Agricultural Nutrient Management Employing the Concept of Ecological Goods and Services:
A Valuation of Ground Water Quality in Abbotsford, British Columbia

by

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Abstract

There is an appetite for market based management mechanisms in agri-environmental policy. The purpose of this study is to explore how the market based concept of ecological goods and services (EGS) can be applied towards the management of an agricultural externality in British Columbia, Canada. Through literature review I establish the importance of valuation in market based management. With an EGS program in mind I identify the City of Abbotsford as a potential ecological service buyer and establish economic value for improvements to water quality in the Abbotsford-Sumas aquifer. I use a replacement cost approach based on the present value costs of the proposed Stave Lake surface water system. My results suggest that if nitrate remediation and/ or nitrate management practices improve water quality and the costs fall below $168 million there are potential net benefits to the City of Abbotsford. This value could form the basis for a city program or EGS trading scheme to encourage farmers to place a higher priority on water quality in their land management practices. A key finding is that lack of information on the degree to which nitrate contamination constrains well field development limits the ability to conclusively evaluate the net benefits of improved nitrate management and proceed with an EGS program.

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Glossary of Terms

Defense expenditure – A revealed preference valuation methodology based on costs incurred by an individual to defend oneself, in some cases, from poor environmental quality. For example, the cost of a water filter or bottled water to avoid poor well water quality.

Ecological goods and services (EGS) – Goods and services supplied through ecological function such as improved water quality rendered by the water filtration of wetlands. EGS are typically under valued or not valued by existing commodity markets (in other words they are non-market goods) and therefore are undersupplied.

Payments for ecological services (PES) – A transaction between an ecological service buyer and ecological service supplier (see page).

Pigouvian tax – A tax on a good or service that is meant to reduce the production of that good or service to a socially optimal level, taking into account the social cost of the negative externality resulting from the production of the good or service.

Replacement cost – A valuation methodology based on costs incurred to replace ecological function with man-made infrastructure. For example, the cost to build a water filtration plant to perform the function of a wetland.

Revealed preference – A valuation methodology that uses surveys and survey data to elicit willingness to pay estimates for non-market goods.

Stated preference – A valuation methodology that uses observations of market activity to estimate willingness to pay for non-market goods.

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Dedication

To my parents for their endless love and support. And to my husband for his love, help and perspective.

Chapter 1 – Overview and Summary

1.1 Introduction

Agriculture is the dominant human use of the Earth's landbase. At 5 billion hectares there is more land under agricultural cultivation than currently covered by forest and woodlands (Robertson and Swinton, 2005). Humans depend upon ecosystems, agriculturally managed or otherwise, to provide life sustaining services. The ability to grow food is one such service. The production of tradeable agricultural commodities is dependent on ecological systems that make the growth of food possible. Water filtration and carbon sequestration provide additional examples of ecological goods and services (EGS) amongst many. Considering the sizable land base under cultivation, agriculture is in an opportune position to contribute significantly to the provision of EGS and conversely to significantly degrade ecosystem services (Robertson and Swinton, 2005).

Arguably, agriculture is currently providing one service (food and fibre) to the detriment of other services it could provide (improved water quality, soil conservation, wildlife habitat, etc.) (Robertson and Swinton, 2005). In the case of agriculture a benefit, a positive externality or provision of an unpriced ecological service is averted damage and conversely, damage, or a negative externality is a missed benefit (Gerowitt et al., 2003). In either case, positive and negative externalities represent market failure. Where the market fails to arrive at a social optimum because environmental externalities exist there is a role for government intervention. Government intervention must be concerned with socially optimal outcomes which means maximizing social benefits and minimizing social costs.

Market-based government intervention can take several forms and depends on the accurate valuation of the externality. In the literature there is preference for market creation over regulation, taxes and subsidies because markets are adaptive and reflective of localized conditions (Gerowitt et al., 2003). The establishment of a market for a non-market good, such as payments for ecological goods and services (PES), is a kind of market-based management mechanism used to internalize externalities. PES requires that a well-defined environmental service or a land use likely to secure that service be bought by an identified service buyer from a service provider (Engel et al., 2008).

This thesis provides some preliminary exploration into how the concept of EGS can be used by policy makers to address externalities arising from agriculture in British Columbia, Canada. The Lower Fraser Valley (LFV) boasts a prominent agricultural industry that shares a bioregion with the urban metropolis of Greater Vancouver. It hosts some of the most fertile land in Canada and ranks amongst the highest concentrated agricultural activity in the country. There is measurable environmental impact documented in the literature that links increased agricultural intensity in the region to nutrient pollution of surface and ground water (Schreier, 1999, Liebscher et al., 1992, Wassenaar, 1995). Projected growth in the agricultural industry threatens to further exacerbate agricultural nutrient management issues in the LFV.

Currently, voluntary measures and regulations are in place to reduce the impact of agricultural activity on the environment but to unsatisfactory effect. Recent incentives to develop nutrient management plans and improve manure storage have not been shown to effectively reduce nitrate pollution of groundwater. Regulations in place to protect drinking water from agricultural pollutants are under enforced.

The British Columbia Ministry of Agriculture and Lands (BCMAL) is interested in the use of market-based management mechanisms to incentivize positive externalities and provide disincentives for negative externalities from agriculture. Market-based management relies on accurate valuation of non-market goods. The research presented in this thesis explores how the market for drinking water in Abbotsford, BC can be used to determine the value of improved nitrate management practices over the Abbotsford-Sumas aquifer. The City of Abbotsford and its public utility, the Abbotsford/Mission Water & Sewer Commission, is identified as a potential buyer of an environmental service from agricultural producers above the aquifer.

One can infer value for an externality by market oberservations of the willingness to pay (WTP) for it. The City of Abbotsford plans to build a new surface water system to increase the quantity and ensure the quality of drinking water available to Abbotsford and Mission residents. In as much as the Abbotsford-Sumas ground water aquifer is being avoided as a future drinking water source due to nitrate contamination, the WTP for ecological water supply services can be inferred from expenditure on a new surface water system.

1.2 Problem Statement and Research Objectives

This research topic was partially funded by the British Columbia (BC) Ministry of Agriculture and Lands and guided by a contract won by University of BC faculty members Dr. James Vercammen and Dr. Sumeet Gulati. To deliver on this contract I explore the concept of ecological goods and services (EGS) deliverable through BC agriculture and present a case study where agriculture is having a negative impact on the environment. This study explores the question: How can the concept of EGS be applied towards the management of an externality arising from BC agriculture?

The study of the concept of EGS as it applies to a particular agri-ecological issue in BC agriculture necessitated the following sequence of research objectives:

i. A review of the current literature on the economic concept of ecological goods and services as it applies to agriculture, policy tools available to elicit EGS and valuation methodology.
ii. Delimitation of a study region of particular interest in BC agriculture, the Lower Fraser Valley.
iii. A review of the environmental issues (negative externalities) that are a result of the agricultural practices, ecological conditions and social preferences of the region.
iv. Selection of an environmental issue of importance to the region and with available data.
v. A valuation exercise in the study region that measures the value of an agri-environmental externality and provides a starting point for further exploration into use of the concept of EGS.
vi. Recommendations pertinent to the study region as a result of the findings of the valuation.

1.3 Methods and Results

In this section I explain how the research presented in this thesis was conducted and I conclude with a summary of the research results. To begin research on this topic I reviewed environmental and public economics literature. I focussed on the externalities that arise from agriculture, the concept of ecological goods and services (EGS) and agri-enivronmental management tools available to policy makers.

There is a continuum of negative and positive externalities that arise from agriculture. The fuzzy distinction between negative and positive externalities has implications for policy intervention. The ideas of EGS and payments for environmental services (PES) move away from a 'polluter pays' principle towards a 'beneficiary pays' principle (Abler, 2004; Baylis et al., 2008; Coase, 1960; Engel et al., 2008; Gerowitt et al., 2003; Gowdy and O'Hara, 1995; Shortle and Horan, 2001; Wasennaar et al., 2006). The literature review also reveals that valuation, to determine a value for environmental externalities that are not currently valued by market forces, is key in the ability to set effective market-based policy.

Findings of the literature review necessitated that I focus on a particular region to research how the concept of EGS be applied towards the management of an agricultural externality. I focus on the Lower Fraser Valley (LFV) in BC, Canada which includes some of the most highly concentrated agricultural activity in the country. I use journal literature and government publications, interviews with government and professional agrologists and farm visits to assess the major agri-environmental issues facing the LFV industry, state of knowledge and potential data sources. As a result of this survey I focus on the externality of nitrate contamination of water, an agricultural issue with significant environmental implications and the subject of current policy intervention. I find that current policy approaches are regulatory but under enforced, or prescriptive and voluntary.

Having identified a region and agri-environmental issue of interest and, given the importance of valuation asserted in the literature, I proceeded to a subsequent research question: how can the economic value of the externality be measured? There are various approaches to valuation, all with their strengths and weaknesses. Some EGS lend themselves to particular valuation approaches (Farber et al., 2002). The replacement cost method is commonly used to evaluate willingness to pay (WTP) for ecological water supply services (de Groot et al., 2002). My replacement cost approach uses market observation of WTP for substitution technology to determine total WTP for an equivalent service supplied by ecological function.

With available data I use a revealed preference replacement cost valuation approach. An advantage to this approach is the identification of a potential service buyer in a payments for ecological services (PES) scheme: the City of Abbotsford. The city plans to expand their surface water (SW) infrastructure to supply future water demand. However, the city sits atop one of the largest and most productive ground water aquifers in the region. Based on scientific literature review, assessment of city documents and interviews with city staff there is reason to believe that the ground water aquifer is being avoided as a significant future source of drinking water because of agricultural nitrates that contaminate the water. Essentially, a new SW system is replacing what the aquifer could provide if it were not polluted. The City of Abbotsford's willingness to pay (WTP) for a new surface water system reveals their WTP for improved water quality and identifies them as a potential ecological service buyer.

I develop a theoretical model that illustrates the cost minimization problem of public utility expansion. The model is used to explore the parameters used in present value (PV) cost analysis, which is the basis of the valuation approach. In PV cost analysis of public projects it is only appropriate to use a social discount rate and sensitivity analyses of the results are imperative. Also relevant to PV cost analysis, the model indicates the impact water demand growth rates can have on optimal public utility expansion.

Lastly, I perform numerical evaluation of *the WTP for improved water quality* based on PV cost analysis of the proposed public utility expansion. I source data from engineering reports, city documents and the city budget. I also assess the PV costs of well field development to attain *partial costs* to deliver aquifer water supply services. I perform sensitivity analysis on the discount and growth rates used in the PV cost analyses. I also compare my findings with one other comparable valuation study. Due to the unknown degree to which nitrate pollution is a constraint on well field development I am unable to estimate total costs of aquifer water supply services.

I estimate a lower bound of *total WTP for improved water quality* at $181 million, or $15 million per year for 38 years. I also estimate partial costs of aquifer water supply services, based on the PV of well field development at $13 million, or $1 million per year for 38 years. I conclude that if nitrate remediation and/or nitrate management practices can secure the provision of improved ground water quality, and the costs of nitrate management fall below $14 million per year for 38 years there is a potential for net benefits to the City of Abbotsford.

However, a lack of information on the degree to which nitrate contamination constrains well field development limits the ability to conclusively evaluate the net benefits of improved nitrate management and proceed with an EGS program. I recommend and detail the research objectives of a hydrogeological study that would provide needed information.

1.4 Thesis Outline

Chapter two is a literature review that reviews the concept of ecological goods and services (EGS) as it is presented in economic literature. The literature review highlights a number of defining parameters for the chapters that follow.

Chapter three is a review of the scientific literature with regards to agricultural impact on the environment in the Lower Fraser Valley, a particularly agriculturally intense region of the province of British Columbia (BC). The issue of nutrient management is a well documented problem in the area and provides an interesting and policy relevant case study of a negative externality in BC agriculture. The policy environment around nutrient management and water quality issues is described.

Chapter four explains the theory underlying the economic evaluation of a negative externality performed in this thesis. The revealed preference replacement cost method is described. I show how this method is relevant in the case of nutrient pollution of the Abbotsford-Sumas aquifer given a number of assumptions. I develop a model to describe the cost minimization problem facing a public utility needing to expand. I use the model to explore the parameters involved in present value (PV) cost analysis which is useful since PV cost analysis becomes a central tool in the replacement cost methodology.

Chapter five presents the numerical evaluation of the total willingness to pay (WTP) for improved water quality of the Abbotsford-Sumas aquifer. The methodology, data, results and sensitivity analyses are presented.

In chapter six I discuss the valuation study findings and how they provide preliminary information towards the development of an EGS program.

Chapter seven concludes the thesis.

Chapter 2 – Literature Review

In the literature review that follows I review the economic theory underpinning the concept of ecological goods and services (EGS). Essentially, many EGS do not have value in existing markets making them externalities. The existence of externalities, a market failure, provides rationale for government intervention. I review agri-environmental policy tools and consider their suitability in various ecological, agricultural and political situations. Established literature asserts the importance of valuation for EGS policy. Valuation and its various approaches are reviewed as well as the valuation work that has already been done in the Lower Fraser Valley (LFV).

2.1 Review of Economic Theory and Market Failure

Modern welfare economics is premised on the fundamental theorem of competitive equilibrium. A market operating under perfect competition will arrive at a socially optimal state given that a number of conditions exist (Blaug, 2007). One of these conditions, or one of the assumptions upon which modern economic theory rests is that externalities and public goods do not exist. However in the real world this is rarely if ever the case. An externality exists whenever there are goods and services that are under or over valued by the market and are consequently under or over produced by the market. The existence of an externality is a case of market failure where the market fails to achieve a socially optimal state (Blaug, 2007).

Since the 1970's, negative environmental effects from agriculture have received considerable attention and have been extensively researched and documented in agri-science literature (Gerowitt et al., 2003). Table 1 summarizes the negative externalities arising from agriculture as presented in recent literature.

Table 1: Negative Externalities Arising from Agriculture

• Loss of species and habitat diversity. • Pollution of ground water, surface water and drinking water with nitrates, phosphorous and pesticides. • Deterioration of soils and soil function through compaction, erosion and accumulation of pollutants. • Pollution of the atmosphere, odours, fuel and greenhouse gas emissions. • Animal welfare. (Gerowitt et al., 2003; Abler, 2004; Peterson et al., 2002)

Negative agricultural externalities are side effects of agricultural practices deemed undesireable by society, or in other words, that decrease social welfare. Because the cost of the negative externality is not included in the cost of production, the good or service of which the externality is a by-product is overproduced. The negative externality varies in severity according to the agricultural practices in a region, agri-environmental conditions and demographics. Whether the activity results in a negative externality and the degree to which it imposes social welfare costs depends on whether the people affected by it perceive it as a problem and would be willing (and are able) to pay to avoid it or (if it is about to take effect) how much they would be willing to accept in compensation.

Positive agricultural externalities, on the other hand, are side effects of agricultural practices deemed desireable by society, or in other words, increase social welfare but go unremunerated. In the mid-1960's studies began to consider the social welfare value of ecosystem services rendered from the environment in its natural state (de Groot et al., 2002). However, only recently have studies started to consider the positive contributions agriculture can make (Baylis et al., 2008). Table 2 summarizes positive externalities that have been attributed to agriculture in the literature.

Table 2: Positive Externalities Arising from Agriculture

• Enhancement of biodiversity, wildlife habitat. • Reduction of nutrient run-off and soil erosion. • Prevention of natural hazards. • Ground water resource recharge. • Greenhouse gas sinks. • Increase in scenic vistas or spiritual and symbolic values of preserving farming heritage. • Open space, recreation and tourism. • Rural community and economic vitality. • "Natural" and organic goods and fiber products. • Food safety and security (Gerowitt et al., 2003; Baylis et al., 2008; Randall, 2007)

Positive agricultural externalities either arise from the normal functioning of an ecosystem, somewhat metered by agricultural activity, or are directly attributable to land used for agricultural purposes in a way valued by society. Because the benefit or unpriced service goes unremunerated the positive externality is underprovided. The positive externality will vary in importance according to the agricultural practices and agri-environmental conditions in a region. Whether the activity results in a positive externality also depends on whether the people affected by it perceive it as a benefit and would be willing to pay (WTP) for it. Further to this point, if benefits are enjoyed by a large population benefits will be multiplicatively higher than in a region with less people (Olewiler, 2007).

According to economic theory, when a market failure occurs such as when an externality exists, a role for government is implied. The study of government intervention is embodied by the field of public finance and is concerned with government program efficiency and effectiveness, minimizing distortionary effects of government programs on the market and policy analysis. Socially optimal government intervention is dependant on an accurate valuation of the externality the policy instrument is designed to internalize. Arriving at a reduced level of pollution that maximizes net benefits is possible only if the social welfare benefits supercede transaction costs of information, monitoring and enforcement (Abler, 2004; Engel et al., 2008).

Currently, public policy is used to address negative externalities from agriculture (Abler, 2004; Baylis et al., 2008). Taxes/subsidies, standards, markets, contracts/bonds and liability rules are mechanisms that have been examined by economists for their suitability in correcting negative externalities associated with agriculture (Shortle, 2001). In regards to positive externalities, recent studies highlight the under provision of non-commodity agricultural outputs and the role of government (Randall, 2007; Gerowitt et al., 2003). All authors of the studies surveyed see a role for government to provide incentives for the optimal production of ecological goods and services (EGS). The following section takes a closer look at the tools available to agri-environmental policy makers.

As a final point on negative and positive externalities, Gerowitt et al. (2003) suggest there is a continuum of negative and positive externalities in agriculture and the fuzzy distinction between the two does not lend itself to simple intervention. Note that the ecologically related positive externalities listed in Table 2 are in direct contrast to the negative externalities listed in Table 1: the enhancement of biodiversity and wildlife habitat versus loss of biodiversity and wildlife habitat for example. Economic literature has in the past argued that remunerable ecological services are positive external effects while the costs of negative externalities are to be borne by the producer (Gerowitt et al., 2003). The latter is the “polluter pays” principle which provides rationale for command and control regulation. In the case of ecological services from agriculture there is continuity between the two distinctions of positive and negative externalities. A benefit, or provision of an ecological service is averted damage and conversely, damage a missed benefit (Gerowitt et al., 2003). Where there is demand for a particular level of environmental quality a more effective policy stance may be the 'beneficiary pays' principle: payments for ecological services (PES), discussed in the next section.

From here, I move away from the theoretical underpinnings of market failure as it relates to agriculture and the environment. Having established a role for government in the provision of EGS I discuss the mechanisms available for government intervention to address market failure.

2.2 Review of Government Intervention Options

A 2004 OECD report states that in Canada eco-efficiency is constrained by inappropriate market signals and that market-based instruments are insufficiently used. The same report states that too much emphasis is given to voluntary guidelines and command and control, otherwise termed regulation (Adamowicz, 2007). In this section I consider various policy options available to government to promote ecological goods and services (EGS) through the agricultural sector. Policy options are summarized in Table 3.

Command and control measures are notably inefficient in delivering cost effective environmental quality. However, they can be highly effective at achieving environmental objectives and tend to be utilized when perceived social costs warrant an outright ban. Inefficiencies arise from the blanket approach that applies the same limits to all situations and regions and incurs costs of monitoring and enforcement irrespective of the benefits attained (Engel, 2008). An agricultural example is explored by Huhtala and Marklund (2008) who look at phosphorus regulation in Finland which sets limits on manure and fertilizer application to farmers' fields in order to minimize water pollution. They find that shadow prices of compliance change depending on the density of manure in particular regions. They also find inefficiencies in production noting that some producers are not constrained by regulations and that environmental impacts could be reduced with no additional abatement cost to the producer. However, there is no incentive to do so. To generalize this study, effective environmental policies need to be reflective of localized conditions (Huhtala and Marklund, 2008).

Pigouvian taxes are a market-based tool that theoretically reduce negative externalities to a desired level. To maximize net social benefits the tax must be based on accurate measures of costs and benefits and as a result taxes have high informational costs. There are also distributional concerns when considering taxes as the cost of environmental protection is imposed on land users rather than service users (Engel, 2008). Huhtala and Marklund et al. (2008) note that in the case of water pollution from manure use monitoring is prohibitively expensive and actual pollution impact cannot be observed. Furthermore, in regions where environmental damage is most urgent "the high shadow price of environmental restrictions on nutrient use indicate that any tax that would effectively restrict production in such areas should be quite high indeed" (Huhtala and Marklund, 2008) with indication that those taxes would not be politically viable.[1] Taxes are not championed with respect to agricultural policy in the literature surveyed in this research.

