on stream function (*i.e.,* whether the stream is stable from a hydrologic, vegetative, and soil perspective) (Prichard, 1998). PFC utilizes 17 criteria to determine stream health, thereby enabling users to rank rehabilitation priorities. The result of the PFC assessment is qualitative indicator identifying whether the system is in Proper Functioning Condition (PFC), Functional-at-Risk (FAR; with an upward or downward trend), or Non-Functional (NF).

When a system is classified as PFC, it is in state of 'resistance/resilience" enabling it to withstand disturbance without "coming apart" during 25-30 year storm events. Functional-at-Risk (FAR; with an upward or downward trend) has elements of a resilient system by remaining in a functional context, but has a soil, water, or vegetative attribute(s) that makes the system susceptible to degradation. A Non-Functional (NF) system is a system that is one that is not clearly able to dissipate energies associated with high flow events (due to a lack of adequate vegetation, landform, *etc*.) and, thus, cannot reduce erosion, improve water quality, provide habitat, *etc*. These ratings provide a means to which areas can be prioritized for rehabilitation so that when development occurs, resources can be allocated to the most critical areas of the system, thereby preventing degradation of healthy areas (Lucey, Pers. Comm.).

The assessment consists of a checklist based upon the *capability* and *potential* of the system characterized by the interaction of the three components, hydrology, vegetation, and soils. Potential is defined as the potential natural community a system could achieve "given *no* political, social, or economical constraints" (Prichard, 1998, pp. 6). Important to the proposed approach, potential can identify if the system is in "balance with the landscape setting" in terms of its ability to perform critical functions (*e.g.,* habitat, reduction of erosion). Furthermore, capability is defined as what *could be* achieved given current political, social, or economical constraints. Therefore, in the context of urban systems, it is recognized that some systems can, or may, be constrained by various limitations that may be insurmountable (*e.g.,* urban land use, major roads, zoning, political barriers); not precluding rehabilitation, but may limit the systems function.

C.2 PFC: What It Is - What It Isn't[76]

PFC is: A methodology for assessing the physical functioning of riparian-wetland areas. The term PFC is used to describe both the **assessment** process, and a defined, on-the-ground **condition** of a riparian-wetland area. In either case, PFC defines a minimum level or starting point for assessing riparian-wetland areas.

The PFC **assessment** provides a consistent approach for assessing the physical functioning of riparian-wetland areas through consideration of hydrology, vegetation, and soil/landform attributes. The PFC assessment synthesizes information that is foundational to determining the overall health of a riparian-wetland area.

The on-the-ground **condition** termed PFC refers to *how well* the physical processes are functioning. PFC is a state of resiliency that will allow a riparian-wetland area to hold together during a wind action, wave action, or overland flow event, sustaining that system's ability to produce values related to both physical and biological attributes.

PFC isn't: The sole methodology for assessing the health of the aquatic or terrestrial components of a riparian-wetland area.

PFC isn't: A replacement for inventory or monitoring protocols designed to yield information on the "biology" of the plants and animals dependent on the riparian-wetland area.

[76] From "A user guide to assessing Proper Functioning Condition and the supporting science for lotic Areas. Riparian Area Management TR 1737-15" by Prichard, 1998, p. 105-106.

PFC can: Provide information on whether a riparian-wetland area is physically functioning in a manner that will allow the maintenance or recovery of desired values (*e.g.,* fish habitat, neotropical birds, or forage) over time.

***PFC isn't*: Desired condition**. It is a prerequisite to achieving desired condition.

PFC can't: Provide more than strong clues as to the actual condition of habitat for plants and animals. Generally a riparian-wetland area in a physically nonfunctioning condition will not provide quality habitat conditions. A riparian-wetland area that has recovered to *proper functioning condition* would either be providing quality habitat conditions, or would be moving in that direction if recovery is allowed to continue. A riparian-wetland area that is functioning at risk would likely lose any habitat that exists during a wind action, wave action, or overland flow event.

Therefore: To obtain a complete picture of riparian-wetland area health, including the biological side, one must have information on *both* physical status, provided through the PFC assessment, and biological habitat quality. Neither will provide a complete picture when analyzed in isolation. In most cases, proper functioning condition will be a prerequisite to achieving and maintaining habitat quality.

PFC is: A useful tool for prioritizing restoration activities. By concentrating on the "at-risk" systems, restoration activities can save many riparian-wetland areas from degrading to a nonfunctioning condition.

Once a system is non-functional, the effort, cost, and time required for recovery is dramatically increased.

Restoration of non-functional systems should be reserved for those situations where the riparian-wetland has reached a point where recovery *is possible*, when efforts are not at *the expense* of "at-risk" systems, or when unique opportunities exist. At the same time, systems that are properly functioning are not the highest priorities for restoration.

Management of these systems should be continued to maintain PFC and further recovery towards desired condition.

PFC is: A useful tool for determining appropriate timing and design of riparian-wetland restoration projects (including structural and management changes). It can identify situations where structures are either entirely inappropriate or premature.

PFC is: A useful tool that can be used in watershed analysis. While the methodology and resultant data is "area based," the ratings can be aggregated and analyzed at the watershed scale. PFC, along with other watershed and habitat condition information helps provide a good picture of watershed health and the possible causal factors affecting watershed health. Use of PFC will help to identify watershed-scale problems and suggest management remedies and priorities.

PFC isn't: Watershed analysis in and of itself, or a replacement for watershed analysis.

PFC is: A useful tool for designing monitoring plans. By concentrating implementation monitoring efforts on the "no" answers, greater efficiency of resources (people, dollars, time) can be achieved. The limited resources of the local manager in monitoring riparian-wetland parameters can be prioritized to those factors that are currently "out of range" or at risk of going out of range. The role of research may extend to validation monitoring of many of the parameters.

PFC isn't: Designed to be a long-term monitoring tool, but it may be an appropriate part of a well-designed monitoring program.

PFC isn't: Designed to provide monitoring answers about attaining desired conditions. However, it can be used to provide a thought process on whether a management strategy is likely to allow attainment of desired conditions.

PFC can: Reduce the frequency and sometimes the extent of more data- and labor-intensive inventories. PFC can reduce time and cost by concentrating efforts on the most significant problem areas first, thereby increasing efficiency.

PFC can't: Eliminate the need for more intensive inventory and monitoring protocols. These will often be needed to validate that riparian-wetland area recovery is indeed moving toward or has achieved desired conditions (*e.g.,* good quality habitat) or simply to establish what the existing habitat quality is.

