Preliminary Damage Estimates for Selected Invasive Fauna in B.C.

Preliminary Damage Estimates for Selected Invasive Fauna in B.C.

Prepared for

Ecosystems Branch
B.C. Ministry of Environment
Victoria, B.C

Prepared by

Donald C.E. Robinson, ESSA Technologies
Duncan Knowler, Simon Fraser University
David Kyobe, Simon Fraser University
Patricia de la Cueva Bueno, ESSA Technologies

ESSA Technologies Ltd.
600 – 2695 Granville Street
Vancouver, B.C. V6J V6H 3H4

March 2013

Citation: **Robinson, D.C.E., Knowler, D., Kyobe, D. and P. de la Cueva Bueno.** 2013. Preliminary damage estimates for selected invasive fauna in B.C. Report prepared for Ecosystems Branch, B.C. Ministry of Environment, Victoria, B.C. by ESSA Technologies Ltd., Vancouver, B.C. 62 p.

Table of Contents

List of Tables 6
List of Figures 7
Executive Summary 8
Acknowledgements 9
1. Introduction 10
1.1 Background 10
1.2 Scope & Objectives 10
1.3 Organization of the Report 11
2. Conceptual Framework 12
2.1 Methods and Approach 12
2.2 Impact Assumptions and Diagrams 12
2.3 Economic Analysis Methodology 14
3. Zebra and Quagga Mussels 17
3.1 Life History 17
3.2 Physiological Limits 18
3.3 Dispersal 19
3.4 Impacts 20
3.5 Potential Control 21
3.6 Unit Economic Damages 22
3.7 Ecological Carrying Capacity 24
3.8 Preliminary Damage Estimate 26
4. European Fire Ant 27
4.1 Life History 27
4.2 Physiological Limits 27
4.3 Dispersal 28
4.4 Impacts 28
4.5 Potential Control 29
4.6 Unit Economic Damages 30
4.7 Ecological Carrying Capacity 30
4.8 Preliminary Damage Estimate 32
5. Asian Carp 33
5.1 Life History 33
5.1.1 Grass Carp 33
5.1.2 Black Carp 34
5.1.3 Bighead and Silver Carp 34
5.2 Physiological Limits 34
5.3 Dispersal 35
5.4 Impacts 37
5.5 Potential Control 39
5.6 Overview of Economic Damages 39
5.7 Potential Damages to the B.C. Economy 39
6. Sitka Black-tailed Deer 41
6.1 Life History 41
6.2 Physiological Limits 41
6.3 Dispersal 41

6.4 Impacts 42
6.5 Potential Control 43
6.6 Overview of Economic Damages 43
6.7 Potential Damages to the B.C. Economy 44

7. European Starling 47
7.1 Life History 47
7.2 Physiological Limits 47
7.3 Dispersal 47
7.4 Impacts 48
7.5 Potential Control 49
7.6 Overview of Economic Damages 49
7.7 Potential Damages to the B.C. Economy 50

8. Recommendations for Decision Makers 51
8.1 General Recommendations 51
8.2 Species Recommendations 51
8.2.1 Zebra and Quagga Mussels 51
8.2.2 European Fire Ant 51
8.2.3 Asian Carp 52
8.2.4 Sitka Black-tailed Deer 52
8.2.5 European Starling 52
8.3 Data Gaps and Future Research 52
8.3.1 Zebra and Quagga Mussels 52
8.3.2 European Fire Ant 53
8.3.3 Asian Carp 53
8.3.4 Sitka Black-tailed Deer 54
8.3.5 European Starling 54

9. References 55

List of Tables

Table 1.1: Species groups considered in this study, and the rationale for choosing each. 11
Table 3.1: Selected unit damage estimates for zebra and quagga mussels in 2012 CAD. FER = Foreign Exchange Rate; CPI = Consumer Price Index..23
Table 3.2: Preliminary damage estimate for B.C. for zebra and quagga mussels in 2012 CAD. 26
Table 4.1: Selected unit damage estimates for European fire ant in 2012 CAD. FER = Foreign Exchange Rate; CPI = Consumer Price Index. 31
Table 4.2: Preliminary damage estimates for European fire ant in 2012 CAD. 32

List of Figures

Figure 2.1: Illustration of a general approach to the economic evaluation of invasive species. The curve shows the date of establishment (when damages begin to occur), extent of invasions now (if any), total damages at each point in time, and eventual ecological carrying capacity. (Source: Frid *et al.* 2009) 14

Figure 3.1: Habitat suitability based on calcium and risk of establishment for zebra mussels in British Columbia. Results for quagga mussels are almost identical. The left panel shows 3 categories of surface water calcium concentration; the right panel shows the corresponding establishment risk. (Thierrault *et al.* 2012) 19

Figure 3.2: Distribution of sightings of quagga and zebra mussels in North America (Source: U.S. Geological Survey. Non-indigenous Aquatic Species Database 2013). 20

Figure 3.3: Impact diagram for zebra and quagga mussels (*D. polymorpha, D. bugensis*). 22

Figure 4.1: Known and potential distribution of the European fire ant (*Myrmica rubra*) in British Columbia (Source: Higgins 2012). 28

Figure 4.2: Impact diagram for the European fire ant (*Myrmica rubra*) 29

Figure 5.1: Potential distribution of Asian carp species in North America, based on climate suitability. Red areas show high suitability. The green and pink dots represent locations from the U.S. Geological Survey registry, where populations of Asian carp have been established or reported. Top left: grass carp (*C. idella*); top right: silver carp (*H. molitrix*); bottom left: bighead carp (*H. nobilis*); bottom right: black carp (*M. piceus*). (Source: Herborg *et al.* 2007) 35

Figure 5.2: Distribution of grass carp sightings in Canada. (Source: Cudmore and Mandrak 2004) 36

Figure 5.3: Observations of big head carp in Canada. (Source: Mandrak and Cudmore 2004) 36

Figure 5.4: Distribution of Asian carp species in the United Sates: Top left: grass carp (*C. idella*); top right: silver carp (*H. molitrix*); bottom left: bighead carp (*H. nobilis*); bottom right: black carp (*M. piceus*). (Source: USGS 2013) 37

Figure 5.5: Impact diagram for 4 species of Asian carp. 38

Figure 6.1: Distribution of Sitka black-tailed deer (Source: Bunnell 1990) 42

Figure 6.2: Impact diagram for Sitka black-tailed deer (*Odocoileus hemionus sitkensis).* 43

Figure 7.1: Distribution of European starling (*Sturnus vulgaris*) (Source: Cornell Lab of Ornithology 2012)....48

Figure 7.2: Impact diagram for the European starling (*Sturnus vulgaris*). 49

Executive Summary

There may be as many as 1,000 introduced and native invasive species in B.C. Most introduced species do not result in a pervasive cascade of ecological changes, but there is the potential for significant impacts when a novel species is able to successfully enter and thrive within an ecosystem, replacing endemic species and possibly altering pre-existing trophic relationships. When significant ecosystem changes take place, there is the added potential of direct impacts to human populations, as well as impacts upon the ecosystem services which human communities may depend upon.

The goal of the project was to begin to frame an ecological and economic analysis of invasive animal species; selecting case studies which represent different taxonomic groups and a range of issues (see Section 1.2). Preliminary impact hypotheses (*e.g.* Figure 4.2) were created for 5 invasive groups: zebra and quagga mussels, European fire ant, Asian carp, Sitka black-tailed deer and European starling; but special quantitative emphasis was placed on the first two groups, since they were expected to provide the strongest foundation of recent empirical economic research. Zebra mussel and quagga mussel are closely related species which have not yet been observed in B.C., while European fire ant is now found in some locations, mostly in southwestern B.C.

If zebra and quagga mussels eventually occupy all water systems where they could physically survive and thrive, zebra and quagga mussel are estimated to cause annual damages of 2012 CAD 21.7 million (Table 3.2). Of this amount, 57% is due to damages to recreational boaters, 38% through damages to water supplies and 29% through damages to power generation infrastructure.

In the case of European fire ant; if they are to eventually occupy all areas where they could physically survive and thrive, this species is estimated to result in annual damages of 2012 CAD $100 million (Table 4.2). Of this amount, 85% is due to damages to households, with golf courses and schools comprising 10% and 5% of damages, respectively. On a per-unit basis however (Table 4.1), annual damage per household is under $150, while damage per golf course is over $62,000.

General and specific recommendations were also considered as part of this report. In the area of general recommendations there is a need for more primary research in the valuation of damages from Sitka black-tailed deer, European starling and European fire ant. This information is currently lacking, preventing a full-scale analysis. More detailed recommendations were also proposed for each of the five species groups, to address ecological and economic knowledge gaps. These recommendations can be found in Section 8.

Acknowledgements

We express our great appreciation to the following individuals, who generously provided us with their expertise in the course of this study,

Zebra and quagga mussel	
Kevin Aquino	Media Relations & Issues Management BC Hydro
Steven Bourne	Natural Resources Specialist, FrontCounter BC South Coast Region, Ministry of Forests, Lands and Natural Resource Operations
Mike McArthur	Natural Resource Specialist – Water License Requirements BC Hydro
Christopher Seeley	Transport Canada
Sheila Street	FortisBC Environmental Program Lead
European fire ant	
Robert J. Higgins	Department of Biological Sciences Thompson Rivers University
Magda Hogewoning	Executive Assistant Federation of Independent School Associations in BC
Asian carp	
Ron Brooks	Fisheries Division Director Kentucky Department of Fish and Wildlife Resources
Becky Cudmore	Manager Centre of Expertise for Aquatic Risk Assessment Fisheries and Oceans Canada
Kevin S. Irons	Aquaculture and Aquatic Nuisance Species Program Manager Illinois Department of Natural Resources
Kaitlin Steiger-Meister	External Affairs, Public Affairs Specialist Great Lakes Restoration Initiative & Asian Carp U.S. Fish and Wildlife Service, Midwest Region
Sitka black-tailed deer	
Ken Lertzman	Professor & Director: The Hakai Network for Coastal People Ecosystems and Management School of Resource & Environmental Management Faculty of Environment, Simon Fraser University
Sean Muise, RPF	Stewardship Forester, Haida Gwaii District
Tyler Peet	Resource Manager, Authorizations Ministry of Forests, Lands, Natural Resource Operations Haida Gwaii Natural Resource District
European starling	
Connie Bielert	Administration Services BC Grapegrowers' Association
Amanda Doerksen	BC Blueberry Council
Mark Sweeney	Berry Industry Specialist BC Ministry of Agriculture
Douglas Ransome	Fish Wildlife and Recreation Technology British Columbia Institute of Technology

1. Introduction

1.1 Background

A recent review notes that in British Columbia the number of introduced and native invasive species may be as large as 1,000 (Voller and McNay 2007), spanning all phyla including viruses. Most introduced species do not result in a pervasive cascade of ecological changes, but there is the potential for significant impacts when a novel species is able to successfully enter and thrive within an ecosystem, replacing endemic species and possibly altering pre-existing trophic relationships. When significant ecosystem changes take place, there is the added potential of direct impacts to human populations, as well as impacts upon the ecosystem services which human communities may depend upon.

The interactions which result in significant ecosystem and economic impacts are often not well understood in advance, making management and mitigation of these risks very challenging. One useful rule of thumb is to review species that have been shown to be invasive in ecosystems that are broadly similar to those in B.C. and to focus attention on how lessons learned in those jurisdictions might be applied locally.

1.2 Scope & Objectives

The goal of the project was to begin to frame an ecological and economic analysis of invasive animal species, selecting case studies which represent different taxonomic groups and a range of issues: species that are native to B.C. but have been introduced and are invasive in other parts; species that are already present but possibly at low enough levels that mitigation might be practical; species that are not yet present in B.C. which have a high potential for impact; and species that make good cautionary tales of what to avoid.

Ordinarily, a study like this would begin with an iterative process to identify and prioritize invasive species. Selection and evaluation would be founded on factors like:

- prior knowledge of the possible geographic extent of each species, based on physiological and ecological constraints;
- the depth of understanding of their life history, including dispersal mechanisms;
- ecosystem dynamics and control methods;
- the range of management options and costs; and
- current impacts in B.C. or other locations, and the possible extent and significance of ecological and economic impacts.

However, given the limited resources and time for this step, we selected zebra and quagga mussels and European fire ant (see Table 1.1) for a more in-depth analysis. These two groups were selected for special attention because of our prior expectation that zebra mussel (*Dreissena polymorpha*) and red imported fire ant (*Solenopsis invicta)* – a closely related ally of the European fire ant – would likely have already received considerable scrutiny in other locations within North America and would therefore be the best source of economic and ecological literature. To cover the broad range of goals, we also selected 3 other species (lumping four carp species into one group) for a more qualitative analysis of the ecological and economic impact. With these constraints in mind, this study examines the potential economic and ecological of five invasive animal groups shown in Table 1.1.

Table 1.1: Species groups considered in this study, and the rationale for choosing each.

Common name	Scientific name	Rationale
Quagga, zebra mussel	*Dreissena polymorpha,* *Dreissena bugensis* *(= D. rostriformis bugensis)*	High risk of introduction with potential impacts to infrastructure, ecosystem structure and function and recreational fisheries. Biology is similar to zebra mussel, which has an extensive literature.
European fire ant	*Myrmica rubra*	Currently in the Lower Mainland and Vancouver island; they share some characteristics with other invasive fire ants
Asian carp Bighead carp Black carp Grass carp Silver carp	 *Hypophthalmichthys nobilis* *Mylopharyngodon piceus* *Ctenopharyngodon idella* *H. molitrix*	Not yet present, but high potential impact if introduced, a reasonably good base in the literature
Sitka black-tailed deer	*Odocoileus hemionus sitkensis*	A native invasive, endemic to mainland B.C. and introduced on to Haida Gwaii, where they alter forest structure (including recruitment of western redcedar) which cascades to ecosystem function
European starling	*Sturnus vulgaris*	Widespread in B.C., with impacts to agriculture; a possible precautionary tale for a species that has become firmly established

1.3 Organization of the Report

This report is based upon a repeating sequence of analytical steps for each species. In general, the pattern we follow is based on our review of the available literature and describes what is known of the life history, followed by the known physiological limits which may be relevant to adult life, reproduction and dispersal. Each ecological description is accompanied by an Impact Diagram (for example, see Figure 3.3); each of which is a set of hypotheses proposing possible linkages between the invasive species and a range of human/economic components, such as fisheries, recreation, water supply and ecosystem services. The goal of the Impact Diagrams is to begin to quantify the way in which each species ultimately affects a valued component. We do not present the diagrams as proven, but rather as plausible hypotheses: a way to provoke discussion about mechanisms of interaction, management options and possibly as a framework for eventually testing management options through simulation or experiments.

Our presentation of the life history and Impact Diagrams is followed by an analysis of the economic impacts which might be expected from the species. For those species that are not analysed in depth, this part of the analysis is given a more cursory treatment. For quagga mussel and European fire ant, we have carried out a comprehensive review of the economic literature available to us, based on a broad survey from a variety of jurisdictions; applying those data and costs to British Columbia in a way that we think is reasonable. Further details of the economic analysis and rationale are found in Section 2.3.

The report concludes with a brief set of recommendations and identification of key data gaps identified during the study, and possible areas of further study for each of the five invasive species or species groups.

2. Conceptual Framework

2.1 Methods and Approach

Several approaches can be used in establishing damages from the invasive species in question. Ideally, we would analyze what would happen if there was no control of the invasive species. Thus, we would assess the probable dispersal and damages of the species on the assumption that this occurs without human interference. The analysis is hypothetical, since most of the invasive species in question have either not invaded B.C. yet or if present in B.C., have received some measure of control or treatment to inhibit their spread (or elsewhere, if we have used damage estimates from another location and transferred these to B.C.). The advantage of estimating the "no control" level of damages is that these values can be used later in determining the net benefits of proposed management and control programs in a more formal cost-benefit analysis where we would compare the "no control" and "control" levels of damages. Unfortunately, the "no control" approach is not always possible since few such estimates exist and if they do not, then sophisticated models are required to generate them (*e.g.,* see Frid *et al.* 2009).

Instead, many of the estimates that we were required to use involve some degree of control (with associated costs) and, therefore, consist of these control costs plus the residual damages remaining after control actions. Such estimates might be described as a "total damage cost" to society, assuming at least some action has been taken to control the invasive species but not all damages are eradicated. This type of estimate cannot be used directly in a cost-benefit analysis (since it includes program costs for control). As a result, care should be taken with interpreting some of the damage estimates and in considering them for use in management program evaluations.

In addition, this study is preliminary and does not take the analysis to its full and complete endpoint, nor are all potential damages from each invasive species quantified. Information on the ecological attributes for each species and their possible impacts were drawn from the literature. The economic analysis of damages considers those impacts for which information was available or could be transferred from the literature; generally these were impacts related to commercial activities (*e.g.* forestry, fishing, agriculture) or households (*e.g.* health, property damage, recreation). But most invasive species can have significant, even profound, effects on local ecosystems and their flora and fauna, as well as on the resilience of these natural systems and their structure and biodiversity. However, no estimates were available related to the ecosystem impacts of the invaders so this remains an area for further research.

Finally, a full and complete economic analysis would examine the dispersal and establishment of the invasive species over time, considering the level of damages at each point as it disperses. This allows the analyst to generate a value that better captures the concept of total damages that can be used in a cost-benefit analysis (see below). Instead, our analysis presents a single point estimate for damages that assumes full establishment or occupation of the invasive species' potential range within B.C; it is an annual value expressed at full dispersal of the invasive species within its ecological carrying capacity. Further clarification of this point is provided in the next section.

2.2 Impact Assumptions and Diagrams

We structured the analysis to capture the impact of each species in two ways. First, impact diagrams were used to illustrate the various impact pathways through which the chosen species affects or could affect the ecological, social and economic environment in B.C. These diagrams provide a simple visual tool for summarizing the current state of knowledge regarding how these invasive species affect the resource and tourism industries, environmental attributes and human health; as well as any other impacts. Even though we were not able to generate quantitative estimates for the potential damages from the secondary list of invasive species we considered (Sitka black-tailed deer, Asian carp and European starling), we provide the Impact Diagrams for these species to aid future damage valuation efforts.

The economic damages for each species would normally be estimated using the impacts outlined in the Impact Diagrams as a starting point. However, to bring an economic perspective to the analysis, it should be noted that these impacts can be "mapped" on to a suitable valuation taxonomy such as Total Economic Value (TEV; Pearce and Turner 1990) or the Millennium Ecosystem Assessment classification of ecosystem services (Millennium Ecosystem Assessment 2005), allowing the use of standard welfare analysis to calculate economic values. TEV distinguishes between use and non-use values, the former being somewhat self-explanatory and the latter referring to values that rely merely on the continued existence of a species or ecosystem and are unrelated to use. Use values are grouped according to whether they are direct or indirect values (ecosystem services). The MEA classification considers provisioning, regulating, supporting and cultural services from ecosystems. For this analysis we do not consider indirect use or non-use values (*i.e.* regulating, supporting and cultural values using the MEA approach), because of the difficulties in obtaining such information and because the coverage of indirect use or ecosystem service values is very limited at this time. In contrast, we are able to do much more with respect to direct use or provisioning values (*e.g.* power generation, agricultural and forestry values). Thus, in the end our estimated economic damages consist of only some of the impacts outlined in the diagrams, all of which comprise direct use (provisioning) values.

In a more advanced analysis (see above), we next would estimate logistic curves[1] showing "area invaded" and "economic damages" for each of the species at each point in time. In explaining this approach we can better situate our more limited analysis and provide guidance for what would be needed to take our analysis to the next level. This "logistic curve" approach assumes that the area invaded and economic damages initially increase exponentially over time until they approach an ecological carrying capacity (Figure 2.1). In keeping with the use of a logistic model, our damage curves would consider three stages in the dispersal of the invasive species. At the first stage, the area invaded and economic damages increase relatively slowly. At the second stage, the rate of dispersal increases and the invasion proceeds more quickly. At the third stage, dispersal begins to slow as the area invaded approaches the carrying capacity for the invasive species. Economic damages along the curves are established by taking the area invaded in each year and multiplying this figure by an estimate of the damages per hectare per year.

A model of dispersal to project damages of invasive species into the future is required for this analysis, including such key ecological parameters as the intrinsic rate of growth. Figure 2.1 illustrates how the logistic curve approach can be used to depict past damages experienced by society due to the invasive species, potential future damages under alternative management strategies and uncertainties associated with these measurements. A key point in understanding our damage estimates in this study is that they represent an end point estimate of the damages at the ecological carrying capacity or limit and, therefore, constitute a maximum value over the period of dispersion of the invasive species. In reality, most of the invasive species we consider in this study have not yet invaded B.C. (or only minimally) and their current level of damage would be near zero and situated close to the origin along the damage curve in Figure 2.1

It should be noted that extensive raw data on the dispersion and economic damages for invasive species are not readily available. Thus, the estimates presented in this report provide a somewhat incomplete picture in that only a selected set of damage components could be assessed, and data on the rate and extent of dispersion are estimated from the literature rather than derived from field observations. Carrying

[1] The validity of our argument does not depend on this particular equation form. Other cumulative relationships could be applied besides the logistic function.

out our analysis required assumptions about the underlying economic and ecological carrying capacity conditions. These assumptions are described in the next section.

