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Report #12

Economic Impacts of Invasive Plants in British Columbia

Final Project Report

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Executive Summary

According to the most recent Millennium Ecosystem Assessment (2005), invasive alien species are one of the five "most important direct drivers of biodiversity loss and change in ecosystem services" around the globe. Little has been done in British Columbia (BC) to put a dollar value to these impacts within the province; the results of this project help to fill that information gap, and identify the costs and trade-offs of different invasive plant management strategies.

The analysis involved two phases: (1) understanding the impact of invasive plants in the absence of any management action and (2) a cost-benefit analysis of alternative invasive plant management strategies. In Phase 1 of the work, spatial extent and economic damages were assessed over time for seven invasive species present in BC. These species are: purple loosestrife (*Lythrum salicaria*), diffuse knapweed (*Centaurea diffusa*), hawkweed (*Hieracium* sp.), cheatgrass (*Bromus tectorum*), Scotch broom (*Cystius scoparious*), Eurasian watermilfoil (*Miriophyllum spicatum*) and Dalmatian toadflax (*Linaria dalmatica*). Without intervention, the economic damage caused by each one of the selected invasive species was estimated to range from 1 to 20 million dollars in 2008, increasing to between 5 and 60 million dollars by 2020 (based on 2006 Canadian dollars). The total expected damages, in the absence of any management, were estimated to be a minimum of $65 million in 2008, rising to $139 million by 2020. These values are likely underestimates as economic data were not available for all potential impacts. These estimates served to establish a baseline against which to evaluate the costs and benefits of alternative control and management actions.

In Phase 2 of the project, a cost benefit analysis of alternative management strategies was conducted on a subset of the species from Phase 1. The analyses include consideration of alternative conventional management actions, biological control, escalation of control costs and increased management along utility and highway corridors. The species selected for these analyses were: diffuse knapweed, hawkweed, Scotch broom and Eurasian watermilfoil.

An ecological model was developed to quantitatively account for the ecological and economic effects of a suite of management actions. This model is based on a simple logistic growth model, and partitions the landscape into five alternate states that are tracked quantitatively over time. Transitions between states can occur as a result of natural processes or management actions. Based on our species selection rationale, at least one of the alternatives for diffuse knapweed and hawkweed includes biocontrol, at least one of the alternatives for hawkweed also includes delaying actions to a future time, and the analysis for Scotch broom is specific to utility corridors.

For diffuse knapweed, the historic economic viability of BC's biocontrol program was assessed using data from government files and from published papers reviewing various aspects. The Net Present Value (NPV) for the program in BC was estimated to be $17.4 million under baseline assumptions, and the Benefit-Cost Ratio (BCR) was 17.0 dollars gained for every dollar spent on the program. Also for diffuse knapweed, an economic analysis of a hypothetical chemical control alternative to biocontrol was conducted. Based on a treatment budget of $180,000 CDN per year, economic analysis indicates a negative NPV. The BCR fluctuates between 0.78 and 1.05. A negative NPV for the treatment program indicates that the chemical treatment of diffuse knapweed is not economically viable. This outcome highlights the importance of a biological control program and the need to continue efforts to develop biological control agents for other species of invasive plants in the province.

For hawkweed, we projected the potential future benefits of a biocontrol program. Based on an initial upfront research and development investment of $2.5 million CDN, recurrent program costs of $192,000 CDN, an inventory budget of $100,000 CDN per year and a release budget of $100,000 CDN per year,

the estimated NPV for biocontrol of hawkweed is $1.7 billion and the BCR is 185.5 dollars gained for every dollar spent on the program. These results suggest that the biocontrol of hawkweed is economically viable and could generate significant benefits to society. Also for hawkweed, an economic analysis of a hypothetical future conventional treatment program was conducted. Based on a treatment budget of $180,000 CDN per year and an inventory budget of $100,000 CDN per year, economic analysis indicates that both the NPV and the BCR for the treatment of hawkweed are much lower than the estimates for biocontrol. These results imply that conventional treatment is an economically viable option for controlling hawkweed, but the benefits gained from this approach are much lower than those potentially gained from a successful biocontrol program. Overall, despite positive net returns to society, conventional treatment approaches are not predicted to bring hawkweed permanently under control; they only slow its progress towards eventually occupying its entire potential ecological range by the year 2080.

For Scotch broom, the project focused on a localized problem: invasion by Scotch broom along a representative highway corridor on Vancouver Island. Using a limited baseline budget of $20,000 CDN per year, given the small-scale treatment program, economic analysis indicates that the NPV for mechanical treatment of Scotch broom is negative. Indeed, the NPV remains negative and the BCR never exceeds 0.20 for all treatment budget scenarios. These results imply that the treatment of Scotch broom is not economically viable at a local site level when only the benefits we have captured are considered. Sensitivity analysis to the ecological limits suggested that for a control program along a highway corridor to be viable, there must be a large surrounding area that is vulnerable to invasion. In this case control at the corridor can have a large economic benefit by preventing spread into neighbouring areas.

For Eurasian watermilfoil, analysis focused on a hypothetical mechanical treatment program at the provincial level, and was based on parameters from an established treatment program in the Okanagan Basin. Based on a treatment budget of $500,000 CDN and an inventory budget of $200,000 CDN, economic analysis indicates that the NPV for the mechanical treatment of Eurasian watermilfoil is positive. The NPV can, however, be negative when the inventory budget is low or the treatment budget high. The BCR varies between 0.9 and 1.5 for all scenarios. This result indicates that, in general, both the conventional inventory and treatment budgets generate net benefits to society, and inventory should be a key component of a control program.

The study provides 13 recommendations, which are summarized in an abbreviated version below:

1. Efforts should be continued to develop a set of successful bio-agents for hawkweed.
2. Future biocontrol programs should include a plan for evaluation at multiple spatial scales: individual plants, release sites, and both regional and provincial.
3. Further analysis is needed to evaluate the trade-off between releasing less-effective agents sooner and delaying releases until more effective biocontrol agents are discovered.
4. Future participation by the province in the research and development of biocontrol agents undertaken by similar consortia is a worthwhile investment.
5. Ensure that sufficient resources are available for conducting field releases as early as possible in a biocontrol program, without compromising the prevention of non-target effects.
6. Economic evaluation of the cost of invasive plant species should be made prior to the release of biological control agents as a baseline on which success can be evaluated.
7. Standardized monitoring procedures should be developed to track changes in the densities of the target invasive plant, the biological control agents, and the plant community. These data should be made available through regular reports or on websites so that they can be publicly accessible.
8. Efficacy testing should be part of the development of biological control agents to improve the success rate of introduced agents in reducing plant density and to reduce the number of exotic species being introduced.

9. Land management actions, such as grazing management and seeding, are an important component of an invasive plant control program.
10. The management of invasive plants along utility and transportation corridors requires prioritization of corridors that have the potential to impact the surrounding area.
11. A key aspect of a control program against Eurasian watermilfoil is the allocation of resources towards inventory and education aimed at preventing the infestation of currently un-invaded, but vulnerable, lake systems.
12. More primary research is required into the valuation of damages from invasive plants in BC. As an example, a small research program could be sponsored that would fund student research at the Masters and PhD levels.
13. The impacts of climate change on the distribution of the important invasive plant species be considered for future analysis.

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1. Introduction

1.1 Background

According to the most recent Millennium Ecosystem Assessment (2005), invasive alien species are one of the five "most important direct drivers of biodiversity loss and change in ecosystem services" around the globe. Lee (2002) reports that 25 percent of Canada's endangered species, 31 percent of its threatened species and 16 percent of its vulnerable species are in some way at risk because of alien species. Almost one-quarter of the total number of vascular plant species in Canada are exotic (CESCC 2006). The most recent report on the general status of wild species in Canada highlighted the issue of the large number of non-native species in Canada (16 percent of the 7732 species assessed), and noted that of the taxonomic groups covered in the report, vascular plants have the highest proportion of exotic species nationally at almost 24 percent (CESCC 2006).

Significant work has been done on valuing the economic impacts of invasive plants in the United States (Pimentel et al. 2000, Duncan and Clark 2005, Barbier and Knowler 2006), but little has been done in British Columbia (BC) to put a dollar value to these impacts within the province (RNT Consulting Inc. 2002, Colautti et al. 2005). The ESSA project team was contracted by the Invasive Plant Council of British Columbia to help to fill that information gap, and to help identify the costs and trade-offs of different invasive plant management strategies.

1.2 Purpose of this Report and the Intended Audience

The purpose of this report is to summarize our methodology and results, and to provide recommendations based on what has been learned. These recommendations fall into two categories: recommendations for managers regarding invasive plant management strategies, and recommendations for researchers who may wish to extend these methods to other places or species. It is intended for the Invasive Plant Council of British Columbia to provide a full record of our methods, results, and conclusions. In its current form, it is not intended for peer review or publication, nor is it suitable for a non-technical audience. A condensed version which focuses on recommendations would be more suitable for invasive plant managers, and an abbreviated version without technical language which focuses on the results is available to serve a public audience.

1.3 Overview of Methods

Figure 1.3.1 shows an outline of the analysis process as we applied it. An initial phase (1) required the development of criteria and selection of species that would be carried through the analyses. Species were selected in consultation with the project's technical committee using the following criteria: as broad a representation as possible in impacts of different economic sectors, ecosystems, and areas of the province; suitability for the analyses outlined in the original proposal; and availability of ecological and economic data. For the first phase of the project we took the species selected and (2) developed qualitative impact diagrams that identify the damage pathways for each selected species. These diagrams served to clearly illustrate our current understanding of the impacts of selected species and to guide (3) the gathering of data and (4) economic quantifications of the current damages and costs associated with each of the selected species. These damage estimates fed into (5) the quantification of the dispersal curves of the selected invasive plants in order to understand the damages incurred over the past and into the future. A final outcome of the first phase of the project was an estimate of the damage trajectories for selected invasive plants if no management action is taken. These economic damages and costs then fed into a final phase of the project that quantified future damages and the Total Economic Value to society of alternative management strategies against invasive plants. This final phase included (6) the estimation of how alternative management strategies (including the delay of management actions) will alter the dispersal curves for the selected species and (7) a calculation of the area under the damage trajectories to calculate the full benefits and costs of each strategy.

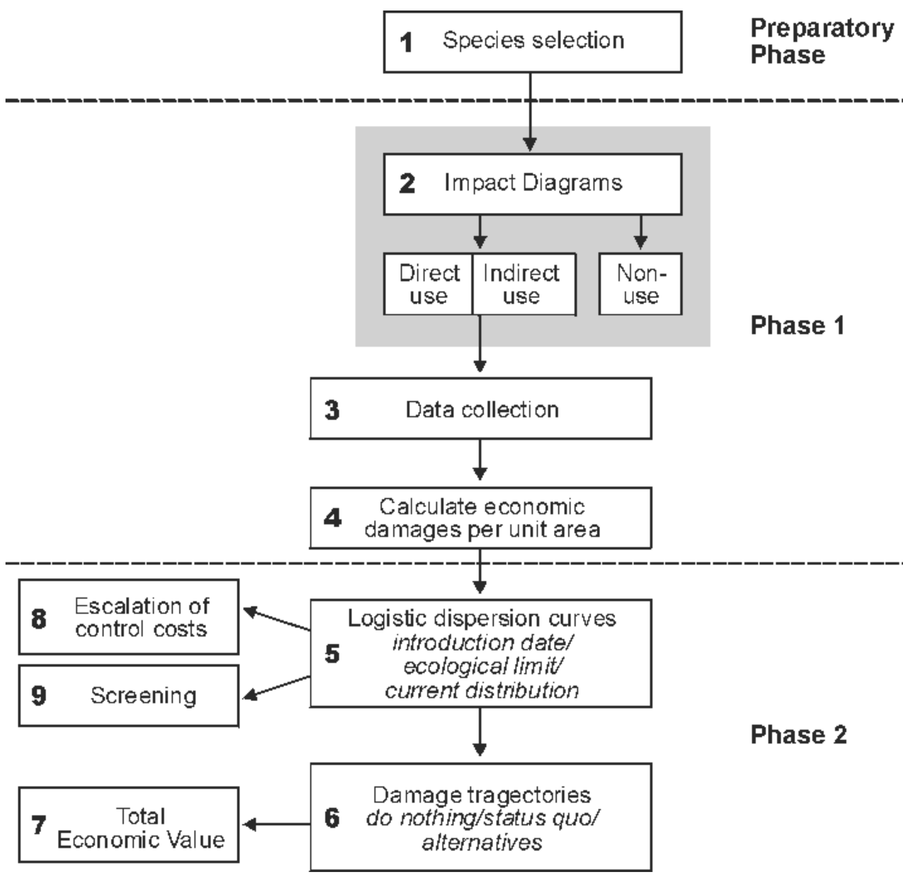

Figure 1.3.1. Overview of Project Approach.

A common aspect to all of the analyses in the project was the use of a model of dispersal to project damages of invasive plants into the future. Figure 1.3.2 illustrates how these curves were used to quantify past damages experienced by society due to the introduction of invasive plants, potential future damages under alternative management strategies, the total economic value of alternative management strategies, the escalation of control costs, and any uncertainties associated with these measurements. A detailed description of the methodologies that were applied in this project is outlined under the next two sections.

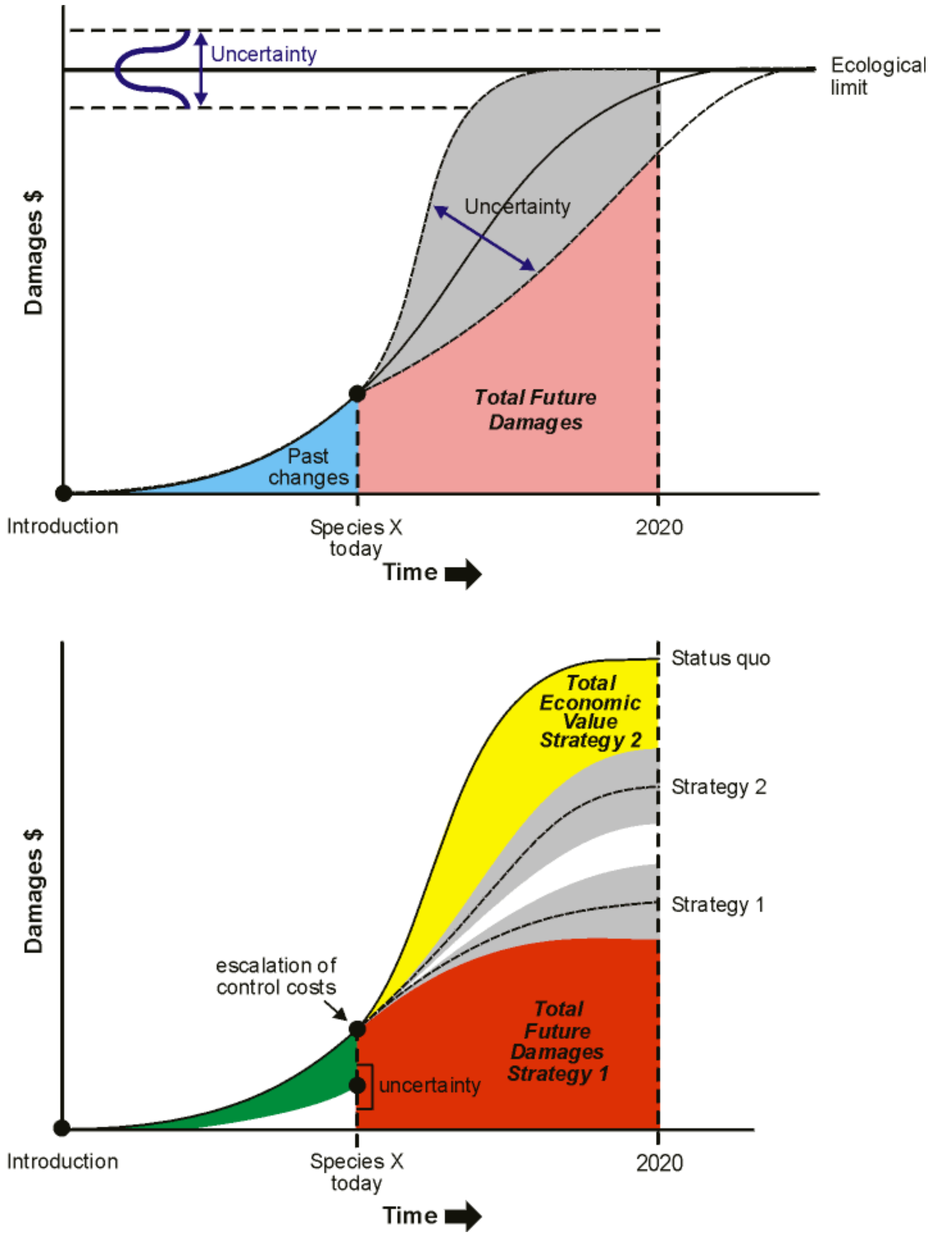

Figure 1.3.2. Illustration of our General Approach to the Economic Evaluation of Invasive Plants. The analysis involves the use of dispersal models.
These curves are derived from estimates of the date of establishment (when damages begin to occur), extent of coverage now, unit damages (per ha) using a Total Economic Value approach and eventual ecological carrying capacity.

It should be noted that extensive raw data on the dispersion and economic damages for invasive plants 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.

2. Phase 1 – Damages if No Management Action is Taken

In this section we present our assessment of the spatial extent and economic damages over time for seven invasive species present in BC. These species are: purple loosestrife (*Lythrum salicaria*), diffuse knapweed (*Centaurea diffusa*), hawkweed (*Hieracium* sp.), cheatgrass (*Bromus tectorum*), Scotch broom (*Cystius scoparious*), Eurasian watermilfoil (*Miriophyllum spicatum*), and dalmatian toadflax (*Linaria dalmatica*).

2.1 Methods and Approach

We first analyzed what would happen if there was no control of the invasive species in question. Thus, we assessed 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 six species in question have received some measure of control or treatment to inhibit their spread. However, this approach serves to establish the baseline from which further investigation of control and management and its costs and benefits was carried out (presented in section 3).

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 affect the ecological, social and economic environment in BC. These diagrams provide a simple visual tool for summarizing the current state of knowledge of the "big picture" regarding how these invasive plant species affect the resource and tourism industries, environmental attributes, and human health, as well as any other impacts.

Next, logistic curves showing "area invaded" and "economic damages" for each of the species were estimated. We assumed that the area invaded and economic damages increase exponentially over time until they approach a carrying capacity. The estimated logistic curves demonstrate the growth pattern of area invaded and economic damages for the period from the date of initial introduction to the year 2100. Economic damages were established by taking the area invaded in each year of the simulation and multiplying this figure by an estimate of the damages per hectare per year invaded.

In keeping with the use of a logistic model, our damage curves show 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 plant. We constructed three curves for each species (high, medium, low) showing the area invaded, and did the same for damages, to accommodate a range of input assumptions and to allow for the uncertainty in our models.

Generating the logistic damage curves required assumptions about the underlying economic and ecological conditions. These assumptions are described below.

1. The **economic damage per hectare** was estimated for each species using the impacts outlined in the Impact Diagrams as a starting point. However, to bring an economic

perspective to the analysis, we structured our components of loss loosely around the Total Economic Value (TEV) framework (Pearce and Turner 1990), and used standard welfare analysis to calculate economic values. The following briefly describes these approaches.

1. 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. For this analysis we do not consider non-use values, because of the difficulties in obtaining such information and because the coverage of indirect use values is somewhat limited. In contrast, we are able to be much more complete with respect to direct use values (e.g., agricultural, forestry values).

2. 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 from earlier, primarily US, studies and converted the estimates to the present Canadian setting using foreign exchange rates and price indices. Such an approach is referred to as “benefits transfer”, whereby valuation results from a site located elsewhere is used (with appropriate modification). There are now accepted protocols for doing benefits transfer, such as transferring a functional relationship, rather than a single value, which we were able to do in several cases. To be consistent with economic theory, we represented all values as consumers or producers surplus and made adjustments to secondary data where necessary.