Command and control as well as Pigouvian tax policies are used to address negative externalities. But what about positive externalities? Engel et al. suggest that from the perspective of recipients payments for ecological services (PES) act like environmental subsidies: "a payment aimed

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[1] A shadow price is the change in an optimized value (the solution to an optimization problem) by relaxing a constraint by one unit. For example, the marginal cost to the farmer of a strengthened manure application limit.

at inducing increases in environmentally beneficial activities" (Engel, 2008). Subsidies suffer from potential inefficiencies. Among them are the potential lack of additionality (subsidizing desireable activities that may have occured anyways), the inadvertent creation of perverse incentives and the potential to raise profitability of one activity displacing preferred outcomes (Greenhalgh et al., 2005, Peterson et al., 2002, Abler, 2004). Subsidies, like taxes, also involve informational, monitoring and enforcement costs. Agricultural subsidies are certainly a common policy tool within developed countries and are already in use in the EU and US to provide incentives for the provision of EGS (Randall, 2007). Taken hand-in-hand with an evaluated market demand PES appears less like direct subsidies and more like a functioning market for EGS.

It must be noted that the agricultural sector worldwide is already widely distorted by subsidies and trade barriers that may be working in opposition to environmental objectives (Randall, 2007, Huhtala et al, 2008). Any effective government effort at creating a market for EGS would first address the effects of existing policies on the provision of EGS before considering targeted incentives. Randall (2007) goes on to consider how commodity pricing and EGS pricing could be reconciled by disposing of commodity price supports in favour of EGS pricing. "Multifunctional agriculture policy bears more relationship to public goods policy than to commodity price policy. It is aimed first at delivering green products and services efficiently to the public, and only instrumentally at delivering money to farmers who supply those green things" (Randall, 2007). As it is with ongoing trade negotiations traditional agricultural subsidies are under threat. There is concern that payments for ecological services (PES) are a repackaging of domestic subsidies. However, well designed programs could avert such suspicion (Baylis et al., 2008).

There appears to be consensus and enthusiasm in the economics literature for the creation of markets for EGS in agriculture. In favour of markets for EGS and speaking to the role of government, Gerowitt et al. (2003) states:

> According to the majority of economists, it is also desirable to profit from the advantages of a market economy when supplying the population with agri-environmental goods. Markets for these collective goods are not as easy to establish as markets for private goods; governmental intervention must be stronger. This strategy can only lead to the desired benefits (provision of goods for members of society at lowest cost) when certain requirements are fulfilled and essential constitutive criteria are considered (Gerowitt et al., 2003).

Gerowitt et al. embarks on an early exploration of what these requirements and criteria might be. More recent literature, cited by Engel et al. embraces similar ideas and present the formal definition of payment for ecological services (PES) as:

> (a) a voluntary transaction where
> (b) a well-defined environmental service (or a land use likely to secure that service)
> (c) is being 'bought' by a (minimum one) service buyer
> (d) from a (minimum one) service provider
> (e) if and only if the service provider secures service provision (conditionality)
> (Wunder, as cited by Engel et al., 2008).

Randall adds that "green prices should be targeted ultimately to local conditions as they affect demand and supply, with farm-level monitoring so that payments can be linked to demonstrated farmer performance. Taking these concerns seriously places a heavy burden on the valuation task" (Randall, 2007). Once the social benefits of EGS are determined through the valuation task contracts can be awarded by the government through bidding processes taking advantage of farmers' own knowledge of their production possibilities. In this way green prices, or the price of EGS can be determined. Such a framework for cost-effective delivery of EGS takes advantage of efficiencies of the market as contracts will be awarded to whoever is able to deliver the most value (Randall, 2007, Baylis et al., 2008).

Literature exploring PES in agriculture is emerging but is still largely confined to theory lacking empirical assessment of mechanisms at work. Few agricultural PES mechanisms have been carefully documented (Engel et al., 2008). One documented program is the agriculturally focussed watershed management program in the state of New York, developed and implemented since the 1990's (Smith and Porter, 2010; www.nyc.org/dep). The program incentivizes on-site practices, such as whole farm planning on private land to deliver safe drinking water. As described in the next section on valuation, the city has avoided having to build a water filtration plant for the Catskill and Delaware watersheds because improved land use has delivered the ecological service of clean drinking water. Smith (2010) documents a number of learned lessons from the program including the importance of cross-sectoral stakeholder involvement, a target-based approach for water quality based on science and adequate research, an adaptive management plan and enabling policy and regulations.

Baylis et al. (2008), Randall (2007) and Engel et al. (2008) explore and compare existing agri-environmental government programs noting that a mix of policy tools including control and command and EGS markets may be desirable to ensure additionality. Baylis et al. (2008) suggest that “cross-compliance linkages, by requiring producers to meet minimum environmental standards before becoming eligible for any farm payments, play an important role in ensuring the efficient delivery of environmental services” (Baylis et al., 2008). In a review of 26 active water quality trading programs mostly within the USA and one in South Nation river watershed, Ontario, Canada, policy driven nutrient limits proved essential to market creation (Selman, 2009). The review also suggests that agricultural producers meet minimum performance standards to qualify as sellers of ecological services. The developing literature on water quality trading, where a cap on emissions is set and ecological service providers earn credits saleable to other emitters, provides insight into the development of markets for agricultural EGS and the important role of government (Lal et al., 2009; Connor et al., 2009; Selman et al., 2009; Mariola, 2011, O'Grady, 2011).

In summary, the success of government intervention to address environmental externalities in agriculture depends on an accurate measure of the actual costs and benefits to society, taking into consideration both demand and supply conditions. Government intervention that is flexible in its response to demand and supply and reflects localized conditions will be most efficient in its delivery of EGS. Equally important are the transaction costs including informational requirements (valuation), administration, monitoring and enforcement. The actual value of costs and benefits to society and transaction costs themselves have implications for whether government should intervene at all if benefits are to be accrued. The degree of scale and targeting needs to be considered to achieve government program efficiency (Abler, 2004, Shortle and Horan, 2001, Engel et al., 2008, Randall, 2007).

Table 3 summarizes the above discussion with a list of the tools available to policy makers charged with the task of reducing agri-environmental externalities. Further attention is given in the table than in the above discussion to the appropriate use of measures and potential issues that may arise in the case of each policy instrument.

Table 3: Summary of Policy Instruments Available to Achieve Agri-Environmental Objectives

Policy Instrument	Appropriate Use	Studies
Education *also known as* Moral suasion, technical assistance, public information. ***Examples:*** Promotion of beneficial management practices e.g. Environmental Farm Plan program, Canada Extension agents. Fact sheets. Shortle and Horan, 2001	When other means are politically unviable.	Kemkes et al., 2009
	When changes in production methods increase both social and private benefits.	Gowdy and O'Hara, 1995
	Asymmetric information about on-farm environmental parameters between farmers and regulators e.g. Government scientists more aware of environmental impacts on farms than farmers because in current markets the farmer's primary concern is commodity production.	Shortle and Horan, 2001
	Potential Issues	**Studies**
	Voluntary take-up of environmentally beneficial technologies can lead to underprovision of environmental services particularly where opportunity costs are high and implementation efforts are costly. An incentive-based mixed policy approach could link payments and cost-sharing to educational initiatives. See *subsidies/ payments*.	Wasennaar et al., 2006; Kemkes et al., 2009
	"Noncompliant 'offenders' in highly vulnerable settings have the potential to endanger overall environmental and water quality objectives."	Wasennaar et al., 2006
Policy Instrument	**Appropriate Use**	**Studies**
Public research & development programs *a.k.a.* Research & development, public services, public investment, public funding ***Examples:*** Experimental farms, Canada. Land grant universities, USA. Extension agents. Technical Support. Marketing services.	When social benefits of research outweigh the public cost incurred to fund the research. Large body of literature shows that public agricultural research increases social welfare.	Hamilton et al, 1998
	To increase income and reduce uncertainty/ variability of annual farm revenues. Public services directed at agriculture found to be a substitute policy for direct payments to farmers in the Netherlands.	Feinerman et al., 2007
	Initial outlay for environmental quality indices can lead to long-term efficiency of agri-environmental programs (e.g. development of the Environmental Benefits Index as part of the US conservation program) minimizing transaction costs after the program has been established.	Baylis et al., 2008
	Potential Issues	**Studies**
	Lack of information transfer to farms.	
	Difficult to ascertain value of research before it is conducted.	

Policy Instrument	Appropriate Use	Studies
Regulation *a.k.a.* command and control, incentive blocking, blanket approach. ***Examples:*** Performance based standards, cap (e.g. emission or emissions proxy standards: restrictions on fertilizer application rates/ timing or modeled nutrient loadings) Technology based standards, prescriptive rather than performance based (e.g. mandatory pollution control practices: restrictions on livestock stocking rates, manure storage capacity) Registration (e.g. Pesticide registration with Health Canada) Shortle and Horan, 2001; Greenhalgh et al., 2005	When the externality represents exorbitant social costs and is deemed to warrant an outright ban, e.g. habitat protection for endangered species, pesticide ban. Can be highly effective in achieving environmental objectives.	Shortle and Horan, 2001; Kemkes et al, 2009
	When the source(s) of the externality can be identified and metered with a reasonable degree of accuracy at reasonable cost.	Shortle and Horan, 2001
	Information symmetry: where the cost functions of the producer is known to both the producer and regulator. In cases of asymmetric information see *taxes* and *subsidies*.	Shortle and Horan, 2001
	As a cross-compliance mechanism/ part of a mixed approach. E.g. Farmers must adhere to a minimum set of environmental practices to be eligible for income support payments in parts of the EU; Required pesticide registration serves as a basis for further management mechanisms.	Shortle and Horan, 2001; Engel et al., 2008; Baylis et al., 2008
	Potential Issues	**Studies**
	Tradeoff between informational/ transaction costs and efficiency: • Accurate information incurs high costs particularly in the case of non-point pollution. • Inefficiencies or even negative net social benefits if the standard is based on inaccurate information.	Shortle and Horan, 2001; Greenstone and Gayer, 2009; Greenhalgh et al., 2005; Gerowitt et al., 2003
	Financial burden on regulators to keep abreast of new technological advances rather than providing incentives for private innovation that could surpass expectations.	Greenhalgh et al., 2005
	Same as taxes/ subsidies: Which point of intervention? Inputs, practices, emissions proxy (nutrient balance) or ambient concentrations. Has implications for who to target and enforcement costs depending on the size of target group e.g. pesticide manufacturers vs. farmers, or farmer groups with largest externalities How information is gathered e.g. self-reporting, may create conditions for moral hazard and adverse selection.	Shortle and Horan, 2001
	Technology based standards: Inadvertent creation of perverse incentives due to imperfect understanding of ecological processes and the effects of agricultural practices rather than allowing producers to find their own solutions to comply with standards.	Shortle and Horan, 2001; Gerowitt et al., 2003
	Coercive measures tend to have low political support.	Kemkes et al., 2009

Policy Instrument	Appropriate Use	Studies
Taxes *a.k.a.* Pigouvian taxes, incentive adjustment, market-based incentives to decrease production of negative externalities, the "stick". ***Examples:*** Input based taxes on factors of production responsible for negative externalities (e.g. fertilizers and pesticides) Output based taxes on outputs responsible for negative externalities (e.g. phosphorous content of manure) Shortle and Horan, 2001	When the negative externality is observable, measurable and **priceable** through valuation. "One need only place explicit values on the non-commodity outputs so that farmers... are penalised for producing those that impose social costs."	Peterson et al., 2002
	In the case of assymetric information: when the producer is more knowledgeable regarding their production/ abatement cost function than the regulator. The main way a tax differs from a regulation: individual agents (e.g. Farmers) make decisions based on their marginal costs and have an incentive to be innovative.	Shortle and Horan, 2001
	To take advantage of government economies of scale that effectively reduces transaction costs between multiple agents by performing lump sum transfers (when compared with a decentralized user-financed/ market-based system), or to enact a policy that has multiple outcomes.	Coase, 1960; Abler, 2004; Engel et al., 2008
	Potential Issues	**Studies**
	Impractical in an agri-environmental context since taxes require that externalities, a.k.a non-market goods be easily observed, measured and **priced.** Dependent on accurate valuation of costs and benefits (observable, measurable & priceable) of non-market goods.	Peterson et al., 2002
	Monitoring costs to ensure ongoing efficiency.	Engel et al, 2008
	Politically unviable where environmental damage is most urgent (where taxes would be very high and seriously impede agricultural activity).	
	Inadvertent creation of perverse incentives due to imperfect understanding of ecological processes and the effects of agricultural practices.	Shortle and Horan, 2001; Gerowitt et al., 2003
	Same as regulation: Which point of intervention? Inputs, practices, emissions proxy (nutrient balance) or ambient concentrations. Has implications for who to target and enforcement costs depending on the size of target group e.g. pesticide manufacturers vs. farmers. How information is gathered e.g. self-reporting, may create conditions for moral hazard and adverse selection.	Shortle and Horan, 2001
	Distributional and equity concerns. Service providers pay rather than service users.	Gowdy and O'Hara, 1995; Engel et al., 2008

Policy Instrument	Appropriate Use	Studies
Subsidies/ Payments *a.k.a.* Pigouvian subsidies, incentive adjustment, market-based incentives to decrease (increase) production of negative (positive) externalities, Payments for Ecological Goods and Services (PES), price/ income supports, quotas, collaborative exchange, market creation, tradeable permits, reverse auctions, the "carrot". ***Examples:*** *Prescriptive:* Best practices adoption subsidies e.g. BC Environmental Farm Plan manure storage infrastructure cost-sharing, US Conservation Reserve Program Direct Payments e.g. organic acreage in Netherlands *Performance Based:* Tradeable Permits, Auctions, cap & trade (a mix of regulation and market creation) Greenhalgh et al., 2005; Feinerman et al., 2007	See *taxes* above. Subsidies are effectively a positive tax. Can be considered a payment from service users (taxpayers) to service providers (farmers): • When a positive externality is observable, measurable and priceable through valuation (including the demand of the service) • There is asymmetric information • There are government economies of scale • There is marginal cost of damage to the ecosystem and opportunity costs of provision are low	Shortle and Horan, 2001; Peterson et al., 2002; Engel et al., 2008; Coase, 1960; Abler, 2004; Kemkes et al., 2009; Steele, 2009
	Increasingly accepted as a cost-effective approach for achieving environmental quality goals potentially at lower social cost than command and control.	Shortle and Horan, 2001, Greenhalgh et al., 2005
	As a method of easing into compliance with regulations. Makes transition from support payments to agri-environmental programs politically feasible.	Baylis et al., 2008; Engel et al., 2008
	When there is a monopsony buyer (e.g. hydroelectric company, or government on behalf of service users) and service providers have opportunity costs (e.g. agriculture is the next best land use).	Kemkes et al., 2009
	To reduce transaction costs where infrastructure is already in place (status quo) – ag price support programs are estimated to have 1-5% transaction costs, whereas agri-environmental programs 5-50%, something that can be addressed by targeting agri-environmental programs. Also termed "automaticity".	Abler, 2004; Hackl et al., 2007; Kemkes et al., 2009; Steele, 2009
	Found to support the provision of agri-environmental goods and services when used in concert with minimum environmental standards (regulations).	Baylis et al., 2008
	Potential Issues	
	Subsidies tied to commodity production is not a first best solution to provide agri-environmental goods and services and has at times exacerbated agri-environmental issues (e.g. Providing incentive to bring marginal lands into production that otherwise would not have been). Directly targeting agri-environmental goods and services is more desireable.	Peterson et al., 2002; Abler, 2004; Brunstad et al., 2005
	Lack of additionality. How to establish a baseline and avoid subsidizing desireable activities that may have occured anyways.	Greenhalgh et al., 2005; Engel et al. 2008; Baylis et al., 2008

Policy Instrument	Potential Issues	Studies
Subsidies/ Payments *(continued)*	Leakage: shifting undesireable environmental impacts elsewhere.	Engel et al., 2008
	Inadvertent creation of perverse incentives due to imperfect understanding of ecological processes and the effects of agricultural practices. If prescriptive of practices subsidies do not allow individual agents to choose their best course of action to achieve a goal.	Greenhalgh et al., 2005; Gerowitt et al., 2003; Engel et al., 2008
	If there is a monopoly service provider or where an ecosystem threshold exists – in which case *regulation* or *education* may better policy options.	Kemkes et al., 2009
	Tradeable Permits: Establishment of the system - who to endow with polluting rights? Informational requirements e.g. what is the optimal ratio of trading between point source and non-point source polluters?	Shortle and Horan, 2001

2.3 Review of Valuation

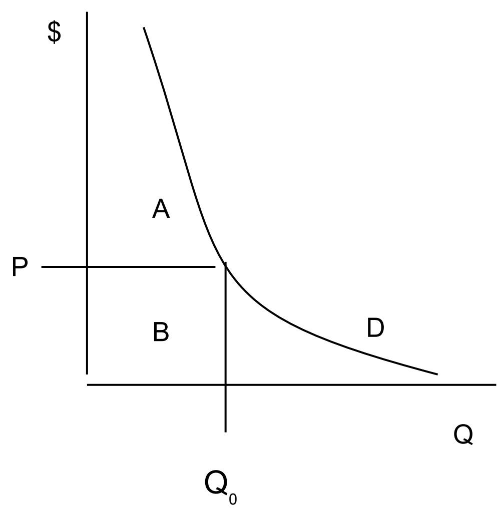
In this section I present a brief overview of valuation: its purpose, underlying theory and variations on methodology. I do not deliberate at length the strengths and weaknesses of different approaches to valuation. Brander et al. (2006) and Randall (2007) are examples of recent literature that do this well. Rather, I provide reasoning for the replacement cost valuation methodology used in this thesis and theoretical context for comprehending the results. I conclude the section with a summary of the valuation studies already performed in the Lower Fraser Valley (LFV) so the reader is aware of the range of valuation work done in the region.

Valuation is key in government provision of ecological goods and services (EGS) because it enables the design of incentives based on explicit prices for non-commodity outputs. It is also valuable in the assessment of policies and programs and whether they achieve their intended benefits cost effectively (Randall, 2007). In this thesis I explore the possibility of determining an explicit price for a non-commodity output.

Valuation measures people's utility-based value for goods and services (Farber et al., 2002). Neo-classical economic theory assumes people maximize welfare by allocating their resources (time and money) across goods and services that suit their preferences best. People allocate on the margin, meaning that by trading a unit of time or money for one unit of good or service, they are giving up a unit of something else. A person's (or group's) marginal willingness to pay (WTP) (a measure of their marginal utility value) can be observed in how they react to a change in price for a good or service.

The aggregation of individual (or a group's) WTP forms the market demand for a good or service. Demand curve *D* illustrated in Figure 1, represents marginal WTP for each unit of good or service, with quanity *Q* on the horizontal axis and price *P* on the vertical axis. 'Total' WTP for Q_0 units of the good or service is the area beneath the demand curve, represented by the aggregated areas of *A* + *B* (Farber et al., 2002). The terms utility value, WTP and 'benefit' are used interchangeably here.

Figure 1: Utility and Exchange Based Value of Goods and Services (Farber et al., 2002)



Where markets exist price *P* (in Figure 1) is the marginal exchange value of the good or service. 'Total' exchange value of Q_0 is $P \times Q_0$ (Farber et al., 2002). Where markets do not exist the goal of valuation is to determine the price *P* of non-market goods. In the case of ecological goods and services economists have developed several ways to estimate utility value indirectly.

The two main categorizations of valuation are 1) stated preference and 2) evidence from existing markets, or otherwise termed, revealed preference. Stated preference, or the contingent valuation (CV) method uses surveys to elicit WTP or willingness to accept (WTA) compensation. Hypothetical situations are posed to a sample of respondents to determine how they value a certain level of good or service. WTP is estimated when respondents are asked how much they would pay to attain an unpriced good or service that they do not already have. WTA is estimated when respondents are asked how much they would accept in compensation to give up a service they already enjoy. Estimates of WTP or WTA are inferred from statistical analysis of the responses.

Revealed preference methodology is based on market observation. De Groot et al. (2002) further classify revealed preference methods into observation of direct market pricing (where a market for EGS has already been established) and indirect market pricing. Establishment of a market for EGS first depends on indirect market valuation using one of the following approaches: avoided cost, replacement cost, factor income, travel cost or hedonic pricing. I focus my attention on an example of the replacement cost method, the valuation approach used in this thesis, which in the example was then followed by market creation and direct market pricing.

New York City provides an example of indirect market pricing which led to EGS market creation. New York City evaluated WTP for the ecological function of watersheds using a replacement cost approach. A water filtration plant and the filtration services of a watershed are substitute technologies both able to fulfill the preferences of New York City residents for safe water. By using the filtration services of mostly undeveloped watersheds the city does not need to build a $6 billion water filtration plant. This cost of the city's next best option to attain a level of water services reveals their total WTP for water filtration services using replacement cost methodology (de Groot et al., 2002).