PFC is: A qualitative assessment based on quantitative science. The PFC assessment is intended for individuals with local, on-the-ground experience in the kind of quantitative sampling techniques that support the checklist. These quantitative techniques are encouraged in conjunction with the PFC assessment for individual calibration where answers are uncertain or where experience is limited. PFC is also an appropriate starting point for determining and prioritizing the type and location of the quantitative inventory or monitoring that is necessary.

PFC isn't: A replacement for quantitative inventory or monitoring protocols. PFC is meant to complement more detailed methods by providing a way to synthesize data and communicate results.

See the PFC user's guides for more details on the PFC process http://www.or.blm.gov/nrst/Tech_References/tech_references.htm.

C.3 PFC Process and Checklist[77]

Proper Functioning Condition (PFC) is a qualitative method for assessing the condition of riparian-wetland areas. The term PFC is used to describe both the assessment process and the condition of a riparian wetland area. The methodology was developed by a national

[77] From *Colquitz River Watershed Proper Functioning Condition Watershed Assessment: Appendix* (p.3-5) by Buchanan *et al.,* 2007, Victoria, B.C: Aqua-Tex Scientific Consulting Ltd. Copyright 2007 by Aqua-Tex Scientific. Reprinted with permission of author.

interagency team and documented in a series of Technical References (TR 9 through 16) (Prichard, 1993 through 1999 *et al*.). See the PFC user's guides for more details on the PFC process http://www.or.blm.gov/nrst/Tech_References/tech_references.htm.

The process involves the following steps:
1. Review existing documents--including maps, files and aerial photos.
2. Analyze the PFC definition--assess riparian/wetland based on a riparian area's capability and potential.
3. Assess Functionality--through document and field review. The rating is based on team discussion.
4. Institute the process--incorporate the information collected into a management plan.

The minimum standards are achieved by using a standardized checklist. The PFC assessment, using the checklist, should work for most sites as long as the procedure is followed and definitions understood. This is because the PFC was founded from rigorous science and is performed in an interdisciplinary setting.

The lotic (stream/moving water) checklist contains 17 items, which are qualitatively assessed by the Team (see checklist below). The lentic (lake/wetland) checklist contains 20 items. The appropriate form is used by the ID Team to assess riparian-wetland conditions. Items on the checklist relate to stream channel stability and/or wetland functionality, and receive "yes" or "no" answers. In some cases, "not applicable" is used.

In addition to the checklist it is helpful to fill out the supplemental field form, developed by John Anderson (a copy of the form follows), which includes information on vegetation community type, restoration measures, stream bank conditions, geomorphology/soils, floodplain availability/size, and grazing.

The checklist and its summarization, which can be done quickly, are used to classify the health or state of physical processes of the riparian-wetland area or reach being studied into one of four categories:

Functional – At Risk (FAR)
Non-functional (NF)
Proper Functioning Condition (PFC)
Unknown

The preponderance of "yes" and "no" responses help the ID Team determine the proper classification, however there is no set number of "yes" and "no" answers to determine into which category a water body falls. Team discussion is an important part of classification.

The significance of the classification categories are:

PFC: The stream channel, floodplain, and/or wetland have the physical characteristics that provide stability through various frequency events. This resiliency allows an area to produce desired values such as fish and wildlife habitat over time.

A riparian-wetland area is considered to be in Proper Functioning Condition when adequate vegetation, landform, or large woody material is present to:

- Dissipate stream energy associated with high water flows, thereby reducing erosion and improving water quality;
- Filter sediment, capture bedload, and aid floodplain development;
- Improve flood-water retention and ground-water recharge;
- Develop root masses that stabilize stream banks against cutting action;
- Develop diverse ponding and channel characteristics to provide the habitat and the water
- depth, duration, and temperature necessary for fish production, waterfowl breeding, and other uses; and,
- Support greater biodiversity.

FAR: The stream or wetland is functioning but is lacking enough vegetation, soils or landform characteristics to withstand various frequency events without significantly damaging the riparian corridor. FAR is the only category that is further stratified by trend (up, down, not apparent). A downward trend rating indicates deteriorating conditions that could become NF. Deteriorated conditions can be transmitted both up and downstream. Trend that is not apparent requires further study.

NF: The stream or wetland is not stable because it lacks most of the stabilizing physical characteristics and may continue to deteriorate. The degraded area or reach cannot sustain long-term desired values and return to proper-functioning condition without intervention (change in management).

Unknown: Sufficient information to make a rating is lacking. Additional study or data collection is necessary.

C.4 Lotic/Lentic PFC Checklists[78]

C.4.1 Lotic Checklist

Name of Riparian-Wetland Area:

Date: Segment/Reach ID:

ID Team Observers:

Potential Riparian-Wetland Vegetation: Potential Channel Characteristics: Rosgen = " "			
Yes	No	N/A	HYDROLOGICAL
			1) Floodplain above bankfull is inundated in "relatively frequent" events
			2) Where beaver dams are present are they active and stable
			3) Sinuosity, width/depth ratio, and gradient are in balance with the landscape setting (i.e., landform, geology, and bioclimatic region)
			4) Riparian-wetland area is widening or has achieved potential extent
			5) Upland watershed is not contributing to riparian-wetland degradation
Yes	No	N/A	VEGETATION
			6) Diverse age-class distribution of riparian-wetland vegetation (recruitment for maintenance/recovery)
			7) Diverse composition of riparian-wetland vegetation (for maintenance/recovery) (*species present*)
			8) Species present indicate maintenance of riparian-wetland soil moisture characteristics
			9) Streambank vegetation is comprised of those plants or plant communities that have root masses capable of withstanding high streamflow events (*community types present*)

[78] From "A user guide to assessing Proper Functioning Condition and the supporting science for lotic Areas. Riparian Area Management TR 1737-15" by Prichard, 1998, p. 63-64.

			10) Riparian-wetland plants exhibit high vigor
			11) Adequate riparian-wetland vegetative cover present to protect banks and dissipate energy during high flows (*enough*)
			12) Plant communities are an adequate source of coarse and/or large woody material (for maintenance/recovery)
Yes	No	N/A	EROSION DEPOSITION
			13) Floodplain and channel characteristics (i.e., rocks, overflow channels, coarse and/or large woody material) adequate to dissipate energy
			14) Point bars are revegetating with riparian-wetland vegetation
			15) Lateral stream movement is associated with natural sinuosity
			16) System is vertically stable (*not downcutting*)
			17) Stream is in balance with the water and sediment being supplied by the watershed (i.e., no excessive erosion or deposition)

Remarks

SUMMARY DETERMINATION

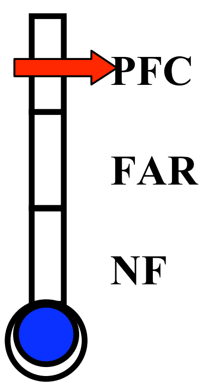
Functional Rating

__Proper Functioning Condition

___ Functional - At Risk

___ Non-functional

___ Unknown



Trend for Functional - At Risk:

__ Upward

___ Downward

___ Not Apparent

Are factors contributing to unacceptable conditions outside the control of the manager?