2. Various **ecological assumptions** were required, such as the date of introduction, spread rates, and ecological limits; these were derived from secondary data and expert opinion. Since few data exist on the spread rates of selected species in BC, we obtained data on spread rates from studies conducted chiefly in the US. Using this information, we then set three spread rates (high, medium, low) for our selected species for the BC situation. For most species, we set the ultimate ecological carrying capacity for each individual species based on the ecological limits data in the Gap Analysis report (Miller and Wikeem 2005), with 25 percent confidence intervals between high carrying capacity and low carrying capacity. We also made assumptions on the susceptibility to invasion by the target invasive plant for biogeoclimatic variants (high, medium, and low) based on the current distribution of each individual species in the province of BC.

The resulting assumptions used to construct the logistic damage curves are presented in Table 2.1.1.

Table 2.1.1. Summary of Ecological/Economic Data for Selected Invasive Species.

Parameter	Hawkweed	Diffuse knapweed	Dalmatian toadflax	Cheatgrass	Scotch broom	Purple loosestrife	Water milfoil
1. Unit damage cost (C$ per ha)							
	165.04	21.09	n.a.	20.09	39.51	110.0	954.95
2. Year of Introduction							
	1922	1900	1952	1904	1850	1897	1970
3. Dispersal Rate (per year)							
High	18%	16%	16%	18%	10%	15%	20%
Medium	14%	15%	13%	14%	7.5%	12.5%	17.5%
Low	10%	9%	10%	10%	5%	10%	15%
4. Ecological Limit/Carrying Capacity (ha)							
High	10,903,099	1,421,411	1,677,631	651,937	2,295,493	413,668	26,250
Moderate	8,722,479	1,137,129	1,342,105	521,550	1,836,394	330,934	21,000
Low	6,541,859	862,854	1,006,579	391,163	1,377,296	248,201	15,750

2.2 Results for Seven Invasive Species

2.2.1 Purple Loosestrife

Impact Diagram

Purple loosestrife (*Lythrum salicaria*) is a perennial forb native to Eurasia. It is adapted to seasonally wet soils and occurs commonly in wetlands and riparian areas where it out-competes native plants. The degree to which purple loosestrife has impacted native vegetation communities has been disputed in the past, but recent evidence suggests it has significant impacts (Blossey et al. 2001). These impacts can have effects on various ecosystem components including those shown in Figure 2.2.1.1.

Purple loosestrife out-competes plants that provide nesting habitat and forage for waterfowl, songbirds, ungulates and cattle (*arrow 1*). This forage loss can reduce wildlife populations and therefore reduce hunting and viewing opportunities for these species. These effects ultimately result in a loss of land values for recreational activities (*arrow 5*) (Ogrodowyczk and Moffit 2001 cited in Duncan and Clark 2005). Losses in forage to cattle can lead to direct economic losses to ranchers (*arrow 10*) (ATTRA 1997 cited in Duncan and Clark 2005).

Purple loosestrife can clog up spawning habitat for fish and may alter the aquatic invertebrate community composition favouring smaller invertebrate species (Gardner et al. 2001 cited in Duncan and Clark 2005) and thus affect the availability of food for fish species (*arrow 2*). Reductions in the population sizes of fish species as a direct or indirect effect of purple loosestrife could lead to reduced catch for both recreational and commercial fisheries (*arrow 9*).

Purple loosestrife can form nearly monospecific stands by replacing native vegetation communities and can therefore reduce the biodiversity of plant species in infested wetlands (*arrow 3*) (DiTomaso and Healy 2003, Welling and Becker 1990 *Cited in Duncan and Clark*). In other parts of North America there is evidence that purple loosestrife has had negative impacts on rare and endangered species, such as Longs Bulrush and the Bog Turtle (*arrow 7*) (Bury 1979, Kiviat 1978 cited by Duncan and Clark 2005). In British Columbia, the impact of purple loosestrife on the endangered species *Sidelcia hendersonii* was weak, although over 20 years, the frequency of *S. hendersonii* declined by 50% while that of purple loosestrife increased by 20% in transects measured in Ladner Marsh (Denoth and Myers 2007).

A major impact of purple loosestrife can be in drainage and irrigation ditches where it can grow so densely that it will change water flow (*arrow 4*) (Skinner et al. 1994 cited by Duncan and Clark 2005). This increases maintenance costs for irrigation systems (*arrow 8*). In natural waterways, changes in water flow caused by purple loosestrife can alter erosion patterns and therefore affect bank stability and soil loss processes (*arrow 11*) (Dixon and Johnson 1999).

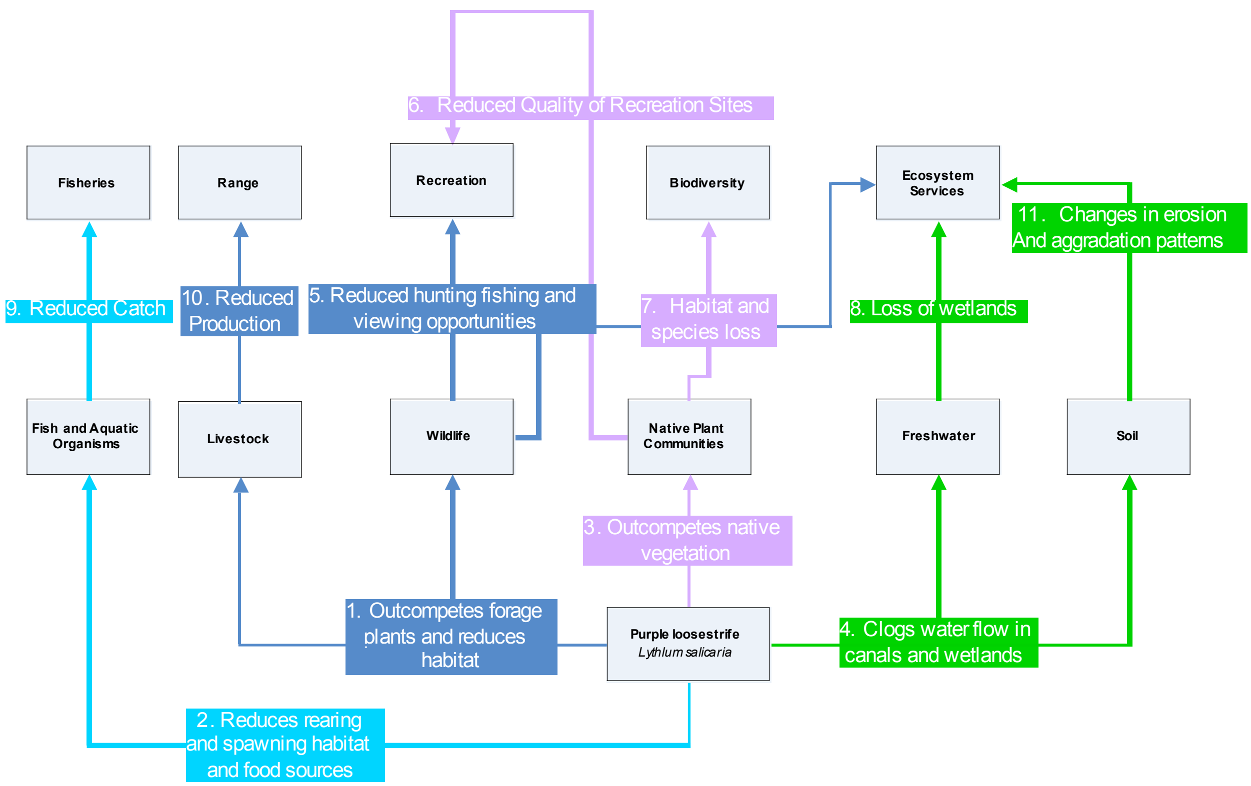

Figure 2.2.1.1. Impact Diagram for purple loosestrife.
Boxes represent valued ecological and socioeconomic components affected by the plant.

Economic Damage per Hectare

While some data exist as to total damages from purple loosestrife in the US (ATTRA 1997), the source of these values is unknown, so their use in a BC context is uncertain. Instead, we chose to develop our own damage values using a predictive equation for the ecosystem services provided by wetlands as a basis, along with an estimate of potential forage losses in riparian areas. We expect that coastal and freshwater wetlands will be the primary land types affected by purple loosestrife, followed by pasture and hay land, as noted in US damage estimates. Per-hectare economic damage for Canada in 2006 prices was estimated at $110.00. Detailed calculation of this unit damage estimate is shown in the table below.

Purple Loosestrife Unit Damage Estimate:

Impact	**Original Estimate**				**Adjust -ments**	**Final Estimate 2006** ($/ha)	**Notes**
	Year of estimate	*Total Damage ($ million)*	*Area Infested (ha)*	*Unit Damage ($/ha)*			
1. Wetlands (impacts on various ecosystem services)	1990 (USD)			123.70	Foreign Exchange Rate (FER): X 1.160 Consumer Price Index (CPI): X 1.390 Share: X 0.50	100.00	1/
2. Forage loss	2006				Share: X 0.20	10.00	2/
Total						110.00	

Notes:

1/ We used an equation predicting the value of non-market services provided by wetlands from a statistical analysis of 39 studies of individual wetland values in North America (Woodward and Wui 2001). We entered the mean values of all the explanatory variables into the equation. These variables covered a range of ecosystem services potentially provided by wetlands. We then adjusted the predicted value of an average wetland to 2006 Canadian prices. The resulting value was about $2000 per ha (see Appendix). We then assumed that infestation by purple loosestrife would reduce these values by 10% and that wetlands represent 50% of the infested area.

2/ We assumed that pasture/hay land would represent about 20% of the ultimate area infested, based on a review of information for areas subject to long term infestation (e.g. Eastern Canada, various US). We then used hay land crop budget information to estimate the net revenue value of hay land at about $100 per ha per year (Malmberg and Peterson 2006). We further assumed this was reduced by 50% by purple loosestrife infestation.

Ecological Assumptions

We set the ultimate ecological carrying capacity at 330,934 hectares based on the ecological limits data in the Gap Analysis report (Miller and Wikeem 2005), with 25 percent confidence intervals between 248,201 ha and 413,668 ha. This estimate assumes that highly susceptible biogeoclimatic variants could be 3 percent infested, mid susceptibility variants could be 2 percent infested, and low susceptibility variants could be 1 percent infested. This ecological limit is based on 4 percent of the land area of BC represented by wetlands (Crown Registries and Geographic Base Branch [ILMB] 2006). Our assumption is that highly susceptible biogeoclimatic (BEC) variants would, on average, have 75 percent of their wetlands being susceptible to invasion (50 percent for moderately susceptible variants and 25 percent for variants with low susceptibility). We assume that the BEC zones that are vulnerable to purple loosestrife invasion are the Bunch Grass, Coastal Douglas-Fir, Coastal Western Hemlock, Interior Cedar Hemlock, Interior Douglas-Fir, and Ponderosa Pine. In future, a more accurate assessment would consider the proportion of wetland area within each of the susceptible BEC variants identified in the gap analysis.

The date of introduction of purple loosestrife into British Columbia was set at 1897, based upon a herbarium specimen collected in Stanley Park. The rate of spread was reported to be 14.8 percent in the United States (Duncan and Clark, 2005), but other studies put this somewhat lower, around 11 percent (Barbier and Knowler 2006). Since we do not have spread rate data for BC, we set three values for spread rates (low, medium, and high, at 10, 12.5, and 15 percent, respectively) to take into account the uncertainty in the estimation of damage curves for purple loosestrife. Identifying current estimates for the distribution of purple loosestrife would be helpful in validating the spread and damage curves for this species.

Damage Curves for Purple Loosestrife

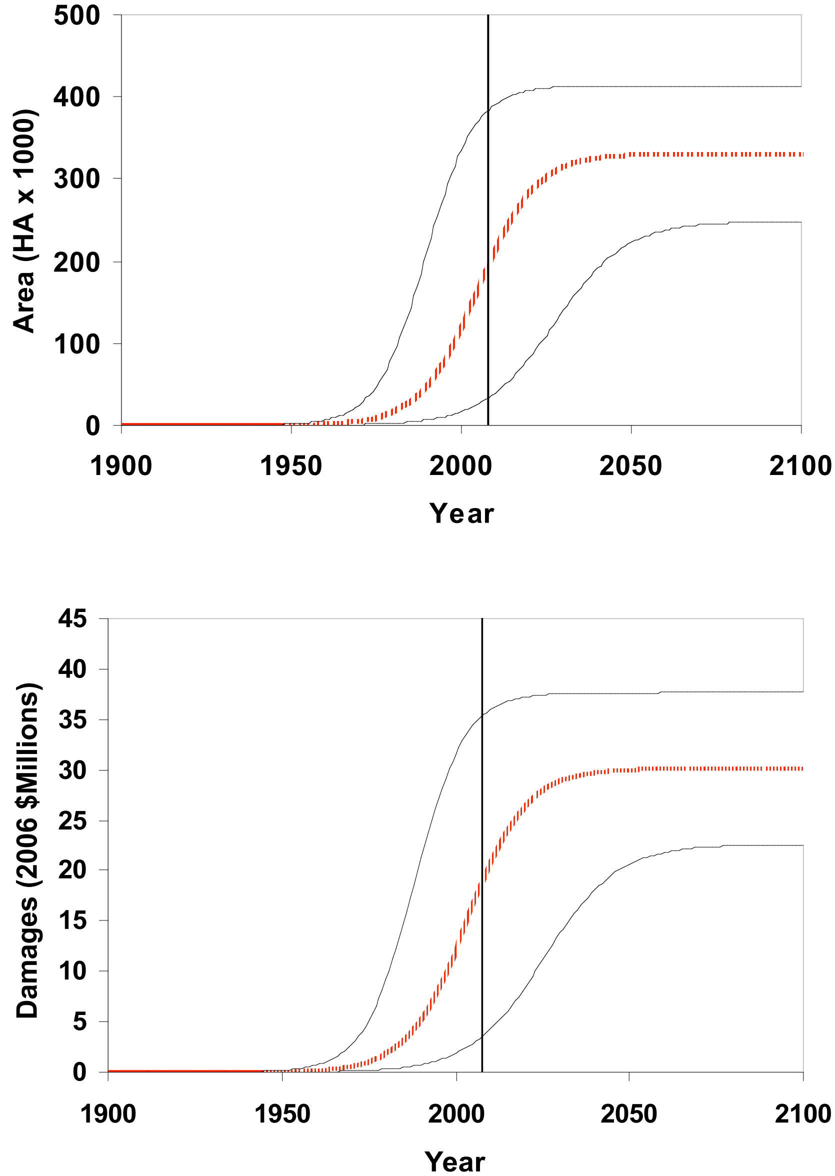

Figure 2.2.1.2. Estimated Area (a) and Economic Damages (b) from purple loosestrife in British Columbia Over Time.
The red line uses the mid range rate of spread (12.5%) and ecological limit (330,934 ha) estimate and the upper and lower lines use the fastest (15%) and slowest (10%) spread rates and highest (413,668 ha) and lowest (248,201 ha) ecological limits respectively. The shaded area represents our confidence in the estimates. We assume a logistic growth pattern with an initial infestation size of 1 ha in 1897. The vertical line represents the current year 2008.

2.2.2 Diffuse Knapweed

Impact Diagram

Diffuse knapweed (*Centaurea diffusa*) is a rangeland weed introduced to North America from the Mediterranean region and western Asia. Both managed rangelands and disturbed sites are susceptible to invasion by diffuse knapweed (Duncan and Clark 2005). Where present, diffuse knapweed can have impacts on various ecosystem components including those shown in Figure 2.2.2.1.

Diffuse knapweed out-competes more nutritious forage plants (*arrow 1*). Where present, it can reduce grass production by up to 50 percent (Myers and Berube 1983). Loss in forage plants can lead to a decrease in rangeland production per unit area of land (*arrow 4*, see Economic Damages Per Ha below). Loss of forage plants also affects wildlife species (*arrow 1*). Decreases in the population of game due to forage loss could lead to loss of recreation values for hunting and wildlife viewing (*arrow 5*). However, there is evidence that diffuse knapweed can provide nutritional value to some wildlife species such as bighorn sheep and deer (Duncan and Clark 2005).

Because of its taprooted morphology, diffuse knapweed can lead to the formation of surface crusts and reduced infiltration. This increases surface runoff rates, soil erosion (*arrow 2*) and sedimentation of nearby waterbodies (*arrow 8*) as well as reducing the moisture content of soils (Lacey et al. 1990 cited by Duncan and Clark 2005).

Diffuse knapweed is highly competitive and can form monotypic stands, out-competing native vegetation including species at risk (*arrow 3*). Species loss can lead to local and regional reductions in biodiversity (*arrow 7*) as well as reduced quality for recreation in terms of viewing and experiencing natural ecosystems (*arrow 6*).

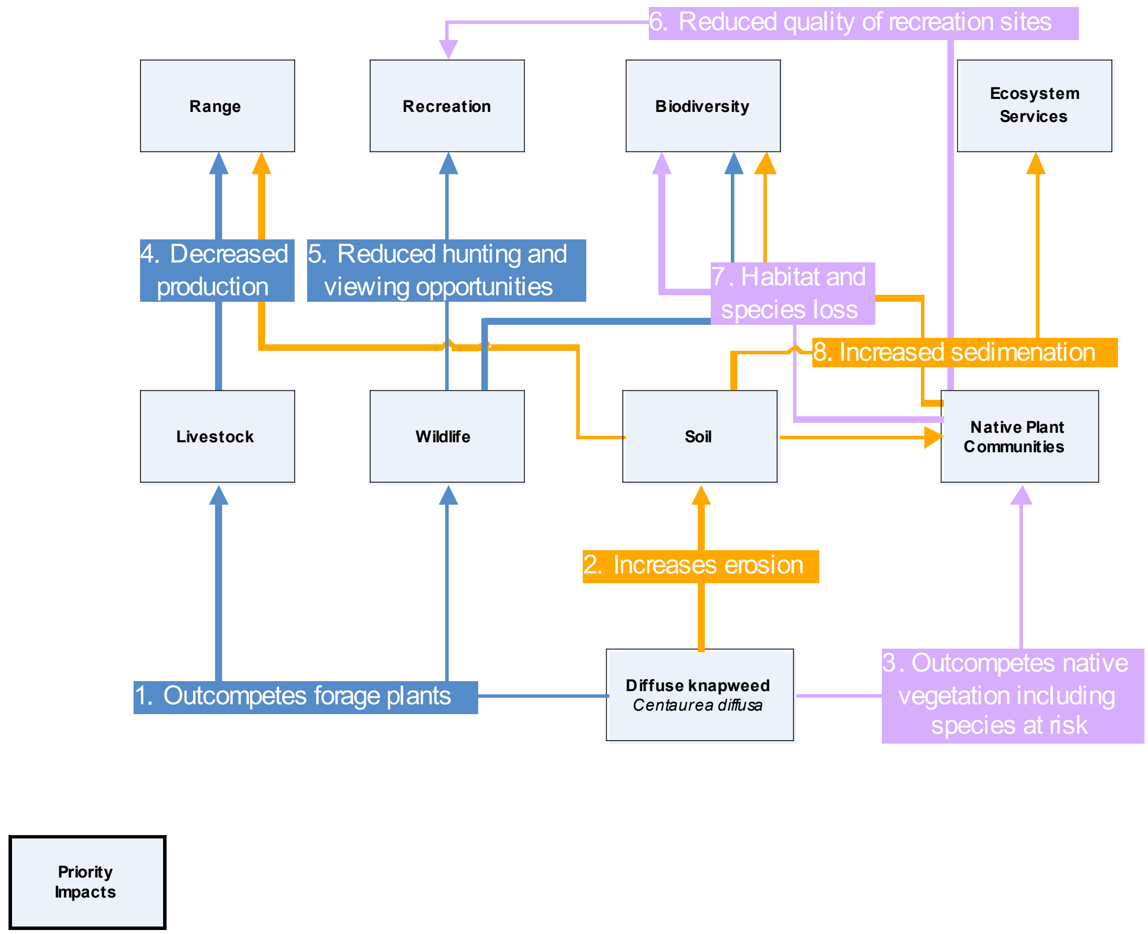

Figure 2.2.2.1. Impact Diagram for diffuse knapweed.
Boxes represent valued ecological and socioeconomic components affected by the plant.