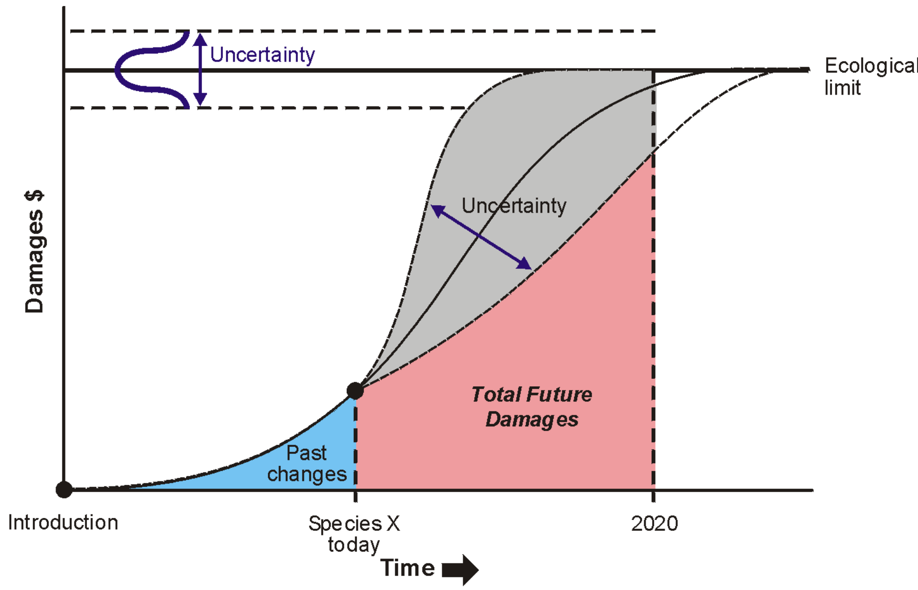

Figure 2.1: Illustration of a general approach to the economic evaluation of invasive species. The curve shows the date of establishment (when damages begin to occur), extent of invasions now (if any), total damages at each point in time, and eventual ecological carrying capacity. (Source: Frid *et al.* 2009)

2.3 Economic Analysis Methodology

Economic analysis is the main analytical tool employed in this study, and in this section we outline the basic approach and methodology. Economists have developed a systematic approach for assessing whether projects or activities are worthwhile from society's standpoint, and in its fullest form this is known as cost-benefit analysis (CBA). However, it is not the purpose here to carry out comprehensive CBAs of the selected management approaches for each of the invasive species in our study. Cost-benefit analyses involve calculation of the present values of benefits and costs and taking the difference between the two, the net present value (NPV), as an indicator of an action's viability in economic terms. As noted above, a full CBA would consider the distribution of the benefits and costs from invasive species management strategies, in addition to the impacts on aggregate welfare. Instead, our analysis employs components of CBA in a more limited form. This is admittedly a more limited task than a full CBA or project appraisal, but still benefits from discussion of some basic economic concepts.

Economists use a "with/without" criteria to refer to the case with and without a given project or management intervention. What this means for economic analysis involving invasive species is that we must calculate the damages from a scenario where no management intervention is made and compare this to one where we calculate the damages with the management intervention in place. This approach provides the measure of benefits attributable to the management intervention in isolation. From this measure of damages avoided from the intervention we subtract the incremental costs to society, or opportunity costs, of the resources employed in the management intervention. These opportunity costs include investment costs (*e.g.* screening or filtering equipment) and field operations costs (*e.g.* mechanical removal of invasive aquatic species). In this study we generate only limited information for use in a CBA; specifically, we try to provide an estimate of the baseline damages prior to management or control of the invasive, but with some exceptions, as noted in a previous section.

As also highlighted in the previous section, the long gestation period during which invasive species disperse and expand their range (and are often inconspicuous due to low population density) raises the issue of time and discounting. When economists evaluate benefits and costs that extend over more than one time period they take this into account with a discount rate. The discount rate is used to weight benefits and costs occurring in different time periods, similar to the use of an interest rate to calculate interest payable on bank accounts. Since we would prefer having a sum of money in the present, compared to waiting until a later time period for it, we place a greater emphasis (weight) on current values than on ones in distant periods. To accomplish this, we use a discount factor which incorporates the selected discount rate. Weighting a series of future benefits or costs and summing these yields a present value.

As discussed above, we do not carry out discounting in this study but we do use an amortization procedure to place investment costs on a consistent basis with annual control costs for some damage estimates (*e.g.* power plants and quagga or zebra mussels). For these calculations the challenge arises in selecting an appropriate discount rate for discounting or amortization. Since there is little consensus among economists regarding the correct value of the social discount rate, we considered a baseline value of 3.5 percent and an alternative of 7 percent.

We recognized that some values are "non-market" values, meaning that market prices would not exist. In these cases we relied on non-market valuation estimates, primarily from Eastern Canadian or U.S. studies, and converted the estimates to the present B.C. setting using foreign exchange rates (if needed) and price indices. Such an approach is referred to as "benefits transfer," whereby valuation results are used from a site located elsewhere (with appropriate modification). There are now accepted protocols for doing benefits transfer, such as preferring to transfer a functional relationship, rather than a single value (which we were unable to do here). Ideally, to be consistent with economic theory we would present all values as consumers or producers surplus but this was not possible in cases where damages are reported using the "total damage cost" approach (see above). We avoid using changes in property values and do not present this as a separate category of loss since in many cases reduced property values simply reflect the capitalization of direct damages to individuals into the value of property. Thus, double-counting of losses can occur if care is not taken.

Various ecological and physical assumptions were required to estimate the potential impacts and ecological limit for our two main species: quagga mussel and European fire ant. These were derived from secondary data and after consulting with experts. Primarily, we worked from information about areas of potential high risk for invasion from various risk assessments and data generated by experts. This information defined the geographical limits for invasion and we then proceeded to determine the main impact targets for damage by the species within this geographical zone (*e.g.* schools, golf courses, power plants, *etc.*). By accessing regional or provincial data bases, or consulting with experts, we were able to determine the numbers of physical structures, households, land area or other impact targets that coincided with our unit economic damage values. Multiplying these two sets of values provided the estimate of total damages (or total damage costs) reported in the sections for our two main invasive species.

The economic information used in the damage calculations for each of these two invasive species are presented in a set of two tables in each section. The first provides the estimation procedure for the unit damage value, including any adjustments for consistency, updating, foreign exchange, *etc.* We followed a standardized step-wise procedure to place all estimates on a comparable basis in 2012 Canadian dollars ($2012 CAD), regardless of the origin (Canada or USA) or year of the original estimate from the literature. First, we converted the value (if not in $ CAD) to Canadian prices using the foreign exchange rate (FER) for that year. Then we updated the converted estimate in Canadian dollars to 2012 prices using the Canadian consumer price index (CPI).

This unit damage value is then carried forward to the second table where we present the final damage estimate consisting of the unit damage value multiplied by the physical units affected at the ecological carrying capacity. The final aggregate estimate therefore represents the point estimate for damages assuming the invasive species has dispersed to its full ecological limit. All assumptions and data sources are listed as footnotes below these tables. Finally, all financial values in the economic analysis are presented in Canadian dollars at 2012 prices.

3. Zebra and Quagga Mussels

The zebra and quagga mussel (*Dreissena polymorpha* and *Dreissena bugensis*) are small freshwater bivalve molluscs indigenous to the Dnieper River basin in Ukraine, in the Ponto-Caspian Sea region (Mills *et al.* 1996). Both species are members of the Dreissenidae family. Based on phylogenetic studies (Therriault *et al*. 2004), *D. bugensis* is considered as a freshwater race of *D. rostriformis*, a mesohaline mussel species which tolerates salinity levels ranging between 12-13‰ and whose geographical range is limited to the Middle and South Caspian Sea. The invasive quagga mussel presently found in North America was initially identified as *Dreissena bugensis* (May and Marsden 1992) and two taxonomic denominations: *D. rostriformis bugensis* and *D. bugensis* can be found in the literature referring to the same species. For simplicity and compatibility with other reports, we refer to quagga mussel as *D. bugensis* in this report.

Like the zebra mussel (*D. polymorpha*), quagga mussels became established in the Great Lakes basin (Lake St. Clair) by the mid-1980s (Mills *et al.* 1996) and since then both species have spread westward in the U.S. (Benson *et al.* 2013). Populations of *D. bugensis* have recently become established in reservoirs in Nevada, Arizona, California and New Mexico. They inhabit freshwater rivers, although preferring to colonize lakes and reservoirs (Benson *et al.* 2013).

A unique aspect of quagga mussel populations in North America is the presence of two phenotypes. Although genetically similar, the two phenotypes prefer vastly different habitats. In the Great Lakes, one phenotype (*D. bugensis* “sensu stricto-eplimnetic”) is found exclusively in warm shallow bays and basins. The other phenotype (*D. bugensis* “profunda”) – which has not been specifically reported in European waters – is normally found in deep cold offshore regions but less commonly in near shore areas above the thermocline (Nalepa 2010).

While not currently present in British Columbia or in adjacent watersheds, zebra and quagga mussel have the potential to colonize some drainage basins in B.C. Members of the genus *Dreissena* are highly polymorphic and have a high potential to rapidly adapt to novel environmental conditions through the evolutionary selection of favorable allelic frequencies and recombination; possibly leading to significant long-term impacts on North American waters (Mills *et al.* 1996).

3.1 Life History

Both zebra and quagga mussel can exhibit many different morphs. Adults may reach a length of 4 cm (Mills *et al.* 1996). Although genetically and morphologically distinct, the two species share similar life histories and distribution ranges. Dreissenids are dioecious (male and female sexes are separate) with external fertilization. Fully mature females are capable of producing up to one million eggs annually. After fertilization, microscopic veliger larvae develop within a few days and soon acquire minute bivalve shells. The pelagic and free-swimming veligers drift with the currents for three to four weeks sustained by ciliary feeding, and eventually locate a suitable substrate to settle, securing themselves by byssal threads. Mortality in the transition from planktonic to settled life may exceed 99% (Bially and MacIsaac 2000).

Data from Lake Mead, Nevada, where a subtropical climate allows year-round reproduction (Wong *et al.* 2012), indicate that colonization rates of 1,000-100,000 mussels/m^2/month are possible. This study showed that quagga mussels are highly heat sensitive, and growth was observed to be slower during the summer months. In more temperate environments such as the Great Lakes and Europe, the growing season extends from late spring to summer; with no growth from autumn to early spring as temperature is a key factor controlling growth (Wong *et al.* 2012). *D. bugensis* colonizes soft substrates in water depths exceeding 40m and silt-sand substrates between 10-30m (Mills *et al.* 1996).

Cross *et al.* (2011) developed a model to estimate the carrying capacity of the Boulder Basin of Lake Mead based on food levels (chlorophyll *a*).Their model estimates that for average food concentration levels, a maximum density of 100,000 adults/m^2 could be sustained. At Parker Dam (Lake Havasu) in Arizona, densities of up to 35,000 adults/m^2 were reported in 2010 (Benson *et al.* 2013).

The biology and ecology of zebra and quagga mussels is constrained by environmental factors such as water temperature, and Wong *et al.* 2012 observed that at temperatures <13°C veligers are present in very low densities (*e.g.* <1.1 individuals/l). As noted above, in the Colorado River basin relatively warmer water temperatures allow mussels to spawn year-round, complicating management (Mann *et al.* 2010).

3.2 Physiological Limits

Calcium, water velocity and water temperature appear to be the main physical factors limiting successful mussel colonization (Mann *et al.* 2010). Calcium may limit potential growth rates and densities, water velocity may limit micro-locations for attachment, and temperature may constrain spawning timing (Mann *et al.* 2010). Current knowledge about the ecology of the quagga mussel is limited and although related, the zebra mussel may not always be an exact analog (Whittier *et al.* 2008). Compared to the zebra mussel, the quagga mussel is more energy-efficient and can live and spawn in cooler, more oligotrophic conditions (Baldwin *et al.* 2002).

Salinity tests (Spidle *et al*. 1995) on both quagga and zebra mussel have showed an upper limit of 5‰ for both species, with no specimen of either *D. bugensis or D. polymorpha* surviving this salinity level in laboratory conditions. Exposure to higher levels of salinity can increase survival times although it does not create absolute tolerance above the 5‰ level. Temperature also plays a role in salinity tolerance; it seems that at warmer temperatures (>20°C), both species have a much shorter survival time at all salinity levels.

The upper thermal tolerance limit of the North American quagga mussel lies around 30°C (Mills *et al.* 1996), with reduced thermal tolerance following spawning (Wong *et al.* 2012). In this regard they are less heat tolerant than zebra mussels. However, quagga exhibit a greater tolerance to low temperatures than zebra mussels, as may be inferred by their occurrence in the deeper waters of the Great Lakes which remain cold (<10°C) year-round. In Lake Ontario, both quagga and zebra mussels coexist at depths of 8-110m, but the relative abundance of quagga mussels increases with depth, with quagga predominating at depths below 65m and being the only dreissenid found at depths of 130m (Mills *et al.* 1993).

Water velocity is another factor which potentially limits mussel establishment and abundance. Chen *et al.* (2011) found that velocity is a key factor affecting veliger settlement, with highest settlement rates at low velocities (<3 cm/s). Veligers may be unable to initiate attachment at water velocities above 2 m/s. Occasional periods of high velocity may strip mussels from structures, preventing infestations from occurring. Zebra mussels are better able to tolerate higher velocities than quagga (Peyer *et al.* 2009).

If all other conditions (*e.g.*, nutrient supply, turbidity, flow, temperature and substrate) are optimal for zebra and quagga mussel growth and reproduction, then calcium, required for basic metabolic function as well as shell building, begins to become a limiting factor. Whittier *et al.* (2008) determined that current mussel distributions in North America are correlated with calcium concentrations and defined invasion risk categories based on the calcium concentration in surface waters. Calcium concentrations above 28 mg/l would result in a very high risk of successful establishment, while conditions of 20 mg/l provide a high risk of establishment. At calcium concentrations below 12 mg/l the risk of successful establishment is very low.

A recent ecological risk assessment (Therriault *et al*. 2012) for three dreissenid species (including both zebra and quagga), has been conducted for freshwater ecosystems in the western Canadian provinces, Ontario, and Quebec. This assessment evaluates, at the sub-drainage watershed level, the probability of establishment of quagga mussels populations based on habitat suitability (calcium concentration) and the likelihood of invasion considering the distance to established populations. The results of this risk assessment (maps of habitat suitability and probability of establishment) for the quagga mussel in the province of British Columbia are shown in Figure 3.1.

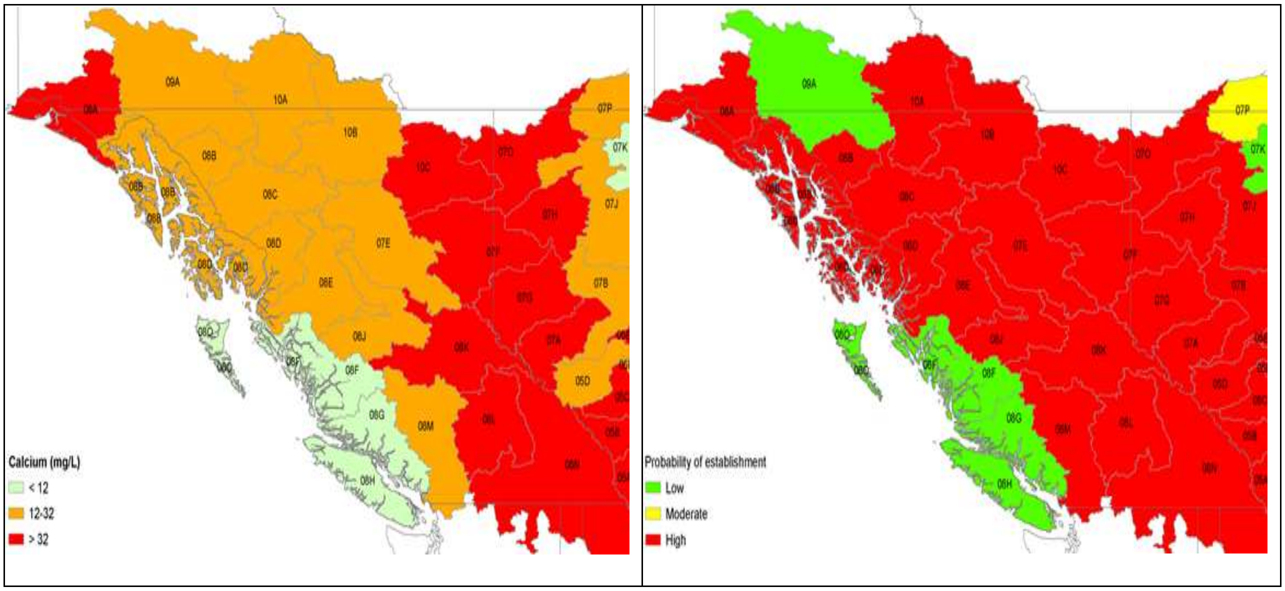

Figure 3.1: Habitat suitability based on calcium and risk of establishment for zebra mussels in British Columbia. Results for quagga mussels are almost identical. The left panel shows 3 categories of surface water calcium concentration; the right panel shows the corresponding establishment risk. (Thierrault *et al.* 2012)

3.3 Dispersal

The introduction of *D. bugensis* into the Great Lakes appears to be the result of ballast water discharge from transoceanic ships that could have been carrying veligers, juveniles or adult mussels (Wilson *et al.* 1999). The genus *Dreissena* is both prolific and highly polymorphic, giving it the potential ability to rapidly colonize and adapt to new environments (Mills *et al.* 1996). Once established, larval drift in river systems or transport through fishing and boating activities allows for overland and inter- basin spread (Benson *et al.* 2013). Within an indigenous river system in Ukraine, Mills *et al.* (1996) report a spread rate of up to 20 km/yr.

Despite some differences in their patterns of colonization, the radiation of both *D. bugensis* and *D. polymorpha* has been exceedingly rapid following their initial introduction (Wilson *et al.* 1999). The species share two key characteristics that contribute to their rapid spread: First, they have a planktonic stage in which free-living veliger larvae can remain suspended for up to several weeks. Second, they attach by byssal threads to substrates (Mills *et al.* 1996). Once attached, this allows them to accompany water craft overland and disperse to geographically disparate regions. A study on dispersal strategies of

D. bugensis (Wilson *et al.* 1999) suggests that long-distance dispersal, possibly mediated by boater movement patterns, may be having a significant impact on the spread of quagga mussel. Other studies by Padilla *et al.* (1996) and Johnson *et al.* (2001) have concluded that recreational boats fouled with quagga mussels on trailers have proved to be key pathways for infestation. A current record of North American quagga and zebra mussel observations is shown in Figure 3.2.

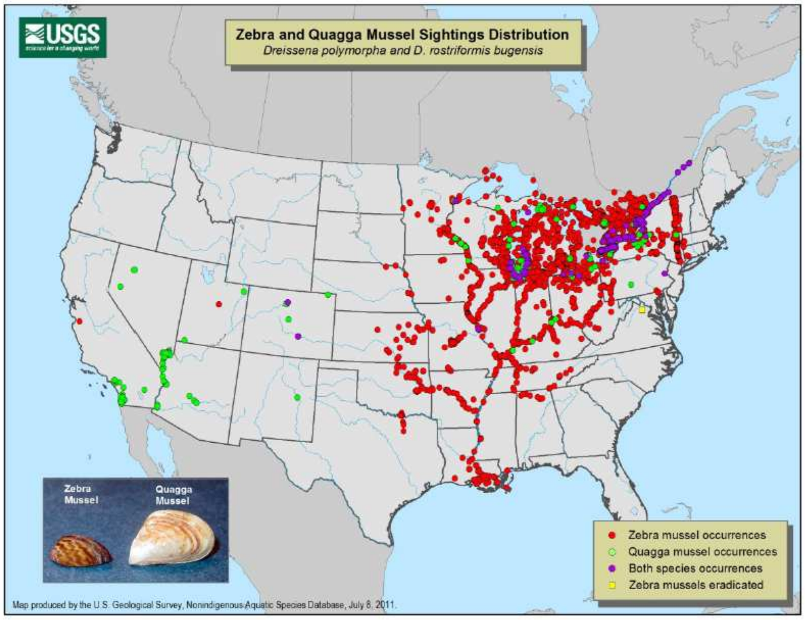

Figure 3.2: Distribution of sightings of quagga and zebra mussels in North America (Source: U.S. Geological Survey. Non-indigenous Aquatic Species Database 2013).

Quagga mussels have been observed to spread more slowly than zebra mussels although once established, they tend to replace local populations of zebra mussels (Mills *et al.* 1996, Benson *et al.* 2013) and become the dominant species. Physiological and developmental differences between these two related species may explain the initial more rapid spread of zebra mussels in North America followed by their subsequent slow displacement by quagga mussels.

3.4 Impacts

Ecological impacts of quagga and zebra mussel depend on the achieved densities and characteristics of the invaded system (Strayer *et al.* 2006, Nalepa 2010). Mussels filter prodigious volumes of water, removing substantial amounts of phytoplankton and suspended particulates. By removing phytoplankton, quagga mussels decrease the primary food source of zooplankton and other benthic organisms, therefore altering the food web (Benson *et al.* 2013). Where conditions are favorable and dreissenids are abundant, fundamental changes in energy and nutrient cycling occur. In the Great Lakes, a reduction in the density of the benthic crustacean *Diporeia spp.* has been linked to the dispersal of quagga mussel into Lake Michigan (Watkins *et al.* 2007).

The establishment of quagga and zebra mussels has also been linked to changes in the availability of inorganic nutrients in the water column (Turner 2010). Substantially higher post-invasion concentrations of soluble reactive phosphorus have been measured in lakes and rivers. Ammonium and nitrate concentrations have also increased in many systems, and oxygen concentrations have declined.

Impacts associated with the filtration of water also include increases in water transparency, decreases in mean chlorophyll *a* concentrations and accumulation of pseudofeces (a mixture of mucus and non-edible particulates). Increased water clarity results in deeper light penetration which can cause a proliferation of aquatic plants, eutrophication events and blooms of cyanobacteria (Bootsma *et al.* 2008). Pseudofeces released by quagga mussels can also accumulate and create a foul environment in which oxygen is consumed through decomposition, pH is reduced and toxic byproducts are produced. Quagga mussels also accumulate organic pollutants within their tissues to levels more than 300,000 times greater than concentrations in the environment. These pollutants can also be found in pseudofeces and passed up the food chain, increasing wildlife exposure to organic pollutants (Snyder *et al.* 1997).

The impacts on fishes depend on the habitat requirements, diet and the population state of the particular species, but zebra and quagga mussel impacts on fish communities are now becoming more apparent as quagga mussels increase and expand into new habitats (Nalepa 2010). Overall, their impacts on the food web cause near-shore regions to become more nutrient enriched as benthic productivity increases, whereas offshore regions become more nutrient starved as pelagic productivity declines. The impacts of zebra and quagga mussels on native freshwater mussels are still unclear but could be severe (Ricciardi *et al.* 1998). The recent SARA listing of Rocky Mountain ridged mussel (*Gonidea angulate*) was due to the potential for zebra and quagga mussel invasion into the Okanagan basin.