Yes ___

No ___

If yes, what are those factors?

___ Flow regulations

___ Mining activities

___ Upstream channel conditions

___ Channelization

___ Road encroachment

___ Oil field water discharge

___ Augmented flows

___ Other (specify)

C.4.2 Lentic Checklist

Name of Riparian-Wetland Area:

Date: Segment/Reach ID:

ID Team Observers:

Acres/Hectares:			
Potential Riparian-Wetland Vegetation:			
Yes	No	N/A	HYDROLOGICAL
			1) Riparian-wetland area is saturated at or near the surface or inundated in “relatively frequent” events
			2) Fluctuation of water levels is not excessive
			3) Riparian-wetland area is widening or has achieved potential extent
			4) Upland watershed is not contributing to riparian-wetland degradation
			5) Water quality is sufficient to support riparian-wetland plants
			6) Natural surface or subsurface flow patterns are not altered by disturbance (i.e. hoof action, dams, dikes, trails, roads, rills, gullies, drilling activities)
			7) Structure accommodates safe passage of flows (e.g., no headcut affecting dam or spillway)
Yes	No	N/A	VEGETATION
			8) There is a diverse age-class distribution of riparian-wetland vegetation (recruitment for maintenance/recovery)
			9) There is a diverse composition of riparian-wetland vegetation (for maintenance/recovery)
			10) Species present indicate maintenance of riparian-wetland soil moisture characteristics
			11) Vegetation is comprised of those plants or plant communities that have root masses capable of withstanding wind events, wave flow events, or overland flows (e.g. storm events, snowmelt)
			12) Riparian-wetland plants exhibit high vigor
			13) Adequate riparian-wetland vegetative cover is present to protect shoreline/soil surface and dissipate energy during high wind and wave events or overland flows

			14) Frost or abnormal hydrologic heaving is not present
			15) Favorable microsite condition (i.e. woody material, water temperature, etc.) is maintained by adjacent site characteristics

Yes	No	N/A	EROSION DEPOSITION
			16) Accumulation of chemicals affecting plant productivity/composition is not apparent
			17) Saturation of soils (i.e. ponding, flooding frequency, and duration) is sufficient to compose and maintain hydric soils
			18) underlying geologic structure/soil material/permafrost is capable of restricting water percolation
			19) Riparian-wetland is in balance with the water and sediment being supplied by the watershed (i.e., no excessive erosion or deposition)
			20) Islands and shoreline characteristics (i.e. rocks, coarse and/or coarse woody material) are adequate to dissipate wind and wave event energies

Remarks

SUMMARY DETERMINATION

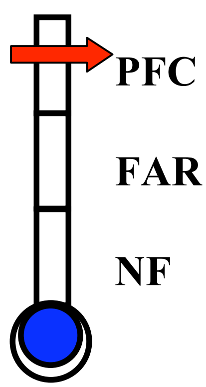
Functional Rating

__Proper Functioning Condition

__ Functional - At Risk

___ Non-functional

___ Unknown



Trend for Functional - At Risk:

__ Upward

___ Downward

__ Not Apparent

Are factors contributing to unacceptable conditions outside the control of the manager?

Yes ___

No ___

If yes, what are those factors?

___ Flow regulations

___ Mining activities

___ Upstream channel conditions

___ Channelization

___ Road encroachment

___ Oil field water discharge

___ Augmented flows

____ Other (specify)

Appendix D. BPP Sieve Analysis and Design Principles.[79]

Existing and Future Development Context

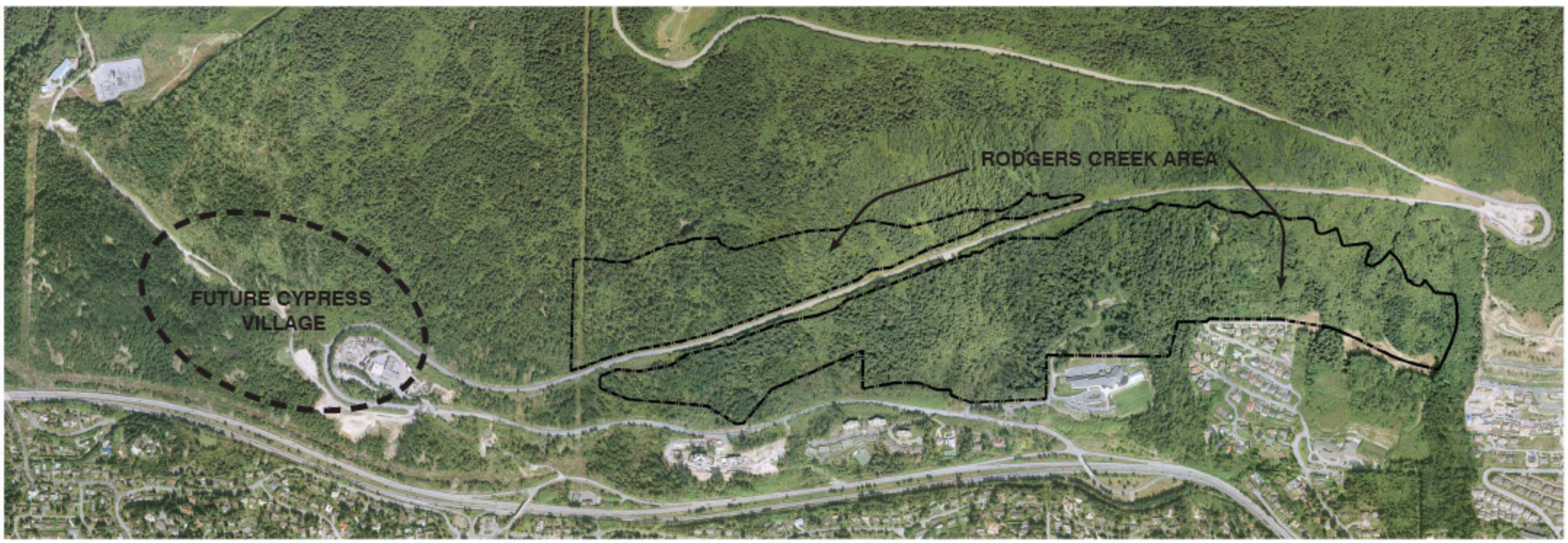
The Rodgers Creek Area is located immediately west of the Marr Creek corridor which is the conservation area on the western edge of Whitby Estates. Originally, the first phase of Rodgers Creek was to be Taylor's Lookout, the area between Marr Creek and the established subdivision around Chairlift Road. Taylor's Lookout was approved in advance of the Area Plan and is currently under development.