Economic Damage per Hectare

We were able to locate economic loss estimates for three components of total loss: recreation, soil erosion and forage from US and Canadian sources, although these generally referred to all knapweed species taken together. Unit damage cost per ha from diffuse knapweed was estimated at $21.09 for BC in 2006 prices. The details of this calculation are shown in the following table.

Diffuse knapweed Unit Damage Estimate:

Impact	Original Estimate				Adjust -ments	Final Estimate 2006 ($/ha)	Notes
	Year of estimate	*Total Damage ($ million)*	*Area Infested (ha)*	*Unit Damage ($/ha)*			
1. Recreation loss	2004 (USD)	0.36	315,000	1.14	FER: X 1.32 CPI: X 1.14	1.72	1/
2. Soil Erosion	1993 (USD)			6.06	FER: X 1.30 CPI: X 1.27	10.01	2/
3. Forage loss	2006					9.36	3/
Total	2006					21.09	

Notes:

1/ Total values were calculated for consumers' surplus from recreation on Montana wild lands and this was adjusted for losses due to knapweeds per ha. Total consumers' surplus from hunting was estimated from Loomis (2005) and Fish and Wildlife Service (1991) for Montana at 1.94 million resident days X $48.55 per day. For the non-consumptive use value, the figures were 1.64 million days and $37.24 per day. Total consumers' surplus from both activities in 2004 was $155.26 million. We multiplied the following factors from Hirsch and Leitch (1996) to determine the loss in consumers' surplus: (i) reduction in wild land habitat value from infestation (0.35), (ii) area of wild lands affected (0.023), (iii) proportion of recreation (hunting/non-consumptive) occurring on wild lands (0.69), and (iv) activity lost due to reduced opportunities (0.42). Multiplying these factors gives a total loss from knapweeds of 0.23% in total consumers' surplus from hunting and non-consumptive activities in Montana. Total loss was divided by the infested area from Hirsch and Leitch to get the initial unit damages, which were then adjusted to a Canadian value in 2006 prices.

2/ From Hirsch and Leitch (1996). Based on an average reduction in soil and water conservation benefits of 25% on infested land and an estimate of total benefits of $9.80 per acre in 1993.

3/ While Harris and Cranston (1979) report losses from diffuse knapweed for BC of 1.1 AUM per ha, Duncan and Clark (2005) report a loss of 0.8 AUM per ha in Washington State and Hirsch and Leitch (1996) cite losses of .65 AUM per ha infested by all knapweeds in Montana. However, these figures seem too high for our purposes. Data from the 1994 BC Range Analysis (BC Ministry of Forests 1995) suggests Crown range productivity is as low as 0.29 AUM/ha. This is based on 376,000 AUMs on 1.3 million ha of Crown range in the Kamloops Region. If we include private rangeland as well, with possibly some improved pasture, a rough average for BC's main grazing areas might be closer to 0.5 AUM/ha. Following Harris and Cranston (1979), we assume this productivity declines on average by 43% under diffuse knapweed infestation, producing a figure of 0.215 AUM/ha as damages. We valued this loss two ways. First, we assessed the market rental value of an AUM at $25/AUM[1] providing one value for the lost forage of $5.38/ha infested. Second, we considered the cost of replacement forage; it would take 2.22 tones of hay to replace one AUM, based on a forage requirement of 450 kg/AUM. Pricing hay at $138/tonne (field data), this results in a value of replacement forage of $62/AUM and forage loss from diffuse knapweed at $13.33/ha infested. For our calculations, we took the mean of these two approaches.

[1] George Geldhart, BC Ministry of Forests and Range, Kamloops, e-mail communication October 16, 2008

Ecological Assumptions

Harris and Cranston (1979) estimate the ecological limit for diffuse knapweed at 1.1 million ha based on the soil types in its native habitats. We set the ultimate ecological carrying capacity at 1,132,374 hectares based on the ecological limits data in the Gap Analysis report (Miller and Wikeem 2005), with 25 percent confidence intervals between 849,280 ha and 1,415,467 ha. This is assuming that highly susceptible BEC variants could be 22 percent infested, mid susceptibility variants could be 15 percent infested, and low susceptibility BEC variants could be 6 percent infested. This estimate is also supported by the initial Harris and Cranston estimate of 1.1 million ha. The BEC zones with susceptible variants are: the Bunchgrass, Interior Cedar Hemlock, Interior Douglas-Fir, Montane Spruce, and Ponderosa Pine.

E-Flora BC (http://www.eflora.bc.ca/) shows a record of diffuse knapweed occurring in the southern end of the province between 1900 and 1925. Therefore, we set the date of introduction for diffuse knapweed at 1900.

The rate of spread was 16 percent in the United States (Duncan and Clark 2005). Muir (1986) identified the rate of spread for BC at 9 percent. Based on this variation we set three values for spread rates (low – 9 percent, medium – 15 percent, and high – 16 percent) to take into account the uncertainty in the estimation of damage curves for diffuse knapweed. Note that the medium rate was calibrated based on two independent estimates of area infested by knapweed in 1974 (Watson and Reney 1974) and 1983 (Muir 1986).

Damage Curves for Diffuse Knapweed

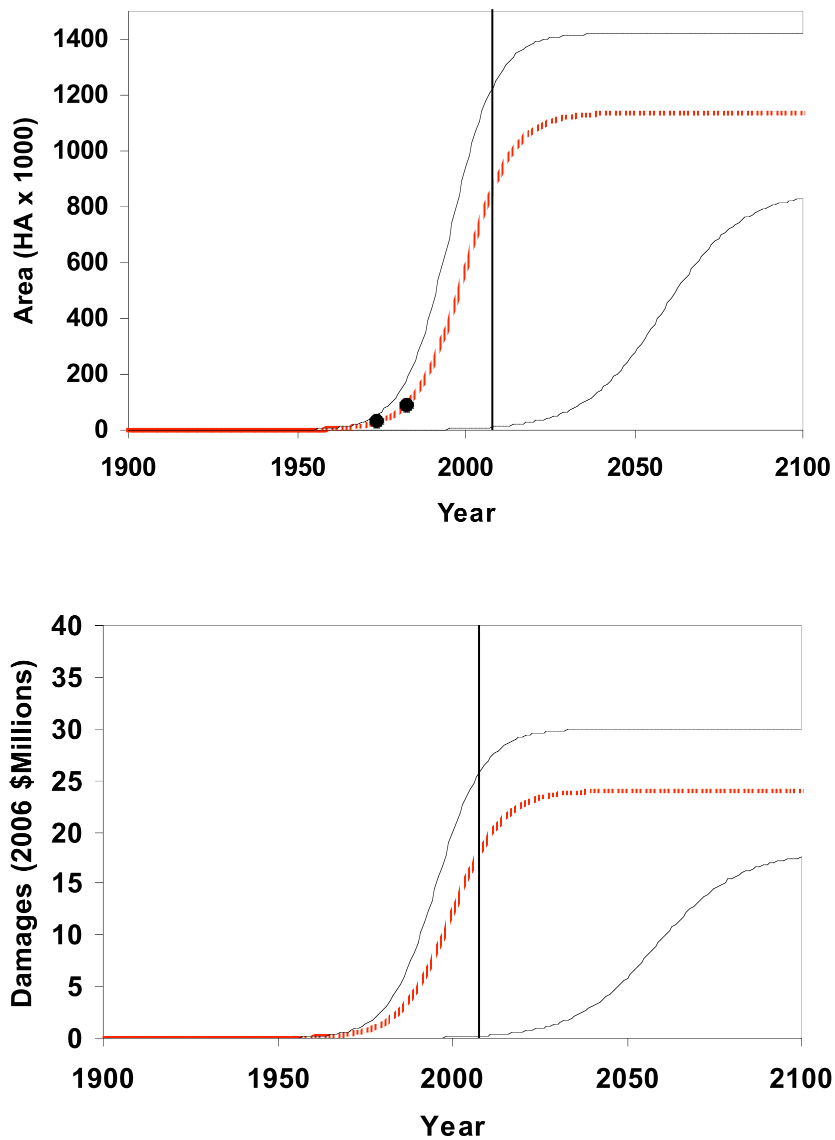

Figure 2.2.2.2. Estimated Area (a) and Economic Damages (b) from diffuse knapweed in British Columbia Over Time.

The red line uses the rate of spread (15%) and ecological limit (1,132,374 ha) estimate and the upper and lower lines use the fastest (16%) and slowest (9%) spread rates and highest (1,415,467 ha) and lowest (849,280 ha) ecological limits respectively. The shaded area represents our confidence in the estimates. We assume a logistic growth pattern with an initial infestation size of 1 ha in 1900. Note that the rate for the red line was calibrated with 2 black dots in the top figure representing estimates of area invaded by knapweed for 1974 (Watson and Reney 1974) and 1983 (Muir 1986). The vertical line shows the year of 2008.

2.2.3 Hawkweed

Impact Diagram

Hawkweed (*Hieracium* sp.) is a member of the sunflower family introduced to North America from Europe as an ornamental. Since its introduction, hawkweed has escaped into the landscape and become a noxious weed in various parts of the US and Canada. When present, it has various negative impacts on ecosystems. The impacts include those shown in Figure 2.2.3.1 (Wilson pers. comm.).

Hawkweed out-competes forage for both livestock and wildlife. It also out-competes crops, particularly hayfields in the Interior of BC (*arrow 1*). These decreases in forage and crop plants lead to decreases in production in the agriculture and range sectors of the economy (*arrow 6*) as well as in reduced opportunities to for hunting and wildlife viewing (*arrow 7*).

Hawkweed out-competes young seedlings in plantations (*arrow2*). There is some concern about incursion of hawkweed into areas affected by the mountain pine beetle epidemic. Invasion by hawkweed could prevent reforestation of sites where there is high beetle mortality (*arrow 8*, Wilson pers. comm.).

Hawkweed out-competes native vegetation communities including species at risk (*arrow 3*). It creates very dense clonal populations that exclude all other forms of vegetation. Restoration is difficult and requires reseeding because all plants are excluded from the area. These dense clonal populations of hawkweed can grow well on high-elevation alpine meadows, out-competing native plants. These sites are valuable to hikers and other recreationists (*arrow 9*). The formation of large clonal populations also results in impacts to wildlife from forage and habitat loss. Native plants experience competition and the loss of functional mycorrhizal associations (*arrow 4*) ultimately resulting in a reduction in biodiversity (*arrow 10*). The loss of natural mycorrhizal communities will result in decreased likelihood of recovery of native plant communities after shifts in species composition (*arrow 11*).

Hawkweed causes an intense hayfever response and can affect up to 80 percent of people exposed to it (*arrow 5*). Increases in the rate of allergic reactions will results in increased visits to physicians, increase in missed work days, and decreases in productivity of working peoples' lives (*arrow 12*).

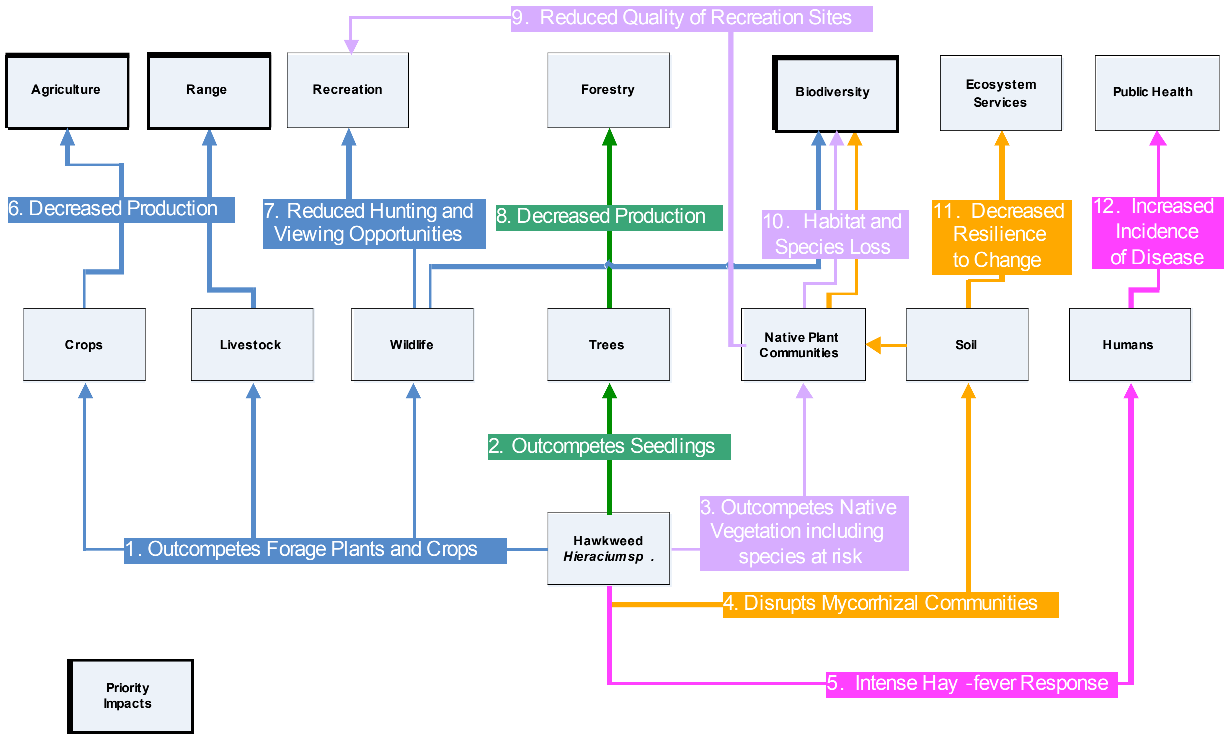

Figure 2.2.3.1. Impact Diagram for hawkweed.
Boxes represent valued ecological and socioeconomic components affected by the plant.

Economic Damage per Hectare

Wilson (2002) reports economic damages from hawkweed species as US $ 222 per ha in the United States. According to Linda Wilson (pers. comm.), these economic damages include losses to grazing, forestry, recreation (hunting and hiking), real estate, and wildland values due to hawkweed, but may represent only higher valued lands infested by it. It is also mentioned that the estimate comes from studies of leafy spurge damages, so it must be seen as 'speculative', as suggested by Duncan and Clark (2005). Nonetheless, there is no other information available so we converted this per-hectare damage cost to Canadian prices in 2006. The estimated economic damage was $165.04 per ha. The detail economic damage calculation is shown in the table below.

Hawkweed Unit Damage Estimate:

Impact	**Original Estimate**				**Adjust -ments**	**Final Estimate 2006** ($/ha)	**Notes**
	Year of estimate	*Total Damage ($ million)*	*Area Infested (ha)*	*Unit Damage ($/ha)*			
1. Grazing, forestry, recreation, real estate & wild land losses	2003 (USD)	58.2	262,162	222.00	FER: X 1.408 CPI: X 1.056 Adjust.: X 0.50	$165.04	1/
Total						$165.04	

Notes:

1/ From Wilson (2003). We reduced Wilson's damage estimate by 50% to obtain an estimate of the per ha value for BC. This reduction is based on the assumption that Wilson's original estimate is based on infestations of lands that are of high value to range production and recreation rather than all lands that are susceptible to hawkweed invasion, and to avoid any possible double counting involving real estate values.

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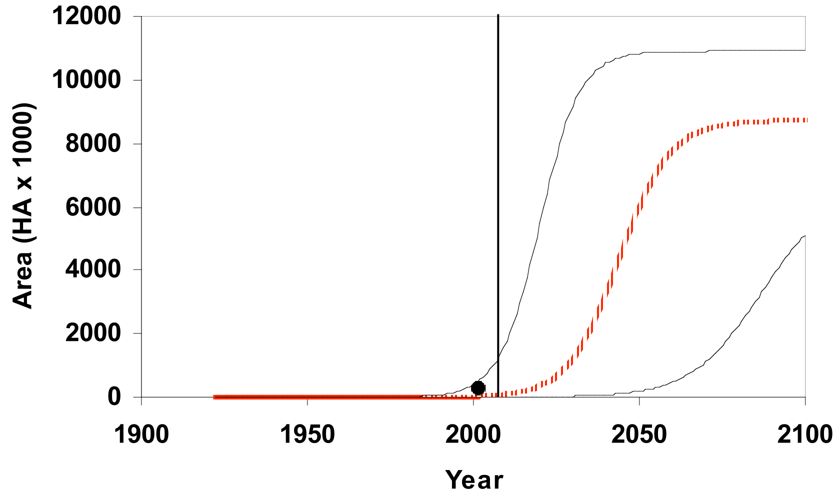
Ecological Assumptions

Based on the ecological limits data in the gap analysis report (Miller and Wikeem 2005) and on communications with Linda Wilson, we set the ultimate ecological carrying capacity at 8,722,479 hectares**,** with 25 percent confidence intervals between 6,541,859 ha and 10,903,099 ha. The BEC zones with susceptible variants were identified as: Ponderosa Pine, Interior Douglas-Fir, Interior Cedar Hemlock, Montane Spruce, Sub Boreal Pine Spruce, Sub Boreal Spruce, Englemann Spruce–Subalpine Fir, and the Coastal Western Hemlock. We assume that highly susceptible BEC Variants could be 30 percent infested, mid susceptibility variants could be 15 percent infested, and low susceptibility BEC variants could be 5 percent infested. The date of introduction of hawkweeds into BC was based upon on the earliest record of its presence in Canada in 1922.

The rate of spread was 11 percent in the United States (Duncan and Clark, 2005). Wilson and Callihan (1999) estimated the rate of spread at 16 percent. Since we do not have spread rate data for BC, we set three values for spread rates (low 10%, medium 14%, and high 18%) to take into account the uncertainty into our model estimation.

Damage Curves for Hawkweed



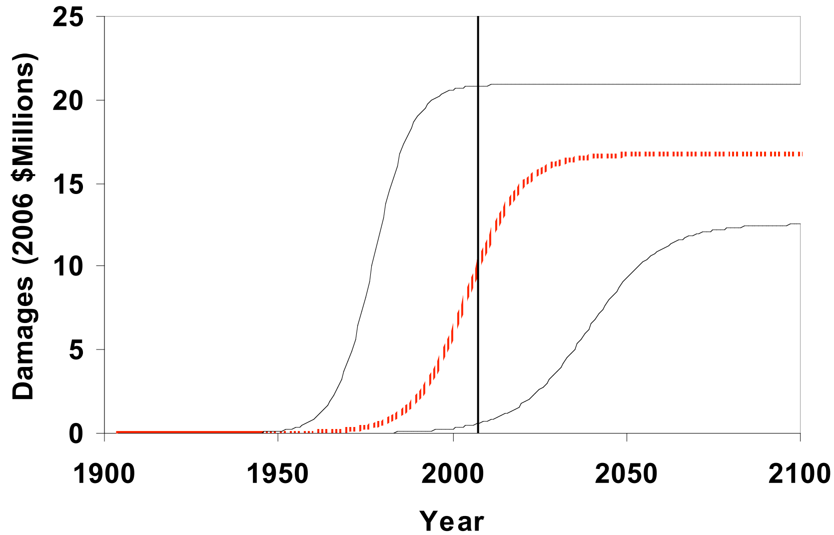


Figure 2.2.3.2. Estimated Area (a) and Economic Damages (b) from hawkweed in British Columbia Over Time.
The red line uses the rate of spread (14%) and ecological limit (8,722,479 ha) estimate and the upper and lower lines use the fastest (18%) and slowest (10%) spread rates and highest (10,903,099 ha) and lowest (6,541,859 ha) ecological limits respectively. The shaded area represents our confidence in the estimates. We assume a logistic growth pattern with an initial infestation size of 1 ha in 1922. Note that the black dot shows the estimated area infested in BC in 2002 (Wilson 2002).