New York City's objective is to minimize cost to attain safe water and will only buy filtration services of watersheds if the cost falls below their total WTP for a filtration plant ($6 billion). Through a wetland trading program the exchange value, or cost $P \times Q_0$ of attaining watershed services from landowners was revealed to be below their total WTP. As a result of market creation the exchange value of wetland services in New York City and more specifically an explicit price for the water filtration service of a watershed is observable through direct market valuation (de Groot et al., 2002). This concludes my brief explanation of stated and revealed preference valuation methods.

Academics are pursuing experimental economics, sophisticated survey design, meta-analyses, construct validity tests, standardization and classification to refine valuation techniques and improve the comparability and applicability of valuation estimates (de Groot et al., 2002; Johnston, 2003; Brander et al., 2006; Randall, 2007). Each valuation technique has its strengths and weaknesses. Contingent valuation (CV) surveys (stated preference methods) where people are asked their WTP rather than observing actual behaviour were at first regarded as suspect but has since been generally accepted (Laughland, 1996). CV remains the only method to estimate non-use values and hence total economic value of ecological goods and services (Brander et al., 2006). An example of a non-use value (or indirect utility value) is people's WTP for conserving an ecological area not because they intend to visit but because they enjoy knowing it exists.

Economists prefer measures of value derived from observed market behaviour. People's actions speak louder than words. However, revealed preference methods have the difficult task of finding market based observations where markets do not exist often with the use of a proxy. Where there are ready proxies particular valuation approaches have been developed. For example, where there are man-made alternatives such as a water filtration plant that provides ecological function the replacement cost approach is commonly used (de Groot et al., 2002). Criticism of the replacement cost method is two fold. As in the New York example above, only partial value rather than total economic value is captured because the cost of a substitute does not capture the passive use or existence (non-use) values of an ecosystem function (Hauser and van Kooten, 1993; Brander et al., 2006). The result is downward bias in estimates generated from the replacement cost method.

On the other hand, the replacement cost method is criticized for its predisposition to inflated estimates because the total replacement cost of natural capital may not represent the optimal level of restoration by which society attains an optimized benefit (Garrod et al., 2001; Farber et al., 2002). This inflationary criticism of the replacement cost method highlights the difference between estimating total utility and marginal utility value. As described in the New York City example above, replacement cost valuation followed by market creation (or a model of market creation) can determine the marginal value (or the exchange value/ explicit price) of an ecological service. Randall concludes there is a need to improve all measures, given they measure different ecological services and sees progress in the use of correlation between CV estimates and market-based estimates to test validity of either method (Randall, 2007).

Aside from the effectiveness of each method in a given circumstance, the choice of which valuation method to use is influenced by costs to conduct studies and data availability. The objective of valuation is to determine an explicit price of an ecological good or service based on its marginal utility value. It would be ideal but is particularly challenging to design a study where the researcher can estimate marginal utility based on a change in utility (benefits) to a change in environmental quality, and further, to make a direct link between that marginal change in environmental quality to a change in agricultural practice. Environmental economics is fraught with data limitations since environmental data itself is costly to acquire. Cost of valuation and the data it requires represents transaction costs in the government provision of ecological goods and services.

Similar to the New York City example, the valuation performed in this thesis evaluates the City of Abbotsford's total WTP for water supply services based on the cost of a substitute. Where New York City showed WTP for the ecological service of water filtration, the City of Abbotsford has a WTP for the ecological service of water supply. Where New York City's total WTP for filtration services was based on the cost of a filtration plant, the City of Abbotsford's WTP for water supply services are based, in this thesis, on the cost of a surface water supply system (which includes a filtration plant). The surface water supply system is substitute technology for a natural groundwater aquifer that lies beneath the city. This thesis evaluates total WTP for the services of a natural aquifer based on the cost of its replacement. The rationale behind this methodology is further developed in section 4.1.

As in the New York City example, an estimate of total WTP could provide a basis for a trading program with other aquifer service users, in this case farmers, if the cost of the aquifer services are found to fall below the city's total WTP. However, determining the total cost of aquifer services falls beyond the scope of this thesis. Once the cost of aquifer services deliverable through agriculture are discoverable through market creation, the exchange value and an explicit price for the ecosystem service can be determined through direct market pricing.

A number of valuation studies have already been done in the Lower Fraser Valley (LFV) and are summarized in Table 4. The studies estimate different values relevant to agriculture, ranging from the benefits of wetland ecological services to the economic impact of pesticides on salmonid habitat. Only one other study performed in this region, Hauser and van Kooten (1993), estimates the benefits of improvement to Abbotsford-Sumas aquifer water quality, also the subject of the valuation exercise presented in this thesis.

The Hauser and van Kooten (1993) study uses both revealed and stated preference methodology. The revealed preference methodology used by Hauser and van Kooten estimates WTP based on private household (defense) expenditure on bottled water and water filters (determined by survey). Hauser and van Kooten consider the estimate generated from defense expenditure to be a validity test for the WTP estimates which are a more accurate estimate of total economic value. The stated preference methodologies used by Hauser and van Kooten estimate WTP based on contingent valuation survey and fuzzy pair wise comparisons that reveal what Abbotsford residents would pay for improved water quality. As stated in the study, the revealed preference estimate provide a lower bound estimate for the WTP survey estimates and validates the construct of the stated preference methodology.

The results of the Hauser and van Kooten (1993) study, after adjusting for inflation, are compared with the results of the valuation exercise performed in this thesis in section 6.1. Aside from methodology, other differences between the Hauser and van Kooten valuation and the valuation performed in this thesis is that this thesis explicity identifies a service buyer, the City of Abbotsford, and measures *partial cost* of aquifer water supply services.

The other studies listed in Table 4 measure various attributes of farmland or ecological services upon which agriculture has an impact. Their values are not directly comparable to the results of the valuation exercise performed in this thesis since they do not measure the benefits of improvements to aquifer water quality. However, the studies do provide an overview of the agri-ecological valuation work done in the region and give some context for the present study. The cited studies also provide specific examples of stated and revealed preference methodology at work.

Table 4: Previous Agri-Environmental Valuation Studies in the Lower Fraser Valley

Location	**Valued**	**Benefits**	**Costs**	**Methodology**
		(in dollar amounts of the year in which the study was conducted)		
Abbotsford-Sumas aquifer, BC	Improvements to aquifer water quality Assumes 6,300 households in the Abbotsford region	**\$70-\$284/ yr/ household,** with the lower bound estimate based on defence expenditure and upper bound deemed the more appropriate measure of total economic value based on WTP survey with fuzzy pair wise comparisons **\$1.8 million/ yr**		**Stated & Revealed Preference:** contingent valuation survey, fuzzy pair wise comparisons & defence expenditures based on purchases of bottled water and water filters (Hauser and van Kooten, 1993)
South Thompson River spawning grounds (tributary to the Fraser River) and Strait of Georgia catch area, Southwest BC	Spawning and rearing habitat conservation to the commercial and recreational coho salmon fishery	**\$0.93 – \$2.63/ ha** of drainage basin **\$1322 – \$7010/ km** of salmon stream length		**Revealed Preference:** market (production function) valuation that links environmental quality to production relationships (Knowler, 2003)

Location	Valued	Benefits	Costs	Methodology
		(in dollar amounts of the year in which the study was conducted)		
Lower Fraser Valley, BC	Waste treatment value of wetlands, nitrogen and phosphorous (relatable to agriculture through set-asides on farm properties for wetland restoration)	**$452 - $1,270/ ha/ yr** **$18 - $50 million/ yr** for 40,000 ha of LFV wetlands		**Revealed Preference**: substitute cost method based on the costs of removing phosphorous and nitrogen in Vancouver's primary and secondary waste treatment plants (Olewiler, 2004)
various locations; Lower Fraser Valley, BC	Total economic value of wetlands: fish, shellfish, waterfowl, mammal and reptile habitat; water supply; erosion, wind, wave barrier; storm and flood control (relatable to agriculture through set-asides on farm properties for wetland restoration)	**$5,792 - $24,330/ ha/yr** **$231.7 million/ yr** for 40,000 ha of LFV wetlands Burns Bog: **$60 million/ yr** ($31,375 ha/yr* 1,890 ha)		cites several studies of varying methodology (Olewiler, 2004)
Abbotsford, BC	Public Amenity benefits of preserving farmland, annuity in perpetuity (value stream) Based on: 26,055 ha of agricultural land	**$65,527/ ha** WTP to preserve farmland; **$150 million** for existing agricultural land in Abbotsford; **$20,782/ha** (high estimate) for wildlife preservation + Ecological services: **$2,421/ ha** riparian habitat for fish; **$14,853/ ha** ground water recharge	**$9,931/ ha** nuisance (odour)	**Stated & Revealed Preference**: contingent valuation survey & market (production function) valuation/ benefit transfer based on Knowler, 2003, assumes 3m of riparian areas are retained on most farmland with 20% stream degradation; ground water recharge market valuation assumes 63% more pervious groundcover of agricultural land relative to urban uses (BCMAL, 2007)
Metro Vancouver, BC	Public Amenity benefits of preserving farmland	**$143,321/ ha/ yr** WTP to preserve farmland		**Stated Preference:** contingent valuation survey as in BCMAL, 2007 with larger population base of metro vancouver (Robbins, 2009)

In summary, the findings of this literature review provide a basis for subsequent research steps undertaken in this thesis. The literature reviewed for the purposes of this thesis covered the economic theory underlying externalities that arise from agriculture, the role of government and possible means for government intervention. The literature emphasizes the importance of valuation as a basis for policy or market-based government intervention to address market externalities.

Chapter 3 – Case Study

This chapter first describes the physical setting and agricultural industry of the region chosen as a case study, then explores the prominent environmental impacts from agriculture focussing on water pollution resulting from surplus nutrients. Lastly, I explore the policy that exists to date to address agricultural nitrate contamination of drinking water.

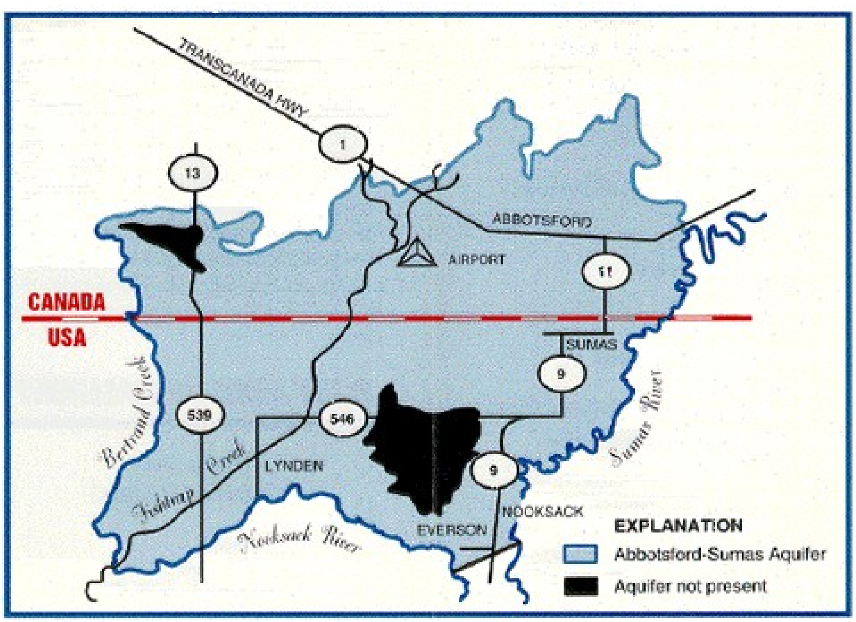
The Lower Fraser Valley of British Columbia, encompassing the Lower Mainland and Metro Vancouver, hosts some of the most highly concentrated agricultural activity in Canada (Schreier et al., 1999). Accompanying intense agriculture are documented impacts on the environment. Where there exists demand for environmental quality there exists a market for ecological goods and services.

The Fraser River headwaters are in the Rocky Mountains near Mount Robson and the eastern border of the province of British Columbia. The river traverses the province eventually travelling south and west before flowing into the Pacific Ocean. From its headwaters the Fraser River canyons are narrow without adjoining lowlands until the District of Hope, after which they open up into a wide river valley stretching from Chilliwack, past Abbotsford and numerous other municipalities through the metropolis region of Vancouver and into the Georgia Straight. The river valley floodplains and adjoining lowlands from Chilliwack to Vancouver make up the Lower Fraser Valley.

The focus of the valuation exercise in this thesis is the Abbotsford-Sumas aquifer. The 200 km^2 aquifer is formed by a layer of underground water-bearing permeable rock that creates ground water reserves. Elevation gradient of the Abbotsford-Sumas aquifer results in an underground southerly flow from the City of Abbotsford on the south side of the Fraser River into Whatcom County, Washington of the USA, as depicted in Figure 1. The Abbotsford-Sumas aquifer provides around 10,000 people in Whatcom and 100,000 people in Abbotsford and Langley with water (Allen, per. comm., 2010). Other aquifer water users include a hatchery, industrial use and agricultural irrigation.

In the sections that follow, there is also reference to the geographical region: the Lower Mainland – Southwest. The Lower Mainland - Southwest refers to Census Agricultural Region (CAR) 590200000, as delimited by Statistics Canada. It includes Census Divisions 9, 15, 29 and 31, the Fraser Valley, Greater Vancouver, Sunshine Coast, and Squamish-Lillooet respectively. The majority of farms and farmland found within these Census Divisions are in the Fraser Valley including Abbotsford and Greater Vancouver, which includes Langley, Surrey, Delta, Richmond, Burnaby, Pitt Meadows, and Maple Ridge.

Figure 2: Abbotsford-Sumas Aquifer, MoE.



3.1 Lower Fraser Valley Agricultural Industry

In 2006, the Lower Mainland-Southwest region hosted 27% of the farms in British Columbia second only to the Thompson-Okanagan region and produced 63% of the gross farm receipts. That is over half of the province's farm receipts on 4% of the agriculturally operated acreage in the province (Stats Can, 2006). The industry practices intensive agriculture and has experienced continuous growth over the past century on a landbase restricted by geography and under pressure of urban development.

Historically, from the 1880's on Lower Fraser Valley (LFV) agriculture was largely in dairy production. Dairy farms operated extensive agriculture on large parcels that produced their own feed: forage (corn) and grass crops. Manure from the dairy herd would be applied to the cropland and the nitrogen would be taken up by the crop with little excess available for nitrate leaching into water bodies.

The past few decades have seen substantial intensification of agriculture in the LFV and a shift away from extensive dairy production. Currently the agricultural industry produces berries, greenhouse vegetables, mushrooms, swine, eggs, poultry and dairy (Statistics Canada, 2006). Over the Abbotsford-Sumas aquifer in particular are raspberry fields and poultry production (Liebscher et al., 1992).

Concentration of raspberry and poultry production in the LFV can be attributed to a number of factors (Sutherland, pers. comm., 2010). Raspberry is a high value crop providing positive returns in an otherwise beleaguered Canadian agricultural industry. Rent streams from pasture land in the Abbotsford region run from $0-$50/acre whereas in the prairie region over the Abbotsford-Sumas aquifer particularly suited for raspberry production rent streams are reportedly $400-$1000/acre (Robbins, pers.comm., 2010). The existence and hydrogeology of the Abbotsford-Sumas aquifer provides advantageous circumstances for raspberries. The moist mild climate makes this region the number one producer of raspberries in Canada. The Abbotsford-Sumas aquifer provides water for irrigation and also well drained gravelly soils essential for avoiding root-borne diseases in raspberry canes (Sutherland, pers. comm., 2010).

Intensive poultry production came to the LFV after a change in the national quota system. Originally, quota was spread out amongst regions throughout the province and the country (Sutherland, pers. comm., 2010). After the regional system was disbanded poultry farmers concentrated in the LFV for economic reasons: improved (less costly) access to a large market, feed imports, chick hatcheries and poultry processors (Sutherland, pers. comm., 2010). The poultry industry tends to be intensive housing birds in production barns without substantial landbase whereas the dwindling dairy industry has maintained its landbase and remains extensive in terms of maintaining pasturelands and silage crops. As a result, dairy farms have land on which to apply their manure whereas in most cases, poultry farms do not. Dairy manure, with a higher moisture content is not as transportable as poultry manure. There have been numerous studies investigating marketing, processing and alternative disposal options for LFV poultry manure (Timmenga, 2003; Arthur, 2009).

3.2 Agri-Environmental Issues

In the literature review environmental effects of agriculture are catalogued as:

1. Loss of species and habitat diversity, removal and disturbance of non-agriculturally used margins within agro-ecosystems, e.g. hedgerows, ditches, grassy margins, etc.
2. Pollution of ground water with nitrates, pesticides and their metabolites leached from farmed land.
3. Deterioration of soils and soil function through compaction, erosion and accumulation of pollutants.

4. Pollution of surface water with nitrate, phosphorus and pesticides due to farming procedures.
5. Pollution of the atmosphere through the gaseous emissions from farming procedures.

(Gerowitt et al., 2003)

Each of these environmental effects, to varying degrees have been documented in the Lower Fraser Valley (LFV). What follows is an inventory of documented environmental impacts from agriculture in the LFV. Note that this review is not a comprehensive listing of all environmental impacts from agriculture in the LFV but rather a selection of studied issues that may lend themselves to valuation.

Loss of species and habitat diversity. The waterways of the Fraser Valley have historically provided important breeding grounds for several salmon species which are the resource base of a fishing industry and an integral component of the local ecology (Knowler et al., 2003). There are numerable pressures on salmon stocks the culmination of which are causing declines in salmon populations. Deterioration of salmon habitat by agricultural practices is one such pressure (Hall and Schreier, 1996) as is urbanization. Agriculture has two major impacts on water quality: nutrient loading and pesticide residues.

Nutrient loading can cause the eutrophication of waterways, reduce available oxygen and thereby compromise salmon habitat, impacts that have been documented in the LFV (Hall and Schreier, 1996, Schendel et al., 2004). With regards to pesticides a 2005 report by Environment Canada uses salmonids as an indicator species of environmental quality and finds that acute historical pesticide residues are found at levels detrimental to salmon populations in the LFV (Wan et al., 2005). A 2009 multi-pesticide residue study established the presence of pesticides common to agriculture today in

LFV waters however the pesticides have not yet been linked to salmon habitat or impacts on salmon populations (Woudneh, 2009). A study in 2003 tackles the valuation task of land use decisions that improve fish habitat (by land conservation) in terms of its benefits to the fishing sector. They find a value of C$1322 to C$7010 per km of salmon stream length (Knowler, 2003).

Deterioration of soils. Soil quality is monitored through varied indicators. At one site in the LFV compaction was shown to occur over a five year period (Kenney et al., 2003). Persistent historical pesticide contamination of soil is reported at several sites (Wan et al., 2005). pH levels of soil are altered by excess nutrients overapplied through fertilizers and manure leading to soil acidity. Acidity, already a problem in LFV soils due to heavy rainfall, makes soil inhospitable to certain crops unless lime is applied (BC Ag Soil pH infosheet, 2001). Overall soil loss is also occurring as agricultural lands are lost to urban encroachment in the region (Campbell, 2006).

Gaseous emissions and air pollution. According to Environment Canada, agriculture is destined to be the second biggest contributor of air pollution in the LFV by 2020, after marine vessels (Levelton and Golder, 2004). While light-duty motor vehicle emissions are on the decline as a result of increasing controls, emissions from agriculture are relatively unchecked and are on the rise. Of major concern are fine particulate matter and ozone, two precursors to smog. Both can be formed from primary emission of nitrogen oxides and ammonia. A 2004 study recommends a number of voluntary beneficial management practices to reduce air pollution from agriculture, several of which overlap with beneficial management practices that improve water quality: management of riparian areas and field margins, changing livestock feed rations, improved application of manure and nutrient management (Levelton and Golder, 2004). Emissions are increasingly being studied by government and industry agencies, particularly in light of climate change and the monitoring of greenhouse gases. A national emissions inventory is forthcoming from Environment Canada, partly based on numbers gathered by Agriculture and Agri-Food Canada.

Odour is another air pollutant in the LFV. Complaints are continuously lodged with the Fraser Valley Regional District and Ministry of Environment. One study assesses the public cost of odour in Abbotsford at $4019/ acre (Olewiler, 2007).