The Rodgers Creek Area is limited at its uphill boundary by the 1200 foot contour line as set by District policy. The downhill boundary is generally established by existing development, including Mulgrave School, and by the Cypress Bowl Road. Cave Creek West, a BC Hydro powerline corridor and highway road allowance form the western boundary.

Future Cypress Village

The location for a future Village to serve the Upper Lands has been identified in the Official Community Plan generally to the west of the Rodgers Creek Planning Area. The current District Works Yard, one of the areas of moderate topography in the Upper Lands, has the potential to become redeveloped as part of the Village, should the District choose to relocate the current uses at some time in the future. Other sites suited to mixed use development are located on the west side of Godman Creek on a series of terraces with superb view opportunities in the area east of Cypress Falls Park.

The first component of the Village, McGavin Field, is currently under construction. As the Village expands, a fieldhouse, children's playground, and other recreational amenities will be considered for this vicinity. Other land uses expected in the Village include: an elementary school, locally-oriented stores and services, a staging area and support services for mountain biking and hiking trail users, and residential development of a wide variety including different types of seniors housing, rental housing, units over retail, live/work, townhouses, and apartments.



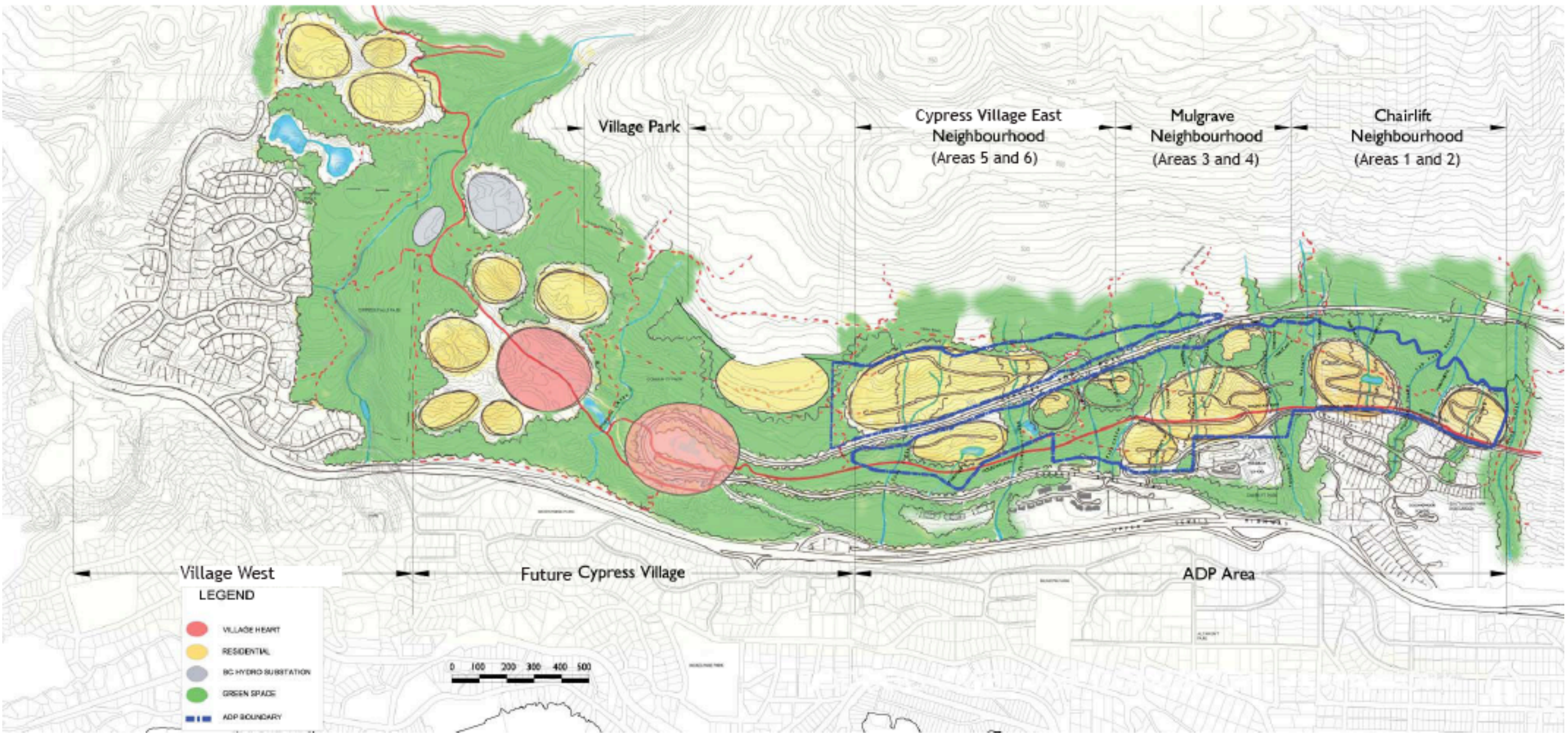
[79] From *Rodgers Creek Area Development Plan (ADP): Overview Report* (p.2-13) by British Pacific Properties, 2008, Vancouver, B.C: British Pacific Properties. Copyright 2008 by British Pacific Properties. Reprinted with permission of author.

Context

The Rodgers Creek Area Development Plan is prepared in the context of the Upper Lands -- their long-range development opportunities and their environmental and recreational resources.