2.2.4 Cheatgrass

Impact Diagram

Cheatgrass (*Bromus tectorum*) is an invasive annual grass introduced to North America from Eurasia. It is highly invasive in semi-arid rangelands, particularly in the Great Basin region of the United States. Where present, cheatgrass can have the ecological and economic impacts shown in Figure 2.2.4.1.

Cheatgrass can out-compete native perennial grasses and crop species (*arrow 1*), leading to decreased production of range and agricultural products (*arrow 5*). A reduction in forage for wildlife can reduce opportunities for hunting and wildlife viewing (*arrow 6*). Cheatgrass can also out-compete native vegetation (including species at risk, *arrow 2*). This can lead to a reduction in the quality of recreation sites (*arrow 7*) and biodiversity (*arrow 8*). Perhaps one of the greatest impacts of cheatgrass is its effect on fire regimes (*arrow 3*). Cheatgrass creates a higher continuity of fine fuels, not normally present in semi-arid rangelands. It is highly adapted to fire and will come to dominate a site following the occurrence of fire. Once cheatgrass dominates a site, the risk of fire is very high (*arrow 9*) and the habitat value for wildlife species (*arrow 8*), such as sage grouse, is very low.

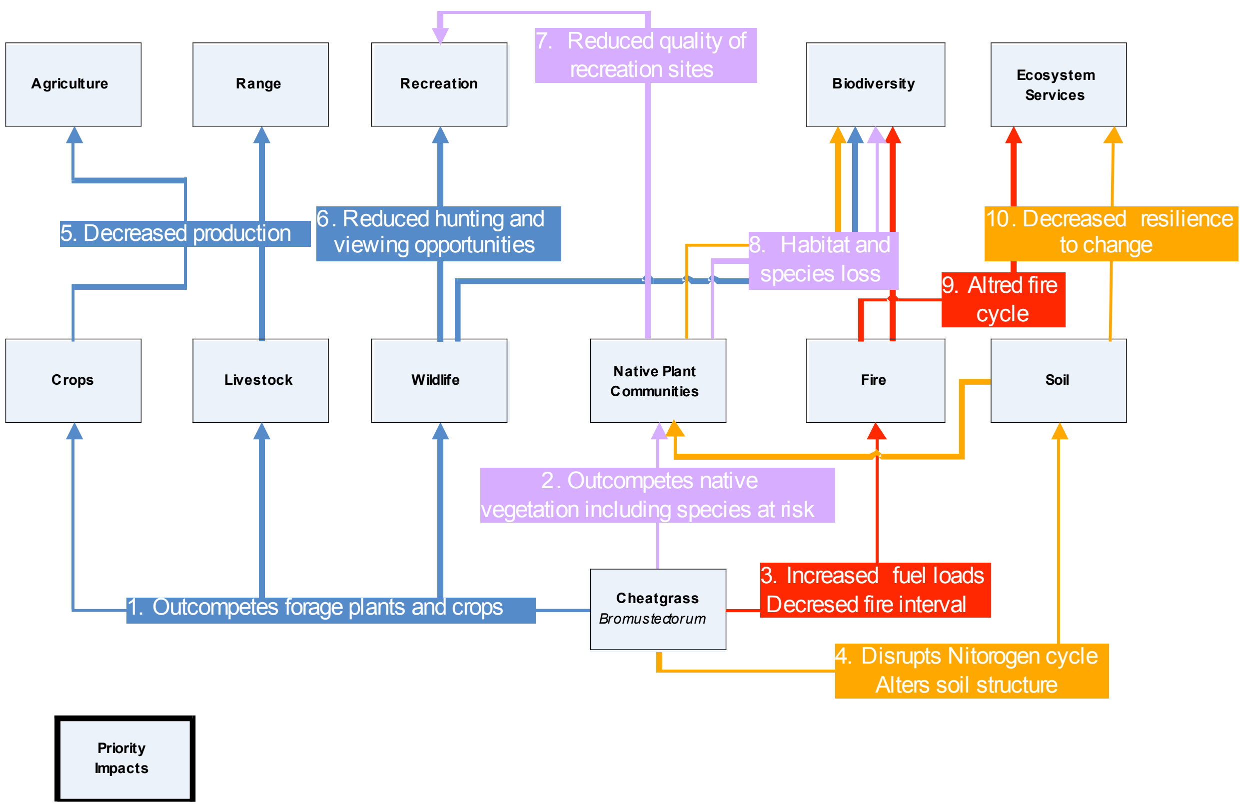


Figure 2.2.4.1. Impact Diagram for cheatgrass.
Boxes represent valued ecological and socioeconomic components affected by the plant.

Economic Damage per Hectare

We calculated unit economic damages from cheatgrass for BC using data from Knapp (1996), who estimated economic damages from wildfires due to cheatgrass in the Great Basin of the US, and from Haferkamp (2001), who investigated cheatgrass impacts on forage production, measured as detrimental physical impacts on livestock. In the former case, we converted Knapp's figure to the Canadian setting but reduced it by 50 percent to account for a lower fire risk in the equivalent Canadian areas of infestation. Hafercamp (2001) considered reduced weight gains for livestock grazing on cheatgrass-invaded pastures but only considers the case of control of cheatgrass, so that we adjusted these figures to damages in newly invaded areas. For other species invading rangelands, we have assessed damages as either the rental value of lost animal unit months (AUMs), or the value of replacement hay. However, we use Hafercamp's approach here since it represents a preferred economic measure (value of lost weight gain). The estimated damage is $20.09 per ha from cheatgrass in BC, and the details of the calculation are shown in the table below.

Cheatgrass Unit Damage Estimate:

Impact	**Original Estimate**				**Adjust -ments**	**Final Estimate 2006** ($/ha)	**Notes**
	Year of estimate	*Total Damage ($ million)*	*Area Infested (ha)*	*Unit Damage ($/ha)*			
1. Fire losses on rangeland	1991 (USD)	10,158,460	3,627,000	2.80	FER: X 1.14 CPI: X 1.310 BC: X 0.50	2.09	1/
2. Forage loss	2006					18.00	2/
Total						20.09	

Notes:

1/ From Knapp (1996). Average wild fire damages over 5 years are estimated for the Great Basin, based on, plus fire suppression and management costs. It is assumed that 50% of these losses are attributable to cheatgrass. Fire damages per ha invaded are based on infestation rates for the sagebrush-bunchgrass (20%) and shadscale zones (1%) where cheatgrass is dominant. Since various experts have argued that fire damages are lower in BC, we reduced the damage figure by 50% and then adjusted it to Canadian prices in 2006.

2/ From Hafercamp (2001). Annual reported livestock weight gain increases by 12 kg per ha when cheatgrass is suppressed. Weight gain/loss is priced at $1.50 per kg on the basis of livestock budgets for the interior of BC (Malmberg and Peterson 2006).

Ecological Assumptions

We assumed that cheatgrass can invade in the Coastal Douglas-Fir, Bunch Grass (BG) and Ponderosa Pine (PP) BEC zones. Assuming that it could ultimately invade 100 percent of the Bunchgrass zone, half of the Ponderosa Pine and 5 percent of the Coastal Douglas Fir zone, we estimate the ecological limit to be 521,550 hectares with 25 percent confidence intervals between 391,163 ha and 651,937 ha. Our assumptions of such high coverage for the BG and PP zones are based on densities in the Great Basin where entire landscapes have been converted to cheatgrass monocultures as a result of the interaction between heavy grazing and altered fire regimes. Although nothing like this has ever been experienced in BC, experts are concerned that climate change could make this scenario more likely (Don Gayton pers. comm.). The first record of cheatgrass in BC is from 1904. The spread rate of cheatgrass in the US is measured at 14 percent annually in Utah (Duncan and Clark 2005). We assumed a range of spreads (10, 14, and 18 percent) to take into account uncertainty.

Damage Curves for Cheatgrass

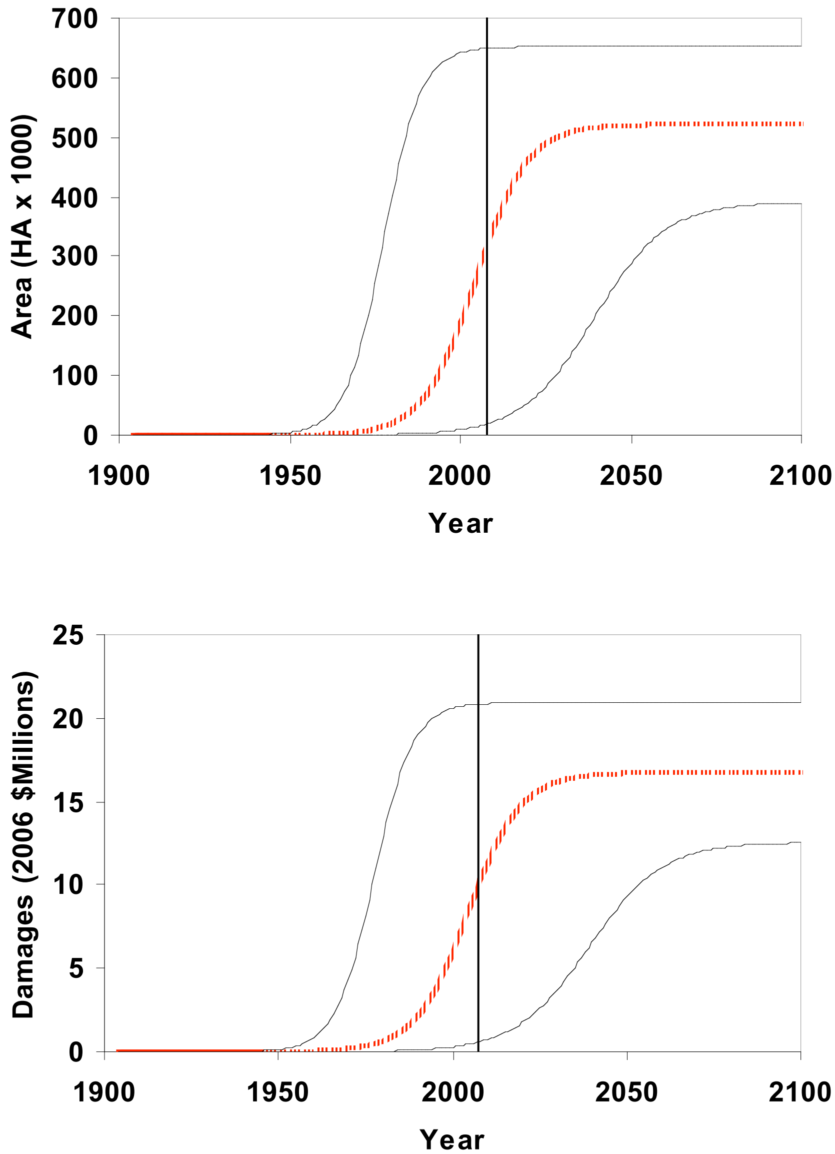


Figure 2.2.4.2. Estimated Area (a) and Economic Damages (b) from cheatgrass in British Columbia Over Time.
The red line uses the rate of spread (14%) and ecological limit (521,550 ha) estimate and the upper and lower lines use the fastest (18%) and slowest (10%) spread rates and highest (651,937 ha) and lowest (391,163 ha) ecological limits respectively. The shaded

area represents our confidence in the estimates. We assume a logistic growth pattern with an initial infestation size of 1 ha in 1904. The bold line represents the year of 2008.

2.2.5 Scotch Broom

Impact Diagram

Scotch broom (*Cystius scoparious*) was introduced to Vancouver Island from the Mediterranean area of Europe. Its first known record from Vancouver Island is for 1850. Since then, it has expanded its range extensively in coastal areas of the province, particularly Vancouver Island and the Gulf Islands, where it poses a threat to endangered Garry oak ecosystems. Where present, Scotch broom can have various impacts on the ecosystem and economy. These are described in detail in Figure 2.2.5.1.

In Oregon, Scotch broom has been shown to out-compete forage for both wildlife and livestock (*arrow 1*). Reduced forage for livestock ultimately results in reduced production (*arrow 8*). Reduced forage for wildlife may result in reduced hunting and viewing opportunities (*arrow 9*) and reduced biodiversity *(arrow 13*). Scotch broom has been shown to reduce tree growth in commercial tree plantations, and in some cases entirely prevent the growth of seedlings (*arrow 3*). This ultimately results in reduced timber production (*arrow 11*). Scotch broom increases fuel continuity (*arrow 4*) and can alter fire regimes (*arrow 12*), increasing the risk of fire for timber plantations, utility corridors, and private property, as well as for sensitive ecosystems. Broom out-competes native vegetation, significantly altering the composition of native plant communities (*arrow 5*) and reducing the quality of recreation sites and resulting in habitat loss for native species (*arrows 10 and 13*). This is particularly significant in the sensitive Garry oak ecosystems of southwestern BC. Finally, Scotch broom may change the nitrogen composition of the soil (*arrow 7*) and therefore possibly alter the potential vegetation composition of a site even after Scotch broom has been removed (*arrow 14*).

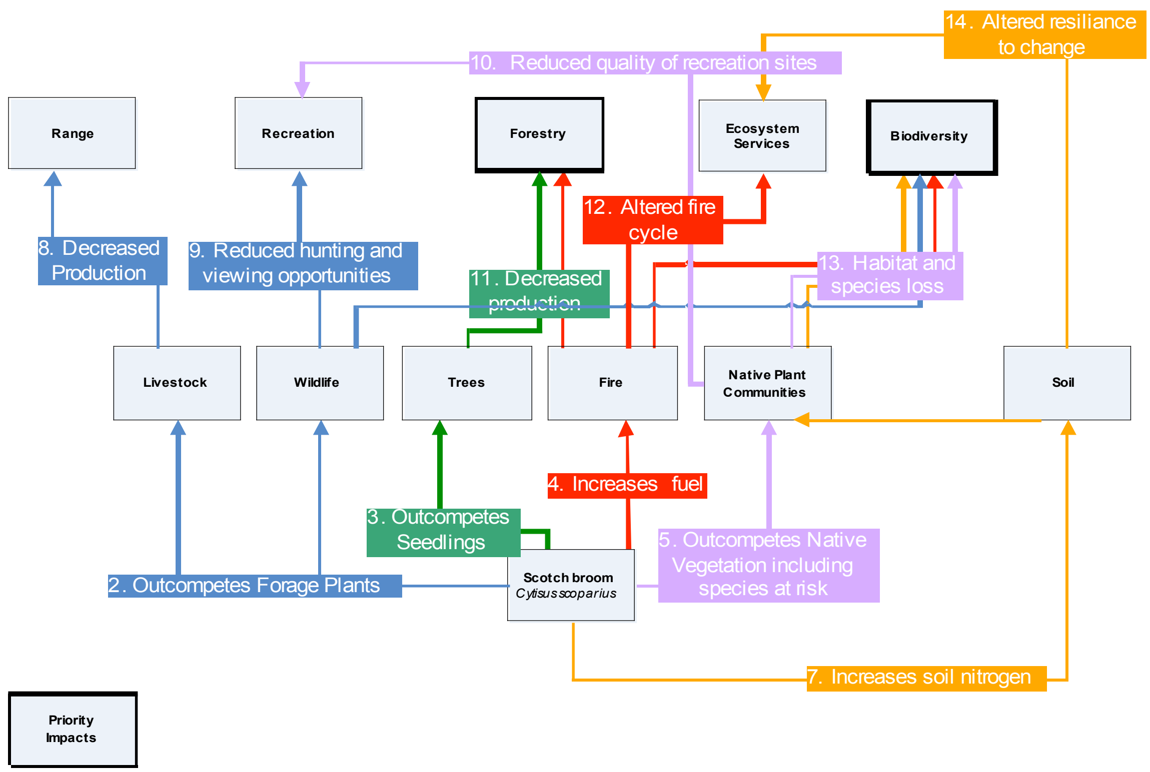


Figure 2.2.5.1. Impact Diagram for Scotch broom.
Boxes represent valued ecological and socioeconomic components affected by the plant.

Economic Damage per Hectare

No studies of the economic damages from Scotch broom have been undertaken in Canada. Instead, we used data from Oregon to estimate the unit damage cost for BC. The Oregon Department of Agriculture (2000) estimated the total economic damage and the total infested area for Scotch broom in Oregon. They developed a theoretical framework that takes into account biophysical impact, direct, and indirect economic impact of invasive species infestation in measuring economic damage per hectare. We used the benefit transfer method to measure unit damage cost per hectare based on this theoretical framework. The cost components for economic damage estimates were selected based on the discussions with experts. According to this report, measurable economic damage from Scotch broom consisted of lost agricultural production (forage), timber sales, and wildlife feed. The US figure was adjusted with the exchange rate to convert to Canadian prices and then inflation was taken into account. The adjusted economic damage value is $39.51 per ha in Canadian dollars. Details of the economic damage estimation are shown in the table below.

Scotch Broom Unit Damage Estimate:

Impact	**Original Estimate**				**Adjust -ments**	**Final Estimate 2006** ($/ha)	**Notes**
	Year of estimate	*Total Damage ($ million)*	*Area Infested (ha)*	*Unit Damage ($/ha)*			
1. Agriculture, timber & wildlife losses	2000 (USD)	14,221,200	607,300	23.42	FER: X 1.48 CPI: X 1.14	39.51	1/
Total						39.51	

Notes:

1/ From Oregon Department of Agriculture (2000). Damages are measured as loss in economic value, with reduced timber growth being the most important component. Unrealized timber production is set at 0.125 million board feet (MBF) per acre per year, priced at $500 per MBF. Agriculture and wildlife losses are due to reduced forage production, based on productivity of 2 acres per AUM. Affected lands are assumed to be split evenly between agricultural/wildlife use and timber. Further details were not provided in the Oregon study.

Ecological Assumptions

Ecological data were based on the information collected by Lisa Scott at Eco-Matters Consulting from Dave Polster (pers. comm.) and Linda Wilson at the Ministry of Agriculture and Lands (pers. comm.). We assume that Scotch broom can invade four biogeoclimatic zones: Coastal Douglas-fir, Coastal Western Hemlock, Interior Cedar Hemlock, and the Interior Douglas-fir. We set the date of introduction of Scotch broom into BC at about 150 years ago based upon the earliest recorded of its presence in Canada in 1850, which was said to take place on Vancouver Island.

We set the ultimate ecological carrying capacity at 1,836,394 hectares based on the ecological limits data in the gap analysis report (Miller and Wikeem 2005), with 25 percent confidence intervals between 1,377,296 ha and 2,295,493 ha. This is assuming that highly susceptible biogeoclimatic variants could be 15 percent infested, mid susceptibility variants could be 10 percent infested, and low susceptibility variants could be 5 percent infested.

Data from the Invaders Database (Rice 2008) on the number of US Pacific North West counties with Scotch broom present suggests a spread rate of 5 percent. Since we do not have spread rate data for BC, we set three values for spread rates (low 5%, medium 7.5%, and high 10%) to take into account the uncertainty in our model estimation. Better information on the current distribution of Scotch broom would help improve our estimates.