Dreissena species' ability to rapidly colonize hard surfaces causes serious economic problems. They tend to overgrow and clog water intake pipes of power stations, municipal water supplies and agricultural raw-water intakes (Padilla *et al.* 1996, Johnson *et al.* 2001).These biofouling effects reduce pumping capabilities for power and water treatment plants. Recreation-based industries and activities are also affected as docks, breakwaters, buoys, boats, and beaches become colonized. Figure 3.3 shows some possible damage pathways likely to operate in the aquatic environments invaded by zebra and quagga mussels.

3.5 Potential Control

The most common treatment for localized control has been prechlorination, but if this method is used to control dreissenid mussels the amount of chlorine used may reach hazardous levels (Grime 1995). Potassium permanganate has been used as an alternative treatment, especially for drinking water sources, even though chemical controls are not the most environmentally sound solution. A recent study has found that thermal treatment of residual water in boats and other water craft is the recommended option for managing quagga and zebra mussel spread. Increasing temperature and exposure time was found to increase the level of veliger mortality (Craft and Myrick 2011). Hot water treatments based on 10 seconds of pressure washing at 60°C or 70 seconds of flushing at 55°C are being standardized across the western United States (Zook and Phillips 2012) and are also recommended by the BC government (Herborg, *pers com*).

Researchers have been studying zebra and quagga mussel for the past 25 years, to learn more about their life cycle and environmental and physiological tolerances, with the hope of developing environmentally safe controls that can be used to control dreissenids. Effective biological control has so far proven to be elusive. Predation by migrating diving ducks, fish species, and crayfish may reduce mussel abundance, but these effects are temporary (Bially and MacIsaac 2000). Other biological controls being studied include selectively toxic microbes and parasites, which might someday play a role in the management of *Dreissena* (Molloy 1998). Other prospective approaches to controlling *Dreissena* may be to disrupt the reproductive process by interfering with the synchronous spawning of males and females (Snyder *et al.* 1997). Another possible approach would be to inhibit the planktonic veliger from settling, since this is the most vulnerable stage in the life cycle (Kennedy 2002).

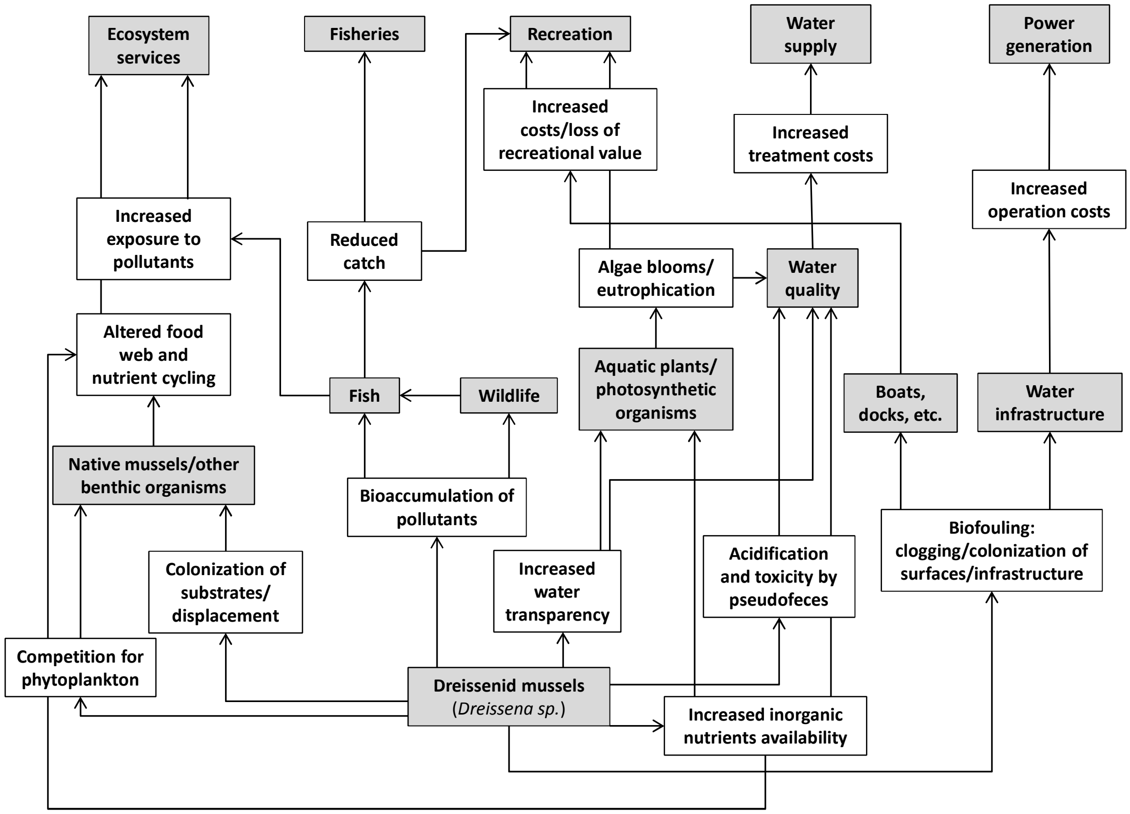

Figure 3.3: Impact diagram for zebra and quagga mussels (*D. polymorpha, D. bugensis*).

Prevention of zebra and quagga colonization through active operational programs has been implemented in most western states in the U.S. since the discovery of quagga mussels in Lake Mead on the Colorado River in early 2007 (Mann *et al.* 2010). These programs usually include mandatory inspections at border crossings and boat launches, public education, and coordination among state, local and tribal agencies. Under the 100th Meridian Initiative, a cooperative effort involving different U.S. agencies and British Columbia to prevent and monitor the spread of zebra mussels and other aquatic nuisance species in the Columbia River Basin (Heimowitz and Phillips 2008) has resulted in the development of an Interagency Invasive Species Response Plan to delineate, contain, and when feasible, eradicate zebra, quagga, and other dreissenid populations if they are introduced in the Columbia basin waters.

3.6 Unit Economic Damages

Where available, Canadian damage estimates are reported; otherwise, data from other jurisdictions, primarily in eastern Canada and the eastern United States, is used. Forecasts of potential economic damages vary widely across studies. O'Neill (1997) estimates total damages of over USD 69 million in 35 states and 3 provinces using surveys to obtain the damages and control costs incurred by industries that draw raw water directly from infested waters (*e.g.* electric power generation facilities, water treatment plants and others). The U.S. Fish and Wildlife Service estimates a potential economic cost to the Great Lakes region of USD 5 billion from 2000 to 2010 (USGS 2000), while Pimentel (2005) estimates USD 1 billion worth of annual damages to water infrastructure and related industry in the United States. Many of the ecological impacts translate to economic impacts, especially in the recreational and commercial fishing industry (Pimentel 2005), although the latter does not exist in the freshwater portions of B.C. Vilaplana and Hushak (1994) estimate the economic impacts on recreation activities in Lake Erie. In that study, the authors surveyed residents and obtained 109 responses from boat owners and of these, 13% reported damages from mussels. However, this value represents current damages and not maximum potential damages. In addition, Limburg *et al.* (2010) estimate the loss in lakefront property values in the Great Lakes region as a result of mussel infestation. While care is required in using property values since they may simply "capitalize" direct damages and lead to double-counting of losses, these studies provide a good starting point for the present study. Economic data were unavailable for the impacts of dreissenid mussels on agriculture (irrigation facilities) and ecosystem services.

Using transferable damage estimates from peer reviewed and gray literature sources and facility count data for B.C., damage estimates are calculated for B.C. hydropower generating facilities, water supply facilities, recreational boating and golf courses. Thus, our estimate excludes several categories of damage captured in Figure 3.3. For example, agricultural water intakes are an additional category that may be impacted, but such data are not readily available. While estimates of the total value of freshwater recreational fishing are available on a regional basis we had no basis on which to adjust these values to include them in our values. Future research should be able to address this shortfall. In addition, there would be extra maintenance to docks, reduced recreational value due to sharp shells on the shoreline and in the water and other damages that we do not capture but which might be quantified in a more thorough analysis. Ecosystem impacts may be more difficult to estimate (*e.g.* effects on salmonid rearing).

Table 3.1: Selected unit damage estimates for zebra and quagga mussels in 2012 CAD. FER = Foreign Exchange Rate; CPI = Consumer Price Index.

Impacts	Initial Estimate			Adjustment	Final Estimate (CAD/unit)	Notes
	Year	Unit	Damage ($/unit)			
Power generation – General						1
NaOCl system	2007	MW	CAD 122.8	CPI: x1.09	134	
Antifouiling trash rack	2005	MW	USD 346.9	FER: x1.21 CPI: x1.14	478	
					Sub-total: 612	
Power generation – Run of River (ROR)						2
NaOCl system	2007	MW	CAD 122.8	CPI: x1.09	134	
Antifouiling trash rack	2005	MW	USD 346.9	FER: x1.21 CPI: x1.14	478	
					Sub-total: 612	
Water supply						3
	1994	Small facility Large facility	USD 23,000 USD 79,833	FER: x1.37 CPI: x1.42	44,629 154,910	
Recreational boating	1993	Boat	USD 315	FER: x1.29 CPI: x1.42	578	4
Golf courses	1995	Facility	USD 150	FER: x1.37 CPI: x1.39	286	5

Notes

1 Unit damages for the NaOCl control system were obtained from Van Oostrom (2007) and unit damages for antifouling painted trash racks were obtained from Philips *et al.* (2005). The one-time cost of installing a NaOCl system in 2007 prices for a 2,286 MW facility was estimated at CAD 2,332,000, with annual operating costs of CAD 115,480. Painting trash racks with antifouling paint was estimated to be USD 1,711,688 or 12.50 per square foot in 2005 prices for a 1,093 MW facility. The NaOCl control system and antifouling trash rack are

capital costs. The NaOCl has a lifespan of 20 years, and the trash racks have to be repainted with antifouling paint every 5 years. To calculate an annual unit damage, these capital costs need to be annualized over their respective lifespans using the following formula: AIV = iI{(1+ i)^n / [(1+i)^n -1]}, where AIV is the annualized investment value, I= the initial capital cost, i =social discount rate (3.5%), and n= 20 and n= 5 for the respective life span of the NaOCl control system and antifouling trash rack. *Total annual costs= annualized capital costs + annual operating costs.* The social discount rate was decided upon after a review of current literature that generally supports a low discount rate, Moore *et al.* (2004) proposed a social discount rate of 3.5%, and this rate was employed by Marbek (2010) in their analysis of the economic impacts of zebra and quagga mussels in Ontario The annual damage estimate for the NaOCL system includes operating and maintenance costs, but excludes cost of training, monitoring, physical removal of zebra and quagga mussels on flat surfaces or loss of production. Unit damage cost for antifouling trash racks does not include labor costs for removal and reinstallation of the trash racks.

2 Although smaller than mainstream hydropower generation facilities, Run of river (ROR) IPP facilities are liable to zebra and quagga mussel damage. Therefore we apply the same unit damages that we did to the mainstream hydropower generating facilities.

3 Park and Hushak (1999) estimate total costs for municipal water supply facilities in the United States. Total costs include monitoring and control costs. The control cost, expenditures incurred to control zebra mussels, was the sum of i) retrofitting costs, ii) physical removal or mechanical exclusion costs, iii) variable chemical treatment costs, and iv) other treatment costs. Water supply facilities were divided into small facilities (raw water intake capacity of 0 to 10 million gallons per day), and large facilities (raw water intake capacity of over 10 million gallons per day). For the purpose of our analysis the intake capacities were converted to liters per day: small facilities (≤37 million liters/day), and large facilities greater than 37 million liters per day.

4 Vilaplana and Hushak (1994) surveyed boat owners in the Great Lakes region and provide annual damage estimates to boat owners based on direct damage, protective paint, and additional maintenance.

5 O'Neill (1997) provides an average annual damage estimate of USD 150 on golf courses in the United States. Expenditures were incurred through monitoring activities.

Detailed calculations of the unit damage estimates for the identified impact categories are shown inTable 3.1, adjusted to 2012 CAD as described above. Our annual power generation damage estimate of CAD 612 per megawatt is calculated from Ontario Power Generation's Niagara Plant Group operations estimate for the installation and operating cost of a permanent sodium hypochlorite (NaOCl) control system (Van Oostrom 2007). We also used Bonneville Power Administration's cost for painting trash racks with antifouling paint (Phillips *et al* 2005). For water supply facilities, damage estimates of CAD 44,630 for a small water supply facility and CAD 154,910 for a large water supply facility are calculated using data from Park and Hushak (1999). For the fouling of recreational boats, a damage estimate of CAD 578 per boat is obtained from Vilaplana and Hushak (1994), who provide one of the few damage estimates for this category of impact. Finally, a damage of CAD 280 for golf courses is calculated using data from O'Neill (1997), based on the average annual costs incurred by golf courses in the Great Lakes region.

3.7 Ecological Carrying Capacity

The potential range of zebra and quagga mussel in B.C. was ascertained from risk assessment estimates at the sub-basin level for B.C. watersheds (Figure 3.1). We only considered sub-basins that have both a highly suitable habitat based on calcium concentration, and a high vulnerability to establishment. As noted in Section 2, we consider the limiting extreme case where infestation occurs in all surface water bodies in these sub drainages, also known as the "ecological carrying capacity." An inventory of facilities in the potential range of the zebra and quagga mussels was conducted by overlaying the potential range on a map of administrative water districts described in Schedule C of the Water Act (1996).

Several data sources provide count data that can be used to determine the total number of vulnerable facilities that could be occupied or affected by the mussels in the water districts of the potential range. The number of hydropower facilities was obtained from the Ministry of Forests, Lands and Natural

Resource Operations water license database, which provides detailed information on licenses issued for the purpose of power generation.[2] Three license categories related to power generation are used in the database: (a) General power generation, including mainstream dams and run of river (ROR) facilities; (b) Commercial power generation of less than 500 kW; and, (c) Residential power generation. We did not include the 225 Commercial and Residential power facilities in our analysis due their small generating capacity.

License information for mainstream dams and ROR facilities was cross-checked with information provided by B.C. Hydro and Fortis BC. We assume hydropower facilities of all capacities are liable to damage from mussels, including ROR power generating facilities. Our analysis did not consider damages to fish passage facilities.

Independent counts of the water supply facilities within water districts in the potential range were unavailable. Therefore, we relied upon the Ministry of Forests, Lands and Natural Resource Operations water license database to estimate the number of water supply facilities. We counted the individual licensees in the Local Authority Waterworks category that have current licenses allowing them to convey water to five or more dwellings.[3] However, the number of water licenses held by an individual Waterworks Local Authority does not indicate the total number of water withdrawal sites they operate. Due to the scarcity of data we made the assumption that each Waterworks Local Authority that is issued a license for the purpose of conveying water to five or more dwellings has only one water withdrawal site. Water supply facilities were divided into two categories: (a) small facilities with an average daily intake capacity of less than 37 million liters; and (b) large facilities with an average daily intake capacity greater than 37 million liters. As a proxy for intake capacity, we used 90% of the maximum quantity of water that can be diverted, as indicated on the license.

Transport Canada estimated that in 2012 there were 357,143 private licensed vessels over 10 horsepower in British Columbia, of which 30% were for use in freshwater (Seeley *pers. comm.*). Villaplana and Hushak (1994) found in their survey of boat owners in the Great Lakes region that 13% of the 109 owners they contacted had experienced damages from mussels, but this reflects the current stage of the invasion (admittedly much advanced). We use this estimate as a starting point, but given that some limited control of mussels might be occurring in the Great Lakes and that the maximum proportion of boats that could be affected likely would be higher, we adjust it upwards. We therefore assume that a maximum of 20% of the freshwater pleasure vessels in B.C. are vulnerable to zebra and quagga mussel damage. This also captures the notion that owners serve as a dispersal mechanism since they can launch their vessel anywhere in the province, and mussels easily attach to boat hulls and motors (Missouri Department of Conservation 2013). This results in an adjusted value for the maximum number of vessels that potentially could be damaged of 21,429 vessels.

Finally, the number of golf facilities relying on diverted surface water was estimated as the number of individual licenses in the Watering purpose category licenses in this category issued for the purpose of watering golf courses.[4] Each water license holder in the Watering category that was registered as a golf course or a resort golf course was assumed to operate one golf course.

---

[2] http://a100.gov.bc.ca/pub/wtrwhse/water_licences.input

[3] Water conveyed to five or more dwellings by a water district incorporated under an Act, municipality, improvement district, water utility under the Water Utility Act or development district

[4] Water used for watering of golf courses, ornamental gardens

3.8 Preliminary Damage Estimate

Preliminary total damage estimates are shown in Table 3.2. Our total damage estimate of CAD 6,524,532 for power generation includes run of river (ROR) facilities and is more conservative than damage estimates published in the United States. For example, the Aquatic Nuisance Species Taskforce (ANST 2009) estimated that zebra mussels cost the state of Idaho USD 47,242,000 annually. Had our analysis relied upon other data sources, the total damage estimate might have been significantly greater. We instead opted to use more current Canadian damage estimates from the Great Lakes region. As such, our damage estimates are in line with estimates from other Canadian jurisdictions. The economic damage to Ontario power plants was estimated at CAD 7,646,884 in 2009 Canadian dollars (Marbek 2010). Due to uncertainty surrounding the estimates, we made an additional calculation by applying a social discount rate of 7% to the capital costs of the NaOCL treatment system and the antifouling trash racks. A discount rate of 7% resulted in an annual unit damage estimate of CAD 689 per MW, and total annual damages to mainstream hydropower facilities of CAD 7,150,854, and CAD 168,902 to run of river facilities. The annual figure of CAD 6,524,532 could potentially increase if more power generating facilities come online in B.C. Currently 9 run of river projects are within various stages of approval and assessment within the potential distribution of zebra and quagga mussels in B.C.

The annual estimated damage to water supply facilities from zebra and quagga mussels was CAD 9,251,608. Again, this is a conservative estimate due to the lack of data on the total number of individual water supply facilities. Due to the data limitations we assumed that each license holder operated only one water withdrawal site; yet one license holder may operate several sites.

Table 3.2: Preliminary damage estimate for B.C. for zebra and quagga mussels in 2012 CAD.

Impacts	Number of Units	Final Estimate ($/unit, 2012)	Preliminary Damage Estimate
Power generation – General	10,415 MW	612	6,373,980
Power generation – Run of River (ROR)	246 MW	612	150,552
Water supply	183 small	44,629	8,167,236
	7 large	154,910	1,084,372
			Sub-total: 9,251,598
Recreational boating	21,429 vessels	578	12,385,962
Golf Courses	38	286	10,867
Total			**21,648,427**

Recreational boating damages make up the largest portion of our total damage estimate in Table 3.2, even when based on the assumption that only 20% of freshwater vessels over 10 horse power in BC might eventually be damaged by zebra and quagga mussels. While large, our damage estimate is consistent with other jurisdictions. For example, Marbek (2010) estimated annual damages to Great Lakes recreational boaters in Ontario of CAD 44,233,155 in 2009 dollars using a similar methodology and the same Vilaplana and Hushak (1994) study as a basis, but this is based on a much larger vulnerable population of 780,000 recreational vessels. Finally, for comparison we estimate the total annual damages for golf courses at CAD 10,867, whereas annual damages were estimated to be CAD 14,308 in Ontario (Marbek 2010) in 2009 prices and USD 17,100 in Idaho in 1997 prices (ANST 2009).

4. European Fire Ant

The European fire ant (*Myrmica rubra*), a member of the Hymenoptera family, is native to northern Europe and western Asia (Groden *et al.* 2005). The first reports of this species in North America date back to the beginning of the 20th century in the northeast of the U.S. It was reported in Quebec in 1915 and started a significant expansion about 30-40 years ago, when it extended its range to southern Ontario and more recently, to the Maritimes (Groden *et al.* 2005). It is estimated that its introduction in B.C. dates back to 15 years ago (Higgins 2013) and it has since become established in the southern part of the province. This ant inhabits moist grass areas, usually in an urban environment. They can form dense colonies in yards and parks and may render the colonized areas unusable.

4.1 Life History

In North America, *M. rubra* nests are cryptic and difficult to locate, since they do not construct obvious soil mounds. Colonies of the European fire ant typically have more than one queen (reported average 15 per colony), a trait which is common to many ant species at higher latitudes. The number of queens per nest is highly variable both within and between seasons and seems to regulate the population dynamics of *M. rubra* (Groden *et al.* 2005). New virgin queens (winged or possibly non-winged in this species) and winged males are produced in late June (Groden *et al.* 2005). In its native range nuptial flights occur in August or September. However, it seems that in North America the colonies of *M. rubra* reproduce predominantly by budding as the colony population grows. Mature colonies contain approximately 1000 workers with queens reported to lay 200-300 eggs per year. As the colony grows larger, some queens will walk away from the parent colony, accompanied by a group of workers, and establish new nests in the vicinity of the original nest (Groden *et al.* 2005, Higgins 2013). This budding process can occur throughout the foraging season. European fire ants are not strong dispersers and are frequently found as patches containing a high density of nests: up to 4 colonies/m^2, which would typically result in 4,000 individuals/m^2 where conditions are favorable. Nest densities in its native range do not usually reach the high levels observed in introduced populations (Groden *et al.* 2005).

M. rubra thrives in heat and moisture, and protected backyard lawns having fences, trees and shrubs, raised garden beds, frames, greenhouses and lawn clutter are therefore ideal for infestation. All of these local features create prime habitat and provide good insulation at ground level and enough shading to hold moisture (B.C. Inter-Ministry Invasive Species Working Group 2012). The behavior of *M. rubra* changes according to temperature. When it is warm and humid the ants become much more active and are highly prone to swarm and sting (B.C. Inter-Ministry Invasive Species Working Group 2012).

4.2 Physiological Limits

The European fire ant requires areas of high moisture and a mean annual temp >6°C. Precipitation in affected areas is usually found to exceed 1000 mm/yr, although in the Toronto area this ant appears to be thriving in an area where precipitation is recorded at 680 mm/yr. Using the Toronto precipitation data as a minimum, and a minimum mean annual temperature of 6°C, this suggests this ant could spread through disturbed, mostly urban-linked habitat, along the full length of coastal B.C. (Higgins 2013) and one small area away from coastal B.C, near the community of Nelson, as shown in Figure 4.1.