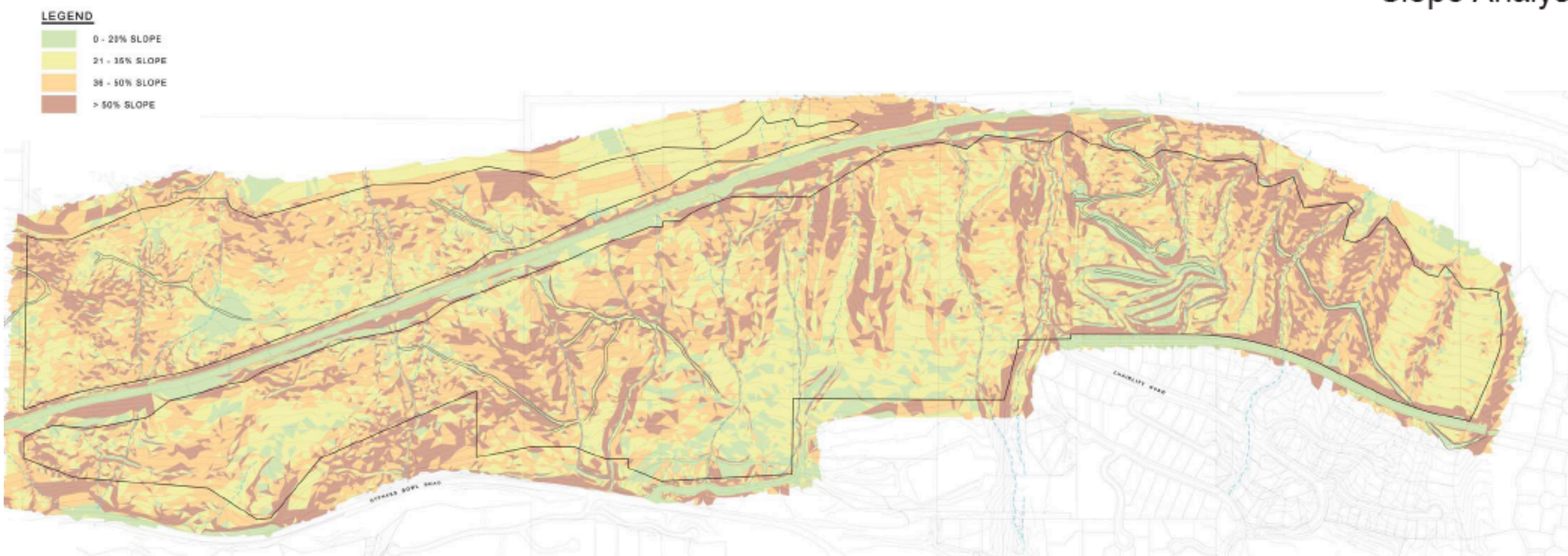
Development of the Upper Lands will occur in the context provided by the Official Community Plan and other relevant District plans, policies, and by-laws.

The images on this page through page 9 are reproduced from the Public Open House display panels of June 20, 2007.



Note: The above and following images are to illustrate the mapping of various land characteristics for the purposes of a sieve analysis and therefore clarity of text is not-significant. See the British Pacific Properties Area Development Plan (ADP) for more information.

Slope Analysis



Survey Work

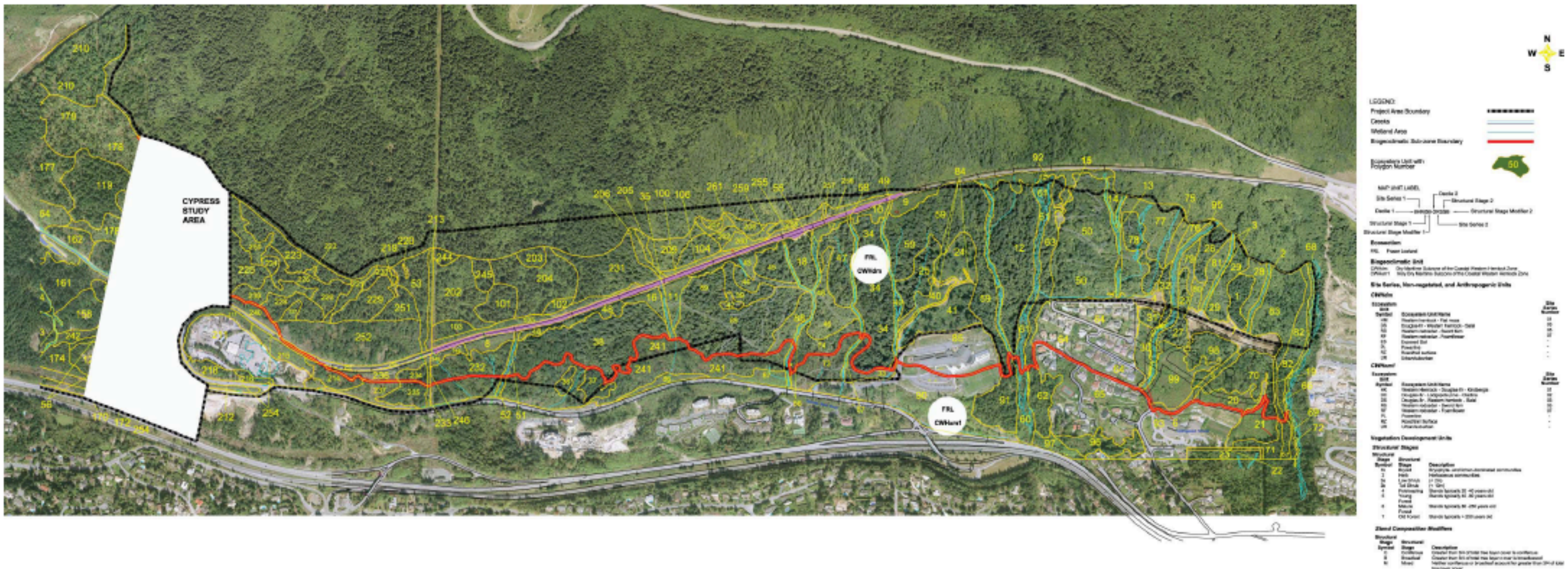
All of the land within the Rodgers Creek Area has been surveyed on the ground to ensure a thorough and accurate understanding of the topography and the features of the site. The detailed survey provided much more detail than the earlier mapping of the Upper Lands that had been done through interpretation of aerial photographs. As a result of the survey work, some areas that appeared to have development potential were found to not be while a large terraced area north of the first switchback on Cypress Bowl Road emerged as a candidate for further study.

Topography

The survey maps were interpreted by computer to reveal the complexity of the topography. The slope classifications used in this slope assessment are based on the categories in the Official Community Plan that identify slopes over 35% as requiring detailed study.