Damage Curves for Scotch Broom

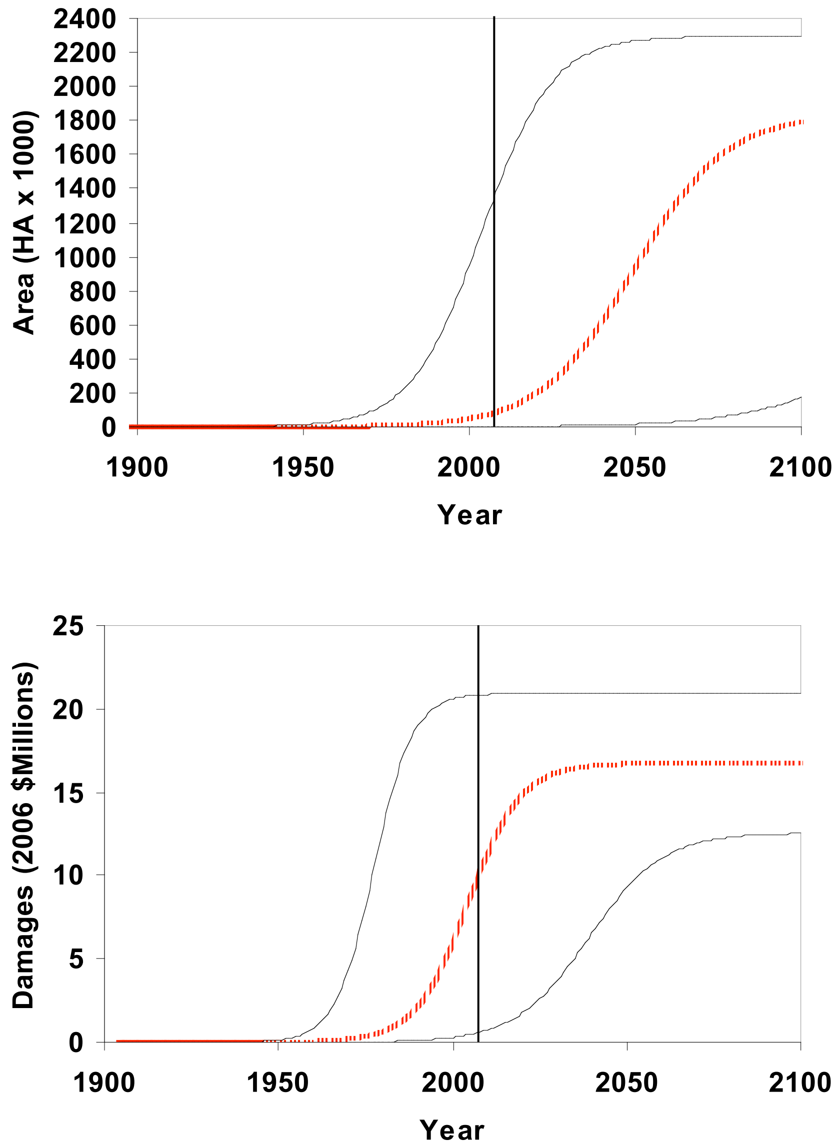


Figure 2.2.5.2. Estimated Area (a) and Economic Damages (b) from Scotch broom in British Columbia Over Time.
The red line uses the rate of spread (7.5%) and ecological limit (1,836,394 ha) estimate and the upper and lower lines use the fastest (10%) and slowest (5%) spread rates and highest (2,295,493 ha) and lowest (1,377,296 ha) ecological limits respectively. The shaded area represents our confidence in the estimates. We assume a logistic growth pattern with an initial infestation size of 1 ha in 1850. The bold line shows the year of 2008.

2.2.6 Eurasian Watermilfoil

Impact Diagram

Eurasian watermilfoil (*Miriophyllum spicatum*) was first detected in BC in the Okanagan Basin in 1970. It spread very rapidly and within four years was well established within all of the main lakes in the Okanagan Basin. Eurasian watermilfoil is an aquatic plant that grows in dense mats in water depths of up to six metres. Where present, it can cause the impacts on people and the environment shown in Figure 2.2.6.1.

Eurasian watermilfoil can increase the amount of habitat for permanent pool mosquitoes (Smith and Barko 1990) which can increase the population size of mosquitoes and, therefore, the exposure risk of mosquito-borne diseases (*arrow 1, arrow 7*). Eurasian watermilfoil can reduce habitat quality for waterfowl (*arrow 2*) and for some fish species (*arrow 3*). These impacts can in turn reduce hunting and viewing opportunities and productivity for fisheries (*arrows 8 and 10*). Watermilfoil can out-compete native plants for light and space (*arrow 4*) therefore reducing biodiversity (*arrow 10*) and the quality of recreation sites (*arrow 9*) for swimming, boating, fishing, and the general aesthetic appeal of the waterfront. Decaying mats of watermilfoil can reduce water oxygen levels (Honnell et al. 1992), alter P:N ratios (Nichols and Keeney 1973), and increase water pH and temperatures (*arrow 5*), therefore reducing water quality (*arrow 11*). Watermilfoil can also clog pipes in irrigation canals, and water and power generation intakes (*arrow 6*), therefore increasing maintenance costs for infrastructure (*arrow 12*) and reducing power generation (*arrow 13*).

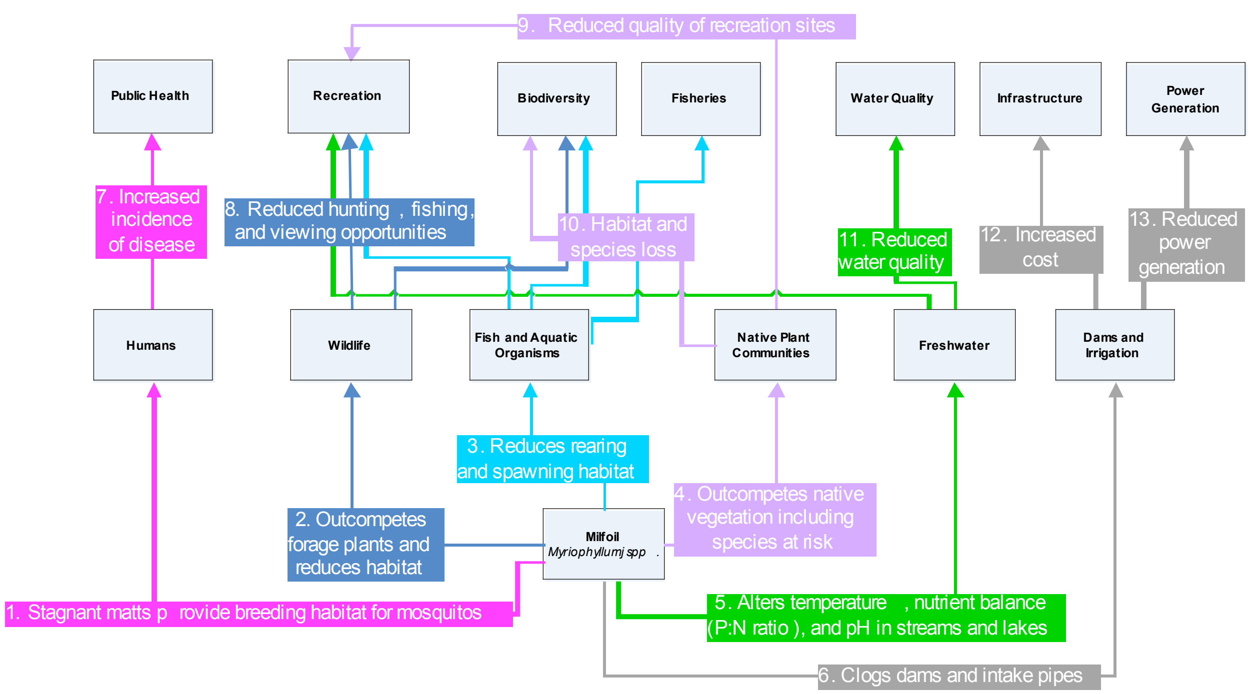


Figure 2.2.6.1. Impact Diagram for Eurasian watermilfoil.
Boxes represent valued ecological and socioeconomic components affected by the plant.

Economic Damage per Hectare

Only limited data on economic damage from Eurasian watermilfoil in British Columbia exist, consisting primarily of a study of a watermilfoil control project on Lake Okanagan carried out by the BC Ministry of Environment (Ference Weicker & Company 1991). Additional studies have been carried out in the US, such as Eiswerth et al. (2000) who estimated the net economic damage to the water-based recreation activities in western Nevada and northeastern California. However, this study was not useful because it only determined the value of recreation in the infested area but has no information about the impact of watermilfoil or the value of damages (reduction in recreational value). Since baseline recreational values tend to be highly site specific, we decided to use the older BC government study and modify and update the figures as required. We also ignore a number of items included in the damage estimates in the BC government study, such as impacts on beachfront property, since they represented double counting. Our estimate of the value of recreation lost from Eurasian watermilfoil in 2006 prices is $954.95 per ha infested. Given growth in the population and numbers of recreational visitors in the region since 1991, this value should be taken as a lower bound.

Eurasian Watermilfoil Unit Damage Estimate:

Impact	**Original Estimate**				**Adjust -ments**	**Final Estimate 2006** ($/ha)	**Notes**
	Year of estimate	*Total Damage ($ million)*	*Area Infested (ha)*	*Unit Damage ($/ha)*			
1. Recreation loss	1991	728,782	1000	728.78	CPI: X 1.31	954.95	1/
Total						954.95	

Notes:

1/ From Ference Weicker & Company (1991). To make our estimate we used (i) the number of beach user days by residents/non-residents in 1991 (8.8 million days), (ii) the percentage of users from a local survey indicating they would be willing to pay more for control of watermilfoil (22%), (iii) the additional amount these individuals would be willing to pay per year for improved control, which we treat as a measure of the damages per affected beach user ($7.86/year) and (iv) the average number of beach days per user (20.88 days). Multiplying these items provides a measure of the total damages incurred in 1991 prices, and this value was then adjusted by the area infested and updated to 2006 prices to give a per ha estimate in 2006 prices.

Ecological Assumptions

Eurasian watermilfoil was first confirmed in BC in 1970 at Okanagan Lake near Vernon (Ferrence Weiker and Company 1991). Once in a waterbody, it can spread extremely quickly, but movement between waterbodies is bound to be slower because of limitations in transport vectors. To estimate provincial scale spread rates, we considered data from Wisconsin and Minnesota. In Wisconsin, data on the cumulative number of counties with watermilfoil present over a 28-year period from 1969 to 1997 indicate an intrinsic spread rate of 15 percent (Buchan and Padilla 2000). Data from Minnesota on the number of lakes affected over a 15-year period indicate an intrinsic spread rate of 18 percent (Roley and Newman *in press*). Based on these numbers and the fact that once within a water system, watermilfoil can spread extremely rapidly

(40 percent in the Okanagan, Ferrence Weiker and Company 1991), we used the following estimates for Eurasian watermilfoil spread in our damage curves (low 15%, medium 17.5%, and high 20%). To estimate the ecological limit of watermilfoil in the province, we considered that in the Okanagan it occupies about 2 percent of the lake area and is likely near its ecological limit. Watermilfoil is adapted to mesotrophic lakes (Madsen 1998) of which the Okanagan Basin is likely over-represented in the province. We therefore assume that Eurasian watermilfoil could invade 0.75 to 1.25 percent of the lake area in BC, or between 16,800 and 28,000 ha.

Damage Curves for Eurasian Watermilfoil

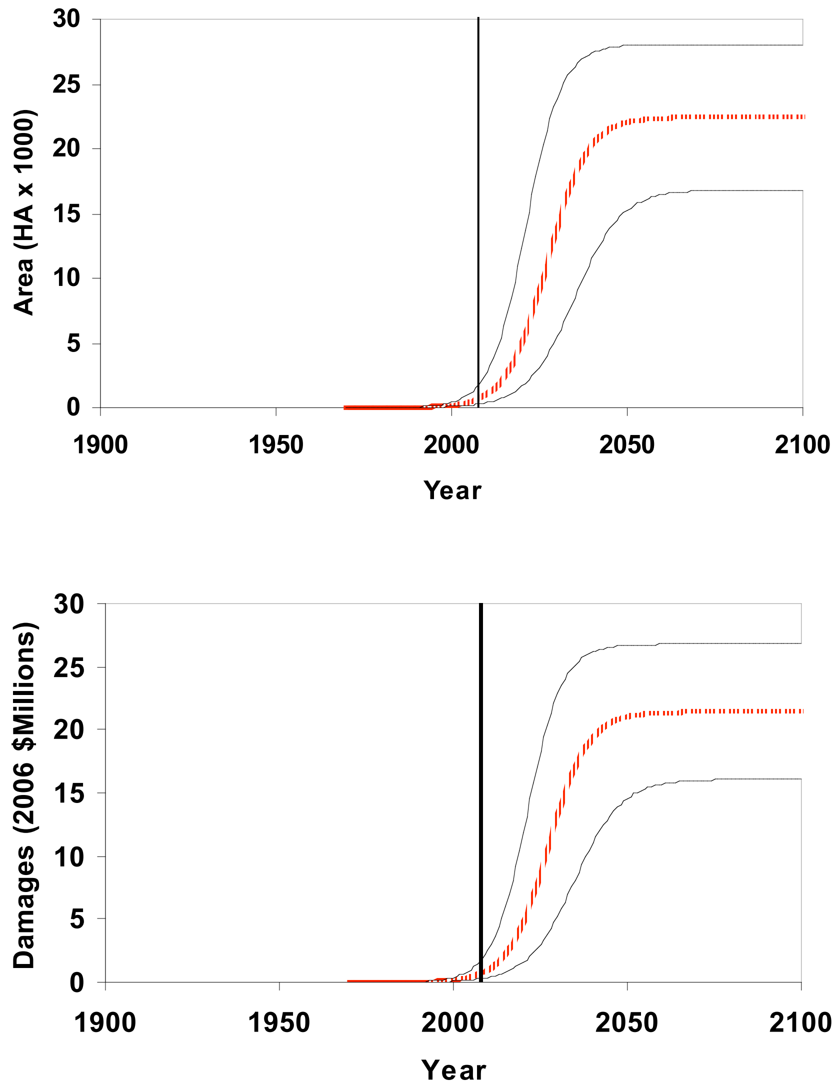


Figure 2.2.6.2. Estimated Area (a) and Economic Damages (b) from eurasian watermilfoil in British Columbia Over Time.
The red line uses the rate of spread (17.5%) and ecological limit (21, 000 ha) estimate and the upper and lower lines use the fastest (20%) and slowest (15%) spread rates and highest (28,000 ha) and lowest (16,000 ha) ecological limits respectively. The shaded area represents our confidence in the estimates. We assume a logistic growth pattern with an initial infestation size of 1 ha in 1970. The bold line shows the year of 2008.

2.2.7 Dalmatian Toadflax

Impact Diagram

Dalmatian toadflax (*Linaria dalmatica* (L.)) is also called broadleaf toadflax and wild snapdragon. This invasive plant lives up to five years with an average lifespan of 3.6 years. The life span of Dalmatian toadflax depends on environmental conditions and the reproductive capacity of individual plants (Duncan and Clark 2005). Dalmatian toadflax has been identified as a restricted weed in three Canadian provinces and 11 US states (Rice 2003 cited in Duncan and Clark 2005). Dalmatian toadflax grows throughout the United States and Canada, with extensive infestation in British Columbia, Alberta, and the northwestern United States.

Dalmatian toadflax was first confirmed in BC in 1952 (Environmental Dynamic Inc. 2006). It spreads very rapidly in areas where summers tend to be dry. Bunchgrass range sites, as well as disturbed areas, roadsides, vacant lots, and cemeteries are particularly susceptible to the invasion of Dalmatian toadflax (Duncan and Clark 2005). Pristine areas and rangelands in good condition can be invaded by Dalmatian toadflax due to seedling establishment within naturally occurring disturbed areas. Where present, it can cause the impacts on the economy and environment shown in Figure 2.2.7.1.

Dalmatian toadflax significantly reduces forage yield and out-competes native vegetation including species at risk (*arrows 1& 3*). Competition between forage plants and Dalmatian toadflax has negative impact on livestock and wildlife population. Invading Dalmatian toadflax in rangelands leads to reduced hunting and viewing opportunities and production of livestock *(arrows 4 & 5)*. This also creates adverse impact on biodiversity (*arrow 7*). Competition between Dalmatian toadflax and native vegetation also leads to loss of habitat and loss of biodiversity (*arrow 7)*. Furthermore, Dalmatian toadflax in native plant communities reduces the quality of recreation sites (*arrow 6*). Dalmatian toadflax also contains toxic compounds *(arrow 2)* and thereby reduces the livestock production and rangeland productivity *(arrow 4)*.

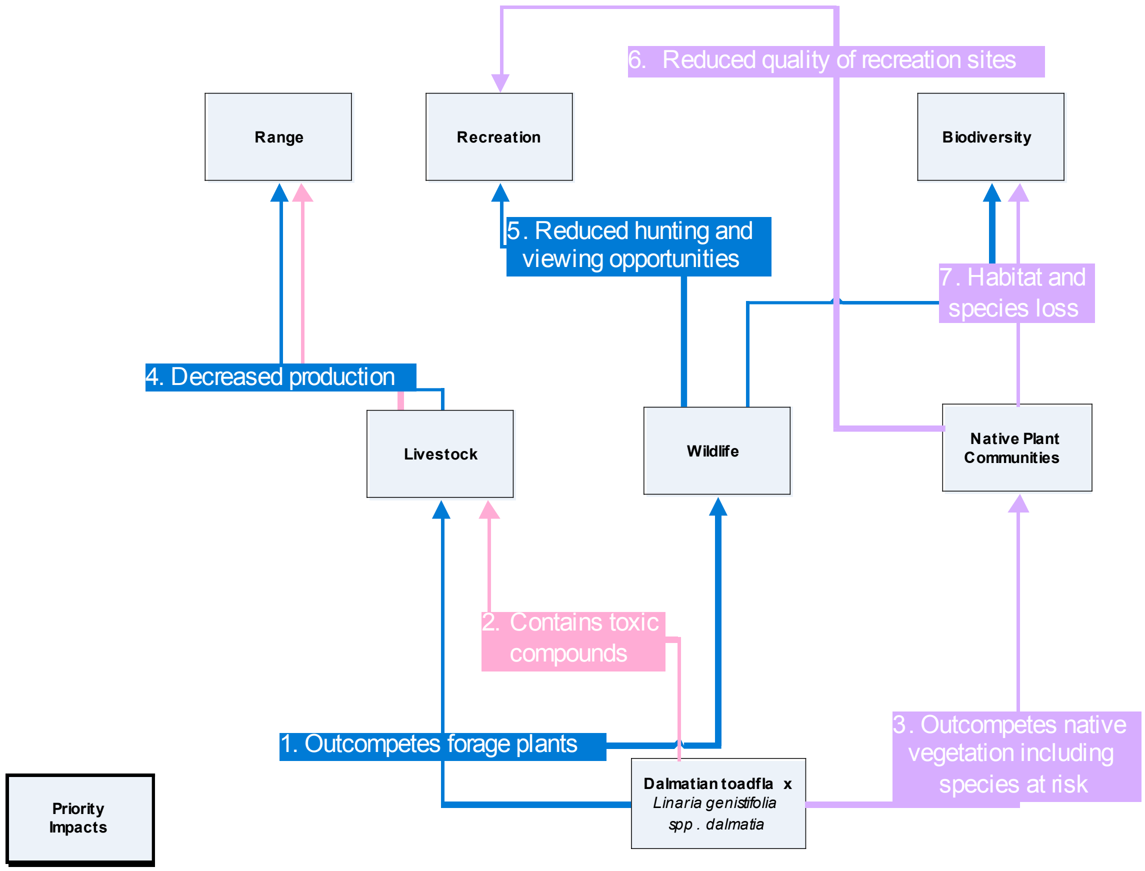


Figure 2.2.7.1. Impact Diagram for Dalmatian Toadflax.
Boxes represent valued ecological and socioeconomic components affected by the plant.

Ecological Assumptions

Rate of spread of Dalmatian toadflax varies throughout the Pacific Northwest. Based on the date of introduction in 1952, the annual rate of spread Dalmatian toadflax is 29 percent in Montana (Duncan 2001). Referring to the study by Rice (2003), Duncan and Clark (2005) showed that the annual spread rate in Oregon state is about 11 percent. According to the study conducted by (USDI BLM 1985), the estimated annual rate of spread is 8 percent for the Pacific Northwest (Duncan and Clark 2005). However, we used the data provided by Val Miller on the infestation of Dalmatian toadflax in the Kootenay region of British Columbia to calculate the annual spread rate. According to Miller's data in the Kootenays, the area invaded by Dalmatian toadflax increased from 3900 ha in 1995 to 8700 ha in 2000. Based on this data, we calculated the annual spread rate for BC as 13 percent. Furthermore, we used following spread rates (low, moderate, and high at 10, 13, and 16 percent, respectively) for estimating damage curves for Dalmatian toadflax.