Ground and surface water pollution. Water pollution is the most documented issue resulting from agriculture in the Lower Fraser Valley to date. To quote the report "The Value of Natural Capital in Settled Areas in Canada," 2004:

> Runoff of agricultural waste products is recognized as the prime contributor to the degradation of ground and surface waters in the Fraser Valley [Lavkulich et al, 1999]. Wetlands and riparian areas adjacent to streams, because of the role they play in filtering these effluents, become increasingly important in maintaining a healthy aquatic ecosystem. The excess manure, in conjunction with the application of chemical fertilizers, has resulted in excessive nutrient loads on substantial areas of farmland in the study area, leading to nutrient-laden runoff into water systems and unbalanced soil conditions. An average of 273 kg of nitrogen, 84 kg of phosphorus and 165 kg of potassium were applied per hectare per year on farms in the Fraser Valley, well above and sometimes twice what was needed for the most nutrient-intensive crops [Schreier et al, 1999]. Of the total nitrogen, 17 to 48 percent found its way into drainage ditches and thus surface water systems [Nagpal et al, 1990]. Excess nutrients that are not absorbed into the water system interact with soil to slow organic decomposition and destabilize nutrient and pH levels, leading to acidity problems. Acidity problems in the study area are affecting more of the Valley's land over time. Nitrate levels in water, while generally below the limit considered unsafe in streams draining agricultural lands in the region, are becoming an increasing concern in the ground water of the Fraser Valley. For example, nitrate levels in half of all well samples of the Abbotsford aquifer were higher than the safe limit. The ground water in both Abbotsford and Langley also contains inorganic chemicals (from herbicides, pesticides and fumigants) in concentrations sufficient to cause concern. "The principle causes of this pollution are inadequate agricultural practices and badly constructed or maintained septic tanks [Elliot et al, 1999]."

3.3 Nitrate Contamination of Ground Water

Nonpoint source water contamination in the Lower Fraser Valley (LFV) dates back to the 1970s when agricultural land use became more intense. Studies show nitrate pollution of surface and ground water particularly in the Abbotsford-Sumas aquifer (Schreier, 1999, Liebscher et al., 1992, Wassenaar, 1995). The Canadian Drinking Water Quality guideline recommends nitrate concentrations be less than 10 mg/L. Higher concentrations can result in the potentially fatal "blue baby syndrome" and increase the risk of cancer (EC, 2004). A surplus of nitrates can also cause eutrophication of creeks, rivers and lakes with an impact on fish populations. The Abbotsford-Sumas aquifer serves the drinking water demand of approximately 10,000 people in Whatcom County and 100,000 in Abbotsford.

The Abbotsford-Sumas aquifer is formed by a layer of underground water-bearing permeable rock that creates ground water reserves. The ground water table sits one to ten metres below the surface to a depth of 15 to 25 metres underground and spans 200 km^2 (Mitchell, 1998). The northern section in Abbotsford, BC sits at a higher elevation than the southern half in Whatcom County, Washington resulting in an underground southerly flow. However, there are likely local variations in the movement of water and complex ground water paths due to significant heterogeneity of hydrostratigraphic units in the aquifer (Scibek, 2005).

Annual rainwater recharges the aquifer at varying rates across the aquifer minus losses to evapotranspiration. The water table is at its highest February to April and at its lowest September to November (Scibek, 2005). Outflows include baseflow to streams, private and municipal wells. Agricultural irrigation and water extraction by a fish hatchery represent minimal ground water outflow since the water is promptly recycled back into the aquifer.

Because the aquifer is surficial and unconfined the ground water is susceptible to contamination. Both Canadian and US regions over the ground water are agricultural, in Abbotsford mostly with broiler (poultry meat) and raspberry production, in the US with raspberries and dairy. Excess nitrates applied to agricultural land in Canada will flow southwards into the US portion of the aquifer. Nitrate contamination of the aquifer is a political issue with international dimension, subject of the Abbotsford-Sumas Aquifer International Task Force to resolve transboundary water quality issues.

There is a definitive link between agriculture in the region and nitrate contamination of the ground water. Liebscher et al. (1992) examined hydrological data from the Abbotsford aquifer since 1955 and found that "the infiltration of nitrates and pesticides from the land surface into the aquifer has caused local deterioration of the quality of water in the aquifer" moreso than from septic systems and natural fluctuations. Wassenaar (1995) used isotopes to determine the material source of nitrate contaminants found in the aquifer and found 54% of the wells sampled had nitrate levels above drinking water limits (10 mg/L). The material source of the nitrates was poultry manure in majority and synthetic fertilizer second (Wassenaar, 1995).

It is important to note that these findings do not mean poultry producers are directly or solely responsible for nitrate contamination of the aquifer. How poultry manure is stored, where it is transported to and how it is used determines the impact of poultry manure on the ground water. With the studies cited, the current state of water monitoring and with the existing policy framework we are 1). unable to determine exact source location and 2). unable to determine the age of the nitrates presently in the aquifer (or when they were deposited) (Sutherland, pers. comm., 2010).

There are also limitations in knowledge regarding the behaviour of nitrates once deposited into the aquifer: 1). how (aside from a southerly flow) the nitrates move laterally and horizontally in the aquifer, 2). the nature of seasonal flows and plumes, and 3). the natural process of denitrification and breakdown of nitrates in the aquifer (Bittman, pers. comm. 2009, Sutherland, pers. comm. 2010).

Limitations of the existing policy framework and government initiatives are discussed in the policy section that follows.

3.4 Policy Framework

In this section, the regulatory framework for water pollution and nutrient management in the Lower Fraser Valley (LFV) is outlined. This, as well as an exploration into current nutrient management policies and programs provides a context in which new market-based mechanisms in the Lower Fraser Valley need to fit. It is also interesting to note, of the policy tools reviewed in Table 3 of the literature review, which ones are currently employed in British Columbia (BC).

The *Environmental Farm Plan Reference Guide* (Part 6) comprehensively outlines the Provincial and Federal regulations pertinent to nutrient management. There are numerous Acts under which nutrient (fertilizer or manure) application and its potential effect on water resources fall: the Provincial Drinking Water Protection Act, Health Act, Environmental Management Act, Wildlife Act, as well as the Federal Fisheries and Species at Risk Act.

The Drinking Water Protection Act prohibits introducing, causing or allowing anything that will result or is likely to result in a drinking water health hazard to a domestic water system. The Environmental Management Act (EMA) deals with agricultural waste products explicitly (manure, compost, vegetation) and states that these materials must be collected, stored, handled, used and disposed of in a manner that prevents pollution. Industry can purchase permits to be in compliance with the Act: $41.13/ tonne of nitrogen and nitrates, $102.91/ tonne of phosphorous or phosphates. However, the Agricultural Waste Control Regulation (of the EMA) exempts agricultural operators from needing a permit as long as they act in accordance with the Code.

The provincial Code of Agricultural Waste Control stipulates conditions on manure storage, timing of application and application in accordance with crop nutrient needs. In practice, manure spreading permits are not required (Bittman, pers. comm., 2009). The provincial Ministry of Environment issues regular reports on industry compliance. A recent compliance assessment over the drinking water aquifers of Abbotsford-Sumas and Hopington was performed between the period of October 2003 and February 2004 and followed up between February and March 2005. 105 hobby and commerical farms were visited during the initial assessment and the follow-up entailed a fly-over to assess early Spring manure application practices. Water quality was not directly measured on the farmsites. Farmers visited during the compliance assessment were notified of their infractions and informed that they were not in compliance.

In general, a softer approach than law enforcement is taken to encourage compliance amongst agricultural producers. Government policies and programs aimed at reducing environmental impact of agriculture come in the form of educational campaigns, volunteer and incentive programs, industry-government task forces and government research. In the case of the Abbotsford-Sumas aquifer application of manure and fertilizers in raspberry production and manure production by landless broiler production were identified as priority issues by the British Columbia Ministry of Agriculture and Lands (BCMAL). Poultry and raspberry producers were targeted through an educational campaign and given incentives to practice effective manure storage (Bertrand, 2001).

The Environmental Farm Plan Program (EFP), a joint provincial-federal government initiative launched in 2003 promotes Beneficial Management Practices (BMP). To improve manure management for example subsidies are provided for building winter manure storage facilities and emission reduced spreading equipment. A cross-compliance requirement is that a nutrient management plan for the farm be completed before accessing funds. The 2008-09 EFP program statistics reveal that across BC over 700 producers participated in the program covering 256,687 acres. The Fraser Valley was one of the most prominent regions for BMP projects and product and waste management one of the most popular project categories. Over $3,500,000 were paid to producers. As yet, environmental outcomes of this program have not been measured. In this case it appears the uptake of BMP's is being used as a proxy by government agencies to demonstrate good environmental practice rather than science-based environmental assessment. This is likely the case because soil and water quality testing is deemed too expensive.

An outcome of the industry-government task force, the Sustainable Poultry Farming Group, and government funded research is the report “Evaluation of Options for Fraser Valley Poultry Manure Utilization” (Timmenga, 2003). The authors conclude that excess manure produced by landless operations, namely the poultry and swine industries need to process their manure and export it. True nutrient flow accounting would indicate that nutrients imported from the prairies to the LFV to feed chickens and hogs needs to be returned to the prairies. There the manure would fertilize grain crops on which high density chickens and hogs are sustained. Following this report it has been within the provincial government's mandate to market chicken manure outside of the LFV. There has been some uptake of this program however it is not known how much and is perceived as not enough (Timmenga, pers. comm., 2009). True uptake of the program is difficult to measure as producers are not required to report what they do with their manure (Bittman, pers. comm., 2009; Sutherland, pers. comm., 2010). A recent government initiative offered truckers to file information on where and how much LFV poultry manure is exported. This program was unsuccessful in attaining the data the ministry wanted (Cavendish, pers. comm., 2009).

Educational efforts have also focussed specifically on nutrient application practices of raspberry producers however information the ministry can offer is limited (Sutherland, pers. comm., 2010). Localized research efforts that would help farmers make environmentally sound decisions regarding fertilization of crops atop a ground water aquifer has not kept pace with the growth of the raspberry industry in the LFV. Raspberries are a low nitrogen demand crop surrounded by a readily available and undervalued high nitrogen chicken manure. There are economic benefits to supplying enough nitrogen to a high value crop and no enforced penalties in applying too much. Abundant rainfall flushes the excess nutrients into nearby waterways and the underground aquifer.

Chapter 4 - Economic Evaluation of an Externality

Based on a comprehensive literature review (Chapters 2 & 3), an assessment of present knowledge and data availability the motivation and conditions for the economic evaluation of an externality emerged. This chapter explains the choice of valuation approach used in Chapter 5: why it is appropriate to use substitute cost methodology to estimate total willingness to pay (WTP) for services from the Abbotsford-Sumas aquifer. The link between total WTP for services from the aquifer and WTP for nutrient management practices is established. Weaknesses of the substitute cost method are discussed and I explain how my valuation approach overcomes these weaknesses.

In the second section of this chapter I put forth an economic model for optimal expansion of a public utility. This model helps to explain present value (PV) cost analysis which is at the core of the substitute cost methodology used in Chapter 5, where the total WTP for aquifer water supply services is evaluated. In the third section I pay special attention, through sensitivity analysis, to three PV cost analysis parameters: growth rate, discount rate and economies of scale.

4.1 Substitute Cost of the Abbotsford-Sumas Aquifer

First, I explain why I use a substitute cost approach in this thesis. The reasons are because 1). the substitute cost method provides foundation for the development of a payments for ecological services (PES) program and 2). the conditions for a substitute cost approach exist and a usable (though not complete) set of data is available.

The purpose of the research component of this thesis is to estimate the value of an externality that could provide basis for the trade of an ecological service deliverable through agriculture. To revisit findings of the literature review payments for ecological services (PES) are:

(a) a voluntary transaction where
(b) a well-defined environmental service (or a land use likely to secure that service)
(c) is being 'bought' by a (minimum one) service buyer
(d) from a (minimum one) service provider
(e) if and only if the service provider secures service provision (conditionality)
(Wunder, as cited by Engel et al., 2008).

I do not, in this thesis, develop a PES program for nutrient managment in the Lower Fraser Valley. I do some preliminary work that may or may not justify further investigation into a PES program that would incentivize improved nutrient management above the Abbotsford-Sumas aquifer. By using a substitute cost approach I lay the foundation for development of a PES program that meets the above characteristics (a) through (e).

To potentially justify the development of a PES program I estimate (as best as possible) the social benefits (WTP) of an environmental service (safe water supply) delivered by the aquifer using a substitute cost approach. To accomplish this I must make some assumptions regarding the state of nitrates in the aquifer, how they limit well implementation and the possible need for remediation (these assumptions are explained in the latter part of this section). Due to a deficit of information on the nitrate compromised ability of the aquifer to supply drinking water services (b) in the list above is not sufficiently well-defined. To be well defined, and to improve the accuracy of the valuation in this thesis, the state of nitrates in the aquifer, how they limit well implementation and the need for remediation (if any) to be able to further exploit the aquifer for drinking water purposes need to be assessed.

Also pertaining to (b) in the list above land uses or practices (linked with effective incentives) to minimize nitrate leaching in susceptible areas of the aquifer are not sufficiently researched. Government led educational initiatives to improve nitrate management over the aquifer have not reduced nitrate levels in the aquifer (to date). Identified effective land uses or practices coupled with meaningful incentives and the ability to enforce (or (e) secure service provision) would help define the land uses likely to secure the environmental service of safe water supply. Research and public and farmer engagement are also needed to pin point (d) potential service provider(s) (farmers and other land users above the aquifer). To develop a PES program multi-disciplinary research that further defines (b), (d) and a mechanism for (e) in the list above would be carried out in the fields of economics, agricultural science, hydrogeology and public policy.

Towards PES program design and through the use of the substitute cost approach I identify in this thesis (c) a potential service buyer: the Abbotsford / Mission Water & Sewer Services (AMWSC). Establishing the potential for a PES program with the AMWSC as the service buyer is a major departure from current policy approaches to the agricultural nitrate contamination issue (see section 3.4). Past studies (Hauser and van Kooten, 1993) arguably identified taxpayers as potential service buyers using a contingent valuation (survey) approach. This thesis is the first study to implicate a municipal level public utility that actively embodies these taxpayers in the agricultural nitrate management issue. This is one major reason that I use the substitute cost approach in this thesis.

The rationale for a PES program and the extent to which the findings of this thesis can be used towards that purpose are discussed in Chapter 6. This concludes the first part of the reasoning behind the substitute cost approach used in this thesis. I continue by explaining the particular agri-ecological and economic conditions around the Abbotsford-Sumas aquifer which make the substitute cost approach possible (given a number of assumptions) and appropriate.

In the next chapter, Chapter 5, I use the subtitute cost method to estimate WTP for water supply services of the Abbotsford-Sumas aquifer. As described in section 2.3 I evaluate the City of Abbotsford's *total WTP* for water supply services of the aquifer based on the cost of a substitute. The substitute technology that will provide safe drinking water is a surface water supply system that is scheduled for development in the near term. The city has considered a number of future water supply options and intends to go forward with the development of Stave Lake.[2] This thesis contends, with the discussion that follows, that the surface water supply system is substitute technology for a natural groundwater aquifer that lies beneath the city. Because the surface water supply system is a substitute WTP for its implementation can be used to estimate the value of ecological services deliverable through the aquifer (safe drinking water).

As revealed in the analysis performed in Chapter 5, the proposed surface water supply system is significantly more costly than it would be to install more city wells to extract more ground water. The proposed surface water supply is Stave Lake. Stave Lake is north of the city and requires a 23.9 km pipeline to transport water from the lake to the city. The Abbotsford-Sumas aquifer lies directly beneath the city where transmission pipes already distribute water from several sources including fifteen (nineteen as of 2010) existing wells. Further costs result from health regulations that stipulate

[2] City council may have to go to referendum to gain support for financing the new surface water system.

surface water be treated in a water treatment plant before being distributed. Ground water, naturally filtered through the gravelly aquifer, requires less treatment.

Why then is the aquifer being avoided as a significant drinking water resource? My review of city documents, engineering reports and interviews with city staff, government officials and university experts reveals several stated reasons why the aquifer is being avoided as a significant source of future water supply. Reasons cited for avoiding the aquifer, further discussed below are: long-term sustainable groundwater yields, impact on baseflows of creeks, well water pumping costs, system redundancy and agricultural nitrate contamination. However, after careful investigation of these issues only nitrate contamination of the ground water stands as a significant constraint on use of the aquifer as a source of safe drinking water into the future. I proceed by investigating the arguments listed above. By wittling them down to the one argument that stands: agricultural nitrate contamination of the aquifer, the conditions for a substitute cost approach are established. If the City of Abbotsford is building a surface water supply system to avoid nitrates the city's WTP for the new system can be used to estimate *WTP for improved nitrate management practices* over the aquifer.

According to the 2006 Water Master Plan, a study commissioned by the City of Abbotsford, there is a large portion of the aquifer that is as yet untapped but is unsuitable for ground water extraction. The reason cited is nitrate contamination resulting from historical agricultural and waste management practices (Dayton & Knight, 2006). The 2010 plan states that city staff are concerned nitrate levels will increase (AECOM, 2010). Given the link between agriculture and nitrates in the aquifer (see section 3.3) the potential for nitrate levels to increase is real and will continue with projected growth in the livestock industry. Nutrient balances for the Lower Mainland estimate 4,447 tonnes of nitrogen above crop needs were applied to the soil in 2010 (Timmenga, 2003; update to Timmenga, 2003 in the Appendix). Three of the fifteen city wells have already exceeded the 10 mg/L recommended drinking water limit of nitrates on more than one occasion.

Other reasons for avoiding the aquifer as a future water source are stated in city documents, however they do not hold up to scrutiny. Water levels of the aquifer and sustainability of well water yields complicated by the future impact of climate change are cited as a concern in the 2010 Water Master Plan. AMWSC staff have doubts about the long-term sustainability of current yields from the Abbotsford-Sumas aquifer (AECOM, 2010). However, there is no evidence of a downward trend in the aquifer water table that would indicate overpumping of the aquifer (Piteau, 2006). A mass balance of the aquifer's flow volume indicates that total inflow remains above outflow over the course of the year, particularly in the winter (Scibek and Allen, 2005).

Experts affiliated with provincial and federal government authorities and universities, when asked, question the direction the city of Abbotsford is taking with its water system and cite the abundancy of the aquifer beneath the city (Graham, pers. comm., 2010; Allan, pers. comm., 2010). In a study commisioned by the city of Abbotsford, the aquifer is reported to be highly productive, recharged annually with 625 mm (19.8 L/s/km^2) of rainwater (Piteau, 2006). With a surface area of 200 km^2 annual recharge is calculated to be 342 million litres a day (MLD). In 2006, approximately 43 MLD (12.5%) was withdrawn by wells (Dayton & Knight, 2006).

Recharge can be thought of as the daily deposit of water into a below ground natural holding tank - the aquifer. The water table, or level of water within the “holding tank” rises and falls about 2.5

metres with seasonal variation in rainwater deposits, peaking in March and at its lowest in late October (Piteau, 2006). Fishtrap creek, west of the Abbotsford airport and running south across the aquifer recharges the aquifer for six months when the water table is low but receives baseflow from the aquifer for six months when the water table is high. Withdrawals occur in the form of well water users and baseflow to creeks. If withdrawals are greater than recharge aquifer "mining" may occur.

To truly understand the capacity of the aquifer to meet water demands a comprehensive hydrogeological study that determines sustainable withdrawal rates throughout the aquifer, throughout the year is needed. The study would likely consider aquifer storage capacity, seasonality of recharge and water demand (inclusive of demand in Washington, USA) and optimal siting of wells so that baseflow to creeks are minimally affected. It is regular practice to commission site specific well feasibility studies where pumping tests determine the potential yield in a particular site. Recent studies supported the 2009/2010 development of the Bevan wells well field. What is lacking is an overall assessment of how many more wells the aquifer could support.

For the numerical evaluation performed in this thesis, I use the 342 MLD average daily recharge rate minus current extraction of 43 MLD as a sustainable expansion guideline (299 MLD). In the valuation that follows it is assumed that the city of Abbotsford could expand their withdrawal capacity to 295 MLD (45 MLD present day capacity + 250 MLD added capacity; see section 5.1). It is important to note that the valuation is based on building capacity to serve peak day demand. Withdrawal would only meet capacity a few days in the summer. By no means have I done a comprehensive hydrogeological assessment of the aquifer to determine the viability of more groundwater wells – no one has. What I contend, on economic grounds alone, is that the aquifer, a valuable underground water reservoir warrants a closer look as a more significant drinking water source.

Climate change modelling of the Abbotsford-Sumas aquifer predicts that from 2010-2039 recharge will decrease 5.6 to 6.3% resulting in a .05 metre to .25 metre fall in the water table (Scibek and Allen, 2005). The higher range of the reduction will occur in the upland regions of the aquifer, whereas lowland regions will see little change in water levels as a result of climate change. From 2040-2069 a 12.7 to 14.6% reduction in recharge is predicted resulting in a .25 metre fall in the water table in the uplands, but again with little change in the lowlands where more wells could potentially be sited (Scibek and Allen, 2005). It is common understanding that surface water features such as lakes will be more significantly affected by climate change than ground water reserves (Allen, pers. comm., 2010). A climate change assessment of Stave Lake, the surface water system being developed by the AMSWC to meet future water demand has not been conducted to my knowledge.

Another cited concern for future use of the aquifer is impact on baseflow of creeks which triggers the need for an environmental assessment. As mentioned above, a comprehensive hydrogeological study of sustainable aquifer withdrawal rates would need to take into consideration the baseflow of creeks. Fishtrap creek peaks a specific concern since it is a transboundary water feature. Potential for impact on baseflow has international political implications. Additionally, wells cited too close to the creek would trigger the need for an environmental assessment, which represents additional costs. This may be acting as a further disincentive to develop well sites south of Marshall Road close to the Abbotsford airport.