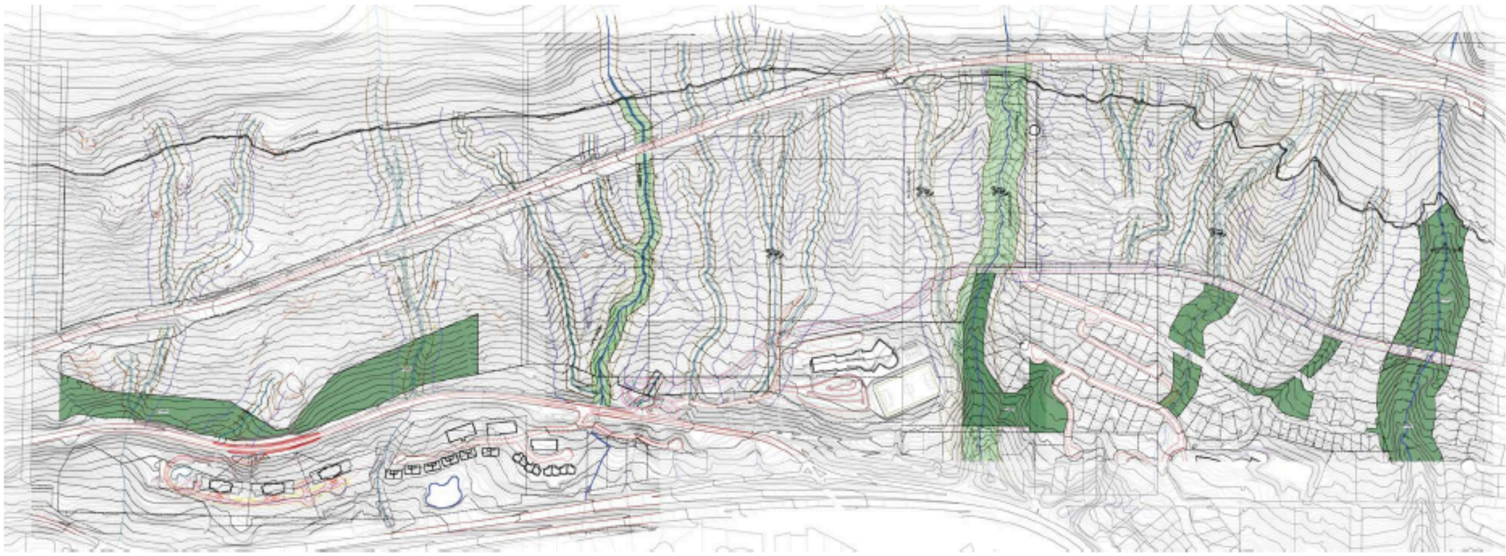
Vegetation Polygons

The slopes of the Rodgers Creek Area have been logged several times over the years. Nevertheless, a few mature trees were found that had escaped logging, mostly by being in the ravines of watercourses. There is a diversity of forest types and characters across the hillside that have been mapped and assessed for their value as habitat. No old growth forest exists within the Rodgers Creek Area and more mature and diverse stands are located above the 1200 foot contour line and to the west.



ECOSYSTEM UNIT LABELS

Watercourses



Watercourse Protection Regulations

The regulations around the protection of watercourses are complex and involve both the District of West Vancouver's by-laws and the policies of senior government agencies. The map on this page illustrates all of the regulations used in preparing the sieve analysis:

- The Riparian Area Regulations (RAR) established by the Provincial government as prepared by Seacor Environmental Consultants and reviewed by District staff (dotted red line and coloured light green)
- The Environmental Development Permit (EDP) areas (dotted blue lines). The District may consider a reduced setback.
- Areas identified as having potential geotechnical stability concerns in a report by Golder Associates (gray dotted lines and light yellow shading).
- Creeks that flow year round are identified as dark blue
- Presence of fish and frogs is noted.

Approach to Stream Protection

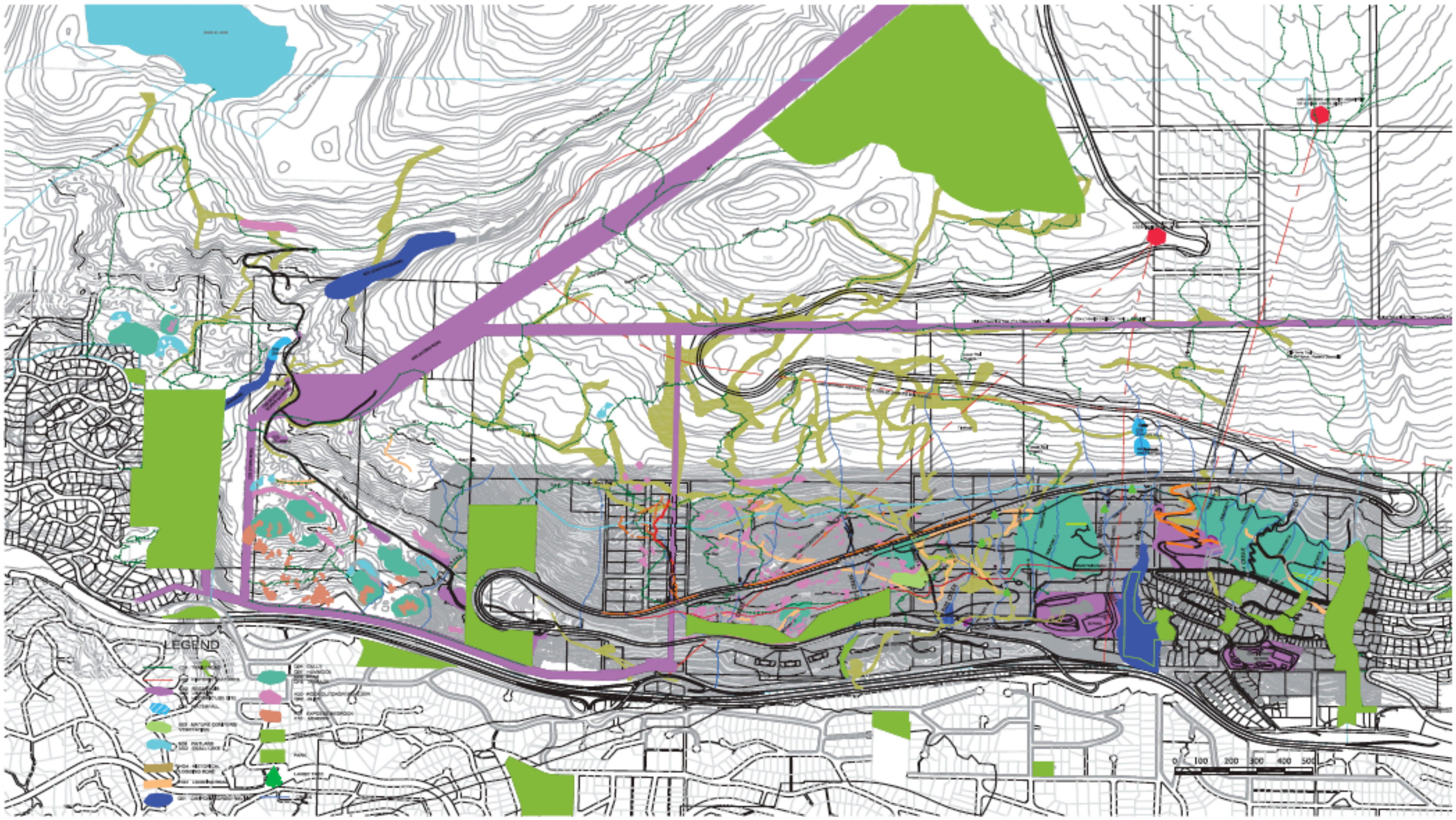
The following seven elements are the basis of the stream protection strategy:

- Watershed based land use planning
- Establish stream buffer network
- Reduce creation of impervious cover
- Limit the disturbance and erosion of soils during construction
- Treat the quantity and quality of stormwater (rainwater) runoff
- Maintain stream protection infrastructure
- Protect sensitive areas from development.