We set the ultimate ecological carrying capacity at 1,342,105 hectares based on the ecological limits data in the Gap Analysis report (Miller and Wikeem 2005), with 25 percent confidence intervals between 1,006,579 ha and 1,677,631 ha. This is assuming that highly susceptible BEC variants could be 20 percent infested, mid susceptibility variants could be 10 percent infested, and low susceptibility BEC variants could be 5 percent infested

Damage Curves for Dalmatian Toadflax

We were unable to find any data on economic damages per unit area for Dalmatian toadflax. We therefore present here only an estimate of the area invaded by Dalmatian toadflax over time.

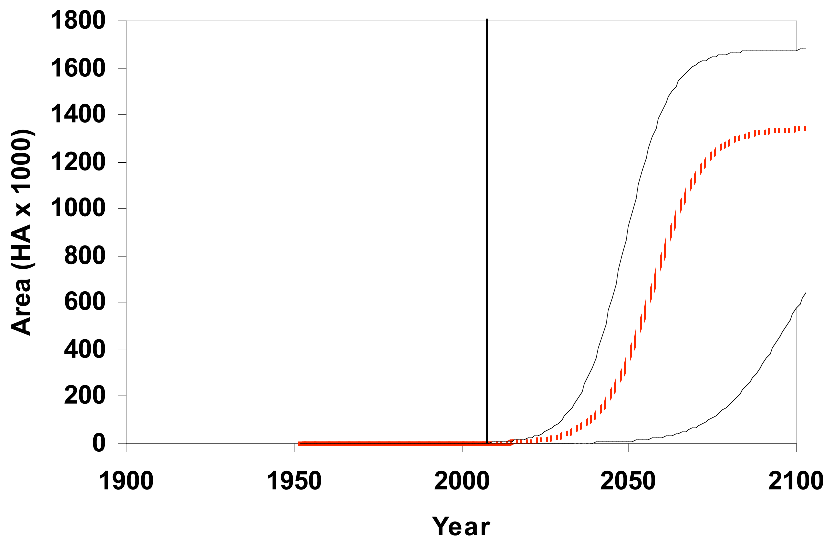


Figure 2.2.7.2. Estimated area invaded by Dalmatian toadflax in British Columbia over time. The red line uses the rate of spread (13%) and ecological limit (1.3 million ha) estimate and the upper and lower lines use the fastest (16%) and slowest (10%) spread rates and highest (1.6 million ha) and lowest (1.0 million ha) ecological limits respectively. The shaded area represents our confidence in the estimates. We assume a logistic growth pattern with an initial infestation size of 1 ha in 1952. The bold line shows the year of 2008.

2.3 Summary of Damages

Table 2.3.1 shows the mean damages predicted from the six species studied for which could calculate economic damages, and the predicted area invaded for Dalmatian Toadflax, for the years 2008 and 2020. We found economic information to substantiate that BC experienced at least $65 million in damages from these six species, since we were able to report only the limited economic information that was available. Most notable is the predicted large increase in damages for hawkweed to the year 2020 because of its relatively recent invasion date and the large ecological limit for this species in the province. These estimates form a baseline against which to evaluate alternative management strategies analyzed in Phase 2 of the project (presented in section 3).

Table 2.3.1. Summary of Estimated Economic Damages and Area Invaded for the Years 2008 and 2020 Across the Seven Species Studied.
These are probably underestimates as we were not able to find economic data on many of the pathways identified in the impact diagram.

Species	Annual Damages (Can $ Millions)	
	2008	2020
Eurasian Watermilfoil	1	5
Scotch Broom	3	8
Cheatgrass	10	15
Diffuse Knapweed	18	23
Purple Loosestrife	20	28
Hawkweed	13	60
Total	**65**	**139**
	Total area invaded (ha)	
Dalmatian Toadflax	9250	123400

3. Phase 2 – Economic Analysis of Alternative Management Actions

3.1 Economic Analysis Methods

3.1.1 Species Selection

The first step in this phase was to identify a subset of species from Phase 1 for which we would conduct the cost-benefit analyses of alternative management strategies. The analyses include a consideration of alternative conventional management actions, biological control, escalation of control costs, and increased management along utility corridors. Table 3.1.1 shows the four species selected and which analyses were conducted for each. "Management alternatives" in Table 3.1.1 include an analysis of biocontrol for diffuse knapweed and hawkweed, utility corridors for Scotch broom, and conventional management for watermilfoil.

Table 3.1.1. Phase 2 Analyses.

Species from Phase 1	Management Alternatives[1]	Escalation Costs
Diffuse Knapweed	✓	
Hawkweed	✓	✓
Scotch Broom	✓	
Purple Loosestrife		
Cheatgrass		
Eurasian Watermilfoil	✓	
Dalmatian Toadflax		

Diffuse knapweed is the species for which we had the best data on dispersal curves. It is also a species for which we had reasonable data on economic damages, including some information for BC. Additionally, diffuse knapweed is a species for which biocontrol has successfully been applied, and it therefore lends itself well to conducting a cost-benefit analysis for a biocontrol program.

Hawkweed is a species that is relatively new to the province and is still in the early stages of invasion. Phase 1 results show that it has huge potential for expansion in the province. This potential for expansion lends itself well to an analysis of the escalation of control costs, as well as an estimate of the net benefit of biocontrol and conventional management treatments.

Scotch Broom is the only species on our list that lends itself to analysis of the net benefits of increased management along utility corridors.

Eurasian watermilfoil is a species for which we had good data on control and management costs for BC, and is well-suited for an analysis of the net benefits of increased management investment.

3.1.2 Identifying Alternative Management Scenarios

In a general sense, management of most invasive plants in BC would likely focus on the following types of actions:

1. Inventory and mapping with the goal of detecting invasive plants at locations where they were previously not known to occur. This activity is critical for Early Detection and Rapid Response, as without it, what starts out as small infestations that could be easily controlled become large, uncontrollable ones.
2. Conventional (chemical/mechanical) treatments of locations where invasive plants are known to occur.
3. Biological control releases at locations where invasive plants are known to occur.
4. Land management to reduce the likelihood of invasion through improved range and development practices. This type of activity is difficult to analyze because there are so many different kinds of things that can be done and a myriad of economic responses.
5. Education of the public to enlist their help in monitoring for invaders (action 1) and preventing their spread (action 4). This is also difficult to analyze. There may be literature on education which could enable some degree of qualitative assessment but not a quantitative analysis.

Resources for invasive plant management are limited and each of the above actions has a cost associated with it. There is a finite amount of each of these activities which can be performed, and there are trade-offs associated with adding more resources to one of these activities at the expense of taking resources away from another. In addition, each of these activities may have a certain level of effectiveness that depends on the resources allocated towards it. Not all monitoring efforts may detect invasive plants that are present, and not all control efforts may eradicate or reduce the size of the targeted population.

Our goal in this part of Phase 2 was to identify the following for each of the selected species[2]:

1. What are the resources available for the management of these invasive plants?
2. How are these resources currently being allocated across the five general management alternatives described above? What are some of the specifics of the current management approach for these species?
3. For each type of activity and each species, what are the costs of implementing these alternative management actions?
4. For each type of activity and each species, what is the level of effectiveness at accomplishing the intended goal? For example, for monitoring, this would be the probability of detection; for control, it would be the percent reduction in population size, and rate of spread.
5. What are some alternative ways of allocating resources towards the management of the selected species?

[2] Note that for Scotch broom the analysis was confined to one or two specific utility corridors.

Based on our species selection rationale, at least one of the alternatives for diffuse knapweed and hawkweed includes biocontrol, at least one of the alternatives for hawkweed also includes delaying actions to a future time, and the analysis for Scotch broom is specific to utility corridors.

3.1.3 Ecological Model of Alternative Management Actions

We developed an ecological model to quantitatively account for the ecological and economic effects of the management actions described above. This model is based on the simple logistic growth model used in phase one of our analysis, but partitions the landscape into five alternate states that are tracked quantitatively over time. These states are: pre-invasion, undetected early infestation, undetected established infestation, detected established infestation, and suppressed by biocontrol. Transitions between states can occur as a result of natural processes or management actions. A graphical depiction of the state and transition model that we used to simulate alternative management strategies is shown in Figure 3.1.3.1. A full mathematical description of the model is given in Appendix 1.

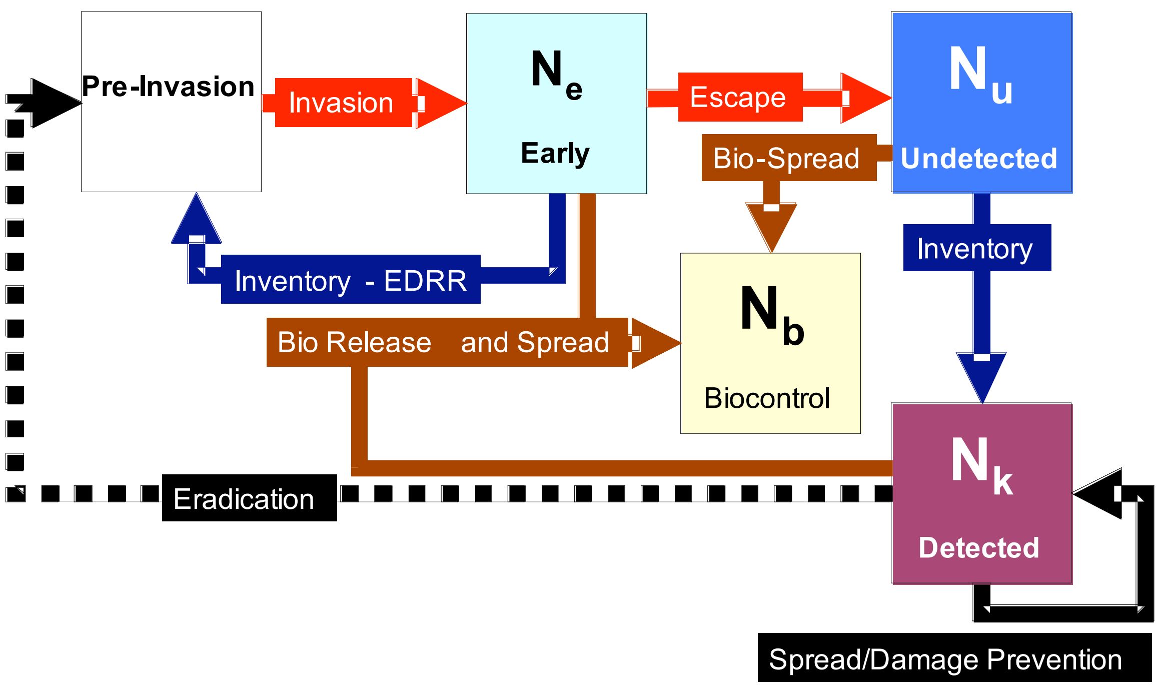


Figure 3.1.3.1: A graphical depiction of the states and transitions used in our model of alternative weed management strategies.

Parcels of land move from the pre-invasion state to an early infestation through the process of spread, which is largely driven by logistic growth from existing infestations on the landscape. The early infestation stage lasts only a year and represents the first generation of plants prior to them having produced seeds or propagules. This stage represents an opportunity for early detection and rapid response. If parcels of land in the early infestation stage are detected through inventory, they can be returned to the pre-invasion stage. However, after the passage of one year, the invasive plants on the parcel of land will set seed and/or produce propagules and move to an undetected established state.

Once in the undetected established state, parcels of land that are detected through inventory cannot be easily moved to the pre-invasion state. Instead, inventory moves parcels of land in the undetected established state to a detected state where conventional control and biocontrol actions can be taken.

Conventional control actions in the detected established state can result in three possible outcomes: a permanent elimination of damages, a temporary reduction in damages, and a failure to reduce damages. If sufficient resources are allocated, control methods are applied correctly over the necessary time period, and follow-up land management actions, such as seeding and adequate grazing management, are taken, conventional control actions can result in almost complete removal of the target plant and a return of the land to a desirable state. This is the highest level of success that can be accomplished.

A lower level of success is the temporary removal of the invasive plants through chemical or mechanical methods that results in a reduction in damages experienced for a short time period. However, due to a persistent seed and propagule source as well as open niches available to the invasive plants, damages return at a future point in time unless management is continuous. As a first approximation, our model assumes that the benefits of this level of success are experienced as a reduction in damages for a one-year period. Future models could take into account longer persistence times experienced due to such factors as residual pesticides, but this would require an age-structured model with a higher level of mathematical complexity.

Conventional control actions—if applied inadequately and/or at the wrong time or under the wrong environmental conditions—may also fail to have any desirable effect on the infested land, and damages from the invasive plants persist as if no treatment had been applied at all.

Biocontrol successfully applied to the detected established state results in a transition to a state in which the invasive plants are suppressed by the biocontrol agents. This state has the following benefits associated with it: the density of the invasive plants is permanently reduced by the biocontrol agents and, therefore, so is the economic damage experienced; and the potential for the infested lands to produce seeds and propagules is suppressed, and therefore spread of the plants across the landscape is permanently reduced. Note that biocontrol agents never cause the complete eradication of the target plant from a site, and therefore there will always be some low level of economic damage experienced in sites where biocontrol is present.

One additional benefit of biocontrol is that like the invasive plants themselves, the bioagent can spread to locations on the landscape where there is suitable habitat (i.e., other invasive plant infestations). We used a logistic model to simulate the spread of biocontrol agents from parcels of land where it has established to parcels of land where the target plants are present.

We developed the model in a Microsoft Excel spreadsheet (Electronic Appendix). The model requires the following ecological input parameters:

1. Starting Conditions:
    a. Date when the analysis begins
    b. Initial area (ha) in the early (first year) of invasion
    c. Initial area (ha) that is established but undetected
    d. Initial area (ha) that is established and detected
    e. Initial area (ha) that is suppressed by biocontrol agents.

2. Biological assumptions about the invasive plant
    a. The intrinsic growth rate determined as part of phase 1.
    b. The ecological limit (ha) determined as part of phase 1.
    c. The duration of the early infestation state (always one year).

3. Assumptions about inventory
    a. Inventory success – the probability that inventory conducted on a site where invasive plants are present will successfully detect them.
    b. Prior knowledge – the probability that the presence of invasive plants at a site will be known in the absence of any inventory action, as a result of reports from an educated public or the landowner.

4. Assumptions about conventional management actions
    a. Eradication success – the probability that management actions at a site where invasive plants are established will result in a permanent elimination of damages from the invasive plant.
    b. Spread prevention – the probability that sites treated with conventional management actions will be suppressed from contributing propagules or seeds for spread to the rest of the landscape during the year when management occurred.
    c. Treatment success – the sum of the eradication and spread prevention parameters.
    d. Damage reduction – for sites where eradication did not occur, the proportion by which economic damages caused by the invasive plant are reduced during the year of treatment.

5. Assumptions about biocontrol
    a. Established success – the probability that release of biocontrol agents at a site will result in their successful long-term establishment.
    b. Eradication success – the probability that, once established, biocontrol agents will lead to the eradication of target invasive plants at a site. We always assume this probability to be zero.
    c. Damage reduction – the proportion by which economic damages are reduced at a site once biocontrol agents have successfully established.
    d. Spread prevention – the proportion by which spread potential is reduced at sites where biocontrol has successfully established.

e. Spread rate – the intrinsic growth rate for biocontrol agents which affects their ability to spread to sites that are infested with invasive plants independent of further anthropogenic releases.
f. Effective coverage – the area (ha) that biocontrol agents can spread to from a single point release during the first year.

Table 3.1.3.1 indicates the values that we used for these parameters for each of the analyses conducted under phase 2.

Table 3.1.3.1. Ecological input parameters used for each of the analyses conducted as part of Phase 2. Analyses conducted include a retrospective analysis of the diffuse knapweed biocontrol program, a counterfactual analysis of diffuse knapweed chemical control, prospective analyses for hawkweed biocontrol and chemical control, Scotch broom mechanical control on the Island Highway, and Eurasian watermilfoil mechanical control.

	Units	Diffuse Knap-weed Biocont.	Diffuse Knap-weed Chem.	Hawk-weed Biocont.	Hawk-weed Chem.	Scotch Broom Mech.	Water-milfoil Mech.
A. START-UP CONDITIONS							
Start Year	year	1967	1967	2008	2008	2008	2008
Start Early	ha	500	500	8,000	8,000	10	50
Start Unknown	ha	5,000	5,000	30,000	30,000	10	100
Start Known	ha	6,000	6,000	40,000	40,000	60	850
Start Biocontrol	ha	0	0	0	0	0	0
B. BIOLOGICAL ASSUMPTIONS (INVASIVE)							
Intrinsic Growth Rate		0.15	0.15	0.14	0.14	0.08	0.19
Ecological Limit	'000 ha	1,100	1,100	8,722.5	8,722.5	0.150	0.0225
Initial Age Class	years	1	1	1	1	1	1
C. INVENTORY ASSUMPTIONS							
Inventory Success		0.85	0.85	0.85	0.85	0.85	0.85
Prior Knowledge		0.00	0.00	0.00	0.00	0.80	
D. TREATMENT ASSUMPTIONS - CONVENTIONAL (CHEMICAL OR MECHANICAL)							
Eradication Success		NA	0.90	NA	0.90	0.00	0.05
Spread Prevention		NA	0.05	NA	0.05	1.00	0.80
Treatment Success		NA	0.95	NA	0.95	1.00	0.85
Damage Reduction		NA	0.05	NA	0.05	1.00	0.79
E. TREATMENT ASSUMPTIONS - BIOCONTROL							
Established Success		0.75	NA	0.75	NA	NA	NA
Eradication Success		0.00	NA	0.00	NA	NA	NA
Damage Reduction		0.75	NA	0.75	NA	NA	NA
Spread Prevention		0.75	NA	0.75	NA	NA	NA
Spread Rate		0.20	NA	0.20	NA	NA	NA
Effective Coverage		0.05	NA	0.05	NA	NA	NA

3.1.4 Economic Assessment of Alternative Management Actions

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. For example, a full CBA would consider the distribution of the benefits and costs from invasive plant management strategies, in addition to the impacts on aggregate welfare. Instead, our analyses employ components of CBA in a more limited form that is more suited to meeting the needs at hand. This is admittedly a more limited task than a full CBA or project appraisal, but still requires coverage of some basic economic concepts.

a) With/Without Criteria

Economists use 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 plants 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 of biocontrol agents) and field operations costs (e.g., chemical spraying or mechanical pulling of invasive plants, and field-level release of biocontrol agents).

b) Discounting

With the long gestation period during which invasive species disperse and expand their range, the issue of time and discounting is an important one. 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, similarly 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 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 discount rate selected. Weighting a series of benefits or costs and summing these yields a *present value*. The challenge arises in selecting an appropriate discount rate. Discussion of the social discount rates selected for use in the current study is deferred to the next section.

c) Decision Criteria

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. An NPV greater than zero implies the action returns positive net economic benefits. Or, the present values of benefits and costs can be calculated and placed in a ratio, referred to as a benefit-cost ratio (BCR). A BCR greater than one indicates that benefits exceed costs and that the action is considered, in balance, favourable. Finally, some analysts consider an additional decision criterion, the internal rate of return (IRR). The IRR estimates the discount rate that yields an NPV equal to zero. This value is then compared to some reference rate, generally the social discount rate that was used for an NPV or BCR calculation. If it is higher than this rate, then the project is viewed as potentially successful, and if lower, the opposite is true. All three decision criteria were estimated for each invasive species analysis in this study, where possible.