Another obstacle to using the aquifer as a water supply source is the possibility of public utility, city staff and engineering firm bias. Public utilities, city staff and engineering firms operating in the Lower Mainland of British Columbia are accustomed to and adept at implementing surface water reservoir systems and therefore may have a bias towards these systems. Pumping costs and the rising cost of fuel present another cited argument against the use of ground water. However, the new Stave Lake water system will also incur energy costs associated with a new pump station. Fuel costs of the respective systems are accounted for in the present value cost analysis that follows.

Finally, redundancy is desireable in a water system to provide back-up when failures occur. This provides rationale for the development of both ground water and surface water system capacity. Currently, AMSWC infrastructure capacity is 45 MLD from fifteen wells and 98 MLD from two surface water sources. The 2010 Water Master Plan advocates 25 MLD added well water capacity from four wells, and 250 MLD added surface water capacity from one new surface water system. A mixture of water sources are also desireable for the respective attributes of different kinds of water.

The above summarizes “quantity” and well infrastructure arguments against increased dependency on the Abbotsford-Sumas aquifer for Abbotsford water needs. However, the quantity of water in the aquifer can meet growing water demand in the near term, for a few days in the summer, up to and within the vicinity of the recharge rate of 342 MLD. Other arguments listed above against the development of wells do not warrant avoidance of the aquifer for future water supply. Only agricultural nitrate contamination of the ground water is a significant constraint on further development of the aquifer for drinking water supply. Which brings us back to the purpose of this thesis: agricultural nutrient management employing the concept of ecological goods and services (EGS).

Because agricultural nitrate contamination constrains aquifer use total WTP for water supply services infers a total WTP for improved nutrient management practices. I estimate total WTP for ecological water supply services of a natural aquifer using a substitute cost approach. This estimate could provide justification for a trading program with other aquifer service users, in this case farmers, if the cost of aquifer services including averted damage from agricultural activity are found to fall below the city's total WTP for drinking water. I conclude this section with a discussion about potential weaknesses of the substitute cost method and how these are minimized in my approach.

The foremost complaint about the substitute cost method is how it measures WTP for an ecological service based on the total cost of substitution, restoration or replacement cost of that service. The socially optimum level of service, or marginal WTP, is not estimated. Therefore the substitute cost method is predisposed to inflated estimates of WTP. In this thesis I am careful to specify the nature of the value I estimate, in this case *total WTP* and differentiate it from the still elusive exchange value. Because the substitution cost method does not establish an exchange value and subsequently an explicit price for an ecological service does not mean the results are not important. They are a step towards establishing market value.

Furthermore in the case of Abbotsford the cost of a substitute technology, a surface water system, represents planned public expenditure. This means there is actual market demand for complete substitution of potential aquifer water supply services. In other cases where the substitution cost method is employed the substitution technology is hypothetical (in Olewiler, 2004 for example). An estimate of *total WTP* for aquifer water supply services based on the cost of substitute technology that is about to be installed is sound since it is based on observation of actual market activity.

Finally, another potential criticism of the substitute cost approach taken in this thesis is that the surface water system represents a high cost substitute technology and because of this, *total WTP* estimates are inflated. Again, I contend that because the high cost substitute surface water system represents revealed WTP from actual market observations this criticism does not apply. It is true that there may be lower cost options to the pending water supply shortfall. For example, water demand management. The city currently employs water demand management mechanisms such as lawn sprinkling bans in the summer and incentives to buy water efficient household appliances. The 2010 Water Master Plan includes water demand management until 2015 when the Stave Lake surface water system will be complete. However, the cost of water demand management programming from 2010 to 2015 is not included in the PV cost analysis of the plan. Nor is water demand management considered for the long term. If the AMSWC chose demand management, the costs of a water demand management program could instead be used to measure the value of improved nutrient management practices, according to revealed preference substitute cost methods.

Further to this point, nutrient management on farmland above the Abbotsford-Sumas aquifer may be revealed in time as a lower cost option to building a new surface water system. It is towards this potential finding that I have undertaken the research in this thesis.

4.2 Public Utility Expansion Model and Present Value Cost Analysis

The City of Abbotsford's total willingness to pay (WTP) for water supply services of an aquifer are estimated in the next chapter based on the cost of substitution technology: a surface water supply system. However, estimating *total WTP* for water supply services falls short of finding an explicit price for water supply services. Only once the costs of aquifer water supply services are known can an explicit price, or exchange value, be estimated. This thesis estimates *total WTP* for water supply services and does not attempt to determine the total cost of aquifer water supply services. However, based on available data the infrastructure related costs of aquifer water supply services (how much it costs to develop wells) are calculated. The result is an estimate of potential social benefits of aquifer water supply services net the cost of building wells, but not net the cost of nitrate management. This section provides an analytical framework for the method used to estimate *total WTP* for water supply services and the cost of building wells. Present value cost analysis is used to estimate both.

Present value (PV) cost analysis is a typical method used to compare projects with different costs spread over varying time horizons in the present. The PV cost of a surface water supply system, or wells, depends on the size of the initial investment and how soon the infrastructure is upgraded. Furthermore, other parameters used in PV cost analysis such as growth and discount rates have their impact on the PV of costs. The public utility expansion model described in this chapter describes the cost minimization problem of the public utility and how PV cost analysis parameters affect the results. The model is not used to determine an optimal capacity development schedule for Abbotsford's public utility since optimal infrastructure is not the purpose of this thesis. Rather the model's powers of parameter analysis are used to illustrate the importance of parameter values used within PV cost analysis as this has implications for the results of the valuation performed in the next chapter. First, the drinking water industry and nature of the public utility is described.

Industries that require transmission networks, such as water and electricity usually result in natural monopolies. Large capital requirements pose a barrier to entry, or if competitors successfully enter inefficiencies and high costs may arise as infrastructure is doubled and utilized under capacity. Often in these cases the sector comes under government management while competition is legally prohibited. A public utility acts on behalf of taxpayers under the direction of a government authority. Dependent on a combination of public funds and user fees the objective of the public utility is to minimize costs. As long as population and water demand grows existing infrastructure will continually need to be expanded. The following model describes the public utility's cost minimization for infrastructure expansion.[3] The model is described, illustrated by a number of graphs and the parameters of the model explored individually.

Population at date 0 is N_0, initial public utility capacity is K_0 and population growth rate is g. The public utility will install more capacity at date 1 when population meets initial capacity of the system:

$$N_0 e^{g T_1} = K_0 \qquad (1).$$

We isolate T_1 to find T_1^*, the time when population catches up to initial capacity and public infrastructure must be expanded to meet demand:

$$T_1^* = \frac{1}{g} \ln\left(\frac{K_0}{N_0}\right) .$$

By substituting T_1^* into (1) we find the population at time of replacement is:

$$N_1 = N_0 e^{g T_1^*} = N_0 e^{\ln \frac{K_0}{N_0}} .$$

Capacity K_1 is installed at time T_1 and more capacity is installed at time T_2 when demand catches

[3] The public utility model presented in this thesis was collaboratively developed by Dr. James Vercammen and myself, Karen Ageson, M.Sc. student of Agricultural Economics, Faculty of Land and Food Systems, UBC.

up to capacity of the system again:

$$N_1 e^{g T_2} = K_1 .$$

As before, we isolate T_2 to find T_2^*, the time when population catches up to capacity for a second time. Substitute $N_1 = K_0$ where demand at date 1 is equal to initial capacity:

$$T_2^* = \frac{1}{g} \ln\left(\frac{K_1}{K_0}\right) .$$

We repeat the above steps to show that in general:

$$T_i^* = \frac{1}{g} \ln\left(\frac{K_{i-1}}{K_{i-2}}\right) .$$

Now, we assume that the cost of expanding existing infrastructure is a function of the ratio $\frac{K_i}{K_{i-1}}$. In other words the system is expanded by a certain percentage each time and the cost increases in equal proportion. This assumption implies that the model has a steady-state solution and that expansion will occur at equal time intervals . If we solve the problem for the first time interval we will know all future time intervals.

Let $k_i = \frac{K_i}{K_{i-1}}$. In steady state the ratio k_i is the same each time expansion takes place. Let $C(k)$ be the cost of each expansion and $\beta = \frac{1}{1+p}$ be the discount factor for each period. Immediately after the first expansion the present value cost of the next expansion is $\beta^T C(k)$. We know $T^*(k)=\frac{1}{g}\ln(k)$ so we can rewrite present value cost as $\beta^{\frac{1}{g}\ln(k)}C(k)$. As a result the present value cost of the current and all future expansions is:

$$PV=\sum_{i=0}^{\infty}\beta^{\frac{\ln(k)}{g}i}C(k).$$

Noting that $\beta=\frac{1}{1+\rho}$ we can rewrite the expression as:

$$PV=\sum_{i=0}^{\infty}\left(\frac{1}{(1+\rho)^{\frac{\ln(k)}{g}}}\right)^{i}C(k)$$

Where $Z(k)=(1+\rho)^{\frac{\ln(k)}{g}}-1$ we can rewrite the expression as:

$$PV=\sum_{i=0}^{\infty}\left(\frac{1}{1+Z(k)}\right)^{i}C(k)$$

This expression is expanded, solved and can be written as:

$$PV=\frac{C(k)}{Z(k)}$$

Next we assume $C(k)=\alpha k^{\Theta}$ where α is variable cost and Θ is economies of scale. The goal of the public utility is to choose k to minimize PV. We take the derivative with respect to k and set it equal to zero to fulfill the first order condition of optimization:

$$\frac{dPV}{dk}=\frac{Z(k)\Theta\alpha k^{\Theta-1}-\alpha k^{\Theta}Z'(k)}{Z(k)^2}=0$$

We solve the first order condition to find $k*$. Appendix 2 shows how to arrive at this solution:

$$k*=\left(\frac{g\Theta}{g\Theta-\ln(1+\rho)}\right)^{\frac{g}{\ln(1+\rho)}}.$$

$k*$ is the optimal proportional increase (and $(k*-1)\times 100$ is the optimal percentage increase) in capacity each time population reaches capacity and must be expanded to meet demand. Coupled with

$T_i^* = \frac{1}{g}\ln(k^*)$, the optimal duration between infrastructure expansion, we have a model for the optimal expansion of a public utility. This public utility expansion model is used to explore PV cost parameters and the impact they have on PV costs.

Model parameters are best introduced by an iteration of the model. Table 5 presents values given to the parameters in a baseline iteration. In the next chapter the behaviour of the parameters are further described through sensitivity analysis. Economies of scale, the discount rate and growth rate are parameters of the model. For the baseline iteration I set economies of scale to two, the reasoning for this explained in the next section. Economies of scale if above one indicate marginal costs increase as production is scaled up or diseconomies of scale. The discount rate is set at Canada's Treasury Board recommended real rate of 8% (Treasury Board, 2007). I use a growth rate of 7% which was the Abbotsford population growth rate between 2001 and 2006.

Table 5: Public Utility Expansion Model Parameters, Baseline Iteration

Parameters	(1)
Economies of scale *(Θ)*	2
Discount rate *(ϸ)*	0.08
Growth rate *(g)*	0.07

The above parameter values are run through the model and the model is solved for k^* and T^*. The baseline iteration (1) is optimal at $k^* = 2.07$ proportional (or $(k^*-1)\times 100 = 107\%$) increase of public utility capacity each $T^* = 10.37$ years between periods of capacity development.

4.3 Parameter Sensitivity Analysis

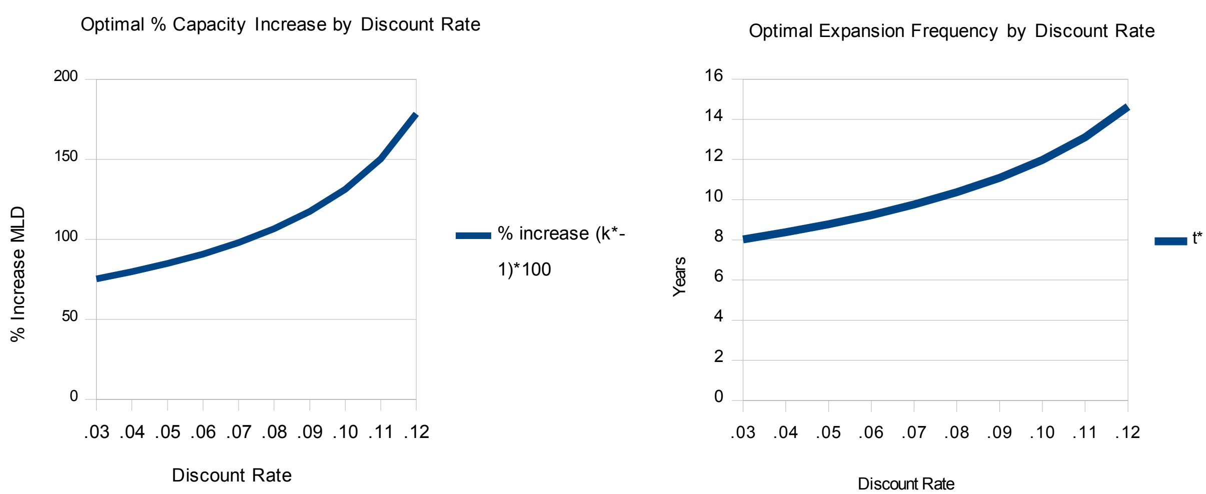
Variations in model parameters illustrate how assumptions regarding those parameters affect the optimal outcome. Comparative statics show us how different discount rates change the optimal proportional increase in capacity and optimal frequency of expansion. Comparative statics of the discount rate, the population (demand) growth rate and a cost curve parameter, economies of scale, follow.

Discount rates are used to render future values comparable to present values, to be able to assess a stream of payments (costs or benefits) over time in the present. The discount rate reflects the opportunity costs of money, that investments can earn a positive rate of return, and time preference, that humans prefer instant gratification to delayed gratification and are uncertain about future events.

In private business ventures the discount rate is used to reflect the opportunity cost of capital (the foregone stream of revenue in alternative projects). This helps assure funds are directed to projects that promise competitive returns on investment. Public opportunity costs and social discount rates are higher than private discount rates since public rates must represent foregone returns at the private rate in addition to foregone tax revenue. Since 2007, Canada's real social discount rate recommended by the Treasury Board of Canada has been 8% and is the rate used in the baseline iteration of this model. The board also recommends that sensitivity analysis of results at 3% and 10% discount rates be performed and reported (Treasury Board, 2007).

Figures 3 show comparative statics of the discount rate in the public utility model. All other parameters are held static, equivalent to the values used in the baseline iteration while the discount rate is varied within the range 3-12%. The first figure shows optimal percentage increase in capacity when costs of expansion ad infinitum are assessed at different rates of discount. The second figure shows how optimal frequency of expansion changes relative to variations in the discount rate.

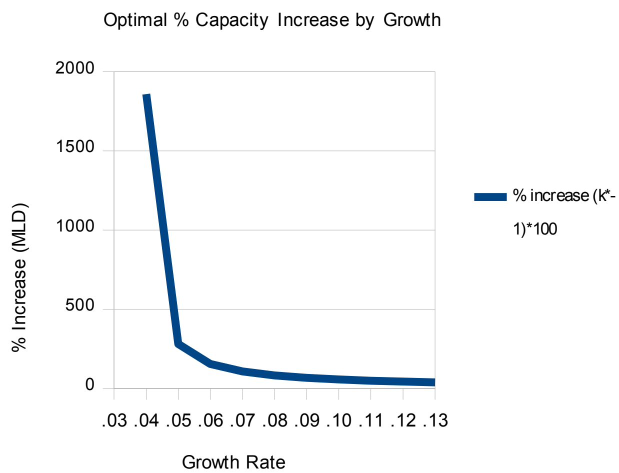
Figures 3: Comparative Statics of Discount Rate in the Public Utility Expansion Model

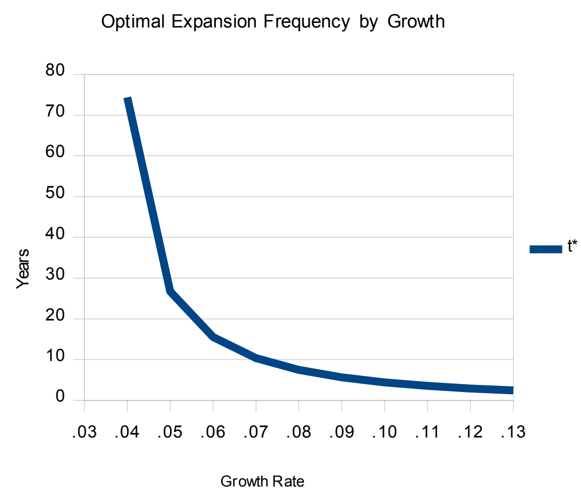


At a low discount rate (3%) the optimal percentage increase in capacity is lower than when the costs of expansion are assessed at a higher discount rate (12%). A low discount rate weighs present day costs more on par with future costs than a higher discount rate. In this model a low discount rate favours smaller expansion (lower costs) in the present and future but with higher frequency. The model suggests that when costs of expansion are assessed with a 3% discount rate capacity is to be expanded in 75% increments every 8 years. It makes sense that with relatively small increases in capacity expansion will occur more frequently to keep up with demand. When assessed with a 12% discount rate capacity is to be expanded in 178% increments every 15 years. When I use a high discount rate, where future costs are more heavily deflated relative to present day costs, the model favours higher cost development more infrequently.

Figures 4 show comparative statics of the growth rate in the public utility model. All other parameters are held static, equivalent to the values used in the baseline iteration while the growth rate is varied within the range 3-13%. The first figure shows the optimal percentage increase in capacity at different rates of demand growth. The second figure shows how the optimal frequency of development changes relative to variations in the growth rate.

Figures 4: Comparative Statics of Growth Rate in the Public Utility Expansion Model





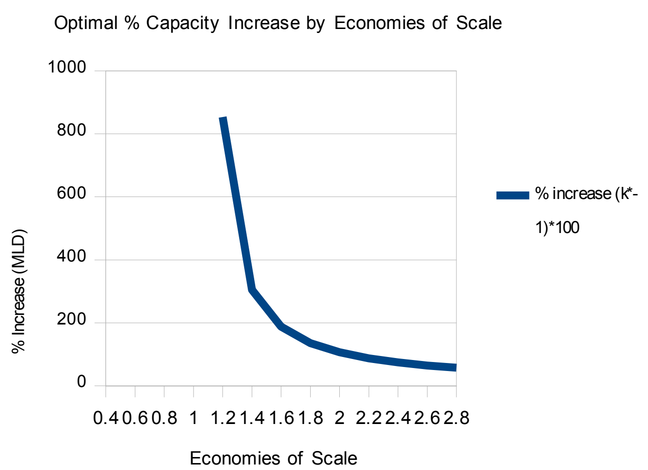
A growth rate of 7% is used in the baseline iteration of the model with the results of an optimal percentage increase in capacity of 107% every 10.37 years. A lower growth rate would be used if for example the growth of demand was expected to decline as a result of water conservation practices. The first figure shows that with a lower growth rate the optimal percentage increase in capacity is higher. The model does not solve with a discount rate of 8% and a low growth rate of 3%. At a 4% growth rate the optimal percentage increase in capacity is 1858%. At the same time, frequency of expansion decreases since expansion is large and it takes longer for slow demand growth to warrant the next instalment of infrastructure (in the case of a 4% growth rate, expansion would occur every 74 years). Faster growth of water demand (13% growth rate) warrants smaller percentage increases in capacity (38% capacity expansion) more frequently (every 2.46 years).

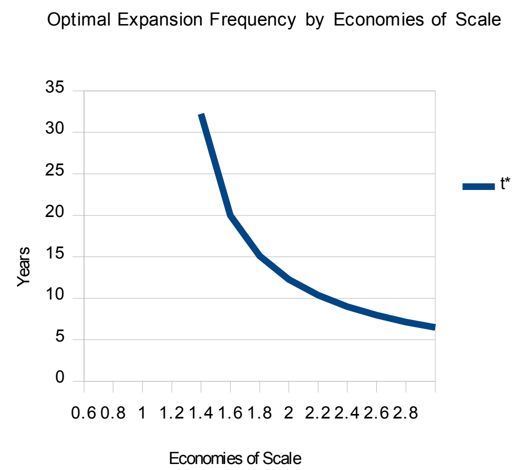
Figures 5 show comparative statics of economies of scale in the public utility decision model. All other parameters are held static, equivalent to the values used in the baseline iteration while economies of scale are varied within the range 0.4-2. Note that there are increasing economies of scale when the economies of scale coefficient Θ is less than one, are constant when the coefficient is one and are decreasing when the coefficient is more than one.