Note: After the original watercourse mapping was completed, the western boundary of the ADP Area above upper Cypress Bowl Road was expanded. The rationale for this expansion in included in Appendix A.

Existing Trails, Recreation and Landscape Features

An inventory of the recreation and landscape resources of the Rodgers Creek ADP Area and a larger context setting was prepared to identify and assess the resources affected by the Plan and to include key resources in the sieve analysis. The context provided the opportunity to compare resources within the Rodgers Creek Area with those in the surrounding Upper Lands to evaluate their prevalence and relative value. The categories used for the Recreation and Landscape Resources Inventory were based on the system developed and used throughout BC by the Ministry of Forests so as to be consistent with current best practices.



Proper Functioning Condition of Watercourses

Assessment of Proper Functioning Condition

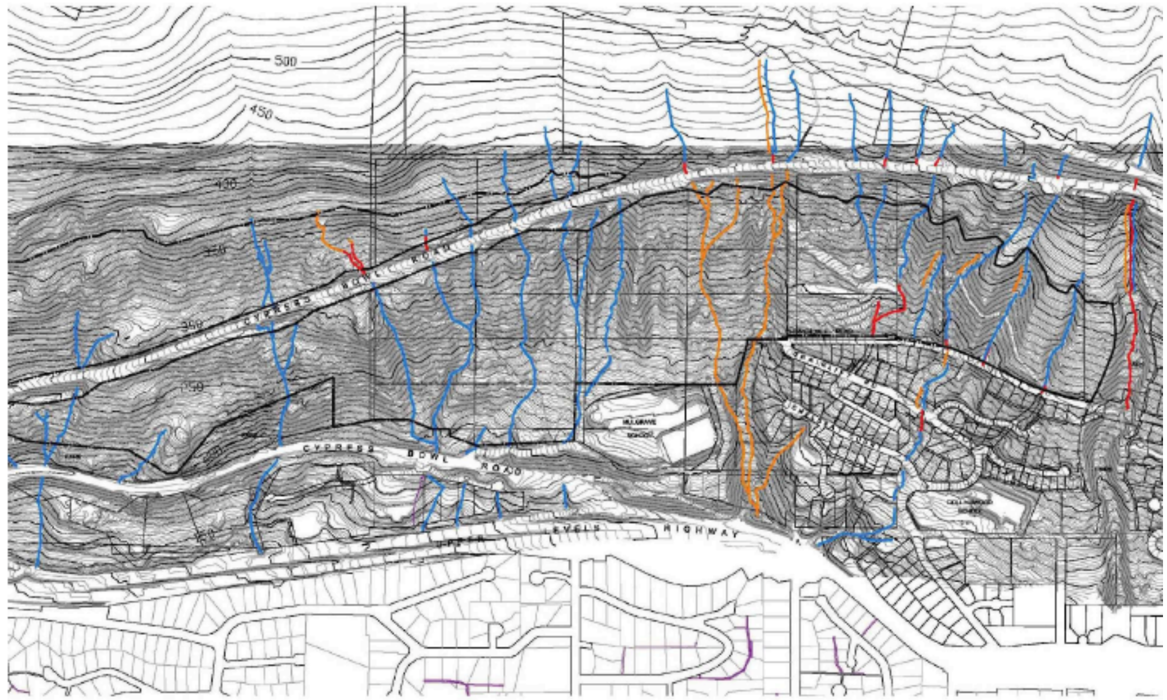
The Proper Functioning Condition (PFC) is a qualitative method for assessing the condition of riparian-wetland areas. The term PFC is used to describe both the assessment process, and a defined, on-the-ground condition of a riparian-wetland area. The PFC assessment refers to a consistent approach for considering hydrology, vegetation, and erosion/deposition (soils) attributes.

The on-the-ground condition termed PFC refers to how well the physical processes are functioning. PFC is a state of resiliency that will allow a riparian-wetland area to hold together during high-flow events with a high degree of reliability.

The PFC for each watercourse was part of the technical information used in assessing their ratings of overall value as H (high), M (moderate), and L (low).

Summary Matrix

The matrix in Appendix B describes each watercourse in the Rodgers Creek Area, including its proper functioning condition.



LEGEND

- Proper Functioning Condition
- At Risk of Losing PFC
- Not in Proper Fuctioning Condition

Sieve Analysis

Sieve Methodology

The sieve analysis has been a work in progress and has been refined a number of times. The original sieve analysis was prepared in collaboration with the land owners and District staff in an intensive workshop session.

Each watercourse was evaluated as a collaborative process at technical meetings, including detailed field review, and categorized with respect to its environmental values as H (high), M (moderate), or L (low).

Note: After the original Sieve Analysis was completed, the western boundary of the ADP Area above upper Cypress Bowl Road was expanded. The rationale for this expansion and the Sieve Analysis for this expanded area is included in Appendix A.

LEGEND

- PROTECTED SENIOR GOVERNMENT SETBACK AREA
- STEEPER SLOPES
- POTENTIAL GEOTECHNICAL CONSTRAINTS - FURTHER INVESTIGATION REQUIRED
- STEEP SLOPES - FURTHER INVESTIGATION REQUIRED
- MUNICIPAL SETBACK AREA - FURTHER INVESTIGATION REQUIRED
- DEVELOPMENT AREA
- LARGE FEATURE TREE
- C04 TRAIL/ ROAD (EXISTING)
- C04 MOUNTAIN PATH (PROPOSED)

Preliminary Planning and Conservation Areas

Working together, District staff and the Rodgers Creek land owners identified areas where development planning should focus. These development planning areas are enclosed in black outlines on the above map. The configuration of these potential development areas depends on the confirmation of road alignments, especially for the extension of the Chippendale connector road. District staff have not approved development within all of the areas shown. Final boundaries will be determined at the Development Permit stage and may be smaller than the areas shown, resulting in more conservation area being transferred to the District. Lands outside of the outlines will not be developed and will be preserved and enhanced.