Some additional considerations were required for our economic analyses of biocontrol programs. We undertook two such analyses, a retrospective, or *ex post*, analysis of diffuse knapweed biocontrol and a projected, or *ex ante*, analysis of a biocontrol program for hawkweed. The two analyses differ because of the historical versus future perspectives involved. Table 3.1.4.1 sets out some of these differences, which concern such issues as the type of data that can be used (actual versus projected or likely).

Table 3.1.4.1. Measuring the Impact of Biological Control Interventions Before and After Implementation: Factors to be Considered for Each Case.

Factor	Before (potential impact)	After (actual impact/benefit)
Area affected	- Measure area at risk of infestation - Predict ultimate distribution of pest - Measure rate of spread	- Measure known area of Infestation
Damage level	- Estimate damage/yield - Loss from crop yield data	- Yield loss assessment with and without biocontrol by field experimentation
Indirect damage	- Estimate likely side effects of pesticide applications: extent of displacement of native organisms	- Assess impact of non-target organisms from before/after data on distribution and abundance
Amenity	- Estimate likely effects on quality of life, human health, environment, social and cultural practices	- Measured environmental, social and cultural benefits following control
Cost of biocontrol	- Assess availability of natural enemies - Estimate cost of exploration, importation, quarantine, release, evaluation - Estimate probability of successful control	- Known cost of biocontrol implementation
Economic loss/benefits accruing	- Estimate benefits/costs to producers and consumers, elasticities - Undertake contingent valuation studies for non-market effects	- Measure actual benefits to producers and consumers, price elasticities

Source: G. Hill and D. Greathead (2000).

There is also an issue with our two biocontrol analyses of modeling a known outcome (diffuse knapweed) versus the unknown outcome of a future program (hawkweed). In the former situation, it may be perceived as self serving to analyze only successful biocontrol programs and ignore those that were not, when programming budgets had funded both outcomes. To address this bias at the broad invasive species management level, an analysis could be carried out where individual invasive species programs are treated as part of a larger program assessment with both successes and failures (Sinden et al. 2004). As that was not possible within the scope of the present study, we have instead assessed the diffuse knapweed program as a stand-alone investment with a known outcome. For the *ex ante* hawkweed case, uncertainty over eventual success can be addressed by developing a probability distribution of possible outcomes and then using this distribution to determine an "expected" value for the project benefits. Instead, we chose to discuss with the technical committee the likely number of biocontrol agents that would need to be screened and released in order to attain successful control. Since the hawkweed biocontrol program in its early stages, this seemed a reasonable and manageable approach, since some information is now available from initial screening.

Economic Parameter Values

Based on the model description in the previous sections and in the appendix, we developed a set of economic scenarios that covered a range of options for inventory and treatment budgets, as well as for program delay (also referred to as "escalation in costs") and the discount rate (Table 3.1.4.2). The budget scenarios were developed from baseline values to include a high and low option. The reasoning behind the starting baseline values is explained under each of the individual analyses below. For each budget parameter we also conducted a sensitivity analysis as outlined in table 3.1.4.2. For the hawkweed analyses (biocontrol and conventional treatment) we examined the impacts of delaying start-up in the respective program (which permits the invasive species to expand its coverage before treatment starts). We considered delays of 5, 10 and 20 years for these programs only. Since there is little consensus among economists regarding the correct value of the social discount rate, we considered the same three scenarios for all analyses; these were a baseline value of 4 percent, and alternatives of 2 and 6 percent.

Finally, the values for the per-hectare damages by each invasive species, which were estimated and presented in the previous section (Phase 1), are summarized in a table in that section and not reproduced here. All financial values in the economic analysis are presented in 2006 prices.

Table 3.1.4.2. Assumptions for Sensitivity Analyses.

	Units	Diffuse Knap -weed Biocont.	Diffuse Knap -weed Chem.	Hawk -weed Biocont.	Hawk -weed Chem.	Scotch Broom Mech.	Water-milfoil Mech.
A. Inventory Budget							
Low	$/year	Data[1]	50,000	50,000	50,000	0	100,000
Medium – Baseline	$/year	Data[1]	100,000	100,000	100,000	0	200,000
High	$/year	Data[1]	150,000	150,000	150,000	0	300,000
B. Treatment Budget – Conventional							
Low	$/year	NA	90,000	NA	90,000	10,000	350,000
Medium – Baseline	$/year	NA	180,000	NA	180,000	20,000	500,000
High	$/year	NA	270,000	NA	270,000	40,000	650,000
C. Treatment Budget – Biocontrol							
Low	$/year	Data[1]	NA	50,000	NA	NA	NA
Medium – Baseline	$/year	Data[1]	NA	100,000	NA	NA	NA
High	$/year	Data[1]	NA	150,000	NA	NA	NA
D. Escalation in Costs Assumptions							
Baseline	Years	0	0	0	0	0	0
Low	Years	NA	NA	5	5	NA	NA
Medium	Years	NA	NA	10	10	NA	NA
High	Years	NA	NA	20	20	NA	NA
E. Discount Rates							
Low	%	2	2	2	2	2	2
Medium – Baseline	%	4	4	4	4	4	4
High	%	6	6	6	6	6	6

[1]"Data" refers to the fact that the budgets used for the knapweed biocontrol retrospective analyses are based on past expenditures and scientist years invested by CABI and the Canadian federal (AAFC) and British Columbia provincial (MFR) governments (Harris 1979; Rose DeClerke-Floate, Susan Turner, and Val Miller, pers. comm.). These numbers are not presented in the table as budgets were not fixed but fluctuated over time. Our analysis took into account these fluctuations in expenditures (see electronic appendix).

3.2 Economic Analysis Results for Four Invasive Plants

3.2.1 Retrospective Economic Analysis of Biocontrol of Diffuse Knapweed

Assumptions

In this analysis we took an *ex post*, or historical, perspective to assess the economic viability of the provincial biocontrol program for diffuse knapweed. Since this analysis is retrospective, it differs from later analyses where we took an *ex ante* perspective and projected benefits and costs on the basis of the best available information. Instead, we have used data from government files on the knapweed program and from published papers reviewing various aspects. Since the knapweed program was funded by both within BC and external governments and agencies, we consider two basic formulations for our analysis that capture these differing "accounting stances." First, we analyze the full costs of the program from a global perspective to determine its overall merit. Second, we take into account only the costs incurred within the borders of BC, to assess the net returns from a BC-only perspective. The next sections describe the assumptions we made for our analysis.

a) Benefits

Benefits for the retrospective analysis were determined using the model described above. Since there has been little, if any, comprehensive assessment of the program at the provincial scale (versus localized studies), we needed to project the benefits using the best information available to us. The biophysical assumptions were detailed in Section 3.1.3 and reflect the start of the program in 1967, when we estimate that relatively little area had been invaded (about 11,000 ha) in comparison to the ecological limit of 1.1 million ha. Since the program developed at least six highly effective agents after screening and releasing 12 contenders, we set the success rate for establishment at 75 percent. Furthermore, discussions with various personnel familiar with the program led us to set the spread prevention parameter to 75 percent and assume that damages in areas experiencing biocontrol were reduced by an average of 75 percent. Finally, we assumed the spread rate (intrinsic growth rate) for diffuse knapweed was 15 percent, while that of the biocontrol agents was set at an average rate of 20 percent. We also conducted a sensitivity analysis of this parameter.

b) Investment Costs

Investment costs were treated separately in the ALL and BC ONLY analyses, and comprised screening, collection/rearing/shipping, and post-release evaluation. Since the investment program was targeted at both diffuse and spotted knapweed, we adjusted the figures using information on the proportion of releases aimed at diffuse knapweed over the period 1987 to 2008, to partition our costs (V. Miller, pers. comm.). The resulting average share of releases was 53 percent.

To determine the all-in (ALL) screening costs, we used the estimated number of 28 scientist-years of input for the investment program from Harris and Cranston (1979) and these included payments to CABI, Agriculture Canada and BC government, and related outlays. Harris (1979) split these inputs about equally between screening and related activities and post-release evaluation. We created a time series to allocate the screening portion of costs to individual years over the program's 25-year screening period using information from more recent biocontrol programs (R. De Clerck-Floate, pers. comm.). We then estimated the value of a scientist year in 2006 prices using Harris and Cranston's value from their study ($64,600 in 1976) and updated this to 2006 using the Consumer Price Index. Allowing for approximately a 6 percent real average increase over the screening period; this resulted in a revised 2006 value of $240,008 per scientist-year. For the BC ONLY analysis, we used actual figures for BC investment in the program provided by the technical committee (S. Turner, pers. comm.) but updated these to 2006 prices.

We were able to develop agent-specific costs for collection/rearing/shipping since we were provided the dates of introduction for all 12 agents released. This portion of investment costs was incurred by agencies funded from sources outside of BC, mostly Agriculture and Agri-Food

Canada. These activities were tied to the research program, and on the basis of discussions with the technical committee (R. De Clerck-Floate, pers. comm.) we estimated that three years were committed to these activities for each species at the level of $7500/year in 2006 prices and beginning the year after initial release of the agent; this was then adjusted for diffuse knapweed only (see above). This expense category was ignored in the BC ONLY analysis.

Post-release study inputs for two of the initial agents released were estimated by Harris (1979) at 2.5 scientist-years per agent. The exact timing of these studies was not known for each species, so we assumed that they took place over eight years for each species, beginning in the fourth year after release (just after collection/rearing/shipping terminates). For the BC ONLY analysis, we had actual data from the technical committee for post-release evaluation and used this information instead (S. Turner, pers. comm.). Since the latter data were deemed to be somewhat incomplete, we arbitrarily increased it by 20 percent to capture this aspect.

c) Field Operations Costs

Field operations costs refer to the propagation, collection, field release, and monitoring activities that result in the biocontrol agents being dispersed into the wider environment. One characteristic of the knapweed program was the slow initial release of agents, under the auspices of the research program itself. In discussions with technical committee members (S. Turner, pers. comm.), we estimated these initial releases at only 5 releases/yr for 1970-81 and 10/yr from 1982-86. During the post-1986 period and up to the present (2008), field releases were carried out by the provincial government on a much expanded scale. For this latter period we used historical data for the number of releases and for the collection, field release and monitoring costs, adjusted to 2006 prices (V. Miller, pers. comm.). Release and collection data for this latter era were partitioned according to target knapweed species (diffuse versus spotted) to determine the proportion of total insects (bioagents) collected and the proportion of the total number of releases targeted at each knapweed species. The data shows substantial variation over time with an average of 53 percent of releases directed at diffuse knapweed and 50% of insects collected for diffuse knapweed. We used the individual annual proportions attributable to diffuse knapweed as multipliers to adjust costs to a diffuse knapweed-only basis for each year. All field release costs were inflated to 2006 prices using the CPI.

The collection, field release, and monitoring information required additional data inputs before field operations costs were complete. The data described above was supplemented with the costs of biocontrol agent propagation for the period 1985 to 2004, which also was provided by the technical committee (S. Turner, pers. comm.). Finally, our release information was configured as "numbers of releases" but our model required this input in area terms (per hectare), so a further adjustment was needed. We assumed exponential growth from a small release area would reach a 20 m radius in five years (V. Miller, pers. comm.). With our baseline spread rate for the biocontrol agents of 20%, the initial area of release associated with this area occupied is about 0.05 ha (radius of 12.6m). As a result, we divided our number of releases by 20 to get the equivalent area in hectares.

Results

The results of our two analyses are suggestive of a program that has been quite successful. Individual results by accounting stance (ALL versus BC ONLY) are discussed below.

a) Including All Program Costs (ALL)

For the all-inclusive analysis of the diffuse knapweed biocontrol program, we estimated an NPV of $16.0 million (all prices 2006) with the baseline set of assumptions and a 4 percent discount rate (Table 3.2.1.1). The baseline Benefit-Cost Ratio (BCR) was 7.6, which is substantially greater than the breakeven value of 1.0. Varying the discount rate had a significant effect on the results. At a 2 percent discount rate, the NPV rises to $86 million and the BCR to 25.5, while the higher 6 percent discount rate results in an NPV of only $2.4 million and a BCR of 2.4. The Internal Rate of Return (IRR), which indicates at what discount rate the NPV falls to zero, was 7.5 percent for this scenario. For this scenario we also carried out a sensitivity analysis. As Figures 3.2.1.1 to 3.2.1.4 show, the results are quite sensitive to discount rate, ecological limit, and unit damages, but not to spread rate. The discount rate is clearly the most sensitive parameter (also see Table 3.2.1.1), while the sensitivity of the ecological limit and unit damage cost falls somewhere between the range exhibited by spread rate and discount rate.

b) Including Only BC Costs (BC ONLY)

When only BC costs are considered, the net economic returns from the program rise, but not excessively (Table 3.2.1.1). The NPV for the program increases to $17.4 million under baseline assumptions and the BCR jumps to 17.0. When the discount rate is raised to 6 percent, the NPV and BCR decrease predictably but the NPV remains positive and the BCR greater than 1.0. When the discount rate drops to 2 percent, the NPV increases to $88.4 million and the BCR increases to 49.8. A good measure of the comparative results for a BC-only accounting stance is the modestly high IRR of 9.9 percent. Although no further sensitivity analysis was carried out for this scenario, results could be expected to be similar to those of the global analysis presented above.

To place the economic analysis results in perspective, we reviewed several studies that compiled economic results for a number of biocontrol projects. In a review of cost-benefit analyses for 33 invasive plant biological control projects in Australia since 1903, McFadyen (2008) found an average BCR of 23.0 with a range of 0 to 312. When prickly pear cactus was excluded from the analyses, the average fell to 12.0, and the range dropped to 0 to 112. Overall, 17 of the programs were considered to be successful. Hill and Greathead (2000) found similar results in their review of 27 biocontrol projects from a range of countries and target species (insects and invasive plants). BCRs ranged from 0.99 to 7405. What is striking about the economics of biocontrol, as indicated by these reviews, is that when it is successful (and not all program are, certainly) it can be very successful.

These reviews also provide a context to assess our results. On this basis, the diffuse knapweed project appears to fall within the lower range of results found for more successful projects. The most likely reason for this more modest result is the large number of agents developed and released and the very low number of field releases in the early years, with benefits substantially delayed until well into the second decade of the project, the benefits are reduced by the discounting procedure. Not surprisingly, the discount rate appears to be the most sensitive parameter.

Table 3.2.1.1. Retrospective Analysis of the diffuse knapweed Biocontrol Program in BC – Total Costs (ALL) and BC Only Costs (BC) with Varying Discount Rate Scenarios.

Treatment Scenarios	Parameter Assumption	NPV ($ M)	BCR	IRR (%)
Biocontrol (ALL)-discount rate (%)				
Low	2	86.7	25.5	7.5
Medium-baseline	4	16.0	7.6	7.5
High	6	2.4	2.4	7.5
Biocontrol (BC)-discount rate (%)				
Low	2	88.4	49.8	9.9
Medium-baseline	4	17.4	17.0	9.9
High	6	3.4	6.1	9.9

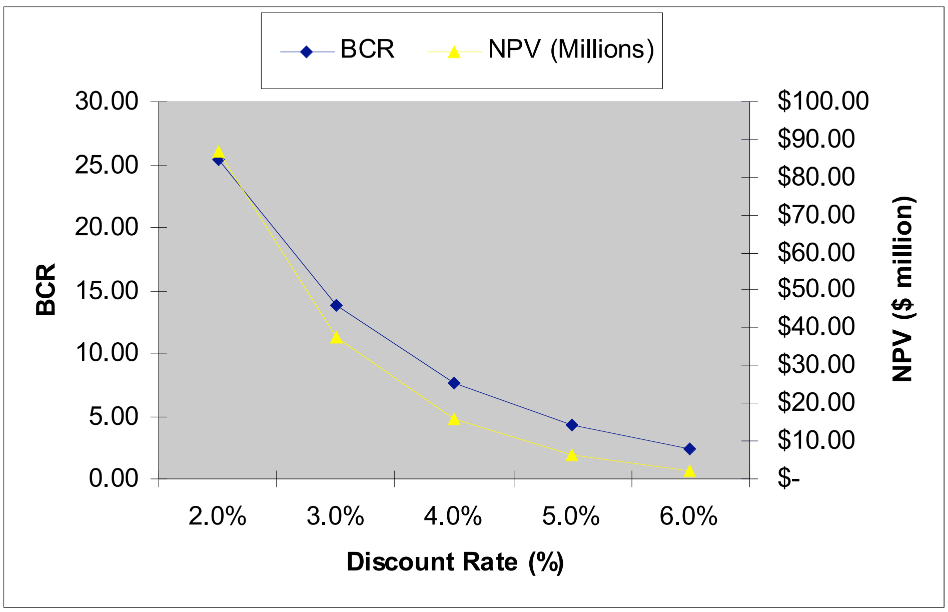


Figure 3.2.1.1. Biocontrol of diffuse knapweed: Sensitivity Aanalysis for Discount Rate and ALL Costs.

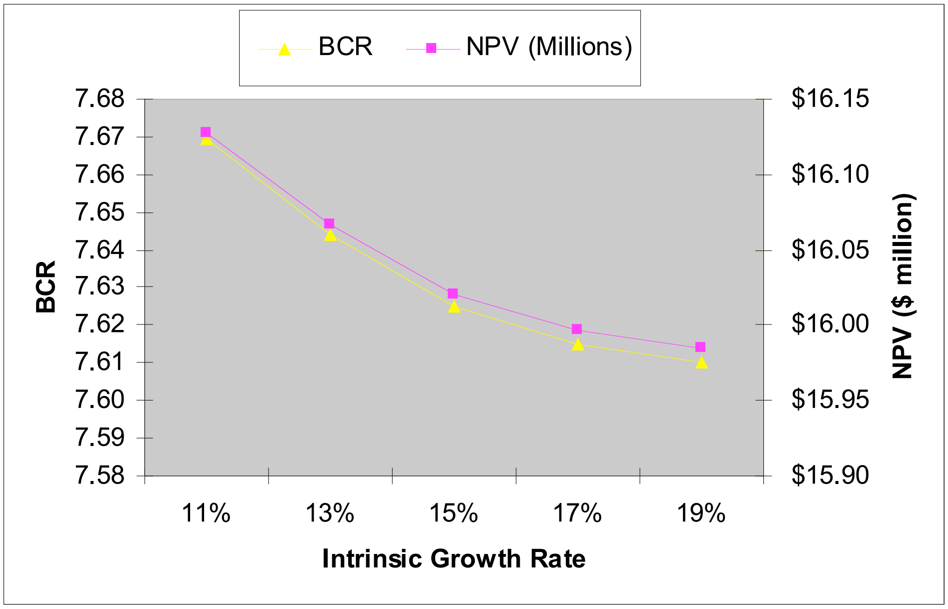


Figure 3.2.1.2. Biocontrol of diffuse knapweed: Sensitivity Analysis for Intrinsic Growth Rate and ALL Costs.

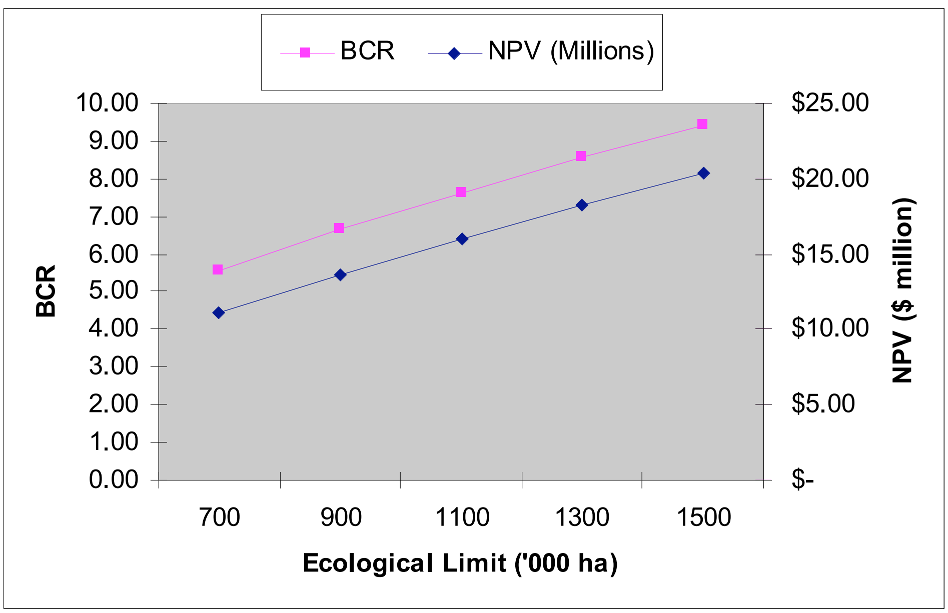


Figure 3.2.1.3. Biocontrol of diffuse knapweed: Sensitivity Analysis for Ecological Limit and ALL Costs.