Typically, increasing economies of scale (decreasing marginal costs with increased capacity) are associated with natural monopolies. Large infrastructure, particularly a transmission network, is more efficiently installed and managed by a single public utility rather than duplicated and managed by several private companies. However, in this model where the public utility must continually source additional natural water features and build filtration plants and pipelines to connect to the established transmission network (where the original economies of scale were derived), the public utility will experience decreasing economies of scale. For the baseline iteration I use a diseconomies coefficient of two by this reasoning. Additionally, the model is constrained in its purpose of comparative statics such that it does not solve when $\Theta \leq 1$. I set $\Theta = 2$ in the baseline iteration to enable comparative statics for discount rates, growth rates and economies of scale above and below the baseline iteration values.

The first figure shows optimal percentage increase in capacity assessed at different economies of scale. The second figure shows how optimal expansion frequency changes relative to variations in economies of scale.

Figures 5: Comparative Statics of Economies of Scale in the Public Utility Expansion Model





Figures 5 show that, given a discount rate of .08 and a growth rate of .07, the model does not solve for economies of scale between 0 and 1. The model solves for economies of scale 1.2 and above. These are diseconomies of scale. At low diseconomies of scale (1.2) the optimal percentage increase in capacity is 854% every 32 years. At higher diseconomies of scale (2.8) the optimal percentage increase in capacity is 57% every 6.48 years.

The findings of this model and its comparative statics illustrate the importance of parameter analysis in PV cost analyses. In the following chapter I present the PV cost analyses that are used in this thesis to estimate WTP for ecosystem water supply services in Abbotsford, BC.

Chapter 5 – Numerical Evaluation

In this chapter I describe the present value (PV) cost analyses used to estimate total willingness to pay (WTP) for ecosystem water supply services in Abbotsford. First I describe generally the series of PV cost analyses performed and the reasoning behind this methodology. In the second section I present detailed methodology and summarize the PV cost analyses in data tables. The series of PV cost analyses are summarized in Table 9 at the end of section 5.2. In section 5.3 I discuss a number of data limitations. In section 5.4 I report the results of the PV cost analyses. Lastly, recalling the importance of parameter sensitivity analsysis the public utility optimization model in the previous chapter I perform sensitivity analysis on a number of the PV cost analysis parameters.

5.1 General Methodology

In this section I describe generally the PV cost analyses performed and the reasoning behind this methodological approach. A detailed description of the methodology and data tables follow. With the initial PV cost analysis I estimate the City of Abbotsford's *total WTP* for ecological water supply services over a 38 year period. To estimate *total WTP* I calculate the PV cost of a substitute technology: a proposed surface water supply system. This system is outlined in the City of Abbotsford's 2010 Water Master Plan. To calculate the PV cost of the system I first establish water demand and supply schedules. I use the city of Abbotsford's growth projections to create a water demand schedule beginning in the year 2010. On the supply side, I base the supply schedule on the city's plans to add capacity in two stages: 200 million litres a day (MLD) by 2015 and 50 MLD by 2024. Knowing the demand schedule and the proposed schedule of supply development I am able to determine when demand will catch up to capacity of the new system. The foreseeable timespan of the proposed infrastructure is 38 years before a next installment will be necessary. Adding the projected fixed and variable costs incurred at different times over the 38 year time period and using a social discount rate of 8% (as discussed in section 4.3) I analyse the PV cost of the new surface water supply system. The result is an initial estimate of the City of Abbotsford's *total WTP* for water supply services over the next 38 years.

However, an estimate of *total WTP* rather than an estimate of *WTP net of total cost* for water supply services falls short of finding an explicit price for the ecological service. Only once the *total cost* of aquifer water supply services are known can an explicit price, by way of an exchange value be estimated (see section 2.3: Review of Valuation). I do not determine *total cost* of aquifer water supply services in this thesis. However, given available information I estimate *partial cost* to deliver aquifer water supply services. With the second PV cost analysis in the series I estimate *partial cost* of aquifer water supply services based on the PV cost of aquifer water supply infrastructure (how much it costs to build and operate wells). I use the same demand schedule and 38 year period of analysis as used in the initial PV cost analysis above. I assume that well capacity is installed 25 MLD at a time based on the most recent Bevan well field development in Abbotsford, BC.[4] I also assume the public utility optimizes by spreading well field development over the 38 year period so as not to incur capital costs all at once which is possible with a less capital intensive technology such as wells. The result of this second PV cost analysis is an estimate of *partial cost* of aquifer water supply services.

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[4] The 2010 Water Master Plan considers 5 new water sources and 17 surface water development options that meet the criteria of sufficient water quantity. Of the 17 options Stave Lake (2) is chosen on the basis of low cost and public perception. Only one other choice, the Fraser River, is found to be lower cost but with public perception issues. The 15 remaining development options to the City of Abbotsford for future surface water supply development entail higher costs than Stave Lake (2).

I then consider the results of these two PV cost analyses together. The initial result is an estimate of the *total WTP* for (or *total benefits* of) ecological water supply services based on the cost of a substitute: a surface water supply system. The second result is an estimate of the *partial cost* to deliver ecological water supply services of the aquifer: well water infrastructure. By subtracting the *partial cost* of ecological water supply services from *total WTP* for ecological water supply services I can estimate *WTP for (or benefits of) ecological water supply services net of infrastructure costs*.

However, as stated above, infrastructure costs are only a partial measure of what it costs to deliver ecological water supply services from the Abbotsford-Sumas aquifer. The cost of nitrate management is another component. Current and ongoing agricultural nitrate contamination of the aquifer has been identified in this thesis as the single most significant constraint on future use of the aquifer for the city's water supply. Nitrate contamination of the aquifer provides the rationale for installing substitute technology: a surface water supply system. Because nitrate contamination exists, it presents a real barrier to using the aquifer for extensive drinking water purposes. Therefore, the cost of nitrate management needs to be accounted for in the total cost to deliver ecological water supply services from the aquifer.

However at this time the cost of nitrate management is unknown. What can be stated is that the difference between the city's *total WTP* for water supply services and *partial cost* to deliver that service by way of the aquifer (cost of wells) can be interpreted as the *total WTP for nutrient management,* or synonymously, *total WTP for improved ground water quality* (through nitrate management) over the next 38 years. If nitrates could be managed in a way that they do not contaminate the aquifer above the recommended guidelines for drinking water and the cost of nitrate management is below *total WTP for improved ground water quality* the City of Abbotsford could realize *net benefits* from the use of the aquifer water supply over the next 38 years. If the cost of nitrate management is above *total WTP for improved ground water quality* there are no net social benefits to accrue.

This summarizes the approach I use to evaluate *total WTP for improvement to ground water quality* of the Abbotsford-Sumas aquifer. The detailed methodology follows including data and a summary table. This is followed by a brief discussion of data limitations. In section 5.4 I report the results.

5.2 Detailed Methodology

In the previous section I reasoned for and described in general terms the methodology used to estimate *total WTP for improvements to ground water quality.* In this section I describe in detail the PV cost analyses performed to arrive at this estimate and the data used. First I explain how I created the demand, supply and expenditure schedules with a discussion of the assumptions made at each step.

The same demand schedule is used in both PV cost analyses that measure *total WTP for water supply service*s and *partial cost* to deliver that service by way of the aquifer. The 2010 Water Master Plan by AECOM and the AMWSC states that in 2007 peak day water demand was 142 MLD and that by 2031 they anticipate a peak day water demand of 297 MLD. They assume a linear growth rate based on commercial growth rates and leakage data provided by the city, and so I also use a linear growth rate to determine the water demand schedule. With peak day demand of 142 MLD in 2007 and

297 MLD in 2031 I calculated the linear growth rate to be 6.4583. This linear growth rate is used to interpolate water demand for the years 2010 to 2031 and extrapolate water demand beyond 2031. Table 6, columns 1-3, shows the demand schedule from 2010 to 2048, the 38 year period of the initial PV cost analysis.

Table 6: Present Value Cost Analysis of Surface Water System, 8% discount rate, 38 year period

1	2	3	4	5	6	7	8	9	10
Period	Year	Peak Demand (MLD)	Added Capacity (MLD)	Supply Capacity (MLD)	Excess Supply (MLD)	Capital Costs ($million)	O&M ($million)	Total Costs ($million)	PV of Costs ($million)
0	2010	161		163	1.63	3		3	**$181.10**
1	2011	168		163	-4.83	27		27	
2	2012	174		163	-11.29	27		27	
3	2013	181		163	-17.75	39		39	
4	2014	187		163	-24.21	60		60	
5	2015	194	200	363	169.33	42	1.68	43.68	
6	2016	200		363	162.88		1.68	1.68	
7	2017	207		363	156.42		1.68	1.68	
8	2018	213		363	149.96		1.68	1.68	
9	2019	219		363	143.5		1.68	1.68	
10	2020	226		363	137.04		1.68	1.68	
11	2021	232		363	130.58		1.68	1.68	
12	2022	239		363	124.13	2	1.68	3.68	
13	2023	245		363	117.67	13	1.68	14.68	
14	2024	252	50	413	161.21	14	2.1	16.1	
15	2025	258		413	154.75		2.1	2.1	
16	2026	265		413	148.29		2.1	2.1	
…									
37	2047	400		413	12.67		2.1	2.1	
38	2048	407		413	6.21		2.1	2.1	
Totals						*227*	*67.68*	*284.17*	

Table 6 in its entirety shows the initial 38 year period PV cost analysis of the proposed Stave Lake surface water system, which I use to estimate the City of Abbotsford's *WTP for water supply services*. With the water demand schedule in place, I next determine the water supply schedule. The system in operation in 2010 is reported in the 2010 Water Master Plan to have a supply capacity of 143 MLD from a mixture of surface and ground water sources. However it was expected that in 2010 the city would get a license to fully employ the new Bevan wells that at full capacity are able to draw 25 MLD. Seeing as 5.4 MLD from Bevan Wells is already counted in the overall supply capacity of 143 MLD an additional 20 MLD takes present day supply to 163 MLD as shown in the first line of column 5, Table 6. This supply schedule starting point of 163 MLD in 2010 is the same for each PV cost analysis performed in the series.

Supply capacity increases as new infrastructure is built. The schedule by which Stave Lake comes online is outlined in the 2010 Water Master Plan. In Table 6, column 4, when Stave Lake phase one is complete 200 MLD of additional supply capacity comes online. In column 5, an increase in supply capacity is seen from the present day supply of 163 MLD to 363 MLD. The second phase of the Stave Lake system is scheduled to come online in 2024 raising supply capacity to 413 MLD. Table 6, column 5 constitutes the water supply schedule over time (columns 1 and 2).

Table 6, column 6 shows the difference between peak day demand and supply capacity, or the supply shortfall. AECOM projects peak day shortfalls from 2010-2014. The 2010 Water Master Plan encourages demand management to help alleviate impacts from these shortfalls until the new surface water source comes online. Cost of demand management initiatives are not reported in the 2010 Water Master Plan nor were they discovered for the purposes of this valuation study.[5]

[5] Demand management is another substitute technology for aquifer water supply services. If costs of demand management initiatives were known along with costs associated with decreased water use (decreased productivity in industry for example if they were a target of demand management initiatives) these costs could present another measure of *total WTP* for water supply services. It is arguable that an estimate of *total WTP* based on the substitute cost of demand management initiatives would be considerably less than *total WTP* based on the substitution cost of surface water supply infrastructure. However, demand management for the long-term is not, to date, the substitute technology

To remain consistent I assume in each of the PV cost analyses that demand management initiatives occur from 2010-2014 and that supply capacity, be it surface water infrastructure or wells, comes online in 2015. By starting each supply schedule in 2015 the unknown costs associated with demand management are endogenized in each PV cost analysis. When *partial cost* estimates (based on well infrastructure development beginning in 2015) are subtracted from *total WTP* estimates (based on surface water infrastructure development beginning in 2015) the hidden demand management costs cancel each other out. Finally, note that excess supply decreases to almost zero in 2048 and would be negative the following year. The years 2010-2048 constitute the 38 year time period in which demand catches up to supply.

With the demand and supply schedules in place I turn to the capital costs of supply capacity development and operation and maintenance (O&M) costs of the Stave Lake system over 38 years. Capital costs and their schedule of expenditure are reported in the 2010 Water Master Plan and are replicated in column 7 of Table 6. It is assumed that the public utility's planned expenditure is cost minimizing since they make decisions in the best interest of their constituents and under public scrutiny.

I deduce the stream of O&M costs by finding the difference between the overall PV cost for the Stave Lake system reported in the plan ($224 million) and capital cost expenditure. I assume O&M costs (which include inspection, administration, treatment and utility fees) are fixed according to

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chosen by the City of Abbotsford, therefore an estimate of *total WTP* based on the cost of demand management would not be a market-based observation or a revealed preference estimate.

supply capacity and unchanged by the level of actual output (determined each year by the level of demand). This is done because the breakdown of fixed versus variable costs is unknown. To refine the numbers this assumption may warrant more attention but for present purposes the error introduced by this assumption is considered negligible.

The 2010 Water Master Plan reports the overall PV cost for the system includes the stream of capital expenditure and O&M costs evaluated over a 31 year period with a 3% discount rate. With these parameters and the PV formula

$$PV = \sum_{t=0}^{n} \frac{FV_t}{(1+i)^t}$$

where n is the total number of periods, FV the future value ($224 million) in period t (year 31) and i is the discount rate (3%), I calculate the schedule of O&M costs. I determine that the first phase of development (a capacity of 200 MLD) would incur O&M costs of $1.68 million a year (see Table 6, column 7) and after the second phase of development (an added capacity of 50 MLD) O&M costs would rise 25% to $2.1 million a year. In Table 6, columns 4 and 5, I show that at the end of each phase of capital cost expenditure supply capacity increases. At the time supply capacity increases operation and maintenance (O&M) costs in column 7 are initialized. Table 6, column 7 constitutes the schedule of water supply expenditure for the Stave Lake system over time (columns 1 and 2).

The next step is to evaluate PV costs, again using the PV formula above. To calculate PV costs of the Stave Lake system I define n number of periods as 38, FV by the total cost figures shown in column 9 and t by the period numbers shown in column 1, Table 6. Finally, a social discount rate of 8% is employed for reasons discussed in section 4.3. With the social discount rate and this time frame the PV cost of the Stave Lake surface water system is $181 million as stated in column 10 of Table 6.[6] The result is a $181 estimate of the *total WTP* for ecological water supply services over 38 years based on the cost of a substitute technology: the Stave Lake water supply system.

However, an estimate of *total WTP* rather than an estimate of *WTP net of total cost* for water supply services falls short of finding an explicit price for the ecological service. To get closer to finding an explicit price I estimate *partial cost* to deliver water supply services by analysing the PV cost of well infrastructure. This value can be subtracted from *total WTP* to arrive at an estimate of *WTP for (or benefits of) ecological water supply services net of infrastructure costs.* The PV cost analysis of ground water infrastructure differs only slightly from the PV cost analysis of the Stave Lake system. Well infrastructure cost estimates are drawn from proposed expenditure on the Bevan Wells development as reported in the 2006 Water Master Plan by Dayton & Knight Ltd. Costs for feasibility study and infrastructure are estimated at $3.33 million for a well field of four. Operation and maintenance (O&M) costs are ascertained from the 2009-2010 AMWSC budget and include treatment, inspections, maintenance and utilities. The values used for each O&M category are shown in Table 7.

Table 7: Annual Operation & Maintenance (O&M) Costs for a Well Field

Well O&M Costs	**4 wells, 25 MLD**
Treatment	$1,000.00
Inspections	$4,100.00
Maintenance	$22,600.00
Utilities	$25,000.00
Total	**$52,700.00**
(in millions)	**0.0527**

Source: AMWSC Budget 2009-2010

[6] This is less than the $224 PV cost projected in the 2010 Master Water Plan due to the higher social discount rate used and longer time frame of analysis.

Table 8 shows the PV cost analysis for well infrastructure over a 38 year period. The water demand schedule remains the same as the first PV cost analysis presented in Table 6. The supply schedule differs since well fields are a lower capacity technology than the proposed surface water system. Whereas the surface water system entails high up-front capacity, well field capacity can be added in smaller increments (25 MLD as opposed to 200 MLD). As demand reaches supply, additional wells are added until the end of the 38 year period. Table 8, column 4 indicates when it is optimal for additional capacity to be added, beginning in 2015. To be consistent with the 2010 Water Master Plan and the initial PV cost analysis detailed in Table 6, I assume demand management initiatives are minimizing the effects of a supply shortfall in the years 2010-2015.

Table 8: *Present Value Cost Analysis of Well Field Development, 8% discount rate, 38 year period*

1	2	3	4	5	6	7	8	9	10
Period	Year	Peak Demand (MLD)	Added Capacity (MLD)	Supply Capacity (MLD)	Excess Supply (MLD)	O&M ($million)	Capital Costs ($million)	Total Costs ($million)	PV of Costs ($million)
0	2010	161		163	1.63				**$12.69**
1	2011	168		163	-4.83				
2	2012	174		163	-11.29				
3	2013	181		163	-17.75				
4	2014	187		163	-24.21				
5	2015	194	50	213	19.33	0.1054	6.66	6.7654	
6	2016	200		213	12.88	0.1054		0.1054	
7	2017	207		213	6.42	0.1054		0.1054	
8	2018	213	25	238	24.96	0.1581	3.33	3.4881	
9	2019	219		238	18.5	0.1581		0.1581	
10	2020	226		238	12.04	0.1581		0.1581	
11	2021	232		238	5.58	0.1581		0.1581	
12	2022	239	25	263	24.13	0.2108	3.33	3.5408	
13	2023	245		263	17.67	0.2108		0.2108	
14	2024	252		263	11.21	0.2108		0.2108	
15	2025	258		263	4.75	0.2108		0.2108	
16	2026	265	25	288	23.29	0.2635	3.33	3.5935	
17	2027	271		288	16.83	0.2635		0.2635	
18	2028	278		288	10.38	0.2635		0.2635	
19	2029	284		288	3.92	0.2635		0.2635	
20	2030	291	25	313	22.46	0.3162	3.33	3.6462	
21	2031	297		313	16	0.3162		0.3162	
22	2032	303		313	9.54	0.3162		0.3162	
23	2033	310		313	3.08	0.3162		0.3162	
24	2034	316	25	338	21.63	0.3689	3.33	3.6989	
25	2035	323		338	15.17	0.3689		0.3689	
26	2036	329		338	8.71	0.3689		0.3689	
27	2037	336		338	2.25	0.3689		0.3689	
28	2038	342	25	363	20.79	0.4216	3.33	3.7516	
29	2039	349		363	14.33	0.4216		0.4216	
30	2040	355		363	7.88	0.4216		0.4216	
31	2041	362		363	1.42	0.4216		0.4216	
32	2042	368	25	388	19.96	0.4743	3.33	3.8043	
33	2043	374		388	13.5	0.4743		0.4743	
34	2044	381		388	7.04	0.4743		0.4743	
35	2045	387		388	0.58	0.4743		0.4743	
36	2046	394	25	413	19.13	0.5270	3.33	3.8570	
37	2047	400		413	12.67	0.5270		0.5270	
38	2048	407		413	6.21	0.5270		0.5270	
Totals			*40 new wells*			*3.4782*	*33.3000*	*44.0508*	

Wells become operational within the same year as drilling and so O&M costs for disinfection and fuel for pumping begin to accrue and supply is increased in the same year as capital costs are incurred (see Table 8 columns 7, 8 and 9). *FV* costs incurred each year (column 9) are evaluated at a social discount rate of 8% and summed to arrive at a PV cost for well field development of $13 million over a 38 year period. The result of this second PV cost analysis is an estimate of *partial cost* of aquifer water supply services.

Table 9 summarizes the PV cost analyses I use to evaluate *total WTP for (or benefits of) improved ground water quality* of the Abbotsford-Sumas aquifer. The first result is an estimate of the *total WTP* for, or *total benefits* of ecological water supply services based on the cost of a substitute: a surface water supply system. The second result is an estimate of the *partial cost* to deliver ecological water supply services of the aquifer: well water infrastructure. I then consider the results of these two PV cost analyses together. By subtracting the *partial cost* of ecological water supply services from *total WTP* for ecological water supply services I estimate *total WTP for (or benefits of) improved ground water quality net of infrastructure costs*. This estimate is reported in Table 10 in the last section of this chapter.