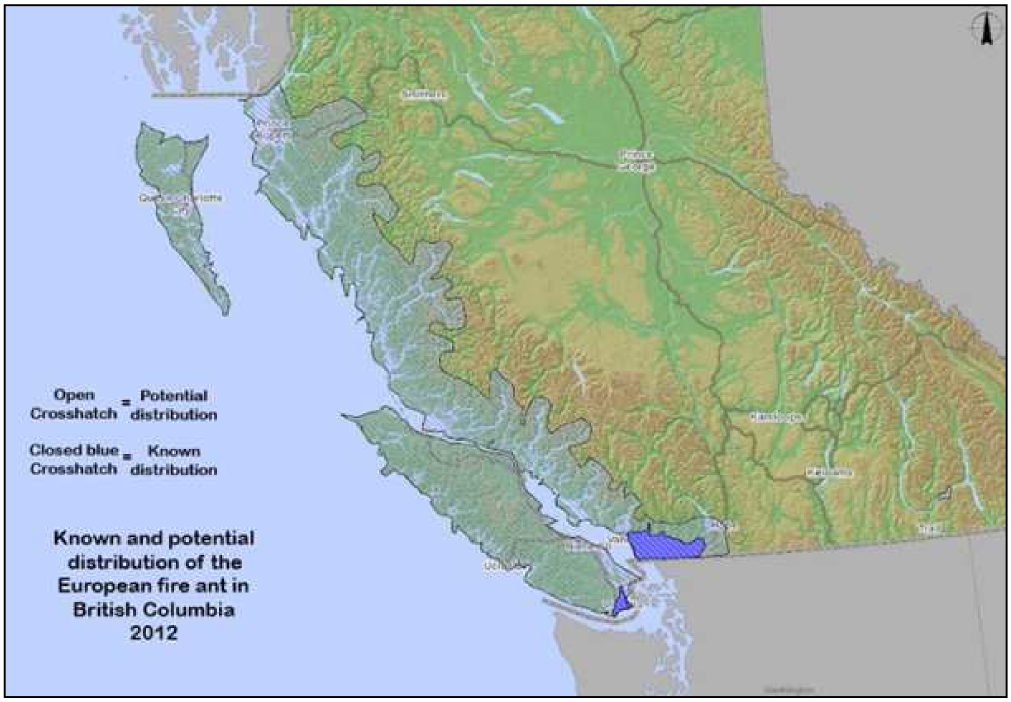

Figure 4.1: Known and potential distribution of the European fire ant (*Myrmica rubra*) in British Columbia (Source: Higgins 2012).

4.3 Dispersal

The European fire ant was introduced to southwestern B.C. at least 15 years ago. The most likely pathway for introduction seems to have been through landscaping plants and the movement of contaminated soil (Higgins 2012). They are now found predominantly in residential areas, parks and community and botanical gardens.

Once established in a new habitat, colonies tend to grow exponentially until reaching their maximum density or carrying capacity, which in B.C. seems to be 4 nests/m^2 (Higgins 2013), representing about 4,000 individuals. Since their habitat tends to be patchy and they reproduce by budding in North America rather than through nuptial flying, the European fire ants tend to remain localized, which limits their ability to spread rapidly (Higgins 2013).

4.4 Impacts

M. rubra can form dense colonies in yards, making such areas difficult for residents to use and extremely unusable for small children or pets. Stinging incidents may result in health costs from people who have received emergency treatment for severe allergic reactions and veterinary bills for pets that have been stung. The presence of ants in the yards and gardens could also potentially reduce property values because of the injuries and other damages attributable to the fire ants. However, care is needed in using such information to determine losses to avoid double-counting as these damages are already captured in estimates of health and related impacts. Other locations of concern for *M. rubra* infestation include high value parks, community gardens and in sweet crops via aphid tending (B.C. Inter-Ministry Invasive Species Working Group 2012).

Infestations of *M. rubra* in B.C. have so far been limited to green areas in the urban environment, but there is a risk that they might invade rural areas and affect crops, as seems to be happening in some community gardens in southern B.C. (Higgins, *pers. comm.*). The European fire ant is also likely to affect wildlife (birds) and could displace native ant species. These effects have yet to be quantified or estimated. Higgins (2012) reports that European fire ants become more aggressive at higher densities and higher temperatures and when disturbed, can swarm rapidly and attack by stinging. In contrast, ant species native to B.C. are less aggressive (Higgins 2013). Although the stings are generally not painful, there have been some cases of serious reaction that required medical assistance (Higgins 2013). Based on these effects, the impact diagram shown in Figure 4.2 show the possible damage pathways likely to operate in the environments invaded by the European fire ant.

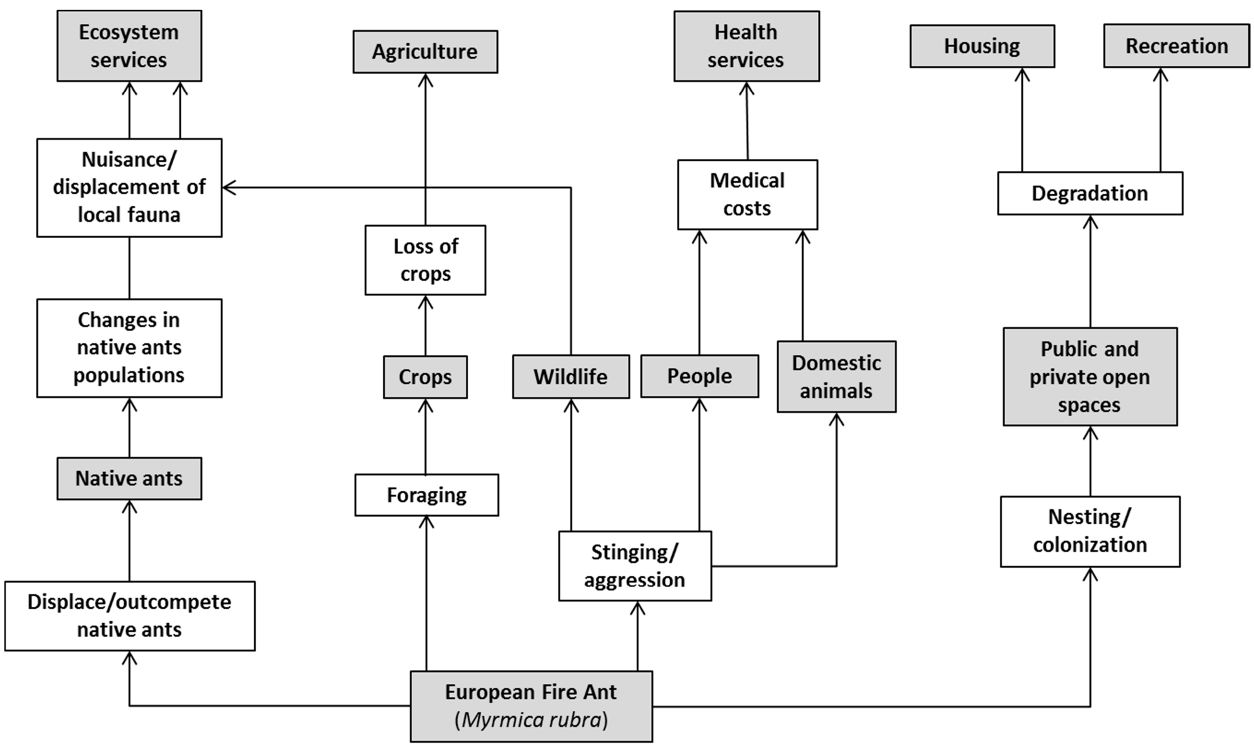

Figure 4.2: Impact diagram for the European fire ant (*Myrmica rubra*).

4.5 Potential Control

No effective control for European fire ants has been yet been found in eastern Canada or the United States. In B.C, pesticides containing boric acid have been found to show some success in controlling *M. rubra* (B.C. Inter-Ministry Invasive Species Working Group 2012). Laboratories studies to develop pesticides have not yet produced conclusive results. Baits containing 0.7% boric acid do not appear to quickly kill colonies although their impact was greater than that seen in untreated control colonies (Higgins 2012).

Additional recommendations for control that can be implemented in yards and other green areas commonly include: cutting the grass short, removing lawn clutter, turning over garden beds in winter to expose nests to cold, and placing paving rock on gravel instead of on soil to avoid facilitating nesting and hiding places (B.C. Inter-Ministry Invasive Species Working Group 2012). The B.C. Inter-Ministry

Invasive Species Working Group also recommends community education and cooperative involvement as key elements for effective control.

Since invasions of *M. rubra* are a fairly recent problem, further research on possible management actions, and a better understanding of the ant's biology, current distribution and rate of spread is needed (B.C. Inter-Ministry Invasive Species Working Group 2012).

4.6 Unit Economic Damages

Studies of the economic damages from European fire ant *(Myrmica rubra*) have not been undertaken in Canada or in any other jurisdiction. However, the economic damages of the red imported fire ant (*Solenopsis invicta)*, which has become established in the southern United States and in the Pacific region, have been extensively documented. For example, the annual economic costs of the red imported fire ant in Hawaii are estimated to be USD 211 million/year (Gutrich *et al.* 2007). Similarly to *M. rubra* the red imported fire ant is found in close proximity to human settlements, and is almost impossible to eradicate once established. Since the facilities affected by *S. invicta* are likely to be similar to those affected by *M. rubra,* we use the economic impacts of *S. invicta* as a surrogate for *M. rubra*. In Texas, estimated annual costs of managing and controlling red imported fire ants exceeded $580 million for households, schools, cities, and gold courses (Lard *et al.* 2002). We use the damage estimates for Texas from Lard *et al.* (2002) as a basis for our unit damage impacts for B.C. We adjust the Texas damage estimates to 2012 Canadian prices, but reduce the estimates by 50% to account for the less aggressive behavior of the European fire ant; *M. rubra* has been described as being 40% less aggressive than the red imported fire ant (Higgins *pers. comm.*) and although this does not directly correspond to a reduction in the potential damages we use it as an indicator.

In making our unit damage estimates for the European fire ant, we use Figure 4.2 as a guide but do not match it perfectly. Instead, we use the damage categories from Lard *et al.* that include the same impact considerations but organized slightly differently (for convenience). In particular, Lard *et al.* include medical costs together with other damages by impact category and not separately (for tractability) and isolate schools, municipal open green space (mostly parks) and golf courses as specific examples of public and private infrastructure vulnerable to fire ant invasion but distinct from household property. Agriculture estimates were not transferable to B.C. because of the different classification of farming activities in other jurisdictions. In addition, economic damage data were not available for the impact of red imported fire ants on ecosystem services.

Using the information from Lard *et al.* and making various adjustments, we estimate annual unit damages for B.C. households of CAD 298 in 2012 Canadian prices, CAD 9,794 for schools, and CAD 106,019 for cities, and CAD 127,502 for golf courses. Details of the calculation of these unit damage estimates are shown in Table 4.1.

4.7 Ecological Carrying Capacity

An estimate of the known and potential distribution of the European fire ant in B.C. shown in Figure 4.1 served as a guide to selecting administrative regional districts that lie within the physiological limits of *M. rubra*. Alberni, Capital, Central Coast, Comox & Strathcona, Cowichan Valley, Fraser Valley, Kitimat-Stikine (Kitimat, Kitimat electoral area C and Terrace only), Metro Vancouver, Mount Waddington, Nanaimo, Powell River, Skeena & Queen Charlotte Squamish-Lillooet (Squamish only), and Sunshine Coast regional districts were identified within the area of known and potential distribution. Count data for all households within the regional districts was accessed through Statistics Canada 2011 census data (Statistics Canada 2013). We assumed *M. rubra* would damage single detached dwellings because they may have access to a yard or a garden, which is a common habitat area for *M. rubra.*

School districts located within the known and potential distribution of *M. rubra* were identified, and a count of the number of public and private schools in each school district was obtained from the Ministry of Education's provincial report on the kindergarten to grade 12 school system (B.C. Ministry of Education 2013). We assume that in defining the maximum range or ecological carrying capacity for *M. rubra,* all schools within these school districts should be included as potentially vulnerable to damages.

The number of municipalities in each regional district was obtained from B.C. Stats 2012 (B.C. Statistics 2012). A municipality was defined as a city, town, village, district municipality, resort municipality, or Indian Government District as per the Local Government Act (1996). Unincorporated areas were not included in the analysis as they are under the jurisdiction of regional districts and may not be responsible for providing municipal services and infrastructure. We further assume that parks, athletic fields, recreational areas, public cemeteries, community centers, public office and building areas are liable to damage from *M. rubra* (but note this is not the same as changes to property "values"). Finally, we assume that all golf courses within the physiological limits of *M. rubra* are vulnerable to invasion, using an inventory of golf courses obtained from the B.C. Golf Guide (2013).

Table 4.1: Selected unit damage estimates for European fire ant in 2012 CAD. FER = Foreign Exchange Rate; CPI = Consumer Price Index.

Impacts	Initial Estimate			Adjustment	Final Estimate (CAD/unit)	Notes
	Year	Unit	Damage ($/unit)			
Households (health and property)	1998	Household	USD 151	FER: x1.48 CPI: x1.33 Other: x0.50	149	1
Schools (health and property)	1998	School	USD 4954	FER: x1.48 CPI: x1.33 Other: x0.50	4,897	1
Municipalities (health and property)	1998	Municipality	USD 53,628	FER: x1.48 CPI: x1.33 Other: x0.50 Other: x0.024	1,267	1,2
Golf Courses (health and property)	1998	Golf Course	USD 63,495	FER: x1.48 CPI: x1.33 Other: x0.50	62,492	1

Notes

1. Data were collected from five metroplex (metropolitan) areas in Texas. Annual expenditure for the management and control of fire ants was selected as an indicator of the annual economic impact. Damage estimates for single detached households, schools, and golf courses were based on fire ant treatment costs which include pesticides, baits and other control measures, repair, medical care and replacement costs. These damage costs were extrapolated to all single detached households in the metroplex. Damage estimates for each of the impact categories were reduced by 50% to account for reduced aggressiveness in comparison to the red imported fire ant (Higgins, *pers. comm.*), but a less conservative 20% value is used later as an alternative. Estimates include both health damages (medical costs) and damages to property for each impact listed.
2. Specific municipal properties considered by Lard *et al.* 2002 include parks, athletic fields, recreational areas, public cemeteries, community centers, public office and building areas. An additional adjustment was made to account for differences in the definition and size of B.C. municipalities (average number of households: 7,435) in comparison to the "cities" defined in the Texas study (average number of households: 305,596) using a simple prorating approach; this resulted in an adjustment factor of 0.024.

4.8 Preliminary Damage Estimate

We estimate the maximum potential economic damage from the European fire ant (*M. rubra*) in B.C. at CAD 100,000,901, at least for the impact categories we have been able to quantify. Details of the calculation are provided in Table 4.2 and show health damages combined with damages to property for each impact type listed in the table. The impact category that provides the most habitat area for *M .rubra* is likely to be households (health and other damages), so it accounts for the most damage. About 50% of the damage to households is treatment costs for lawns and gardens, while medical costs resulting from stings only represent about 10% of the damages. Golf course greens provide an expansive area of suitable habitat for *M. rubra,* requiring extensive and regular treatments, leading to significant annual damages. The modest area allocated to municipal open green space within the fire ant range in B.C. means that potential damages are low in comparison to other impact categories.

Table 4.2: Preliminary damage estimates for European fire ant in 2012 CAD.

Impacts	Number of Units	Final Estimate ($/unit, 2012; including 50% reduction)	Preliminary Damage Estimate
Households	572,511	149	85,282,174
Schools	948	4,897	4,642,218
Municipalities	77	1,267	97,559
Golf Courses	160	62,492	9,998,720
Total			**100,000,901**

To account for the inherent uncertainty surrounding the difference in aggression between *M. rubra* and *S. invicta*, we calculated additional damage estimates that were reduced by only 20% rather than 50%, which represents a less conservative assumption about the differences between the two species. A 20% reduction in damage associated with the European fire ant resulted in total annual damages for B.C. of CAD 160,036,586.

5. Asian Carp

This group of four invasive Cyprinid species, all originally from Asia, includes the bighead carp (*Hypophthalmichthys nobilis*), the black carp (*Mylopharyngodon piceus*), the grass carp (*Ctenopharyngodon idella*) and the silver carp (*H. molitrix*). Collectively, these species are known as Asian carp.

The grass carp, native to southeastern Russia and northwestern China, is an herbivorous species which has been deliberately introduced into many countries for the purpose of controlling aquatic vegetation (Cudmore and Mandrak 2004). In Canada, grass carp escaped from an aquaculture facility in Alberta and were also intentionally released for vegetation control in Saskatchewan. Only a few individuals have been found in the Great Lakes, presumably bought from live fish markets and later released.

The silver and bighead carp, native species from eastern Asia, have also been intentionally introduced into the southern U.S. and the province of Alberta, in this case for the purpose of controlling eutrophication in lakes and ponds (Herborg *et al.* 2007). These species feed on phytoplankton and small zooplankton and may compete with juveniles of many native fishes. These two species have been rapidly dispersing up the Mississippi basin (Mandrak and Cudmore 2004).

The black carp, which is a molluscivorous species, was introduced in the southern United States by fish farmers to control parasite-carrying molluscs (Herborg *et al.* 2007). Subsequently this species also found use in human consumption, in addition to its role as a biological control agent for the control of non-native snails in catfish aquaculture ponds in Arkansas and Mississippi. The species has also been studied as a potential biological control agent for zebra mussel (Buck *et al.* 2010). Of the four Asian carp species discussed in this section, the black carp has the most limited distribution in the U.S. (Buck *et al.* 2010), although repeated occurrences in the Mississippi River (Herborg *et al.* 2007) suggest that there might be an established population in this basin.

5.1 Life History

These four species of Asian carp share many common life history traits. Their reproductive capability, population densities, feeding habits, broad climate tolerance, mobility, and longevity all strongly suggest that the species are highly likely to cause ecological and economic effects once populations become established (Conover *et al.* 2007).

5.1.1 Grass Carp

The typical life span of wild grass carp from their indigenous home in the Amur basin is 5-11 years; although based scale samples indicate that the maximum life span may be as much as 15 years. As with most species, growth in grass carp is a function of age and size, as well as abiotic factors such as density, nutrition, temperature and oxygen. In temperate areas of the United States, grass carp have been observed to be sexually mature at 4-5 years. Water temperature required for the stimulation of sexual maturation and spawning ranges between 20°C and 30°C. However, grass carp have been shown to spawn at water temperatures as low as 15°C.

A well-defined spring and summer spawning period occurs in temperate latitudes. Grass carp prefer to spawn during high water periods in sandy areas of high flow in the primary channels of rivers and canals (Cudmore and Mandrak 2004). Increases in water elevation exceeding 1 meter over a 12 hour period are required. Female fecundity is proportional to length, weight and age and can range as high as 2 million eggs, averaging 0.5 million for a 5 kg female. In the United States, eggs have been observed to hatch in

26-60 hours at 17-30°C (NatureServe 2003). Within a few days of hatching, larvae must enter quiet waters for rearing (Fedorenko and Fraser 1978). Juveniles (age 1-10 years) may move out of these nursery areas and migrate upstream or downstream as much as 1,000 km from the original spawning grounds. Survival is probably low for the early stages from egg to fingerling, particularly in the first week after hatching. Survival of fingerlings in ponds ranged from 23-60%.

5.1.2 Black Carp

Black carp reach maturity at 6-11 years of age in their native Asian habitat (USACE 2012). In their native habitat spawning occurs annually, coinciding with high water levels and an increase in water temperatures to 26-30°C, similar to grass carp. Females are capable of producing up to 1.2 million eggs annually, depending upon body size. Fertilized eggs drift with the current until reaching areas of low flow in areas such as floodplain lakes, smaller streams, and water channels. The early life history of the species has four larval stages two fry stages. The rate of growth rate before sexual maturation is determined especially by quality and quantity of food

5.1.3 Bighead and Silver Carp

Both silver carp and bighead carp are filter-feeders, primarily consuming phytoplankton and zooplankton. These species typically spawn in larger rivers, but inhabit lakes, backwaters, reservoirs and other low-current areas for most of their life cycle. (Buck *et al.* 2010).

A study of silver carp indicates that to reach maturity they require an average about 2,700 degree-days each year over several years. In a northern native population, males normally mature in 4-10 years and females in 6-10 years (Gorbach and Krykhtin 1981); maturity can be delayed by up to 2 years by cold conditions. In North America maturity has been seen as early as 2-3 years (Kipp *et al.* 2011). Once mature, bighead carp require about 650 degree-days above 15°C to reach a spawning condition, with mass spawning seen above 930 degree-days. In its native range, the fecundity of silver carp ranges up to 5.4 million eggs per female (Kolar *et al.* 2007), but in North America, the maximum is slightly lower, at 3.7 million eggs (Kipp *et al.* 2011).

Bighead carp are only known to spawn in rivers and it is believed that a flood event is the primary trigger for spawning. In its native range, the maximum fecundity of bighead carp is about 1.1 million eggs (Kolar *et al.* 2007), with a slightly greater maximum of 1.6 million eggs in North America (Kipp *et al.* 2011). Once eggs are released and fertilized, they remain suspended in the current until they hatch (Kolar *et al.* 2007). Due to flow requirements, unobstructed reaches of 50 km to 100 km in large rivers are traditionally considered necessary for successfully development of eggs (Kipp *et al.* 2011). Hatching time is also related to temperature (Kolar *et al.* 2007). Newly hatched bigheaded carp larvae are around 6 mm in length and they move to productive habitats (*e.g.*, wetlands) for feeding.

5.2 Physiological Limits

Herborg *et al.* (2007) predicted the potential range for Asian carp in North America based upon ecological niche modeling. Their study concluded that the four species of Asian carp can potentially establish widely in North America (see Figure 5.1), indicating their ability to thrive across a wide range of temperatures and accounting for the relative low of importance of temperature as a limiting factor. However, Asian carp did show a preference for humid environments and areas with mean daily precipitations in the range of 5 to 60 mm and with 58-185 days of precipitation/year appeared to be the most suitable areas for these species, indicating their preference for large rivers.