The preliminary planning areas have been numbered and lettered for reference purposes.

Note: After the original Sieve Analysis was completed, the western boundary of the ADP Area above upper Cypress Bowl Road was expanded. The rationale for this expansion and the Sieve Analysis for this expanded area is included in Appendix A.

LEGEND

- PROTECTED SENIOR GOVERNMENT SETBACK AREA
- STEEPER SLOPES
- POTENTIAL GEOTECHNICAL CONSTRAINTS - FURTHER INVESTIGATION REQUIRED
- STEEP SLOPES – FURTHER INVESTIGATION REQUIRED
- MUNICIPAL SETBACK AREA - FURTHER INVESTIGATION REQUIRED
- DEVELOPMENT AREA
- LARGE FEATURE TREE
- C04 TRAIL/ ROAD (EXISTING)
- C04 MOUNTAIN PATH (PROPOSED)

KEY ORGANIZING PRINCIPLES FOR THE RODGERS CREEK AREA PLAN

OCP Community Building Principle 1 for the Upper Lands – Establish a sensitivity and connection to the natural environment and mountain qualities:

1.01 Keep development outside of environmentally sensitive areas (i.e, riparian areas, steep slopes, geotechnical hazard lands) and protect significant natural features; place both environmentally sensitive areas and significant natural features in public ownership wherever possible (OCP Policy page 101 and 103)

1.02 Avoid fragmentation of environmentally sensitive lands by creating large, continuous forested / natural areas throughout the planning area (OCP Policy page 101)

1.03 Avoid wide-scale clearing intended soley to provide uninterrupted, panoramic views, and minimize tree clearing on single family lots

1.04 Employ site sensitive built forms by:
- designing buildings to step into the terrain and using material and colours that harmonize with the forest setting; and
- minimizing footprints and visual impacts (OCP Policy page 108)

1.05 Minimize the need for 'constructed' responses by providing for road layouts, design standards and alignments that are sympathetic to the terrain and minimize site disruption including clearing of entire road right-of-ways, as set out in the Roads Policy 1999

1.06 Watercourses remain open and unimpeded, and are protected from change of course, piping, unnatural erosion and other human impacts.*

1.07 Provide multi-use utility corridors to minimize impact on the landscape

1.08 Trails may be provided along creek corridors, when located so as to minimize the impacts on riparian areas (OCP Policy page 101)

1.09 Natural, undisturbed areas (open spaces) and green connectivity belts are maximized and planned into housing complexes, and horizontal connections are treated as importantly as vertical connections.*

OCP Community Building Principle 2 for the Upper Lands – Create a strong community

2.01 Concentrate higher densities in areas that will foster strong community interaction (including a proposed commercial centre located to the west of the planning area) and outside environmentally sensitive lands

2.02 Ensure that the concerns and impact of new development on existing development adjacent to the planning area are identified and considered

2.03 Provide a 'mountain pathway' defined as:
- an east-west multi-use path, with gentle grades, for future residents of the planning area and the community at large; an
- a path that provides a connection to the natural setting and a physical connection to each neighbourhood within the Rodgers Creek Planning Area, and to a future commercial centre and neighbourhoods to the west; and
- a path that provides a variety of experiences and opportunities for people to meet, interact and connect (OCP Policy page 101)

2.04 Ensure that all destinations and public spaces including the mountain pathway (both its primary and secondary routes) provide for multiple activities by a variety of age groups and capabilities

2.05 Within the future Collingwood and Mulgrave Neighbourhoods, provide activity nodes along the mountain pathway that bring neighbours into regular social contact with each other. In the future development area at the west end of the Rodgers Creek Planning Area, provide community amenity buildings and facilities in addition to activity nodes along the mountain pathway.

2.06 Incorporate cultural heritage (such as logging and skilift history) and natural features (such as viewpoints, boulders and waterfalls) in activity nodes

2.07 Connect pedestrian and vehicle networks (including transit and cycling) into existing networks and with future amenities, including trails to and from the mountain

2.08 Include appropriate vehicle staging areas to ensure access to various public amenities and facilities

2.09 Provide for clear way-finding

2.10 Ensure all residential buildings are integrated into the landscape and have easy access to the mountain pathway

2.11 Continue the 1000-foot connector as the major east-west connecting road above the Upper Levels Highway (OCP Policy page 101)

2.12 Consider potential areas of synergy through the integration of Rodgers Creek Planning Area with future developments west of the Rodgers Creek Planning Area.

2.13 Identify existing recreational activities within and adjacent to the planning area and consider opportunities to retain, enhance and/or connect with these recreational activities.

OCP Community Building Principle 3 for the Upper Lands – Encourage a diverse community

3.01 Facilitate a diverse and more complete community by providing a variety of housing types and unit sizes

3.02 Provide opportunities for accessory housing such as such as coach houses, carriage houses and suites over garages and in the main dwelling, and do so by excluding them from total unit count

3.03 Ensure non-single family housing types include ground-oriented options such as duplexes, triplexes and townhouses

3.04 Ensure single family housing accounts for no more than 20% of the total housing units in the Rodgers Creek Planning Area (OCP Policy page 102: this policy provides for at least 40% non-single family homes in the entire Upper Lands; a higher percentage of non-single family is anticipated in the Rodgers Creek Planning Area)

3.05 Integrate housing with public/quasi public spaces and facilities, and connect with schools within the planning area and with the proposed commercial centre to the west

OCP Community Building Principle 4 for the Upper Lands – Focus on environmental and economic sustainability

4.01 Reduce the car-centric nature typical of new development with a focus on an effective movement system for pedestrian, cyclists and transit (OCP Policy page 101)

4.02 Green / sustainable design and operation standards, to a municipal standard that is being developed, form the foundation for building design

4.03 Strive for innovative, green infrastructure design and operation standards that minimizes immediate and life cycle cost

4.04 Think of rainwater as a resource, not a problem

4.05 Contribute to a resilient natural environment including healthy, properly functioning watercourses. Minimizing impervious surfaces and designing storm water systems to enhance watercourses are examples of measures that should be used to contribute to a resilient natural environment.*

4.06 Ensure that sustainability encompasses social sustainability, along with environmental and economic sustainability

4.07 Fish and fish habitat are conserved and protected*

4.08 Native vegetation is retained*

* From "Principles and objectives for Rodgers Creek Area Plan", adopted by West Vancouver Streamkeeper Society May 2, 2007.