Table 9: *Summary of Present Value Cost Analyses*

Number in Series of Present Value (PV) Cost Analyses	Value Estimated	PV Cost Analysis Period	Surface Water (SW) Supply Capacity Expansion	Ground Water (GW) Supply Capacity (Wells) Expansion	PV of System Expansion Costs
		(years)	(MLD)	(MLD)	(million)
1	***Total WTP for water supply services***	**38**	**250**	**0**	**$181**
2	*Partial cost of aquifer water supply services*	38	0	250	$13

In Table 10 I also report the estimates as annualized values. To find the annualized values I use the following annuity payment formula:

$$PMT = \frac{PV}{\frac{(1 - \frac{1}{(1+i)^n})}{i}}$$

where *PMT* is the annual benefit to Abbotsford taxpayers of improved ground water quality, *PV* is the present value of that benefit over the entire period, *i* is the discount rate used in the analysis (8%) and *n* is the number of periods (38 years). Finally, I calculate the annual benefit per household by dividing the annualized value by Abbotsford households reported in the 2010 Statistics Canada census (34,977).

Next I describe the limitations in this numerical evaluation before presenting the above mentioned results in Table 10, section 5.4.

5.3 Data Limitations

It is important to note some data limitations within the PV cost analyses presented in this thesis. As mentioned above, O&M costs of the surface water system are likely inflated. O&M costs of the new surface water system are assumed to be equally spread out over the time horizon dependant on capacity rather than actual output. At an 8% discount rate this puts upward pressure on PV costs. If O&M costs were less to begin with and got progressively higher a higher proportion of costs would be discounted and result in a lower PV cost overall. There is also an upward pressure on the capital costs of the surface water system. The expenditure figures presented in the 2010 Water Master Plan are the result of a high level analysis where it is general practice to add 25% contingency for costs (Alexander, pers. comm., 2010).

O&M costs of wells are likely downplayed. Budget lines (because they were made available by the City of Abbotsford) rather than actual expenditures are used to approximate the cost of running four new wells. The PV cost anlayses that include well capacity development assume higher extraction levels from all wells on average and on peak days than the AMSWC has historically operated at. Higher extraction levels would put upward pressure on the cost of disinfection and energy used for pumping, but also increase the well fields economies of scale due to fixed expenditure on inspection and administration. An investigation into the correlation between historical well water flows and actual expenditures would help to refine the numbers used here.

However, for the same reasons above, that 25% contingency is added for costs in planning documents, it is likely that the capital costs for well infrastructure are inflated. Furthermore the 2010 Water Master Plan assumes that Clearbrook Waterworks, currently a system operating independently of AMSWC, will no longer be operating. This inflates the projected demand on the AMSWC system and ignores some existing well infrastructure. The result is a further inflated value for the capital expenditure necessary to develop the wells.

One important data limitation of course are the unknown costs of nitrate contamination. This particular data limitation shapes the end result of this research presented in the following section.

5.4 Results

The PV of system expansion costs for surface water and well infrastructure are reported in Table 9 above and replicated in Table 10 below. The PV costs represent *total WTP for water supply services* (based on the PV cost analysis of a surface water system) and *partial costs for aquifer water supply services* (based on the PV cost analysis of well infrastructure). By subtracting *partial costs* from *total WTP* I find an estimate for *total WTP for improved ground water quality net of infrastructure costs* or *total WTP for improved nitrate management* as explained in section 5.1. Results are presented in Table 10.

Table 10: WTP Estimate for Improved Ground Water (GW) Quality in Abbotsford, BC

Number in Series of Present Value (PV) Cost Analyses	Valuation	Period of Analysis	PV of System Expansion Costs	Annualized Values	Annualized Values by household
		(years)	(million)	(million)	(household/ year)
1	Total WTP for w ater supply services	38	$181	$15	$438
2	Partial cost of aquifer w ater supply services	38	$13	$1	$31
	Total WTP minus Partial Cost	*38*	*$168*	*$14*	*$407*

I estimate *total WTP for (or total benefits of) improved ground water quality net of infrastructure costs* to be $168 million over 38 years. That is to say, if assumptions hold, that Abbotsford taxpayers are willing to pay $168 million over the next 38 years to avoid nitrate contaminated water, since this is what they will pay for a new surface water system over and above what it would cost to install wells. This estimate can also be interpreted as *total WTP for (or total benefits of) improved nitrate management* since agricultural nitrates are the barrier to further developing the groundwater aquifer for drinking water purposes.

The second to last column in Table 10 shows the annual benefit to Abbotsford taxpayers of improved ground water quality net of infrastructure costs to be $14 million per year over a 38 year period. The last column shows the $14 million annualized benefit divided up amongst Abbotsford households amounts to a benefit net of infrastructure costs of $407 per household per year over 38 years.

5.5 Results Sensitivity Analysis

In section 4.2 I present a model that illustrates how the public utility expansion cost minimization problem can be solved over an infinite time frame. The purpose of the public utility model is to demonstrate the importance of parameter values in present value (PV) cost analysis. I pay special attention to the following parameters: discount rate, growth rate and economies of scale. In this chapter I find the value of an ecological service through revealed preference substitute cost methodology. Based on revealed preference (demonstrated preference through actual market activity) I use a 38 period PV cost analysis to estimate *total WTP for improved ground water quality (or nitrate management practices)*. 38 years is the amount of time the proposed surface water system expansion meets demand, given the City of Abbotsford's projected growth rate, before further expansion will need to occur. In this section I explain how changes in the PV cost analysis parameters affect the *total WTP for improved ground water quality (or nitrate management practices)* estimate.

The *total WTP* and *partial cost* estimates reported in Tables 9 and 10 are the results of PV cost analyses that use a real social discount rate of 8%. To show how different discount rates impact PV costs Table 11 presents the results using a discount rate of 3% and a discount rate of 10%. The percentage change from the original results (when an 8% discount rate is used) are also shown.

I use a low discount rate of 3% in this sensitivity analysis because this is the rate used to determine PV costs in the 2010 Water Master Plan. The Treasury Board of Canada also recommends that sensitivity analyses report results when a 3% and 10% discount rate is used. When assessed with a discount rate of 3% the PV cost of the surface water system (*total WTP*) increases by 37% to $236 million. The PV cost of ground water infrastructure (*partial cost*) increases by 100% to $25 million.

Table 11: Discount Rate Sensitivity Analysis

Discount Rate			***8%***	***3%***		***10%***	
Number in PV Cost Analyses Series	Description	Period of Analysis	PV of Added Capacity Costs	PV of Added Capacity Costs	% change	PV of Added Capacity Costs	% change
		(years)	(million)	(million)		(million)	
1	Total WTP	38	$181	$236	37%	$151	-10%
2	Partial Costs	38	$13	$25	100%	$10	-23%
	Total WTP minus Partial Costs	***38***	***$168***	***$211***	***31%***	***$142***	***-9%***

Analysis with a low discount rate means that future costs will count for more of the PV cost which explains the increase in both PV cost analyses. Surface water system expansion entails high up front costs with relatively low future costs. The development of well fields on the other hand has costs more evenly spread out over the 38 year period, which at a low discount rate are considered more on par with present day costs. It is not surprising then that PV costs of surface water system expansion increase less than PV costs of well development. When assessed at a higher discount rate of 10% PV costs of surface water system expansion and well field development decrease. Higher up front costs means PV cost of a surface water system decrease less than PV cost of well field development. The cumulative effect, when the *total WTP* and *partial cost* estimates are assessed at a 3% discount rate is an increase in the *total WTP for improved ground water quality net of infrastructure costs*. When assessed at a 10% discount rate the *total WTP for improved ground water quality net of infrastructure costs* is lower.

A 6.45% linear growth rate was used in the PV cost analyses based on city projections of water demand growth. To show how different growth rates impact PV costs Table 12 presents the results using a growth rate of 4% and a growth rate of 13%. The percentage change from the original results (when a 6.45% discount rate is used) are also shown. A change in the growth rate changes the time period of PV cost analysis used in the valuation. A lower growth rate means that the proposed surface water expansion will match demand further into the future. This extends the period of PV cost analysis and the number of years that O&M costs accrue.

In the case of a surface water system all capacity is installed up front and on a schedule independent of the growth rate (200 MLD in 2015; 50 MLD in 2024).[7] PV costs of surface water system expansion increase as a result of additional years of O&M costs. In the case of well field development a lower growth rate and extended time frame of analysis means infrastructure development is pushed further into the future (and discounted more). Additional years of O&M costs accrue but are less than surface water system O&M costs. A lower growth rate results in lower PV costs for well field development. In summary, at a lower growth rate, *total WTP* (PV cost of surface water system expansion) increases whereas *partial cost* (PV cost of well field development) decreases.

[7] Recall that the surface water expansion PV cost analysis is based on the 2010 Master Water Plan in keeping with revealed preference methodology.

As a result, *total WTP for improved ground water quality net of infrastructure costs* increases.

Table 12: Growth Rate Sensitivity Analysis

Growth Rate		***6%***		***4%***			***13%***		
Number in PV Cost Analyses Series	Description	Period of Analysis	PV of Added Capacity Costs	Period of Analysis	PV of Added Capacity Costs	% change	Period of Analysis	PV of Added Capacity Costs	% change
		(years)	(million)	(years)	(million)		(years)	(million)	
1	Total WTP	38	$181	67	$182	1%	20	$177	-2%
2	Partial Costs	38	$13	67	$8	-35%	20	$17	32%
	Total WTP minus Partial Costs	***38***	***$168***	***67***	***$174***	***3%***	***20***	***$160***	***-5%***

At a higher growth rate the period of analysis shortens since added surface water system capacity will be matched by water demand sooner. This reduces the O&M costs in the PV cost of the proposed surface water system and in the PV cost of well field development. However, in the case of the well field development PV cost analysis infrastructure costs occur earlier (and are discounted less). In summary, at a higher growth rate PV cost of surface water infrastructure decreases while PV cost of well field development increases. The overall effect is a decrease in *total WTP for improved ground water quality net of infrastructure costs*.

Economies of scale are explored in the public utility expansion model but are not explored here. Because the model entails a cost function with an economies of scale parameter sensitivity analysis of this parameter is possible. However, there is no economies of scale parameter used in the PV cost analyses which form the basis of the WTP estimates. Rather, economies of scale are embedded in the figures sourced from the 2010 Master Water Plan and City of Abbotsford budget which were then used in the PV cost analyses.

With this I conclude sensitivity analysis of the WTP estimates determined by way of the PV cost analyses performed in this chapter. The above comparative statics show the impact on the results of changes in PV cost analysis parameters. Next, I discuss the WTP estimates, their validity and meaning.

Chapter 6 - Discussion

First, I discuss my willingness to pay (WTP) estimates in the context of a previous valuation study that measures benefits of improving water quality in the Abbotsford-Sumas aquifer. I then consider the limitations of the methodology used in this thesis and the implications that has for the results. I further discuss my results in terms of the rationale for a payments for ecological services (PES) program and the extent to which the findings of this thesis can be used towards that purpose.

I compare the findings of my valuation study with one other comparable valuation study performed in the region (already discussed in my review of valuation section 2.3). It is not my purpose to validate the results of one study or the other. I compare the studies in order to contextualize my results within existing valuation work in the region and to launch into discussion about my results. Recall that the Hauser and van Kooten (1993) study measures benefits of improving water quality in the Abbotsford-Sumas aquifer using defense expenditure, contingent valuation and stated preference fuzzy pair wise comparison methodologies. Table 13 compares my results using a revealed preference replacement cost methodology with the results of Hauser and van Kooten (1993) reported in 2010 dollars.

For comparison sake, the values of Hauser and van Kooten are adjusted to reflect an increase in the number of Abbotsford households since 1993 and for inflation. An assumption here is that the current population of Abbotsford is comparable in terms of income and preferences to the smaller population, a sample of whom were surveyed in 1993. A change in population composition would impact the upper bound estimates based on WTP surveys but not the lower bound values based on defense expenditures where, through regressional analysis, income was not found to be significant (Hauser and van Kooten, 1993). For purposes here, despite likely changes in population composition that would mean changes in the upper bound estimate, the values present a basis of comparison for the results of my study reported in the first row of Table 13.

Table 13: Non-Market Values of Agriculturally Related Ecological Goods and Services in the LFV

Study	**Measured**	**Method**	**Values, in 2010 dollars**
Ageson, 2010	Benefits of improvements to aquifer water quality	Replacement Cost	\$15 million/ yr
Hauser and van Kooten, 1993	Benefits of improvements to aquifer water quality	Defense Expenditure (lower bound) & Fuzzy Pairwise Comparison	\$3.5 - \$14 million/ yr[8]

Note that the value generated by my study and cited in Table 13, \$15 million/ year, is the annualized *total WTP estimate for water supply services* (or synonymously, *total WTP for improved water quality*). This value (and not *total WTP minus partial cost*) is directly comparable to Hauser and van Kooten's lower bound defense expenditure (based on WTP for filters and bottled water) estimate of \$3.5 million/ year. They consider their lower bound estimate (\$3.5 million/ year) a validity test of their stated preference methodologies and not a total economic value. Defense expenditure renders a minimum WTP for water quality void of passive and non-use values associated with ecological services that deliver drinking water. Replacement cost methodology also renders a minimum WTP for water quality void of passive and non-use values.

8 Numbers reported here are based on the values reported in (Hauser and van Kooten, 1993): \$70-\$284/ household/yr, increased by an average inflation rate of 1.87% and an increase in Abbotsford households from 6,300 to 34,977.

It is interesting then that my revealed preference replacement cost estimate of $15 million/ year is much higher than Hauser and van Kooten's defense expenditure estimate. One explanation may be survey error in their study since their defense expenditure data was elicited by survey rather than as observed market behaviour. They also state the defense expenditure estimates may suffer downward bias as a result of taxes being excluded from their analysis. Another explanation is that my estimate is inflated. The inflationary criticism that typically applies to replacement cost methodology does not apply here since my replacement cost estimates are based on market behaviour (the proposed surface water system) rather than total replacement of the ecological amenity. It is possible however that my estimates are inflated because they reflect a city council decision rather than individual Abbotsford households' WTP.

My estimate of $15 million/ year also falls above Hauser and van Kooten's stated preference estimate of $14 million/ year. They use a stated preference fuzzy pairwise comparison methodology where respondents are asked to rank amongst other things what they are WTP for "improving the availability and quality of one's drinking water" (Hauser and van Kooten, 1993). Theoretically, my revealed preference value should fall below their stated preference total economic value since stated preference WTP estimates must exceed revealed preference. Again though, it is possible that their estimates suffer from survey design error. It is also possible that current population composition, depending on their preferences and income, would change their upper bound estimate.
Lastly, as above, it is possible that my estimates overstate WTP for water quality because my estimates are based on the revealed preference of city council, not that of Abbotsford residents.

To summarize the comparison of my study with the Hauser and van Kooten (1993) study, my revealed preference replacement cost estimate falls above their estimates. However, it is important to note that because my estimate is based on market behaviour using replacement cost methodology it does not measure a total economic value. My value of $181 million over a 38 year period ($15 million/ year) is a lower bound measure of WTP since it does not include passive and non-use values of improved ground water quality. Despite being a lower bound value my estimate may be inflated because it reflects the wishes of city council and not those of the tax paying populace.

Improved Abbotsford-Sumas aquifer water quality via improved nitrate management practices would render multiple benefits to multiple users only one group of which is considered in this study: Abbotsford residents. Improved water quality enhances conditions for aquatic life benefitting the fishing sector. Improved nitrate management practices could enhance soil quality, reduce greenhouse gas emissions and result in operational efficiencies. Furthermore, there is the potential of resolving a transboundary political issue. There are a thousand more potential beneficiaries in Washington whose well water is affected by the southerly flow of nitrate contaminated aquifer water from Abbotsford. Consideration of other beneficiaries would increase the estimate of benefits for improved ground water quality and nitrate management practices.

What both my and the Hauser and van Kooten studies do not do is calculate net benefits. The valuation exercise in this thesis takes us so far as to find out how much it would be worth to the City of Abbotsford to improve Abbotsford-Sumas aquifer water quality, and then how much it would cost to develop well infrastructure. Due to a lack of information regarding the behaviour of nitrates in the aquifer no attempt is made in this thesis to determine total costs related to nitrate remediation and/ or ongoing nitrate management practices that would allow for ongoing well field development. A comprehensive hydrogeological study is needed to determine the constraints historical and ongoing nitrate deposition in the aquifer pose to well field development, as well as a mass water balance for the aquifer to determine sustainable rates of water extraction and potential distribution of well sites.

It may be that historical nitrate deposition renders the aquifer unsuitable for future well field development and that the aquifer must be remediated. In California, the Irvine Ranch Water District, serving 50,000, has built a nitrate filtration system that cost $33 million in capital expenditure, with $2.3 million per year in operating and maintenance (O&M) costs (Scott, 2010). If I assume the same nitrate filtration system can be installed to service Abbotsford's 34,977 households and I integrate its costs into the PV cost analysis of well field development over a 38 year period, the PV cost is $55 million. The result is a $126 million (unsusbstantiated) estimate of net benefits or *total WTP for improved ground water quality net infrastructure and nitrate remediation costs*. Water quality improvement through a nitrate filtration system may provide a significant cost-effective alternative to the new Stave Lake surface water system.

On the other hand, it may be that present day nitrate deposition prevents well development and changes in nitrate management practices in susceptible areas of the aquifer would be enough to render the aquifer safe for future well field development. If so, only the cost of nitrate management need be discovered to measure net benefits of improved ground water quality. To consider how the estimated benefits in this study could provide economic rationale for an EGS program that incentivizes nutrient management practices I present a number of calculations. I by no means make a specific suggestion for an EGS program or payments for ecological services (PES) scheme. Any such program would not be coercive or prescriptive, it would not tell farmers what to do. A PES scheme requires careful design concerned with the existing policy regime and policies working at cross purposes, additionality (paying only for what would not have been done otherwise), distributional effects, the possibility of creating perverse incentives, a method to secure service provision and program efficiency (see section 2.2). With the following calculations I only mean to contextualize my estimates and demonstrate their magnitude within the existing economics of Abbotsford agriculture.

First, I calculate per hectare benefits of improved ground water quality in the area south of Marshall Road. This area above the aquifer is agriculturally intense with space for well fields but high nitrate concentrations. The *total WTP for improved water quality net infrastructure costs* ($168 million) is divided by the 1,415 hectares of agricultural land that sits south of Marshall Road. The result is a WTP of $119,018 per hectare over the next 38 years if improved nutrient management practices could be shown to secure the provision of improved water quality in this region of the aquifer. Annualized, the benefit amounts to $10,062 per hectare per year for 38 years.

Consider what a WTP of up to $10,062 per hectare per year for improved ground water quality could mean. In a spot check with the regional agrologist, the agricultural rent streams for Abbotsford low land as pasture were reported to be $0 - $124 per hectare per year while raspberry rent streams were $988 - $2,471 per hectare per year (Robbins, pers. comm., 2010). Pasture used for grazing cattle is relatively environmentally benign since nitrates are more likely to be taken up by the feed crop than to leach into ground water. Localized long-term trials at the Agassiz agricultural research station seem to support this (Sutherland, pers. comm., 2010). Rent streams or land rental rates are measures of the opportunity cost of land in its next best alternative use. In the case presented here I am suggesting that with a WTP for improved ground water quality of $10,062 per hectare per year, $7,591 above the per hectare value generated by the next most profitable crop (raspberries), the best use of this land may be in ecological service delivery of ground water quality through pasture.

Only as a point of illustration not to be mistaken for a policy recommendation: a scheme aimed at reducing nitrate leaching into the aquifer could pay land managers the difference between farming a high (private) value but environmentally damaging crop (a crop with social costs) and a low (private) value but environmentally benign crop (a crop with the social benefit of water quality). In effect, such a payment turns the ecological non-market service of improved water quality into a market good. Given the rent stream values provided by Robbins, the revenue difference between producing raspberries and pasture for cattle is $988 - $2,347 per hectare per year. Based on the estimate of $10,062 per hectare per year (in the area south of Marshall Road) for improved nitrate management practices a $2,347 payment per hectare per year for farmers to improve water quality by raising pasture for livestock instead of raspberries leaves $10.9 million per year for studies and to run the program for 38 years.

Farming an environmentally benign crop is just one method to improve nitrate management practices above the aquifer. Whole farm nutrient management plans, application of manure and fertilizers at levels specific to crop needs, manure composting and manure export provide other potential ways to reduce nitrate leaching into the Abbotsford-Sumas aquifer. An effective market-based EGS program would allow producers to discover in their own production possibilities the most efficient way to deliver ecological services. What is important is that farmers and other land users above the ground water aquifer are provided incentives to achieve specific reductions in nitrate pollution.

Market-based instruments such as PES or water quality trading aimed at research-based nutrient emission reduction targets are likely to be more effective than current voluntary adoption policy approaches. Of utmost importance is multi-level government and stakeholder involvement, funded research and enabling legislation (such as a nutrient application and/ or production limit on land above ground water aquifers). Such legislation was an important precursor to water quality trading programs developed in the United States (see section 2.2). To name a few of the potential partners in the development of a market-based nutrient reduction initiative are the BC Ministry of Agriculture and Lands, BC Ministry of Environment, the BC Sustainable Poultry Farming Group and other farming associations, City of Abbotsford, Fraser Valley Trout Hatchery, households on septic systems and households with groundwater wells. Not to mention partners across the border in the state of Washington, such as the City of Sumas. A market-based nutrient management program is a substantial undertaking that would take time to implement. In the meantime peak day water shortfalls could be met through demand management initiatives, as is occuring under the Stave Lake surface water system development plan until 2015.