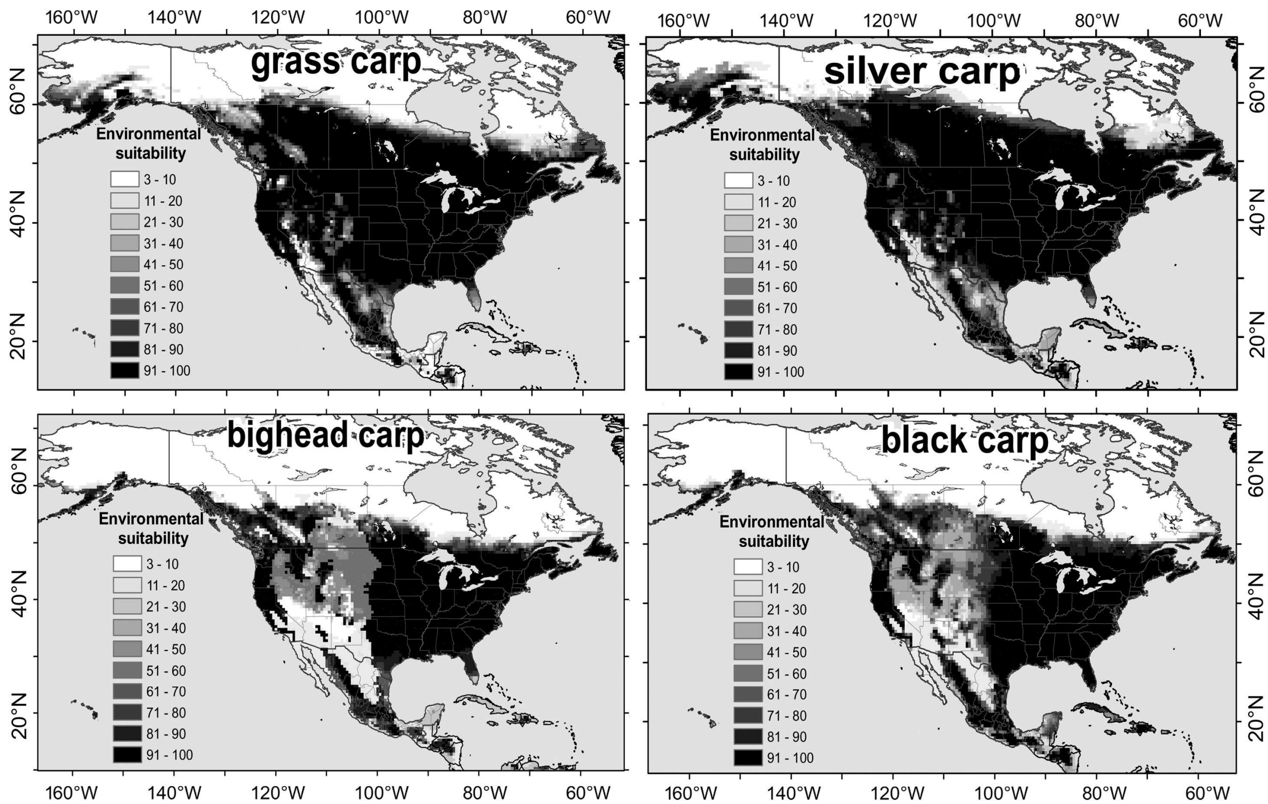

Figure 5.1: Potential distribution of Asian carp species in North America, based on climate suitability. Red areas show high suitability. The green and pink dots represent locations from the U.S. Geological Survey registry, where populations of Asian carp have been established or reported. Top left: grass carp (*C. idella*); top right: silver carp (*H. molitrix*); bottom left: bighead carp (*H. nobilis*); bottom right: black carp (*M. piceus*). (Source: Herborg *et al.* 2007)

5.3 Dispersal

The maximum annual rate of upstream movement by Asian carp in major rivers in the United States has been estimated at approximately 80 km (Minnesota Department of Natural Resources 2004). They can become established approximately two years after the arrival of the first individuals. In the case of grass carp, ten years after stocking in the Mississippi drainage system, individuals were found almost 2,700 km from the original release point, highlighting the strong potential for dispersal in this species (Moyle 1986). Size is a factor in determining movement capability, with larger individuals moving greater distances than smaller ones (Gorbach and Krykhtin 1988, Bain *et al.* 1990). The presence of grass carp in Canada is so far limited to the Great Lakes basin and the provinces of Alberta and Saskatchewan, as shown in Figure 5.2.

Figure 5.2: Distribution of grass carp sightings in Canada. (Source: Cudmore and Mandrak 2004)

Few studies have assessed the movement of bighead carp in natural environments. Movement rates of bighead carp reported from riverine telemetry studies have varied. Peters *et al.* (2006) conducted telemetry studies on the Illinois River during the summer and found an average movement rate of 1.7 km/day but with significant outliers of up to 14 km/day. So far, reports of bighead carp have been confined to the Great Lakes basin, as shown in Figure 5.3.

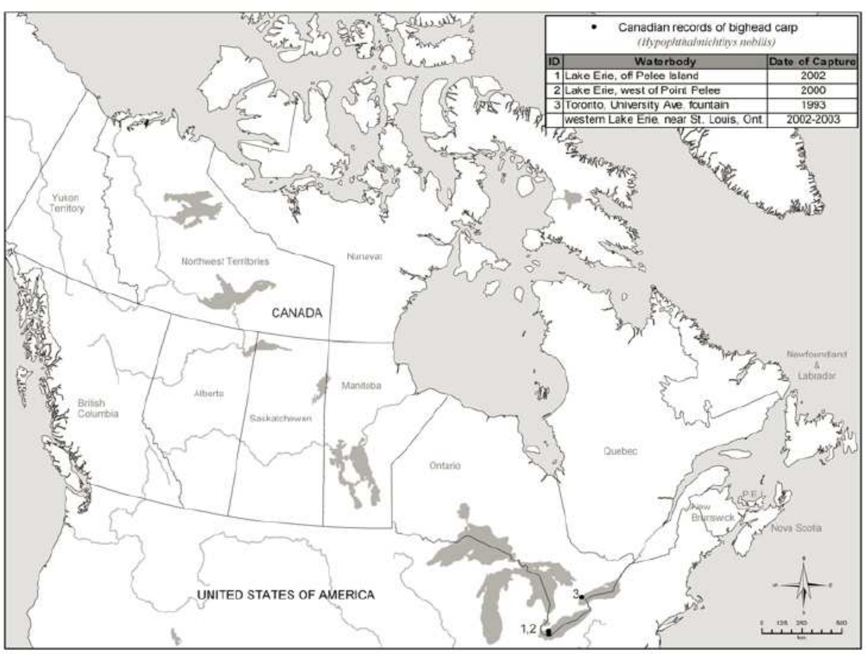

Figure 5.3: Observations of big head carp in Canada. (Source: Mandrak and Cudmore 2004)

Outside of Canada, feral bighead, grass, and silver carp have all established reproducing populations in several major rivers of the United States. By 2007, there were at least 14 confirmed collections of adult black carp by commercial fishers in the United States (Conover *et al*. 2007). The current distribution of Asian carp in the U.S. is shown in Figure 5.4.

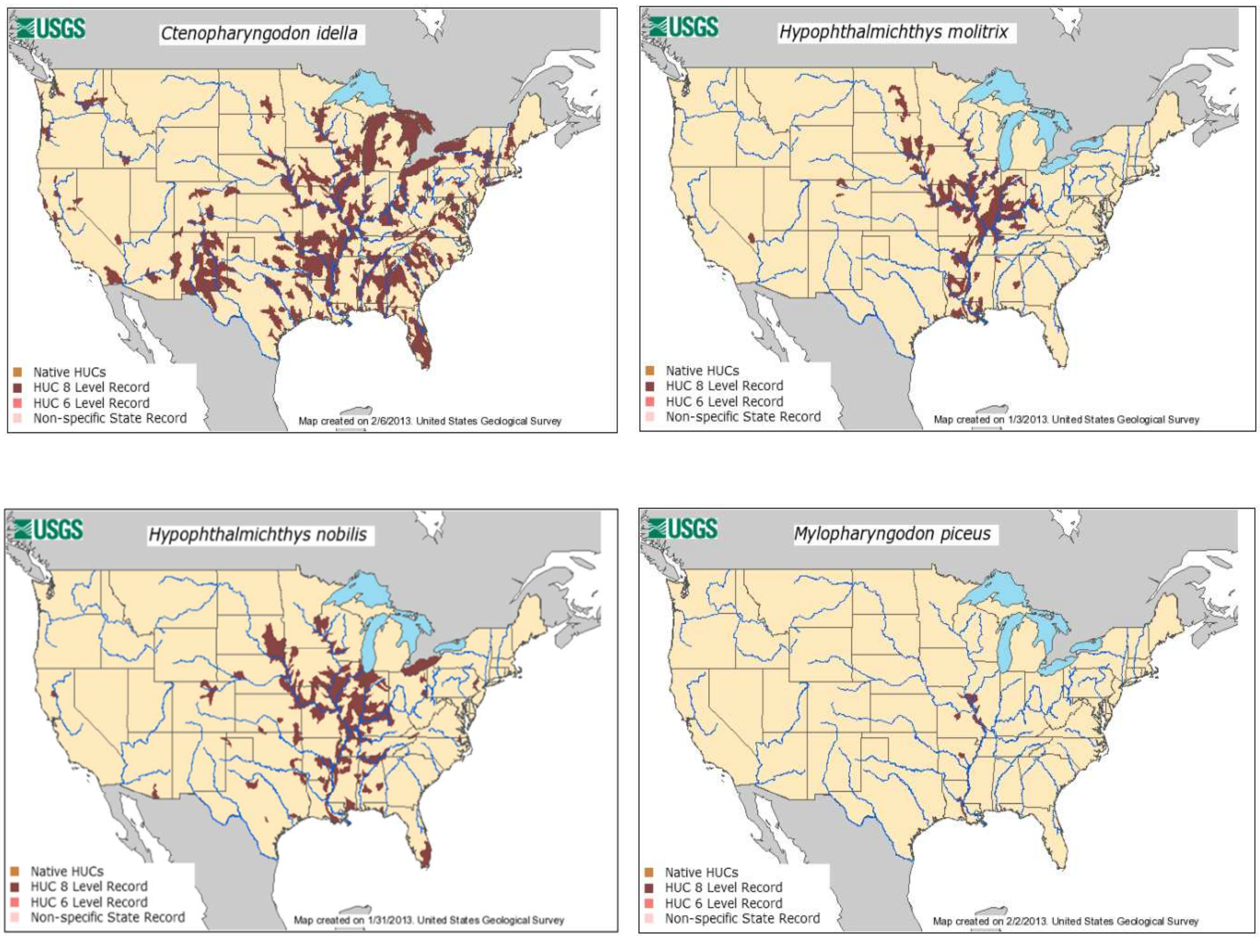

Figure 5.4: Distribution of Asian carp species in the United Sates: Top left: grass carp (*C. idella*); top right: silver carp (*H. molitrix*); bottom left: bighead carp (*H. nobilis*); bottom right: black carp (*M. piceus*). (Source: USGS 2013)

5.4 Impacts

The life history traits of Asian carp (*e.g.*, reproductive capability, population densities, feeding habits, broad climate tolerance, mobility, and longevity) indicate that these four species have a high probability of causing ecological and economic effects where populations become established (Cudmore and Mandrak 2004, Conover *et al*. 2007).

Direct ecological effects are likely to result from the specific diets of each species: silver carp eat phytoplankton; bighead carp eat zooplankton; black carp eat invertebrates such as snails and mussels, and grass carp eat aquatic plants (Buck *et al.* 2010). Each species has the potential to compete with native species for food. The indigenous species at greatest risk include native mussels, other aquatic invertebrates, and fishes.

Based on the results of its introduction around the world, the establishment of the bighead carp in Canadian freshwater ecosystems would likely result in negative impacts on the food web and the trophic structure of these aquatic systems (Mandrak and Cudmore 2004) by inducing changes in the lower trophic levels. Both bighead and silver carps predate on zooplankton and compete with native planktivorous fish species. However, the particular effects on native ecosystems of these two invasive species are still not completely understood (Nico 2013).

As for the grass carp, the introduction of this species into an aquatic system has been shown to directly, or indirectly, impact aquatic macrophytes, water quality and aquatic fauna including plankton, benthic macroinvertebrates, fishes and wildlife (Cudmore and Mandrak 2004). Since adult grass carp generally make up most (95%) of their diet from macrophytes, and can cause significant changes in the composition of macrophyte, phytoplankton, and invertebrate communities

The black carp can negatively impact native aquatic communities by feeding on and reducing populations of native mussels and snails (Nico *et al*. 2005). They can also restructure benthic structure by direct predation and removal of algae-grazing snails.

Figure 5.5 shows the potential impacts and damage paths that could be created through the introduction of Asian carp.

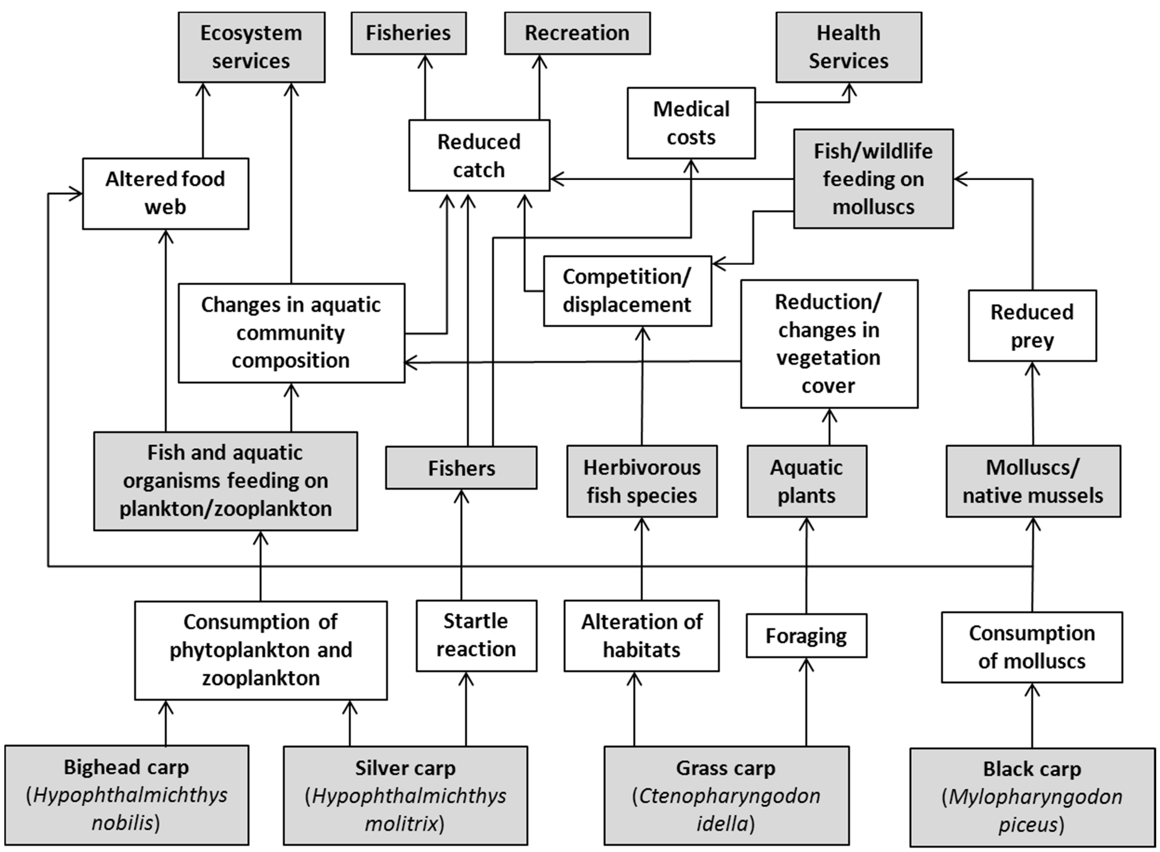

Figure 5.5: Impact diagram for 4 species of Asian carp.

5.5 Potential Control

Common control methods include physical and behavioral barriers. Nevertheless, eradication of non-native species in aquatic environments is difficult and rare, having only occasionally been successful when efforts were focused on small-scale and closed systems like reservoirs, ponds, small locks, and marinas (Buck *et al.* 2010). Besides, the feasibility of eradication efforts such as selective harvest in conjunction with other mitigation efforts has not been evaluated (Garvey *et al.* 2007). An Asian Carp Working Group (Conover *et al*. 2007) has been formed in the U.S. to manage and control the spread of Asian carp in the U.S.

5.6 Overview of Economic Damages

The economic damages resulting from the Asian carp invasion of North American waterways have not been quantified, and gathering economic damage data is extremely difficult (Brooks, *pers. comm.*; Irons, *pers. comm.*). The damages from Asian carp could exceed those caused by previous sea lamprey and zebra mussel invasions. The cost of controlling sea lamprey has exceeded USD 300 million to date (ACRCC 2012). It is expected that invasive Asian carp threaten the recreational fishing and angling fisheries, commercial fisheries. Silver carp are of particular concern as they can leap out of the water when startled, as documented on YouTube. This behavior can damage boats, destroy expensive electronic gear and deter anglers and pleasure boaters (Brooks, *pers. comm.*). Numerous boaters have sustained injuries requiring medical attention as a result of silver carp (Buck *et al.* 2010).

5.7 Potential Damages to the B.C. Economy

Asian carp have not yet invaded B.C., thus the cost of an invasion is unknown. However it is clear that Asian carp will have a negative effect on freshwater recreational fishing in B.C. waters, the magnitude of the effect on ecosystems and the wider economy they support is not well understood. Even in regions of North America that are currently plagued by Asian carp, researchers have been faced with the challenge of making accurate predictions on changes in lake and river species composition. The potential results from the river species composition are even more difficult to predict because ecological and economic systems are complex (Buck *et al.* 2010).

What is clear is the importance of recreational fishing and angling to the B.C. economy. The expenditures on fishing goods and services have direct, indirect, and induced impacts on jobs, income, and the larger economy. Freshwater angling is a year round industry in B.C.'s 200,000 lakes and 750,000 km of streams and rivers (G.S. Gislason & Associates 2009). B.C. anglers target 24 popular species of game fish, with rainbow trout being the most prized species. Angling occurs within all of the Ministry of Environment designated resource management regions. With the exception of the northern regions of the Skeena and Peace Rivers all these regions are within the potential distribution of the Asian carp (Figure 5.1).

The fresh water recreational industry in B.C. will be under threat if Asian carp, which is not a species targeted by anglers, replaces sport fishes. In 2005, the freshwater fishery of British Columbia had license sales of 340,000, nearly 4 million angler-days, CAD 480 million in angler expenditures and about 8.2 million fish caught. The economic impacts from these expenditures resulted in CAD 131.5 million in GDP, CAD 75.1 million in wages, 7,500 jobs and CAD 83.3 million in government taxes (G.S. Gislason & Associates 2009). However, estimates of the "consumer surplus" value of recreational freshwater fisheries are not available and would need to be estimated using stated or revealed preference valuation methods. Consumer surplus represents the difference between what recreationists are willing to pay, and together with the amounts they actually pay, are the correct measure of welfare to use when assessing changes in the quantity and quality of the fishing experience (not industry expenditures).

At this point, there is no information to link how these economic welfare measures might change and the potential extent of an Asian carp invasion. Certainly, the damage that Asian carp may have on recreational fishing will depend on the extent of their displacement of native sports fish. An investigation of fish kills in waters of a National Wildlife Refuge in Missouri discovered Asian carp comprised 97% of the biomass, indicating that Asian carp can potentially displace virtually all other fish (MICRA 2002). In addition, commercial fishers reported Asian carp overwhelming their fishing nets. In the Illinois River, fishers routinely haul catches upwards of 11,000 kg of bighead and silver carp (Irons *et al.* 2007). In Kentucky, it is reported that fishing guides have quit fishing in Kentucky and Barkley lakes' tail waters because of the high number of Asian carp in those waters (Brooks, *pers. comm.*). Asian carp would also be a hazard to boaters and water skiers because of the jumping behavior of the silver carp when startled. Damages will be inflicted on the boats, boating equipment, and passengers, leading to injuries and medical costs.

6. Sitka Black-tailed Deer

The Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) is native to the wet coastal rain forests of Southeast Alaska and north-coastal British Columbia. They were introduced in the late 19th century in the Haida Gwaii archipelago as a source of game (Stroh *et al.* 2008). In the absence of native deer and of their natural predators, and in a context of mild winters with little or no snow on the ground, the introduced deer proliferated and colonized most of the archipelago. Their impact on the regeneration of commercial tree species and on the structure and resilience of the forest ecosystem of Haida Gwaii generally is now well-documented.

6.1 Life History

As with most deer species, Sitka black-tailed deer populations can increase rapidly in number. Females usually do not breed until their second year. After the first reproduction, females usually produce annually throughout an approximate life span of 10 years. Typically, fawns are born in early June, weighing approximately 3 kg at birth. Reproductive rates may be higher on Haida Gwaii, although reproductive data are limited (Gillingham 2008). It is estimated that in Haida Gwaii the onset of reproduction occurs in the first year, with females reproducing every year throughout their approximately 7- to 10-year life span. In addition, most litters tend to be of twins.

In the absence of control mechanisms other than food availability, deer can reach high densities. The current estimated population ranges between 113,000 and 250,000 individuals on 8,500 km^2 of available habitat. Average population density on the larger islands has been estimated at about 13 deer/km^2 (Englestoft 2001). However, on smaller islands covered by old-growth forest density estimates reached 33 deer/km^2 (Daufresne and Martin 1997). These densities are higher than those found on the mainland, where the presence of natural predators and more severe winters are likely to keep deer populations in check.

6.2 Physiological Limits

Several studies have demonstrated that snow can be a major factor influencing winter survival of Sitka black-tailed deer because of the reduction of available forage and the increased energetic costs of moving (Gillingham 2008). The energy cost of moving through 25cm of snow is about 2.5 times that of moving through 10cm and increases as snow packs become deeper. It has been found in southern Alaska (Gillingham 2008) that snow depths higher than 15cm in the open significantly reduce movement (and foraging opportunities) of Sitka deer. However, winters in coastal forest tend to be less severe and usually present a discontinuous snow cover for much of the winter.

The temperatures below which deer are thermally stressed and must begin to increase metabolic rates to maintain an acceptable body temperature vary with seasonal pelage (Parker 1988). In winter, the lower critical temperature for black-tailed deer is –6ºC; in summer, it is +12ºC.

6.3 Dispersal

After their introduction into the archipelago in the North of Graham Island (Masset area) in the late 19th century, Sitka back-tailed deer increased rapidly due to the mild climate, lush vegetation, absence of wolves and cougars, low hunting pressure and a lack of large competing herbivores.

Reimchen *et al.* (2005) estimated a persistent colonization rate to offshore islands in the Haida Gwaii archipelago of one deer/year. As this study was done for the more remote islands in the archipelago, there will be probably even higher migration rates among the more proximate islands.

It should be noted that black-tailed deer are not herding animals (Bunnell 1990) and usually do not move as large groups but in smaller groups of 3-4 animals. Seasonal movements are limited (Gillingham 2008) compared to those of the populations on the mainland and deer in the archipelago do not show a migratory behavior. The current distribution of Sitka deer in British Columbia is shown in Figure 6.1

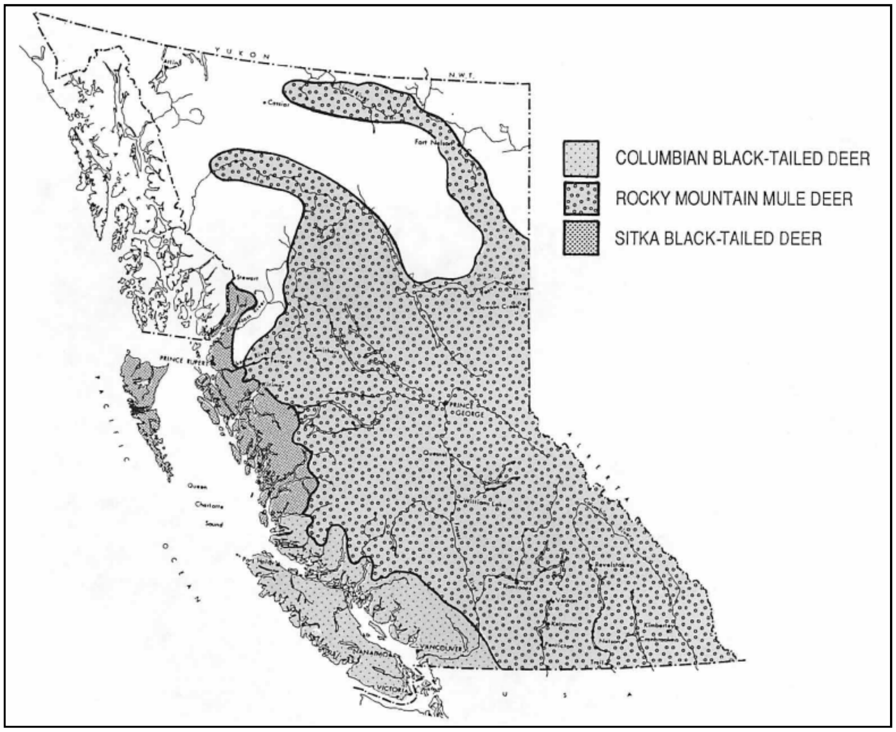

Figure 6.1: Distribution of Sitka black-tailed deer (Source: Bunnell 1990)

6.4 Impacts

The effects that the introduction of Sitka black-tailed deer has had in the Haida Gwaii archipelago has been studied since the 1990s (Golumbia 2000). In particular, deer browsing has reduced abundance and vigor of virtually all species of shrubs and herbs; in extreme cases, the understory structure of the forest is absent. The indirect effects on forest structure affect all animal species using the canopy and the forest floor as habitats for feeding, foraging or nesting. More subtle effects are likely occurring at the forest floor with regard to nutrient cycling, terrestrial arthropods, and soil structure.

Recent studies by Stroh *et al.* (2008) have shown how the introduction of Sitka deer in Haida Gwaii has dramatically affected the regeneration of woody species in both old- and second-growth forests. The reduced recruitment of western redcedar (*Thuja plicata*) seedlings in old-growth forests has been attributed to the effects of black-tailed deer, which have a strong preference for western redcedar (*Thuja plicata*) shoots, one of the three dominant conifers in this coastal maritime forest.

Selective foraging by deer also affects plant species composition through changes to the absolute and relative abundance of woody species, and can significantly alter the forest structure (Rooney 2001).

Stockton *et al.* (2005) tested the long-term effects of high Sitka deer densities on plant cover and species richness in the understorey of forest interior and forest edge habitats of Haida Gwaii. These results of this study suggest that in the absence of predators, deer have the potential to greatly simplify the forest ecosystem.

Figure 6.2 shows the potential impacts associated with the presence of Sitka black-tailed deer.

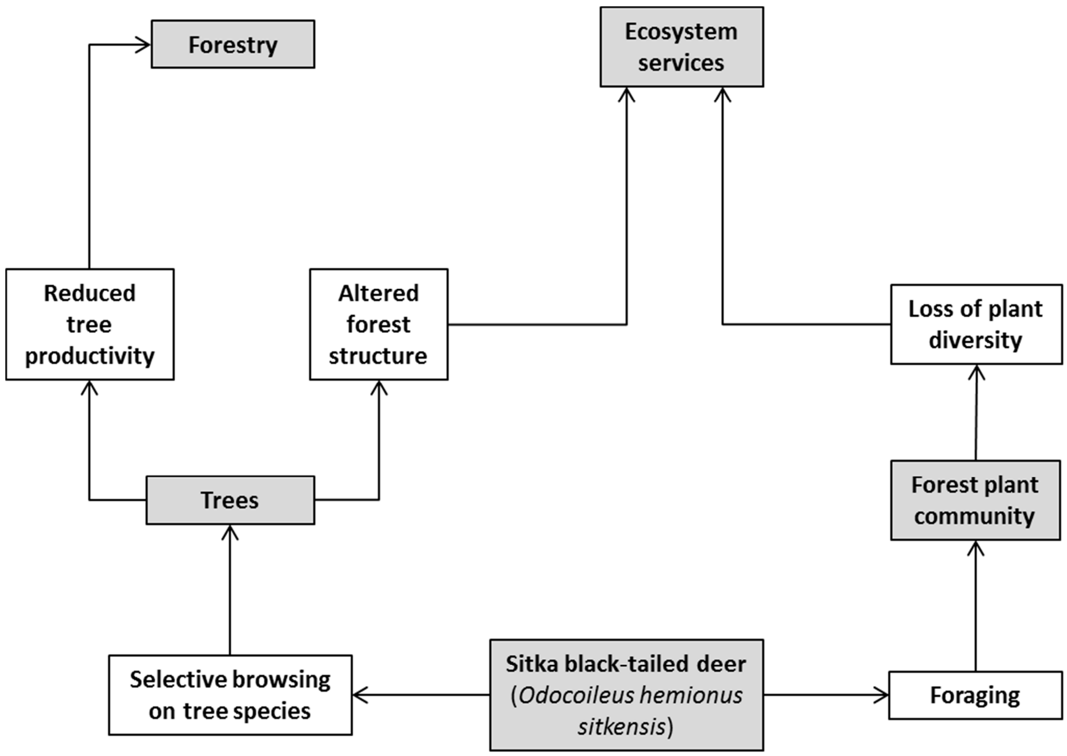

Figure 6.2: Impact diagram for Sitka black-tailed deer (*Odocoileus hemionus sitkensis).*

6.5 Potential Control

Hunting quotas can have a localized effect on deer numbers. These have been in place throughout Haida Gwaii and have shown positive effects on redcedar survival in the areas under hunting (Golumbia 2000). While eradication of Sitka deer in the Haida Gwaii archipelago is not considered socially desirable (Golumbia 2000), effective control seems to be possible only in the islands of the archipelago sufficiently distant from a population source. To test this concept, an experimental deer cull was undertaken in the 1990s on two islands and re-establishment of the herb and shrub layer was reported after only one growing season.

6.6 Overview of Economic Damages

Deer herbivory inflicts damage to commercial forestry by browsing of restocked seedlings, which limits growth, reduces stem quality, and even kills the planted trees (Putman 2012). Additionally, mature stands of forest are damaged by bark stripping and fraying during territorial displays or when bucks clean the velvet from their antlers (Mayle 1999). Forests that rely on natural regeneration are particularly vulnerable, as the impact on seedlings affects subsequent recruitment (Reimoser 2003). These impacts result in losses that are twofold. First, valuable timber is lost through lack of regeneration and, second, costs are incurred attempting to reduce the damage by protecting seedlings by fencing, employing tree guards or culling deer (Putman 2012). In Pennsylvania, foresters incur costs of USD 100-500/hectare to regenerate sites browsed by deer (Redding 1987 cited in Witmer and DeCalesta 1992). Conover (1997) estimated that USD 367 million in 1994 prices occurs on 9% of commercial forestland. The cost on the remaining 91% may be far greater. The situation is especially acute in Haida Gwaii, where intensive deer browsing has resulted in almost no recruitment among unprotected western redcedar seedlings (Muise, *pers. comm.*). The cost to replant and protect western redcedar in Haida Gwaii is approximately CAD 7.00/stem, compared to about $1.50/stem for less vulnerable species (Muise, *pers. comm.*).

In addition to the economic damages affecting commercial forestry, deer browsing has direct non-monetary costs associated with the loss of ecosystem services, e.g. biodiversity loss through elimination of vulnerable species. Daufresne and Martin (1997) document cases where the entire understory of forests is absent. The indirect effects, which are less apparent, include loss of food sources for other species. Changes in the physical forest structure results in loss of habitat for nesting and cover for birds and small mammals (Daufresne and Martin 1997). The habitat of the blue grouse has been affected particularly severely, having ramifications for the threatened northern goshawk that preys on it (Muise, *pers. comm.*). Furthermore, deer browsing may be exerting subtler, unmeasured effects related to nutrient cycling and soil structure (Golumbia 2000). However, measuring the economic value of changes in ecosystem services due to the introduction of the black-tailed deer would be challenging.

6.7 Potential Damages to the B.C. Economy

In this section we make a preliminary estimate of some of the components of a complete estimate of the damages to commercial forestry on Haida Gwaii from black-tailed deer. We make a large number of assumptions and simplifications so that the values must be viewed as notional and not precise. However, it is hoped this will point the way towards more rigorous calculations in future. The basis for the calculation here is that in the worst case the unrestricted browsing by deer leads to total eradication of western redcedar as a commercial timber species. To correctly make such an estimate requires consideration of a host of issues, such as the unrelated conversion of production forests over time to a species mix that is different than at present. For example, as old growth cedar is harvested it may not be replaced by replanting of redcedar, regardless of the presence of deer. Consequently, a different harvest mix might be expected in the future under all scenarios. For now, we assume that the harvest in 2012 is representative of a typical harvest and abstract from the issue of future harvest mix. According to the B.C. government's harvest billing system the harvest of cedar logs in 2012 was 312,387 m^3 from a total log harvest for all species of 545,205 m^3. The simplest estimate of deer-induced losses would be to value the loss of this volume and its replacement with the commercial – and lower-valued – species likely to replace cedar, namely western hemlock and perhaps Sitka spruce. In economic terms the procedure involves determining the loss in producer surplus associated with the shift from harvesting cedar to alternative, lower-valued species.

To make this calculation requires several steps and various assumptions. To generate an estimate of revenue earned through harvest, we first estimated a grade-weighted average price/cubic meter for each species using monthly data for 2012. To accomplish this we used a combination of the Coastal Log Market Reports, which report average prices/m^3 of old growth logs by species and grade, and harvest data for the Haida Gwaii Forest District provided by the Harvest Billing System. While only a minority of logs are bought and sold on the Vancouver Log Market, it is believed that the prices reported in the Log Market Reports accurately reflect value. The Harvest Billing System provides data on the volume/grade and per species harvested within the Haida Gwaii Forest District. We multiplied this volume by the grade and species-specific average prices for 2012, quoted in the Log Market Reports for the coastal region. Our estimated price for old growth cedar is $126.99/$m^3$, hemlock $49.65/$m^3$ and spruce $79.93/$m^3$.

Since our analysis concerns regenerated timber supply we need to consider second and not old growth values. Second growth log prices are lower than that of old growth logs. Log market reports are not available for second growth logs since the harvest is too small. Instead, we used the second growth price conversion parameters provided in the Ministry of Forests Revenue Branch Coast Log Prices report for 2004,[5] established by the Coast Appraisal Advisory Committee. The conversion table provides estimates of what second growth log prices would be in comparison to old growth prices for similar grades. Second growth log prices were obtained by multiplying the old growth prices from the Log Market Reports by the second growth conversion factors derived by the Coast Appraisal Advisory Committee (B.C. Ministry of Forests 2004). A second growth Grade H hemlock log, for example, would be worth about 75% of a similar old growth Grade H hemlock log. The weighted second growth prices are \$105.58/m^3 for cedar, \$44.03/m^3 for hemlock and \$49.97/m^3 for spruce.

Producer surplus is defined as the difference between price and "variable" costs, which are costs that vary with the level of production. In the short run firms will continue operating as long as variable costs are covered, but in the long run firms will only continue operating if all costs are covered. Logging companies in British Columbia have experienced considerable losses within the past decade, so considering costs is especially important when lower-valued second growth is involved. The Coast Forest Products Association has estimates for total harvest costs for 2004 and 2005 and these appear to be the most recent available for the coastal forest region. These cost estimates include harvest costs, such as falling, yarding, loading, and delivery; head office and administrative costs, such as operational overhead and road building; as well as silviculture and replanting costs. The cost estimates do not include government administrative costs or stumpage and other royalties paid by harvesting firms to the government. We use an average of the total harvest for the two years from the Coast Forest Products Association report and update to 2012 using the CPI, yielding a value of \$89.22/m^3. Other data suggests that variable costs are on average about 57.7% of total harvest costs (Knowler and Dust 2008), so that an estimate of variable harvest cost in 2012 is \$51.50/m^3.

A very simple estimate of damages from deer browsing would start with the value of foregone producers' surplus if the current harvest of cedar were replaced with hemlock or spruce. Our assumptions suggest that the difference in price for second growth hemlock versus cedar in 2012 was \$61.55/m^3 (\$105.58 - \$44.03). Multiplying this value by the volume of cedar harvested in 2012 yields \$19.23 million/year (\$61.55 x 312,387 m^3). If cedar was replaced with spruce in 2012 the difference in price is \$62.61/m^3 (\$105.58 - \$42.97). Multiplying this value by the volume of cedar harvested in 2012 yields \$19.56 million/year (\$62.61 x 312,387 m^3). Note that since we assume harvest costs are identical for all species these simply drop out of our calculation and the loss in producers' surplus is the difference in revenues obtained/ m^3.

There are several important issues with this calculation. First, if variable harvest costs are taken into consideration then the harvests of second growth hemlock and spruce appear uneconomic. Although forest companies may have other reasons to harvest despite low profitability the value of producers' surplus would be negative. In the worst case, the converted stands of hemlock and spruce might be excluded from the harvested land base in future.

A second concern is that the current harvest is from unmanaged old growth, while we must consider managed second growth in our estimate. Thus, it would be preferred to know the area of cedar harvested in 2012 and then to determine the amount of second growth that would be available at harvest age in the future for each species. But there are differences in stand density and growth rates when considering the different species. Cedar generally grows in lower densities and more slowly than hemlock or spruce so

[5] ftp://ftp.for.gov.bc.ca/HET/external/!publish/Web/exports/second-growth-conversion-table.pdf

that volume-at-age curves would show differences across species and typically higher values at harvest for a hectare under hemlock or spruce versus cedar. Data for volume-at-age by species are provided in the Haida Gwaii (Queen Charlotte) Forest District Timber Supply Area (TSA) report. Assuming a minimum allowable harvest age of 110 years for all species from the Public Review Period for an Annual Allowable Cut Determination report (Haida Gwaii Management Council 2011), the ratio of stand volume for other species versus cedar are 1.68 for hemlock and 2.45 for spruce. A more credible calculation of the losses from deer browsing would use this information and it likely would result in a smaller measurable loss from deer browsing because overall yields would be smaller than from the current old growth stands and the gap between cedar and other species would be lower as well.

Finally, the harvest from 2012 that we use is only representative of a single year and may not be particularly close to the long term sustainable yield of cedar. The annual allowable cut for Haida Gwaii suggested in the Haida Gwaii TSA report is about double the harvest of 2012 so there seems to be considerable scope for higher yields. But the share in this harvest that is cedar might be much smaller than at present as the old growth stands are slowly liquidated. A preferred approach to the one taken above is to determine the long term AAC, the share that would be cedar and to use this in the calculation, assuming it is replaced by hemlock and/or spruce. The Haida Gwaii Forest District TSA contains a sensitivity analysis involving deer-induced effects on cedar and this might be a starting point for a more rigorous analysis of damages (B.C. Ministry of Forests 2000, p66).

7. European Starling

The European starling (*Sturnus vulgaris*), native to Europe, southwest Asia and northern Africa, was purposefully introduced in New York in the late 19th century (Linz *et al.* 2007) and has since then colonized most North and Central America.

7.1 Life History

Starlings can have two broods a year with four to five eggs in a brood. Incubation of the eggs takes 12 days and the fledglings leave the nest after 25 days. The young leave to join other juveniles and form flocks that move on to other territories.

Eggs are laid from late March to early July, depending on latitude. Most starlings produce 1 to 2 clutches per year of 4-6 eggs each, with birds above 48° N producing only one clutch (Linz *et al*. 2007). The average life span is about 2-3 years, with a longevity record of over 20 years.

Starlings are prolific and have a 48% to 79% rate of nest success. Even so, only 20% nestlings survive to reproduce. Adult survival is much higher, probably around 60% (Linz *et al*. 2007).

European Starlings are dietary generalists, eating a variety of invertebrates, such as snails, worms, millipedes, and spiders, in addition to fruits, berries, grains, and seeds. They forage in flocks year round; flock size depends upon the time of year and availability of food. In winter, they often forage in mixed-species flocks with cowbirds and blackbirds.

7.2 Physiological Limits

Cold and wet weather and extreme hot weather can contribute to mortality of nestlings, with both factors affecting the availability of an important food source, temperature sensitive invertebrates. European starlings are habitat generalists (Linz *et al*. 2007) and have year-round populations established in most of North America, as shown in Figure 7.1.

7.3 Dispersal

Once established at a site, starlings have a high degree of breeding site fidelity (Linz *et al*. 2007). The young-of-the-year disperse widely and find new breeding sites, often far away from their natal site, which may account for their high dispersal potential.

Starlings are strong flyers and can, if necessary, migrate distances of 1,000-1,500 km, especially to escape heavy snow that covers food sources (Linz *et al.* 2007). They can migrate long distances in a single day at speeds of 60-80 km/h.

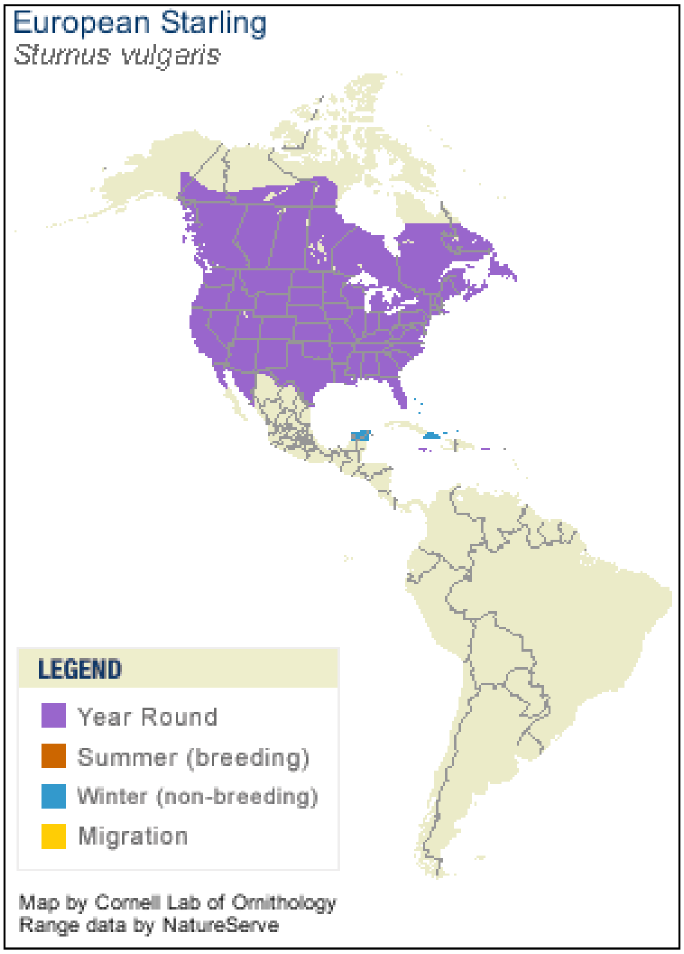

Figure 7.1: Distribution of European starling (*Sturnus vulgaris*) (Source: Cornell Lab of Ornithology 2012)

7.4 Impacts

European Starlings cause damage to agricultural crops. When significant numbers are present starling flocks may descend on fruit and grain crop fields to forage, causing massive damage, and having serious economic consequences. In addition to damaging and consuming tree fruits, grape and berry crops, starlings can be disease vectors for Salmonellosis, Chlamydiosis, Johne's Disease, avian tuberculosis and histoplasmosis. They are also aggressive competitors and are relentless in taking over nesting cavities that would otherwise be used by bluebirds and other native songbirds. (Bielert and Hol 2008).

The use of urban areas by wintering flocks of starlings seeking warmth and shelter for roosting can have serious consequences (Linz *et al*. 2007). Large roosts in buildings and industrial structures cause filth, noise, odor, and health and safety hazards. Additionally, the droppings are corrosive to infrastructure. These impacts are represented in Figure 7.2.