The City of Abbotsford may also consider other policy options in collaboration with the BC Ministry of Agricuture and Lands and Ministry of Environment. Policy options might include increased enforcement of the Environmental Management Act and Drinking Water Protection Act helped through financial contributions from the City of Abbotsford. The city may also consider capabilities within their bylaws, business licensing and permits with advice from the Farm Industry

Review Board on limitations imposed by the Farm Practices Protection (Right to Farm) Act.

On a final note, it may be that the City of Abbotsford does not develop the Stave Lake surface water system with the expenditure schedule analysed in this thesis. The residents of Abbotsford are voting in a referendum in November 2011 on whether they want to borrow funds for the Stave Lake project. If citizens reject the project on financial grounds my WTP estimates, based on the cost of the Stave Lake development, are unfounded. If the city then decides to pursue a less expensive option, may it be a smaller surface water system or demand management initiatives, the new option would provide basis for a new WTP estimate. With this thesis I have developed a framework for analysing the public utility's water service delivery options with a preliminary investigation into the option of water services through ecological function.

In summary, my $181 million estimate of *total WTP for improved water quality* by the City of Abbotsford is a lower bound estimate. However, depending on the nature of the city council decision to develop a costly surface water system and their ability to gauge their constituents' WTP for water quality, my estimate may be inflated. A better understanding of how nitrate contamination is affecting well field development in Abbotsford would validate whether replacement cost is an appropriate measure for WTP. If it is, my results indicate that net benefits are significant enough for the City of Abbotsford to throw some time and money towards the problem of agricultural nitrates polluting the Abbotsford-Sumas aquifer.

Chapter 7 - Conclusions

British Columbian (BC) agriculture has the potential to render ecological goods and services (EGS) and there is a push to utilize market-based mechanisms in agri-environmental policy. Agricultural practices that render extra costs or benefits, such as those that result in pollution or those that improve water quality are unvalued by the market and are therefore over or under supplied. Government intervention is justifiable in such cases of market failure but needs to be carefully considered and effectively designed. An important tool of public policy is the valuation of non-market costs and benefits. Values for non-market goods and services can provide economic rationale for policy and basis for market-based management mechanisms.

In this thesis I explore how the concept of EGS can be applied towards the management of an agricultural externality in BC. I focus on the Lower Fraser Valley, an agriculturally productive and intensive region of Canada where there are documented impacts of agricultural nutrients on the Abbotsford-Sumas aquifer. Voluntary and command and control policy measures to date have been ineffective in reducing nitrate deposition in the aquifer.

To provide groundwork for an EGS program I set out to determine the value of improvements to ground water quality. I identify the city of Abbotsford's public water utility as the service buyer which is a major departure from current agricultural policy in the region. I propose that willingness to pay (WTP) for improved water quality in the Abbotsford-Sumas aquifer can be estimated using revealed preference for a substitute technology: the proposed Stave Lake surface water system.

The valuation methodology presented in this thesis provides a framework for similar valuation work in regions with parallel circumstances. I estimate a lower bound of total WTP for improved water quality at $181 million, or $15 million per year for 38 years. I also estimate partial costs of aquifer water supply services, based on the present value (PV) of well field development at $13 million, or $1 million per year for 38 years. Due to the unknown degree to which nitrate pollution is a constraint on well field development I am unable to estimate total costs of aquifer water supply services. I conclude that if nitrate remediation and/or nitrate management practices can secure the provision of improved ground water quality, and the costs of nitrate management fall below $14 million per year for 38 years there is a potential for net benefits to the City of Abbotsford.

The valuation methodology used in this thesis rests on the assumption that nitrate contamination constrains further well field development. To validate and render more accurate results further study is needed. Hydrogeological study objectives would be to: 1). determine sustainable rates of extraction throughout the aquifer, with respect to seasonal fluctuations in supply and demand, and optimal distribution of wells throughout the surface area of the aquifer and 2). assess drinking water limitations directly attributable to nitrate pollution of the aquifer. Once the optimal restoration level of the aquifer is known more accurate estimates of net benefits will be possible.

It is my hope that the findings of this study are used to shed some light on the economic implications of agricultural nitrates in the Abbotsford-Sumas aquifer and the lack of information regarding their actual (not perceived) impact. If present day nitrate contamination is proven to be what constrains future well field development my estimates suggest that an EGS program which incentivizes improved nitrate management practices warrants a closer look.

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Appendix 1: Lower Mainland – Southwest Nutrient Balance, 2005

Methodology

The methodology for this nutrient balance is based on a report produced by Timmenga & Associates Inc. in 2003, entitled "Evaluation of Options for Fraser Valley Poultry Manure Utilization". This report was commissioned by the Sustainable Poultry Farming Group, a government formed industry organization which includes representatives from the Broiler Hatching Egg Producers' Association, BC Chicken Growers Association, BC Turkey Association and the Fraser Valley Egg Producers' Association. Dr. Hubert Timmenga is a professional agrologist who has worked as an applied scientist and consultant since 1981. Timmenga (2003) does a thorough treatment of both animal manure and crop nutrient uptake estimates, taking into consideration local conditions, and includes three nutrients pertinent to agro-ecological issues: nitrogen, phosphorous and potassium. Nitrogen has been a long-time focus, but phosphorous is becoming more and more of a concern (Kowalenko, G., pers. comm., 2009). The Timmenga (2003) method is accessible when field data collection is not feasible.

First, the crop nutrient demand is calculated for the region and then livestock manure production. The demand and supply are compared to arrive at the nutrient balance.

Data

Data is from the 2001 and 2006 Agricultural Census for the Lower Mainland – Southwest Census Agricultural Region (CAR 590200000) of British Columbia. The actual production years of the nutrient balance are 2000 and 2005 and will be referred to as such in the text and tables that follow. The Lower Mainland - Southwest area includes Census Divisions 9, 15, 29 and 31, the Fraser Valley, Greater Vancouver, Sunshine Coast, and Squamish-Lillooet respectively. Further data requirements for each part of the nutrient budget are explained in their respective sections.

Nutrient Demand

To calculate the crop nutrient demand of animal manure in the Lower Mainland – Southwest, crops are included if in current practice animal manure is used as is the practice by Timmenga (2003). For example, greenhouse vegetable operations are not included in the nutrient demand calculations. Lower Mainland greenhouses are typically "fertigated" meaning they are fertilized through irrigation systems using liquid chemicals. Suspended solids in animal manure even with high moisture content, such as dairy manure, cause blockages in fine irrigation heads. There are two organically certified greenhouses in the region who purportedly use liquid fish fertilizer rather than animal manure further supporting the notion that the use of animal manure in greenhouses is not currently practicable (Timmenga, pers. comm., Oct. 2009).

Muhroom production is also omitted from this balance. The large mushroom industry in the Lower Mainland does use local poultry manure however it is returned to the land base after being used as a growing substrate. The residual substrate, or mushroom manure, has its own fertilizer value and should be accounted for in a regional nutrient balance. It is assumed here, for simplicity and to use the methodology developed by Timmenga (2003), that mushroom manure has little changed nutrient value from its original poulty manure composition so that the mushroom industry can be side-stepped in the analysis. Mushroom acreage is not included in the nutrient demand.

Raspberries are the only berry crop used in the final nutrient balance. Currently chicken manure is widely used in raspberry production. Blueberries, cranberries, and strawberries are not included because animal manure is not used to fertilize these crops in current practice (Timmenga, H., pers. comm., 2009). Grapes are reported in Table 1, as they are in Timmenga's Table 3, but are not included in Timmenga's final nutrient balance since the acreage is negligible. Area of Christmas tree production is also omitted since it is negligible.

Lastly, as per Timmenga (2003), grain crops are omitted from the final nutrient balance since they are seeded in the winter, a time when manure application is restricted. Pulses and legumes as well as green manures are also not included in the nutrient demand in Timmenga (2003). These crops add nitrogen to the soil through fixation and so are typically included in the calculation of crop nutrient supply. More about leguminous crop nutrient deposition in the nutrient supply section.

Table 1 shows Lower Mainland – Southwest crop acreage for 2000, as reported in Timmenga (2003) for comparison sake, and for 2005.

Table 1. Lower Mainland – Southwest Crops 2000 & 2005, in Acres

Selected Crops	**2000**	**2005**
Wheat	1,718	2,029
Oats	1,420	1,395
Barley	1,248	1,538
Mixed Grains	x	286
Corn for Grain	1,696	648
Rye	683	1,192
Subtotal Grains	*6,765*	*7,088*
Corn for Silage	18,348	21,292
Alfalfa & Alfalfa Mixtures	6,458	12,013
Other Tame Hay and Fodder Crops	62,612	54,666
Subtotal Field Crops	*87,418*	*87,971*
Potatoes	6,005	6,374
Tree Fruits*	1,661	1,378
Raspberries	5,269	4,798
Grapes	210	120
Subtotal Fruits & Berries	*5,479*	*4,918*
Field Vegetables	14,135	13,315
Nursery & Sod	7,795	8,576
Improved Land for Pasture and Grazing	19,092	20,098
Total	**248,012**	**249,695**
*includes 'other fruits, berries, and nuts total area' census category		

The coefficients used to calculate crop nutrient demand in the Lower Mainland, reported in Table 3, were drawn from Timmenga (2003), using the nutrient demand values reported in their Table 5 against the actual acreage reported by the 2001 Census.[9] Timmenga & Assoc. consulted BC Ministry of Agriculture and Lands (BCMAL) published recommendations for nitrogen, phosphorous and potassium needs of crops grown in the region.

To verify the suitability of these coefficients against most recent BCMAL recommendations, Table 2 compares the coefficients used in Timmenga (2003) with recommendations made in the BC

9 There are mathematical errors in the totals reported in Table 5 of Timmenga (2003). These do not affect the crop demand coefficients.

Environmental Farm Plan (EFP) Resource Guide (2005). In each case, where EFP Resource Guide recommendations exist the Timmenga (2003) coefficient, here reported in kg/ha, lies either below or within the range recommended by the EFP Reference Guide. The EFP Reference Guide does not include recommendations on the annual uptake of crops of Phosphorous or Potassium. Other nutrient demand recommendations are available from disparate sources and for specific crops, for example by the Oregon State University Extension Service. These recommendations refer to different soils, to particular crop varieties and do not make explicit recommendations for applications of Phosphorous and Potassium. For these reasons, the coefficients determined by Timmenga (2003) are used, since they are within ranges recommended in the EFP Reference Guide, were developed for an analysis of the Lower Mainland – Southwest and make possible a nutrient balance including phosphorous and potassium.

It is important to note that the coefficients used by Timmenga & Assoc. assume poor soil quality and that there are no residual nutrients in the soil. Some regions in the Lower Mainland - Southwest have better than poor soil quality and typically there are nutrients that remain in the soil from year to year. These assumptions will tend to overstate the nutrient demand that actually exists, and understate the surplus. Table 4 reports the crop nutrient demand for the Lower Mainland – Southwest, 2006.

Table 2. Evaluating Timmenga (2003) Coefficients against BC Environmental Farm Plan recommendations for Nitrogen

Crop	N (kg/ha) Timmenga	N (kg/ha) BC – EFP	Timmenga within range?
Silage Corn	128.62	190-250	no, under
Alfalfa	67.73		
Tame Hay	392.53	260-400*	yes
Tree Fruits & Nuts	123.48		
Berries	61.79	50-100**	yes
Vegetables	168.18	80-185**	yes
Nursery & Sod	56.11		
Potatoes	67.49	80-185	no, under
Pasture	67.3		
* listed as Perennial Grass in Table 6.9 of EFP Reference Guide			
** maximum nitrogen application depends on type of berry or vegetable			

Table 3. Nutrient Demand Coefficients, Timmenga (2003)

Crop	N	P2O5	K2O
Silage Corn	0.05	0.05	0.08
Alfalfa	0.03	0.08	0.08
Tame Hay	0.16	0.06	0.08
Orchards	0.05	0.08	0.11
Raspberries	0.03	0.03	0.04
Vegetables	0.07	0.09	0.09
Nursery	0.02	0.05	0.05
Potatoes	0.03	0.06	0.06
Pasture	0.03	0.03	0.03

Table 4. Lower Mainland – Southwest Crop Nutrient Demand 2006

Crops	Acres	N (tonnes)	P205 (tonnes)	K2O (tonnes)
Silage Corn	21292	1108	1061	1639
Alfalfa	12013	329	934	934
Tame Hay	54666	8684	3473	4218
Orchards	1378	69	106	156
Raspberries	4798	125	120	196
Vegetables	13315	906	1209	1209
Nursery & Sod	8576	195	389	389
Potatoes	6374	174	405	405
Pasture	20098	547	547	547
Total	**142510**	**12137**	**8245**	**9693**

* excluding grain crops as these are mostly winter seeded and do not receive manure, most berry crops, christmas trees, pulses & legumes, green manures.

Nutrient Supply

Timmenga (2003), whose purpose is to investigate the potential for poultry manure use in the Lower Mainland, uses data provided by the Sustainable Poultry Farming Group for poultry output estimates. These numbers are perceived to be the best available as they are based on actual manure measurements from the industry itself, but are not publicly available. The dairy, hog and beef production outputs are estimated using census data and literature based coefficients, and are therefore less robust. The manure production coefficients are derived from a mixture of BCMAL guidelines and University of North Carolina coefficients, as per Timmenga (2003), depending on availability of relevant coefficients to the production systems common in the Lower Mainland.

To attain the 2005 nutrient supply, growth rates for each sector of the livestock industry were calculated and applied to the manure outputs reported in Timmenga (2003). Growth rates are reported in Table 5 below. For the four poultry sectors, industry reported outputs by the BC Chicken Marketing Board, BC Egg Producers, BC Turkey Marketing Board and the BC Broiler Hatching Egg Commission, are used to determine growth rates. The growth rates of the dairy, hog and beef sectors are determined using agricultural census livestock head counts. Table 6 reports the nutrient coefficients used in Timmenga (2003), and subsequently here. Table 7 reports the 2005 nutrient supply in terms of manure production and nutrient output.

Atmospheric and biological deposition of nutrients are not estimated in this nutrient budget. AAFC, in their residual nitrogen indicator and using the standardized Canadian Agricultural Nitrogen Budget model (CANB Version 2.0), include leguminous crop nitrogen deposition, based on soil types across Canada. These factors are beyond the scope of this nutrient balance. Here, atmospheric and biological deposition is omitted from the nutrient balance, leading to an underestimation in the final balance.

Note, additional nutrients applied through chemical fertilizer have also not been considered here as per Timmenga (2003) because the interest here is the nutrient surplus created by livestock intensity. Furthermore, the information is not readily available. Fertilizer application data is not available. AAFC, in their residual nitrogen indicator, use fertilizer sales as a proxy weighted by provincial application recommendations.

Table 5. Livestock Sector Growth Rates 2000 - 2005

	2000	2005	Growth Rate
Chicken, live weight, millions kg	186	210	12.90%
Turkeys, eviscerated, millions kg	16.7	19.4	16.17%
Commercial Egg, millions dozen eggs	55.2	52	-5.80%
Hatching Egg, millions of hatching eggs	104.9	91.3	-12.96%
Dairy, head	49,755	52,355	5.23%
Hog, Growers & Finishers head	73,859	50,238	-31.98%
Beef, head	10,014	9,783	-2.31%

Table 6. Nutrient and Moisture Content of Animal Manure, Timmenga (2003)

	N	**P2O5**	**K2O**	**Moisture**
Poultry Manure	3.00%	2.70%	1.45%	31.74%
Dairy Manure	0.34%	0.24%	0.52%	90.00%
Hog Manure	0.29%	0.21%	0.16%	90.00%
Beef Manure	0.46%	0.52%	0.89%	65.00%

Table 7. Manure Production and Nutrient Supply, 2005 (tonnes)

	Manure	**Nitrogen**	**P2O5**	**K2O**	**Moisture**
Poultry Manure	261,839	7,855	7,070	3,797	83,108
Dairy Manure	2,325,696	7,907	5,582	12,094	2,093,127
Hog Manure	349,467	1,013	734	559	314,520
Beef Manure	87,766	404	456	781	57,048
Total	**3,024,768**	**17,180**	**13,842**	**17,231**	**2,547,802**

Results

Table 8 reports the nutrient balance for the production year of 2005. The regional supply of each nutrient surpasses the nutrient requirements of crops that can benefit from livestock manure.

Table 8. Nutrient Balance for the Lower Mainland – Southwest, 2005, in tonnes

	Manure	Nitrogen	P2O5	K2O
Poultry Manure	261839	7855	7070	3797
Dairy Manure	2325696	7907	5582	12094
Hog Manure	349467	1013	734	559
Beef Manure	87766	404	456	781
Total	**3024768**	**17180**	**13842**	**17231**
Crops	**Acres**	**Nitrogen**	**P2O5**	**K2O**
Silage Corn	21292	1108	1061	1639
Alfalfa	12013	329	934	934
Tame Hay	54666	8684	3473	4218
Orchards	1378	69	106	156
Raspberries	4798	125	120	196
Vegetables	13315	906	1209	1209
Nursery & Sod	8576	195	389	389
Potatoes	6374	174	405	405
Pasture	20098	547	547	547
Total	**142510**	**12137**	**8245**	**9693**
Surplus		***5042***	***5597***	***7537***

A nutrient surplus is in keeping with the findings and forecasts reported in Timmenga (2003), Brisbin (1995), Zebarth (1999), Berka (2001), Chesnaux (2007) and the residual nitrate agri-environmental indicator developed by Agriculture and Agri-Food Canada.

Note, nutrients produced in the Lower Mainland – Southwest are not necessarily applied in the local area. For example chicken manure is collected, processed and exported to markets in the Fraser Valley, BC interior and the United States. This was an initiative started in 2004 to decrease the nutrient load in the Lower Fraser Valley and is largely done to reduce the pressure on the Abbotsford aquifer as there have been and continue to be water contamination issues (Paulson, S., pers. comm., 2009).

Some studies estimate the nutrient removal rate at 30% (Chesnaux, 2007). This would set the manure Nitrogen surplus at 3,529 tonnes. Table 9 finds the growth rate of the manure based nutrient surplus assuming a constant use of fertilizer and projects nutrient supply for 2010 at 4,447 tonnes.

Table 9. Nutrient Balance Increase for the Lower Mainland – Southwest, 2005, in tonnes

	2000 (Timmenga, 2003)	2005	Change	2010 Projection
Manure N Surplus	2,801	3,529	+26%	4,447

Appendix 2: Public Utility Expansion Model Optimization

We can simplify the first order condition to:

$$\Theta Z(k) = Z'(k)k$$

Recall that $Z(k) = (1+p)^{\frac{\ln(k)}{g}} - 1$. In general if $y = A^{f(x)}$ then $\frac{dy}{dx} = \ln(A) A^{f(x)} f'(x)$. In our case $Z'(k) = \ln(1+p)(1+p)^{\frac{1}{g}\ln(k)} \frac{1}{gk}$. We rewrite this as $Z'(k) = \ln(1+p)\frac{[Z(k)+1]}{gk}$ and substitute this expression into the first order condition simplified derivative $\Theta Z(k) = Z'(k)k$:

$$g\,\Theta Z(k) = \ln(1+p)[Z(k)+1].$$

Next we solve for $Z(k)$:

$$Z(k)[g\Theta - \ln(1+p)] = \ln(1+p)$$

$$Z(k) = \frac{\ln(1+p)}{g\Theta - \ln(1+p)}.$$

Substitute $Z(k) = (1+p)^{\frac{\ln(k)}{g}} - 1$ into the first order condition and the expression can be written as $(1+p)^{\frac{\ln(k)}{g}} - 1 = \frac{\ln(1+p)}{g\Theta - \ln(1+p)}$ and finally $(1+p)^{\frac{\ln(k)}{g}} = \ln(1+p) + g\Theta - \ln\frac{(1+p)}{g\Theta - \ln(1+p)}$.

Log both sides $\frac{1}{g}\ln(k)\ln(1+p) = \ln\left\langle\frac{g\Theta}{g\Theta - \ln(1+p)}\right\rangle$ and simplify to $\ln(k) = g\ln\left\langle\frac{g\Theta}{g\Theta - \ln(1+p)}\right\rangle\frac{1}{\ln(1+p)}$. We exponentiate $k^* = e^{g\ln\left\langle\frac{g\Theta}{g\Theta - \ln(1+p)}\right\rangle\frac{1}{\ln(1+p)}}$ and find the solution:

$$k^* = \left\langle\frac{g\Theta}{g\Theta - \ln(1+p)}\right\rangle^{\frac{g}{\ln(1+p)}}.$$