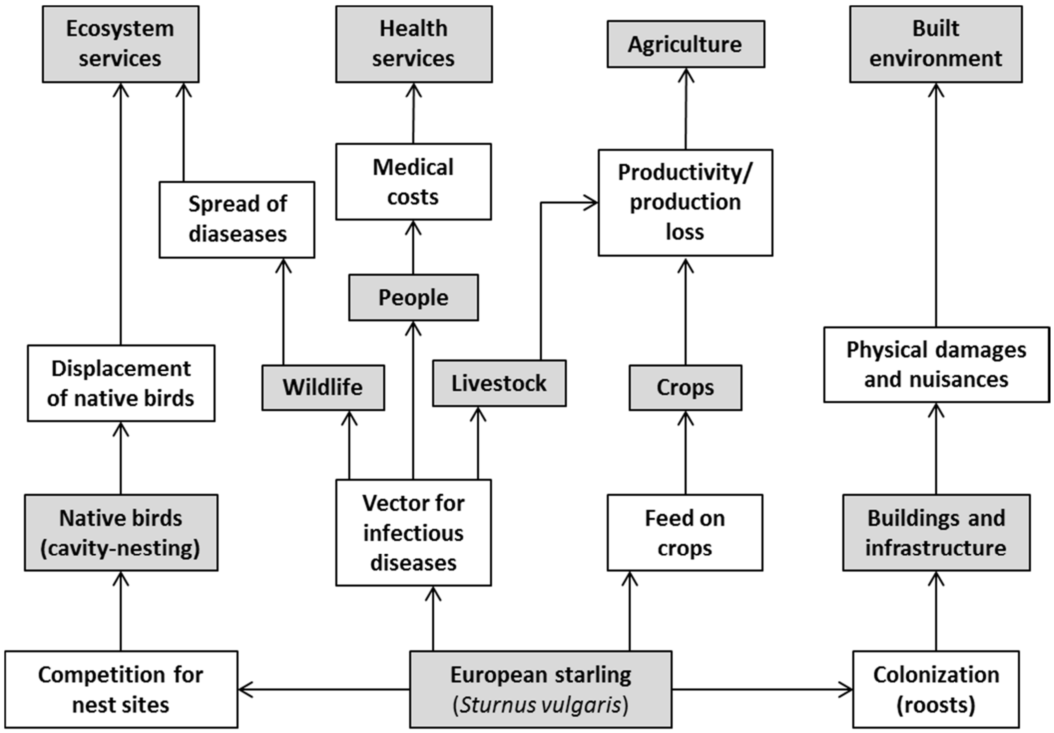

Figure 7.2: Impact diagram for the European starling (*Sturnus vulgaris*).

7.5 Potential Control

Noise deterrents, such as propane cannons or electronic distress calls, and visual repellents such as reflective tape, have been used to frighten starlings away, with decreasing effectiveness. Netting of fields has also been used to prevent crop loss, but it is expensive and not practical for tree fruits.

Some producers hire commercial falconers to assist in the control of starlings in their vineyards and orchards. Native raptor species can be effective in directly preying, or chasing starlings, but consistent presence is difficult to guarantee.

In some agricultural areas, like in the Okanagan- Similkameen region, farmers have joined to implement coordinated control programs. The Starling Control Project, aimed at reducing starling populations through an aggressive trapping program, began in the Okanagan-Similkameen in 2003 (B.C. Grape Growers' Association 2010a) and it is currently applied in three regional districts. The method of control is through traps located in feeding sites, such as feedlot areas and other cattle operations and landfill sites. Once trapped, the birds are euthanized.

Preliminary observations from the data collected for the Starling Control Project suggest that while the breeding population may be declining, the winter population continues to increase, likely due to migration from other geographic areas (B.C. Grape Growers' Association 2010b).

7.6 Overview of Economic Damages

The European starling is one of the most damaging invasive species in North America (Lowe *et al.* 2000). Despite starling population control efforts to avoid economic damages, starlings are responsible for USD 800 million worth of annual damages to feed lot operations, field crops, fruit orchards, and vineyards

(Linz *et al.* 2007). An additional USD 800 million worth of damages is estimated in annual treatment costs for livestock and human due to disease transmission (Linz *et al.* 2007). In 1968 the cost of cattle feed consumed by starlings was USD 84/1000 starlings or USD 554 in 2012 prices.

7.7 Potential Damages to the B.C. Economy

Resident and migrant birds significantly affect vine and fruit growers in B.C. Growers incur damages as the economic value of crops lost and the costs associated with starling control. The Okanagan Similkameen grape growing region experiences an annual crop loss to vineyards of CAD 3.5 million in 2008 prices (Bielert, *pers. comm.*). The value is a conservative estimate and is calculated on the basis of tonnage of crop lost and the market value of wine. The B.C. Fruit Growers Association estimate starlings cause annual damages of CAD 2 million (2008 prices) to tree fruits, with a further annual loss of CAD 500,000 to cherry and apple growers (Ransome 2012). The economic impact of starlings on blueberry production in B.C. has been particularly severe as starlings can destroy over 25% of growers' crops (BCFIRB 2009). On average, blueberry growers lose an estimated 7 to 8% of their annual crop due to starlings (Sweeney, *pers. comm.*). This represents a loss of CAD 5.8 to 6.6 million using the farm gate value of blueberry production of CAD 82.6 million in 2010. Starlings not only reduce the amount of crop that can be harvested, but also damage the remaining crop, which reduces the grade the crop receives from processors, resulting in a lower price, and a further loss of producer surplus (BCFIRB 2009).

The preceding damages are residual because growers carry out a number of starling control techniques in an attempt to reduce these damages, as noted above (Tracey *et al.* 2007). The cost of prevention measures can be viewed as a financial cost to farmers and a social cost to society. In B.C., growers have traditionally relied on deterring starlings through the use of bird scare devices like propane cannons. Propane cannons vary in price depending on configuration; however the average price a grower is expected to pay per system is CAD 957.5 (Margo Supplies 2013). Propane cannon control methods have external societal costs that are not accounted for in the market economy. For example, the noise nuisance is said to have a negative impact on surrounding businesses, and may decrease property values around farms that employ propane cannons. Additionally, residents indicate that the cumulative effect of noise from the propane cannons has a negative impact on the community by diminishing the quality of outdoor activities. Furthermore, the cannons have been cited as a cause of health problems related to stress and lack of sleep from the noise especially among night shift workers who sleep in the day time hours (BCFIRB 2009). Other control techniques utilized in the province include trapping programs that have been shown to be effective in the short term. The B.C. Fruit Growers' Association administers a trapping program that removed 183,105 birds from 2008 to 2010. The average cost of the program in 2009 was CAD 114,500, which covered the cost for a team of trappers, supplies, and administration costs (B.C. Grape Growers' Association 2010a).

8. Recommendations for Decision Makers

8.1 General Recommendations

For the invasive species already present in B.C., *i.e.* Sitka black-tailed deer, starlings and European fire ant, there is a strong argument for allocating resources for more primary research into valuation of damages, as this information is lacking. Until B.C.-specific economic data is available a full-scale provincial analysis may not be appropriate. This information can be used to inform policy decisions on regulatory tools, research budgets etc. To date, too little primary research has been carried out in B.C. A PhD or Masters student could undertake the research.

The following two sections provide more specific detail for future economic impact analyses along with a summary of data gaps, respectively.

8.2 Species Recommendations

8.2.1 Zebra and Quagga Mussels

Economic impacts of mussel invasion on fisheries have been poorly studied and additional research on impacts to recreational, commercial and traditional fisheries is required. More researching of the impacts of mussels on waterfront property values (taking care to avoid double-counting) is also needed, since effects will be largely negative.

Further analysis of alternative cost effective and environmentally-friendly mussel treatments also might be explored. Chlorination is an effective treatment, but requires application over a long period of time with a long dose-response time (10-30 days), and can be toxic to the environment.

Several management recommendations also emerge from our analysis. For example, there may be a need for a B.C.-specific rapid response and management plan. Increased monitoring efforts will allow for an early response with greater likelihood of success. Increased outreach and prevention efforts will allow facilities time to put in place mitigation measures.

8.2.2 European Fire Ant

Examples of further research that could assist with evaluating economic impacts of European fire ants include the completion of mapping of the known distribution of fire ants in B.C. and more study of the different behavior of European fire ants compared to red imported fire ants (*S. invicta*) in relation to dispersal and damages. More research into the control options for *M. rubra,* including research on baits, will also be helpful. Ecological research on key issues, such as how *M .rubra* affect or displace native ants and other ecosystem effects, would be useful.

Collaborating with jurisdictions on the East Coast to assess the economic impacts of *M. rubra* might be fruitful. Creating prevention and management programs that foster coordinated and collaborative action by homeowners should be emphasized, as individual control efforts are rarely successful. Education programs targeted at Government Agencies, landscapers, nurseries, and homeowners might also be encouraged.

8.2.3 Asian Carp

Research on the effect of Asian carp on ecosystems, especially their relationship with sports fish, is a key requirement but of continent-wide importance. A specific focus should be given to the effect of Asian carp on pelagic fish, such as salmon, because of their economic and traditional importance to B.C.

There is also a need for a B.C. management plan that includes monitoring, education and public awareness of Asian carp. The management plan should allow for collaboration with Canadian customs and border security, as imported Asian carp destined for live food markets have been identified as one of the most likely invasion pathways.

8.2.4 Sitka Black-tailed Deer

More analysis of the impacts of deer browsing on non-forestry sectors, such as agriculture, may be required, since little is known of the potential for damages outside forestry. Field agriculture is a fairly minor activity on Haida Gwaii, but reports from other jurisdictions indicate that deer can have localized impacts on agricultural crops.

Analysis of deer-vehicle collisions and the associated economic impacts on Haida Gwaii, including damage to vehicles and human injury, is lacking at present. In contrast, more understanding of the potential economic benefits of deer on Haida Gwaii, as a result of hunting, venison sales, tourism and aesthetic values, would provide a more balanced assessment.

On the management side, a coordinated management strategy for all of Haida Gwaii, including crown land, private timber land and conservation areas, could be initiated. Appropriate management involving direct control of the deer population would reduce impacts. Mitigation needs to be cost-effective and environmentally acceptable, but also ethically acceptable to society. For example, managers could consider the nonlethal strategies such as trap-and-transfer and fertility control.

8.2.5 European Starling

The value of additional damage estimates is perhaps less urgent than for other invasive species that may yet invade but should not be dismissed. Clearly, there are societal tradeoffs between starling damages to agriculturalists and the imposition of nuisance costs on surrounding populations. An effective study might shed light on whether control is worthwhile or simply transfers the social costs from farmers to households.

On the management side, a focus on comprehensive management strategies that reduce the economic damages to farmers in B.C. is sensible. Management actions need to be coordinated at a regional and local level to avoid moving the problem to a neighboring property.

8.3 Data Gaps and Future Research

8.3.1 Zebra and Quagga Mussels

A risk assessment by the Department of Fisheries and Oceans for dreissenid mussels conducted on a sub-basin level provided us with an estimate of the potential range and ecological limit of the zebra and quagga mussels. The topographic sub-basins did not correspond to administrative boundaries such as water or regional districts, resulting in challenges creating an inventory of facilities liable to economic damages from mussel infestation. Information on the potential distribution of zebra and quagga mussel on a lake, river, or administrative boundary scale would have helped to refine the accuracy of our economic damage estimates. On a related note, the lack of data on the number of local authority waterworks, and or municipal water utilities proved to be a challenge. Assumptions about the number of water withdrawal facilities had to be made based on the number of unique licensees that held a license in the Waterworks Local Authority category. Some licensees, such as municipalities, have more than one water license and potentially multiple water supply facilities. Our estimate of the economic damages to water supply facilities could have been more accurate had detailed data on the number of water withdrawal sites been readily available. Our analysis was also unable to place a value on the damages to fish passages. IAEB (2010) reports damages may run into millions of dollars annually. Fish passage infrastructure design and technology is specific to the location due to different water flow rate and the fish species that it is provided passage. Therefore damages to fish passages must be estimated on a dam-by-dam basis.

A more complete economic analysis would have required detailed information about patterns of colonization, dispersal and likely growth rates in B.C. Impacts from mussels can be specific to the locality due to mussel population characteristics (IAEB 2010). For example, in the case of hydropower generating facilities, the density of mussel may influence the selected control methods. The lack of specific economic damages in B.C. means that we were only able to employ a small set of numbers gathered by a few researchers in other jurisdictions. As such, the results we present rely on borrowed information and may not reflect specific details of potential damage within B.C. Furthermore, because of limited information; only damages associated with selected human infrastructure are assigned a value. We were unable to include damages to agriculture or fisheries, ecosystem services or non-use values associated with losses of biodiversity resulting from zebra and quagga mussel invasion. Future research is required to value the attitudes towards pristine versus invaded ecosystems.

8.3.2 European Fire Ant

No analysis of the economic impacts of *M. rubra* was found for other jurisdictions. The behavior of *M. rubra* is known to vary according to whether it is indigenous or introduced, and colonies found in Europe are less aggressive than colonies that have become established in Eastern Canada and the United States (Higgins 2012). Therefore, localized economic impact data and an improved understanding of the species' behavior will be required to conduct a reliable full-scale analysis for B.C. Paradoxically, since *M. rubra* is a relatively recent arrival, a full economic impact assessment may only be available once the ant has become widely established, by which time eradication may not be feasible.

Since no damage information is available for *M. rubra*, our damage estimates are derived by adjusting damage estimates for the red imported fire ant (*Solenopsis invicta)*, which is more aggressive and has a different life history (Higgins *pers. comm.*). Further elaboration of the differences and similarities between the two species would help in refining our estimates. Further, more site-specific analysis is required to understand and quantify the potential impact of *M. rubra* on agricultural land (*vs.* Texas). Finally, due to a lack of economic impact data for ecosystem services and *S. invicta*, we were unable to transfer an estimate of non-market effects on native biodiversity or other ecosystem damages to *M. rubra* in B.C. Further economic studies using stated preference valuation methods for ecosystem services may provide more complete damage estimates.

8.3.3 Asian Carp

Currently, the dearth of information on the economic damages of Asian carp hinders a complete economic assessment. The lack of information is compounded by the fact that Asian carp have not yet invaded B.C., and so no local damage estimates exist. In order to conduct a complete assessment, a benefit transfer approach could be employed if resources are insufficient for equivalent valuation studies in B.C. (Rossi *et al.* 2004). However, ideally (and if no comparable studies exist elsewhere) these economic values should be generated using non-market valuation methods such as contingent valuation, which can estimate the public's preferences for environmental preservation; the hedonic pricing method, which estimates the economic value of environmental services that affect property values, and the travel cost method, which uses information about travel distances and expenses to value activities such as recreational fishing. Impacts to recreational fisheries from invasive species such as Asian carp can then be measured correctly in terms of anglers' willingness to pay and consumer surplus (Shrestha *et al.* 2002).

The date of establishment, extent of invasion, total damages at each point in time, and eventual carrying capacity can determine future research needs (see Figure 2.1). Because Asian carp have not yet invaded B.C., conducting research on invasion paths, potential economic damages, and prevention measures is warranted. A complete economic analysis will require data on the damages caused by Asian carp on aquatic ecosystems, which can include direct and indirect use values, and non-use values. Research is required to understand how Asian carp will affect population dynamics of native and sports fish, because a reduction in the biomass of native fish and sports fish, or a change in angler behavior will have effects on B.C. anglers' consumer surplus from recreational fishing, as noted above. One of the concerns of an Asian carp invasion is the hazard posed to boaters by silver carp, which jump out of the water when startled by boat engines. Economic data on the damages inflicted by silver carp on boats, equipment, and to boaters are not readily available, but will be required for a complete economic assessment, likely using survey methods.

8.3.4 Sitka Black-tailed Deer

Surprisingly little has been undertaken so far on the economics of damages from the introduction of Sitka black-tailed deer to Haida Gwaii. Thus, the values discussed above are notional at best but give an idea of the data and methods to be used in making a more complete and credible estimate of deer browsing losses. To perform such a calculation would require additional time and data. The estimates presented above hint at the maximum losses that would bound the damage curve presented in Figure 2.1 (based on ecological carrying capacity or maximum damages). Thus, correcting these estimates as noted above would provide this "piece of the puzzle." But to do this would require various parameters, species specific forest stand information and modeling outputs to capture the damages as they evolve over time.

In addition, we considered only the damage to commercial redcedar stands from deer but other commercial species may be affected and there may be important recreation losses (deer infested areas show much reduced shrubs and greenery) and the aforementioned losses in ecosystem services. More detailed surveys and biophysical studies would be needed to inform damage estimates in these cases.

8.3.5 European Starling

Since their introduction in New York in the 19th century, starlings have successfully colonized the North American continent. Their abundance and adaptability makes their eradication all but impossible. Even though starlings are well established in B.C., understanding invasions that are mature still has value. A better understanding of the dynamics of the starling population is required to develop economic impact models, which will help determine the overall direct, indirect and induced impacts of starlings in B.C. Such an economic analysis could help in the design of future management programs, ultimately reducing damages to growers. Additionally, research needs to be conducted on the cost to society of starling control programs that use scaring devices such as propane cannons. Non-market valuation techniques can be used to determine how much people value the outdoor (or indoor) activities affected by the noise caused by propane cannons. Once comprehensive management strategies have been developed and all the economic damages have been taken into account, a cost-benefit analysis can be conducted to determine whether starling management programs are worthwhile from society's standpoint. A CBA will highlight the costs and benefits of different management options, thus providing policy makers with useful information to help them decide the most efficient allocation of finances to address invasive species.

9. References

ANST (Aquatic Nuisance Species Task Force). 2009. Washington, D.C. 223 p**.**

ACRCC (Asian Carp Regional Coordinating Committee). 2012. Asian carp control strategy framework. Council on Environmental Quality, Washington, D.C. [http://www.asiancarp.us/documents/2012Framework.pdf].

Bain, M.B., Webb, D.J., Tangedal, M.D. and L.D. Magnum. 1990. Movements and habitat use by grass carp in a large mainstream reservoir. Trans. Am. Fish. Soc. 119:553-561.

BCFIRB (B.C. Farm Industry Review Board). 2009. Review of the use and regulation of propane cannons in the South Coast region. Accessed March 18th 2013. [http://www.firb.gov.bc.ca/reports/Propane/09_May_PropaneCannonReview.pdf].

B.C. Grape Growers' Association. 2010a. Okanagan Similkameen Starling Control Program. [http://www.grapegrowers.bc.ca/pdf/Starling_Control_Program_2010_expanded_1.pdf].

B.C. Grape Growers' Association. 2010b. The Starling Control Program in the Okanagan Similkameen Fact Sheet. [http://www.grapegrowers.bc.ca/pdf/Starling_Program_fact_sheet_Jan_10.pdf].

B.C. Golf Guide. 2013. Course search. Accessed March 18, 2013 from [http://www.bcgolfguide.com/city/city.cfm].

B.C. Inter-Ministry Invasive Species Working Group. 2012. European Fire Ant Planning Session for Governments. [http://www.for.gov.bc.ca/hra/invasive-species/Publications/Fire_ant_planning_session.pdf].

B.C. Ministry of Education. 2013. K to 12-education system, provincial reports. Accessed March 18, 2013 from [http://www.bced.gov.bc.ca/reporting/].

B.C. Ministry of Forests. 2000. Queen Charlotte Timber Supply Area Analysis Report. Accessed April 17th 2013 from [http://www.for.gov.bc.ca/hts/tsa/tsa25/tsr2/analysis.pdf].

B.C. Ministry of Forests. 2004. Revenue Branch, 2004. Coast Appraisal Manual.

B.C. Statistics. 2012. Sub-Provincial Population Estimates. Accessed March 18, 2013 from [http://www.bcstats.gov.bc.ca/StatisticsBySubject/Demography/PopulationEstimates.aspx].

Baldwin, B.S., M.S. Mayer, J. Dayton, N. Pau, J. Mendilla, M. Sullivan, A. Moore, A. Ma, and E.L. Mills. 2002. Comparative growth and feeding in zebra and quagga mussels (*Dreissena polymorpha* and *Dreissena bugensis*): implications for North American lakes. Can. J. Fish. Aquat. Sci. 59: 680-694.

Benson, A.J., Richerson, M.M., Maynard, E., Larson, J. and A. Fusaro. 2013. *Dreissena bugensis.* USGS Nonindigenous Aquatic Species Database, Gainesville, Florida. [http://nas.er.usgs.gov/queries/factsheet.aspx?speciesid=95; Revision Date: 6/28/2012].

Bially, A. and H.J. MacIsaac. 2000. Fouling mussels (*Dreissena spp.*) colonize soft sediments in Lake Erie and facilitate benthic invertebrates. Freshwater Biology 43:85-97.

Bielert, C. 2013. Personal Communication. Administration Services BC Grapegrowers' Association. bcga@nethop.net.

Bielert, C. and R. Hol. 2008. A starling control program for the Okanagan and Similkameen. British Columbia Grapegrowers' Association. [http://www.grapegrowers.bc.ca/starling.shtml].

Bootsma, H.A., Young, E.B. and J.A. Berges. 2008. Water quality management options to control cladophora growth in the Milwaukee region of Lake Michigan. Final Report for the Milwaukee Metropolitan Sewerage District.

Brooks, R. 2013. Personal Communication. Fisheries Division Director, Kentucky Department of Fish and Wildlife Resources. ron.brooks@ky.gov.

Buck, E.H., Upton, H.F., Stern, C.V. and J.E. Nicols. 2010. Asian Carp and the Great Lakes Region. Congressional Research Service Reports. Paper 12. [http://digitalcommons.unl.edu/crsdocs/12].

Bunnell, F.L. 1990. Ecology of black-tailed deer (Chapter 5). *In:* Deer and elk habitats in coastal forests of southern British Columbia. *Special Report Series 5*. B.C. Ministry of Forests, Victoria, B.C.

Chen, D., Gerstenberger, S., Mueting, S.A., and Wong, W.H. 2011. Environmental factors affecting settlement of quagga mussel (*Dreissena rostriformis bugensis*) veligers in Lake Mead, Nevada-Arizona, USA. Faculty Publications (EOH). Paper 14. [http://digitalscholarship.unlv.edu/env_occ_health_fac_articles/14].

Conover, M. R. (1997). Monetary and intangible valuation of deer in the United States. *Wildlife Society Bulletin*, *25*(2), 298-305.

Conover, G., Simmonds, R. and M. Whalen (editors). 2007. Management and control plan for bighead, black, grass, and silver carps in the United States. Asian Carp Working Group.

Cornell Lab of Ornithology. 2012. Range map for European Starling. [http://www.allaboutbirds.org/guide/european_starling/lifehistory].

Craft, C.D., and C.A. Myrick. 2011. Evaluation of quagga mussel veliger thermal tolerance. Final Report – January 2011 Research Session. Report prepared for Colorado Division of Wildlife, Aquatic Section, Invasive Species Program 21 p. [http://www.aquaticnuisance.org/wordpress/wp-content/uploads/2010/01/evaluation_of_Quagga_mussel_veliger_thermal_tolerance-Final.pdf].

Cross, C.L., W.H. Wong and T. Che. 2011. Estimating carrying capacity of quagga mussels (*Dreissena rostriformis bugensis*) in a natural system: a case study of the Boulder Basin of Lake Mead, Nevada-Arizona. Aquat. Invasions: 6:141-147.

Cudmore, B and N.E. Mandrak. 2004. Biological synopsis of Grass carp (*Ctenopharyngodon idella*). Fisheries and Oceans Canada. Great Lakes Laboratory for Fisheries and Aquatic Sciences. Can. MS Rpt. Fish. Aquat. Sci. 2705: v+44p.

Daufresne, T. and J.L. Martin. 1997. Changes in vegetation structure and diversity as a result of browsing by a large herbivore: The impact of introduced black tail deer in the primary forest of Haida Gwaii, British Columbia. Laskeek Bay Res. 7:2-26.

Engelstoft, C. 2001. Effects of Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) on understory in old-growth forests of Haida Gwaii (Queen Charlotte Islands), British Columbia. M.Sc. thesis, University of Victoria.

Frid, L., Knowler, D., Murray, C., Myers, J. and L. Scott. 2009. Economic Impacts of Invasive Plants in BC. Report prepared for the Invasive Plant Council of B.C. by ESSA Technologies Ltd., Vancouver, B.C. 105p.

Federenko, A.Y. and F.J. Fraser. 1978. Review of grass carp biology. Interagency Committee on transplants and introductions of fish and aquatic invertebrates in British Columbia. British Columbia, Department of Fisheries and Environment, Fisheries and Marine Service, Technical Report No. 786. 15 p.

Garvey, J.E., DeGrandchamp, K.L. and C.J. Williamson.2007. Life history attributes of Asian Carps in the upper Mississippi River system. Aquatic Nuisance Species Research Program (ANSRP). ERDC/TN ANSRP-07-1. 13 p.

Gillingham, M.P. 2008. Ecology of black-tailed deer in north coastal environments. Proceedings of a symposium: Lessons from the Islands: introduced species and what they tell us about how ecosystems work. RGIS 2002 Conference, 2-5 October 2002.

G.S. Gislason & Associates Ltd. 2009. Economic Impacts of Freshwater Sport Fishing in BC. [http://www.gofishbc.com/documents/pdf/Freshwater_Sport_Fishing_in_British_Columbia_Sending_Ripples_through_the_Provincial_Economy.pdf].

Golumbia, T.E. 2000. Introduced species management in Haida Gwaii (Queen Charlotte Islands). Pp.327-332. *In:* L. M. Darling, (ed.) 2000. Proceedings of a conference on the biology and management of species and habitats at risk, Kamloops, B.C., 15-19 Feb.1999. Vol. 1. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490p.

Gorbach E.I., Krykhtin M.L. 1981. Rate of maturation in grass carp (*Ctenopharyngodon idella*) and silver carp (*Hypophthalmichthys molitrix*) in Amur River. Journal of Ichthyology 21: 835-843.

Groden, E., Drummond, F.A., Garnas, J. and A. Franceour. 2005. Distribution of an invasive ant, *Myrmica rubra* (Hymenoptera: Formicidae), in Maine. J. Econ. Entomol. 98:1774-1784.

Grime, T. 1995. The Great Lakes' other mussel menace! University of Guelph, Canada. [http://www.uoguelph.ca/research/publications/Assets/HTML_MAGS/oasis/environ4.html].

Gutrich, J.J., VanGelder, E. and Loope, L. 2007. Potential economic impact of introduction and spread of the red imported fire ant, *Solenopsis invicta*, in Hawaii. Environmental Science & Policy 10:685-696.

Heimowitz, P. and S. Phillips. 2008. Columbia River Basin Interagency Invasive Species Response Plan: zebra mussels and other dreissenid species Columbia River Basin Team, 100th Meridian Initiative.

Herborg, L.M. 2013. Personal Communication. Aquatic Invasive Species Coordinator for BC, BC Ministry of Environment. Matthias.Herborg@gov.bc.ca.

Herborg, L.M., Mandrak, N., Cudmore, B. and H. MacIsaac. 2007. Comparative distribution and invasion risk of snakehead (Channidae) and Asian carp species (Cyprinidae) in North America. *Can. J. Fish. Aquat. Sci*. 64:1723-1735.

Higgins, R. 2012. European Fire Ant (*Myrmica rubra*) project: confirming current distribution in B.C.and development of effective control methods. Interim Report (31 Oct 2012) prepared for the B.C. Inter-Ministry Invasive Species Working Group.

Higgins, R. 2013. The European Fire Ant (*Myrmica rubra*) in British Columbia. [http://faculty.tru.ca/rhiggins/myrmica_rubra_index.htm].

Irons, K. 2013. Personal Communication. Aquaculture and Aquatic Nuisance Species Program Manager, Illinois Department of Natural Resources. kevin.irons@illinois.gov.

Irons, K.S., Sass, G.G., McClelland, M.A. and J.D. Stafford. 2007. Reduced condition factor of two native fish species coincident with invasion of non-native Asian carps in the Illinois River, U.S.A. Is this evidence for competition and reduced fitness? Journal of Fish Biology 71(Supplement D):258-273.

Johnson L.E., Ricciardi A. and J.T. Carlton. 2001. Overland dispersal of aquatic invasive species: a risk assessment of transient recreational boating. Ecological Applications 11:1789-1799.

Kennedy, V.S. 2002. Swimming and settlement behavior in the quagga mussel. Maryland Sea Grant. [http://www.mdsg.umd.edu/programs/research/projects/past/R_ZM-03/index.php].

Kipp, R., Cudmore, B. and N.E. Mandrak. 2011. Updated (2006-early 2011) biological synopsis of Bighead Carp (*Hypophthalmichthys nobilis*) and Silver Carp (*H. molitrix*). Can. Manuscr. Rep. Fish. Aquat. Sci. 2962: v + 51 p.

Kolar, C.S., Chapman, D.C, Courtenay, W.R., Housel Jr., C.R., Williams, J.D. and Jennings, D.P. 2007**.** Bigheaded carps: a biological synopsis and environmental risk assessment. American Fisheries Society Special Publication 33. Bethesda. MD.

Knowler, D. and K. Dust. 2007. The economics of protecting old growth forest: an analysis of spotted owl habitat in the Fraser Timber Supply Area of British Columbia. Report prepared for David Suzuki Foundation, Vancouver, B.C. 113p.

Lard, C., Willis, D.B., Salin, V. and S. Robinson. 2002. Economic assessments of red imported fire ant on Texas' urban and agricultural sectors. Southwestern Entomologist 25:123-137.

Limburg K.E, Luzadis V.A, Ramsey M., Schulz K.L. and C.M. Mayer. 2010. The good, the bad and the algae: perceiving ecosystem services and disservices generated by zebra and quagga mussels. Journal of Great Lakes Research 36:86-92.

Linz, G.M., Joman, H.J., Gaulker, S.M., Shannon, M., Penry, L.B. and W.J. Bleier. 2007. European starlings: a review of an invasive species with far-reaching impacts. *In:* Witmer, G.W., Pitt, W.C. and K.A. Fagerstone, (eds.) 2007. Managing vertebrate invasive species: proceedings of an international symposium. USDA/APHIS Wildlife Services, National Wildlife Research Center, Fort Collins, Colorado, USA.

Local Government Act [RSBC 1996] Chapter 323.

Lowe, S., Browne, M., Boudjelas, S. and M. De Poorter. 2000. 100 of the world's worst invasive alien species: a selection from the global invasive species database. Auckland, New Zealand: Invasive Species Specialist Group.

Mandrak, N.E. and B. Cudmore. 2004. Risk assessment for Asian Carps in Canada. Canadian Science Advisory Secretariat (CSAS). Research Document 2004/103.

Mann, R., Radtke, H., Huppert, D.D., Hamilton, J.R., Hanna, S.S., Duffield, J. and N.R.Netusil. 2010. Economic risk associated with the potential establishment of zebra and quagga mussels in the Columbia River basin. Independent Economic Analysis Board. Task Number 159. Document IEAB 2010-1.

Marbek. 2010. Assessing the economic value of protecting the Great Lakes. Final report submitted to Ontario Ministry of Environment. http://www.ene.gov.on.ca/stdprodconsume/groups/lr/@ene /@subject/@greatlakes/documents/nativedocs/stdprod_086942.pdf

Margo Supplies. 2013. Zon LP Gas Bird Scare Cannons and Accessories. [http://margosupplies.com/public/canadian1/scare/zon_mark_iv.htm]

May, B. and J.E. Marsden. 1992. Genetic identification and implications of another invasive species of Dreissenid mussel in the Great Lakes. Can. J. Fish. Aquat. Sci. 49:1501-1506.

Mayle, B.A. 1999. Managing deer in the countryside. Forestry Commission Practice Note 6, H.M.S.O.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: biodiversity synthesis. Washington. World Resources Institute.

Mills, E.L., Dermott, R.M., Roseman, E.F., Dustin, D., Mellina, E., Conn, D.B. and A.P. Spidle. 1993. Colonization, ecology, and population structure of the "Quagga" mussel (Bivalvia: Dreissenidae) in the Lower Great Lakes. Can. J. Fish. Aquat. Sci. 50:2305-2314.

Mills E.L., Rosenberg, G., Spidle. A.P., Ludyanskiy, M., Pligin. Y. and B. May. 1996. A review of the biology and ecology of the Quagga mussel (*Dreissena bugensis*), a second species of freshwater dreissenid introduced to North America. Am. Zool. 36:271-286.

Minnesota Department of Natural Resources. 2004. Feasibility study to limit the invasion of Asian Carp into the Upper Mississippi River Basin. Prepared for the Minnesota DNR in cooperation with the Wisconsin DNR and the U.S. Fish and Wildlife Service (Region 3). [http://files.dnr.state.mn.us/natural_resources/invasives/aquaticanimals/asiancarp/umrstudy.pdf]

MICRA. 2002. Asian carp threat to the Great Lakes. River Crossings: The Newsletter of the Mississippi Interstate. Cooperative Resource Association 11:1.

Missouri Department of Conservation. 2013. Zebra mussel control. Retrieved March 18, 2013 from [http://mdc.mo.gov/your-property/problem-plants-and-animals/invasive-animals/zebra-mussel-control.]

Molloy, D.P. 1998. The potential of using biological control technologies in the management of *Dreissena*. Abstracts from the Eighth International Zebra Mussel and Other Nuisance Species Conference. California Sea Grant, Sacramento, California: 69-70.

Moore, M.A., Boardman, A., Vining, A., Weiner, D. and D. Greenberg. 2004. Just give me a number! Practical values for the social discount rate. Journal of Policy Analysis and Management 23:789-812.

Moyle, P.B. 1986. Fish introductions into North America: patterns and ecological impact. Pp. 27-43. *In:* Mooney, H.A. and J.A. Drake (eds.). 1986. Ecology of biological invasions of North America and Hawaii. Springer-Verlag, New York.

Muise, S. 2013. Personal Communication. Stewardship Forester Haida Gwaii District. Sean.Muise@gov.bc.ca

Nalepa, T.F. 2010. An overview of the spread, distribution, and ecological impacts of the quagga mussel, *Dreissena rostriformis bugensis*, with possible implications to the Colorado River system. Proceedings, Colorado River Basin Science and Resource Management Symposium. Coming Together, Coordination of Science and Restoration Activities for the Colorado River Ecosystem, Scottsdale, AZ, November 18-20, 2008. U.S. Geological Survey Scientific Investigations Report 2010-5135. 372 p. [http://www.glerl.noaa.gov/pubs/fulltext/2010/20100039.pdf]

NatureServe. 2003. NatureServe Explorer: An online encyclopedia of life. Version 1.8. NatureServe, Arlington, Virginia. [http://www.natureserve.org/explorer]

Nico, L.G., Williams, J.D. and H.L. Jelks. 2005. Black Carp: biological synopsis and risk assessment of an introduced fish. American Fisheries Society Special Publication 32, Bethesda, MD. 337 p.

Nico, L.G. 2013. *Hypophthalmichthys molitrix.* USGS Nonindigenous Aquatic Species Database, Gainesville, FL. [http://nas.er.usgs.gov/queries/factsheet.aspx?speciesID=549 Revision Date: 3/15/2012]

O'Neill, C. 1997. Economic impact of zebra mussels: results of the 1995 Zebra Mussel Information Clearinghouse Study. Great Lakes Res. Review 3:35-42.

Padilla D.K., Chotkowski, M.A. and L.A.J. Buchan. 1996. Predicting the spread of zebra mussels (*Dreissena polymorpha*) to inland waters using boater movement patterns. Global Ecology and Biogeography Letters 5:353-359.

Park, J. and L.J. Hushak. 1999. Zebra mussel control costs in surface water using facilities. Technical Summary Series OSHU-TS-028. Available from the Ohio Sea Grant College Program, Columbus, Ohio. 16 p.

Parker, K.L. 1988. Effects of heat, cold, and rain on coastal blacktailed deer. Can. J. Zool. 66: 2475-2483.

Pearce D.W. and R.K. Turner. 1990. Economics of natural resources and the environment. Johns Hopkins Press, Baltimore.

Peters, L.M., Pegg, M.A. and Reinhardt, U.G. 2006. Movements of adult radio-tagged Bighead Carp in the Illinois River. Transactions of the American Fisheries Society 135:1205-1212.

Peyer, S.M., A.J. McCarthy and C.E. Lee. 2009. Zebra mussels anchor byssal threads faster and tighter than quagga mussels in flow. J. Exp. Biol. 212:2027-2036.

Phillips, S., Darland, T. and M. Sytsma. 2005. Potential economic impacts of zebra mussels on the hydropower facilities in the Columbia River Basin. Pacific States Marine Fisheries Commission, Portland, Oregon.

Pimentel, D. 2005. Aquatic nuisance species in the New York State Canal and Hudson River systems and the Great Lakes basin: an economic and environmental assessment. Environ. Manage. 35: 692-702.

Putman, R. 2012. Scoping the economic benefits and costs of wild deer and their management in Scotland. Scottish Natural Heritage Commissioned Report No. 526.

Ransome, D. B. 2010. Investigation of starling populations in British Columbia and assessment of the feasibility of a trapping program in the Lower Mainland. [http://www.al.gov.bc.ca/resmgmt/publist/600Series/6703003_Investigation_of_Starling_Populations_in_BC.pdf]

Redding, J. C. 1987. Impact of deer on forest vegetation and timber production in northern Pennsylvania. Pp. 23-32 *In:* R. La Bared. Proc. symposium deer, forestry, and agriculture. Soc. Amer. For., Warren, Pa.

Reimchen, T.E., Nelson, R.J. and C.T. Smith. 2005. Estimating deer colonization rates to offshore islands of Haida Gwaii using microsatellite markers. Pp. 117-120 *In:* Gaston, A.J., Golumbia, T.E., Martin, J.L. and S.T. Sharpe. 2005. Canadian Wildlife Service Special Publications.

Reimoser, F. 2000. Income from hunting in mountain forests of the Alps. *In:* Price, M.F. and N. Butt (eds.). Forests in sustainable mountain development: a state of knowledge report for 2000. IUFRO Research Series 5:346-352.

Ricciardi A., Neves R.J. and J.B. Rasmussen. 1998. Impending extinctions of North American freshwater mussels (Unionida) following the zebra mussel (*Dreissena polymorpha*) invasion. J. Anim. Ecol. 67:613-619.

Rooney, T.P. 2001. Deer impacts on forest ecosystems: a North American perspective. Forestry 74:201-208.

Rossi, F., Adams, D.C. and D.J. Lee. 2004. The use of cost transfer analysis to estimate the economic impacts of a potential zebra mussels infestation in Florida. Pp.14-18. *In:* Selected paper at the Southern Agricultural Economics Association Annual Meeting, Tulsa, Oklahoma (USA), February, 2004.

Seeley, C. 2013. *pers. comm.*. Transport Canada. Christopher.Seeley@tc.gc.ca.

Shrestha, R.K., Seidl, A.F. and A.S. Moraes. 2002. Value of recreational fishing in the Brazilian Pantanal: a travel cost analysis using count data models. Ecological Economics 42:289-299.

Snyder, F.L., Hilgendorf, M.B. and D.W. Garton. 1997. Zebra Mussels in North America: The invasion and its implications. Ohio Sea Grant, Ohio State University, Columbus, Ohio.http://ohioseagrant.osu.edu/_documents/publications/FS/FS045%20Zebra%20mussels%20in%20North%20America.pdf]

Spidle, A.P., Mills, E.B. and B. May. 1995. Limits to tolerance of temperature and salinity in the quagga mussel (*Dreissena bugensis*) and the zebra mussel (*Dreissena polymorpha*). *Can. J. Fish. Aquat. Sci*. 52:2108-2119.

Statistics Canada. 2013. 2011 Census Profile, British Columbia. Retrieved on March 18, 2013 from http://www12.statcan.gc.ca.proxy.lib.sfu.ca/census-recensement/2011/dppd/prof/searchrecherche/lst/page.cfm?Lang=E&GeoCode=59&TABID=1&G=1&Geo1=CD&Code1=5947&Geo2=PR&Code2=59

Stockton, S.A., Allombert, S., Gaston, A.J. and J.-L. Martin. 2005. A natural experiment on the effects of high deer densities on the native flora of coastal temperate rain forests. Biol. Conserv. 126: 118-128.

Strayer D.L., Eviner, V.T., Jeschke, J.M. and M.L. Pace. 2006. Understanding the long-term effects of species invasions. Trends Ecol. Evol. 21:645-651.

Stroh, N., Baltzinger, C. and J-L. Martin. 2008. Deer prevent western redcedar (Thuya plicata) regeneration in old-growth forests of Haida Gwaii: Is there a potential for recovery? Forest Ecology and Management. 255:3973-3979.

Sweeney, M. 2013. Personal Communication. Berry Industry Specialist B.C. Ministry of Agriculture. Mark.Sweeney@gov.bc.ca

Therriault, T.E., Docker, M.F., Orlova, M.I., Heath, D.D. and H.J. MacIsaac. 2004. Molecular resolution of the family *Dreissenidae* (Mollusca: Bivalvia) with emphasis on Ponto-Caspian species, including first report of *Mytilopsis leucophaeata* in the Black Sea basin. *Molecular Phylogenetics and Evolution* 30:479-489

Therriault, T.W., Weise, A.M., Higgins S.N., Guo, S. and J. Duhaime. 2012 Risk assessment for three dreissenid mussels (*Dreissena polymorpha*, *Dreissena rostriformis bugensis* and *Mytilopsis leucophaeata)* in Canadian freshwater ecosystems. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/174v +88 p. http://www.isdm-gdsi.gc.ca/csas-sccs/applications/publications/index-eng.asp

Tracey, J., Bomford, M., Hart, Q., Saunders, G. and R. Sinclair. 2007. Managing bird damage to fruit and other horticultural crops. Bureau of Rural Sciences, Canberra.

Turner, C.B. 2010. Influence of zebra (*Dreissena polymorpha*) and quagga (*Dreissena rostriformis*) mussel invasions on benthic nutrient and oxygen dynamics. *Can. J. Fish. Aquat. Sci*. 67:1899-1908.

USACE (United States Army Corps of Engineers). 2012. Black carp (*Mylopharyngodon piceus*) Aquatic Nuisance Research Program. [http://el.erdc.usace.army.mil/ansrp/mylopharyngodon_piceus.pdf]

USGS (United States Geological Survey). 2000. Zebra Mussels Cause Economic and Ecological Problems in the Great Lakes. Great Lakes Science Center. GLSC Fact Sheet 2000-6.

USGS (United States Geological Survey)**.** 2013. Non-Indigenous Aquatic Species Database. Maps of current distribution range in the U.S.:
[Grass carp: http://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=514]
[Bighead carp: http://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=551]
[Silver carp: http://nas.er.usgs.gov/queries/factsheet.aspx?speciesID=549]
[Black carp: http://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=573]

Van Oostrom 2007. Establishment of a successful *Dreissena*mussel treatment program: A summary of important considerations and lessons learned. ACWA Fall Conference Indian Wells, CA

Vilaplana, J.V. and L.J. Hushak. 1994. Recreation and the zebra mussel in Lake Erie, Ohio. Technical Summary. OHS-TS-023. Ohio Sea Grant College Program. Columbus, Ohio.

Voller, J. and R.S. McNay. 2007. Problem analysis: effects of invasive species on species at risk in British Columbia. FORREX Research Extension Society, Kamloops, B.C. Series 20. 145p.

Water Act, RSBC (1996, c.483) Retrieved from: [http://www.canlii.org/en/bc/laws/stat/rsbc1996c483/latest/rsbc1996c483.htm].

Watkins, J.M., Dermott, R. Lozano, S.J., Mills, E.L., Rudstam, L.G. and J.V. Scharold. 2007. Evdence for Remote Effects of *Dreissenid* Mussels on the Amphipod *Diporeia*: Analysis of Lake Ontario Benthic Surveys, 1972-2003. *Journal of Great Lakes Research.* 33:642-657.

Whittier, T.R., Ringold, P.L., Herlihy, A.T. and S.M. Pierson. 2008. A calcium-based invasion risk assessment for zebra and quagga mussels (*Dreissena spp*). Front. Ecol. Environ. 6:180-184.

Wilson, A.B., Naish, K.A. and E.G. Boulding. 1999. Multiple dispersal strategies of the invasive quaga mussel (*Dreissena bugensis*) as revealed by microsatellite analysis. *Can. J. Fish. Aquat. Sci*. 56:2248-2261.

Witmer G.W. and D.S. DeCalesta. 1992. The need and difficulty of bringing the Pennsylvania deer herd under control. Proc. East. Wildl. Damage Control Conf. 5:130-137.

Wong W.H., Gerstenberger, S., Baldwin, W. and B. Moore. 2012. Settlement and growth of quagga mussels (*Dreissena rostriformis bugensis* Andrusov, 1897) in Lake Mead, Nevada-Arizona, USA. Aquatic Invasions 7:7-19.

Zook B. and S. Phillips. 2012. Uniform minimum protocols and standards for watercraft interception programs for dreissenid mussels in the western United States (UMPS II). Pacific States Marine Fisheries Commission. Portland, OR. 77p.