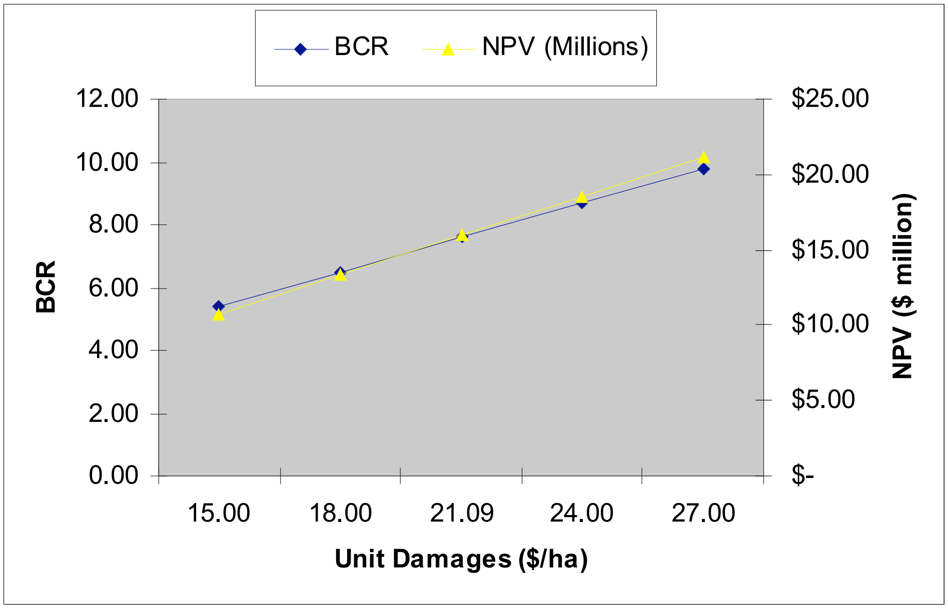


Figure 3.2.1.4. Biocontrol of diffuse knapweed: Sensitivity Analysis for Unit Damages and ALL Costs.

3.2.2 Economic Analysis of a Hypothetical Conventional Treatment Program for Diffuse Knapweed

Assumptions

Since this analysis was carried out as a counterfactual[3] for comparison to the historic biocontrol program, we used assumptions that were as close to the actual biocontrol program as possible. This involved using the same start-up conditions, inventory assumptions, and biological assumptions governing the behavior of the invasive plant (see Section 3.1.4 and 3.2.1). For the per-hectare costs of the spraying program, we adopted the assumptions developed for our hawkweed analysis, described in Section 3.2.4. The treatment program consisted of a three year treatment package that involved spraying and land management (comprising reseeding) that was assumed to be sufficient to provide 90 percent eradication success against the invasive plant. Such a high level of permanent eradication was predicated on the presence of substantial residual effects from spraying, combined with reseeding, so that the opportunity for re-invasion was minimal. The per-hectare cost ($ 2006) for the spraying program was calculated at $1137.50/ha over the entire three-year period and was treated as a single expense for model tractability. See Section 3.2.4 for further details.

As with the hawkweed conventional treatment program, we included environmental costs associated with spraying, as these are a well-documented and important cost element from a societal perspective (Pimentel et al. 1992). The calculated environmental cost, developed from data in Pretty and Waibel (2005) and from USDA information (USDA 2009), was $11.91/ha for our hypothetical spraying program. Again, the assumptions behind our estimates for the three-year treatment package are outlined in Section 3.2.4.

To establish appropriate treatment budget scenarios, we again used the approach developed for our hawkweed analysis. Thus, the approach was one of taking a representative investment cost for the equivalent biocontrol project (about $2 million) and amortizing this over the 100-year project life, and then adding this to the baseline $100,000/year treatment budget we assumed for biocontrol. This approach resulted in a baseline treatment budget of $180,000/year, along with alternative sensitivity scenarios. While the knapweed biocontrol program expenditures were historic and not based on a hypothetical budget, their value, as a constant annual expenditure amortized over the full 100 years, was contained within the range described by our budget assumptions for this analysis (490,000 to $270,000 per year) but is somewhat low in comparison to historical data for the spraying of knapweed in BC, which exceeded $1 million/year in the early 1980s (S. Turner, pers. comm.).

To ensure the spraying program was viable in its entirety for the full 100-year project life, we added a small inventory budget of $40,000/year. This amount was just sufficient to add adequate areas to the know area invaded to permit continued spraying for the entire period. Without this expenditure, the "known" area of knapweed would be exhausted within the project life, causing the spraying program to terminate in our model before Year 100.

[3] i.e., analyzing what might have happened if a hypothetical chemical control alternative to the real biocontrol program (presented in Section 3.2.1) had been undertaken instead

Results

Economic analysis of the baseline scenario for the chemical treatment of knapweed (Treatment budget: CDN $180,000/year) indicates a negative NPV. The BCR was positive for all treatment budget scenarios but it fluctuated between 0.78 and 1.05. A Negative Present Value for the treatment program indicates that the chemical treatment of diffuse knapweed is not economically viable when only the benefits we have captured are considered.

To understand the responsiveness of the treatment budget to the changes in economic and ecological parameters, we carried out a sensitivity analysis. A sensitivity analysis of the discount rate shows that the NPV reaches a peak and then begins to decline rapidly, as the effects of discounting take hold. The BCR also has a negative relationship with the discount rate (Figure 3.2.2.1). The sensitivity analysis for the intrinsic rate of growth indicates that this parameter has a negative relationship with both the NPV and the BCR (Figure 3.2.2.2), which is consistent with our expectation. As the invasive plant's potential to spread increases, the potential benefit from a chemical control program decreases. The ecological limit for diffuse knapweed has a strong positive relationship with both the NPV and the BCR (Figure 3.2.2.3). Not surprisingly, this indicates that the treatment program for diffuse knapweed generates more benefits when the ecological limit expands. As expected, the sensitivity analysis of the unit damage per hectare demonstrates that this parameter also has a positive relationship with both the BCR and the NPV (Figure 3.2.2.4).

Table 3.2.2.1. Economic Analysis of Treatment of diffuse knapweed: Conventional.

Treatment Scenarios	Parameter Assumption	NPV ($ M)	BCR	IRR (%)
Conventional –treatment budget ($/yr)				
Low	90000	-0.7	0.8	NA
Medium-base line	180000	-0.4	0.9	NA
High	270000	0.4	1.1	4.6
Conventional – discount rate (%)				
Low	2	-0.2	1.0	NA
Medium-base line	4	-0.4	0.9	NA
High	6	-0.8	0.8	NA

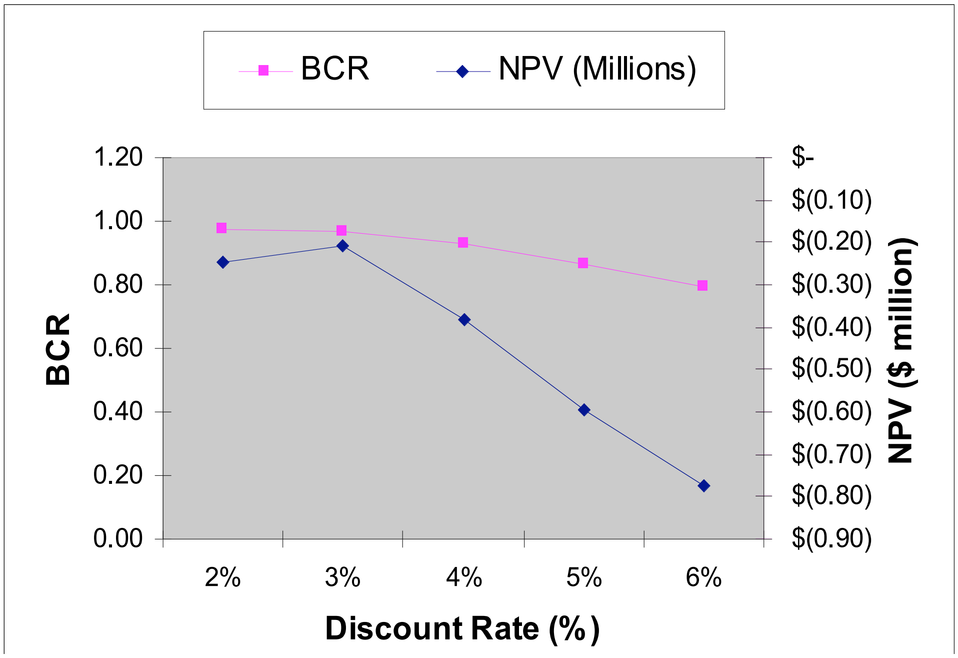


Figure 3.2.2.1. Knapweed Conventional Treatment (with land management): Sensitivity Analysis for Discount Rate.

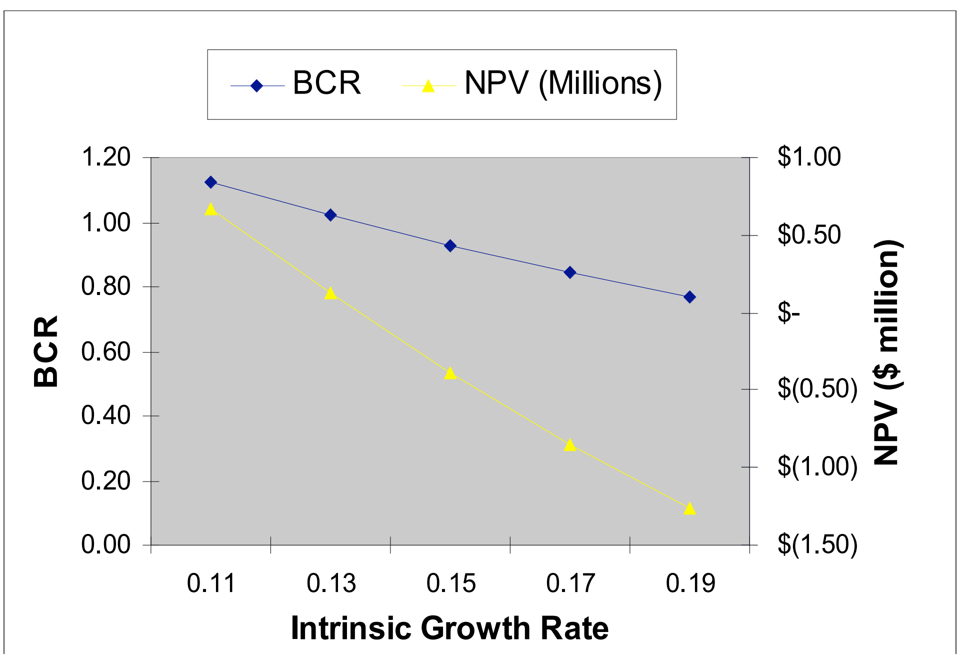


Figure 3.2.2.2. Knapweed Conventional Treatment (with land management): Sensitivity Analysis for Intrinsic Growth Rate.

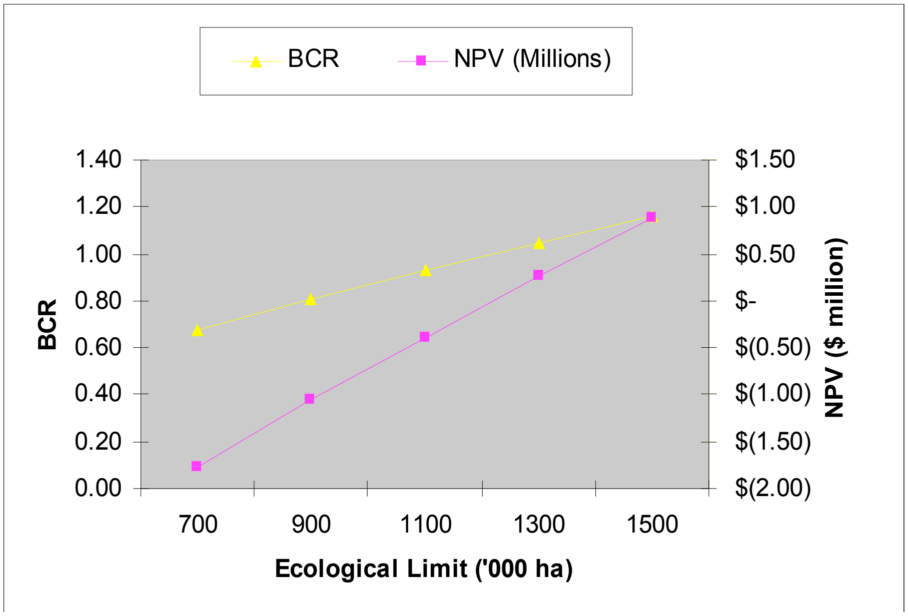


Figure 3.2.2.3. Knapweed Conventional Treatment (with land management): Sensitivity Analysis for Ecological Limit.

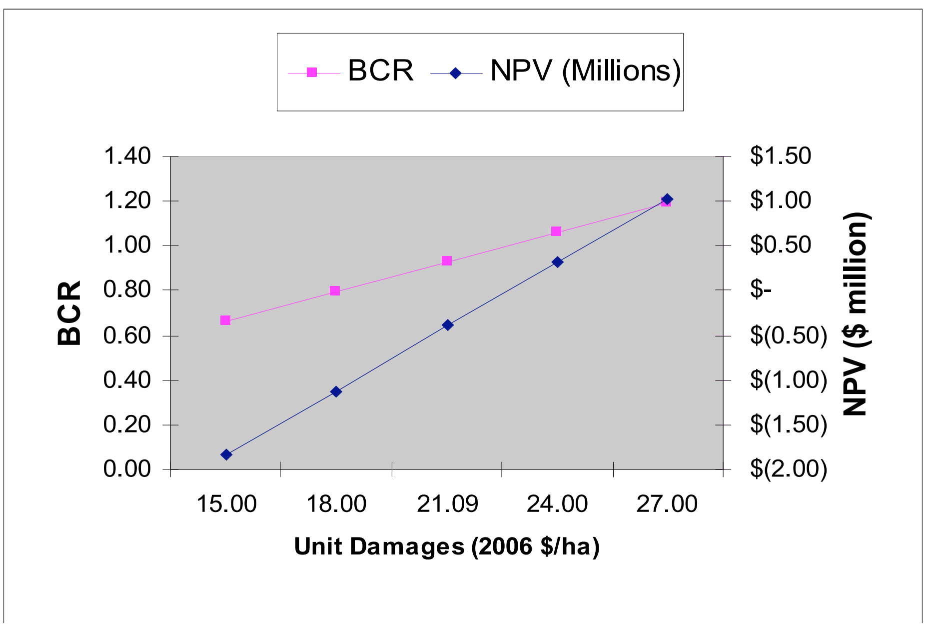


Figure 3.2.2.4. Knapweed Conventional Treatment (with land management): Sensitivity Analysis for Damage Cost from Invasive.

3.2.3 Hawkweed – Biocontrol

Assumptions

For the hawkweed biocontrol analysis, we used a similar approach to that described above for the diffuse knapweed program, recognizing that the present analysis is *ex ante* and, therefore, a projection. For example, the treatment assumptions (eradication parameter, etc.) are identical to those used in the knapweed analysis. However, conditions do differ now – screening can be more prolonged due to greater environmental restrictions and fewer agents are likely to be approved for release. Also, the start-up conditions differ in that the initial area invaded and ecological limit for hawkweed are much larger than for knapweed. Since the hawkweed biocontrol analysis makes use of the full capabilities of our model (no historical data) we were required to set the inventory and treatment budgets, as well as the total screening cost and duration for the screening period.

For this *ex ante* analysis of a new biocontrol project, we discussed the likely number of agents to be released with the technical committee and agreed on five (L. Wilson, pers. comm.). Then the screening costs were developed using information from the well established hound's-tongue and Dalmatian toadflax programs (R. De Clerck, pers. comm.). Since the former has been unusually successful in discovering an effective agent early on, we positioned our hypothetical hawkweed project closer to the toadflax program in terms of screening costs, resulting in values of $2.5 million (2006 prices) for the scientist inputs and a duration of 15 years. Recurrent cost for program administration and management were developed from current records and charged at 0.55 scientist-years annually; to accompany this we used a current estimate for the cost of a scientist year of $350,000.

Field releases were governed in the analysis by the budget set for this purpose. We assumed an annual expenditure of $100,00 for releases as our baseline. To determine the area that these releases would initially affect in per hectare terms, we used the assumptions developed for the knapweed analysis. The average cost of a release was estimated at $295/release based on collection and field release figures for knapweed. We also included an inventory budget of $100,000 in the baseline assumptions, but varied this together with treatment budget to establish alternative scenarios. Finally, to assess the importance of timing in the initiation of a hawkweed biocontrol program, we considered delays in its start of 5, 10, and 20 years to determine the impact on economic returns.

Results

Economic analysis of the baseline scenario (inventory budget of $100,000 and release budget of $100,000) indicates that the NPV for biocontrol of hawkweed is positive (Table 3.2.3.1). Indeed, the NPV remains positive and the BCR is very attractive from an economic point of view for all treatment budget scenarios. These results demonstrate that the biocontrol of hawkweed is economically viable and generates significant benefits to society. To understand the responsiveness of the treatment budget to changes in economic and ecological parameters, we carried out a sensitivity analysis. Sensitivity analysis of the discount rate shows that there is a negative relationship between the discount rate and both the NPV and the BCR (Figure 3.2.3.1).

This occurs because the proportionate change in the present value of the cost of the treatment program is greater than the proportionate change in the present value of the benefits of the treatment program. Sensitivity analysis for hawkweed biocontrol also shows that the intrinsic rate of growth initially has a positive relationship with both the NPV and the BCR. However, this reaches a peak at a 12 percent spread rate and then flattens or slightly declines at higher values (Figure 3.2.3.2). As the spread rate increases initially, the potential for benefits also increases but then eventually declines as the ability of the weed to spread faster than a biocontrol program can contain it increases. The ecological limit for hawkweed has a positive and relatively strong relationship with both the NPV and the BCR (Figure 3.2.3.3). The greater the area that can be potentially affected by hawkweed, the more society stands to gain from a biological control program. As expected, a sensitivity analysis of the unit damage per hectare demonstrates that this parameter has a positive relationship with both the BCR and the NPV (Figure 3.2.3.4). This observation implies that when the unit damages increase, the present value of benefits of avoiding those damages also increases.

Table 3.2.3.1. Economic Analysis of Treatment of hawkweed: Biocontrol.

Treatment Scenarios	Parameter Assumption	NPV ($ M)	BCR	IRR (%)
Biocontrol -inventory budget				
Low	50000	1687.0	214.1	14.6
Medium-base line	100000	1686.1	185.5	14.4
High	150000	1685.3	163.6	14.1
Biocontrol -release budget				
Low	50000	1373.9	164.2	13.3
Medium-base line	100000	1686.1	185.5	14.4
High	150000	1893.1	193.0	15.0
Biocontrol – escalation in costs				
Base line	0	1681.1	185.5	14.4
Low	5	1284.3	172.2	14.2
Medium	10	957.8	156.6	14.1
High	20	475.4	115.9	13.9
Biocontrol-discount rate				
Low	2	7578.5	600.5	14.4
Medium-baseline	4	1681.1	185.5	14.4
High	6	409.8	57.8	14.4

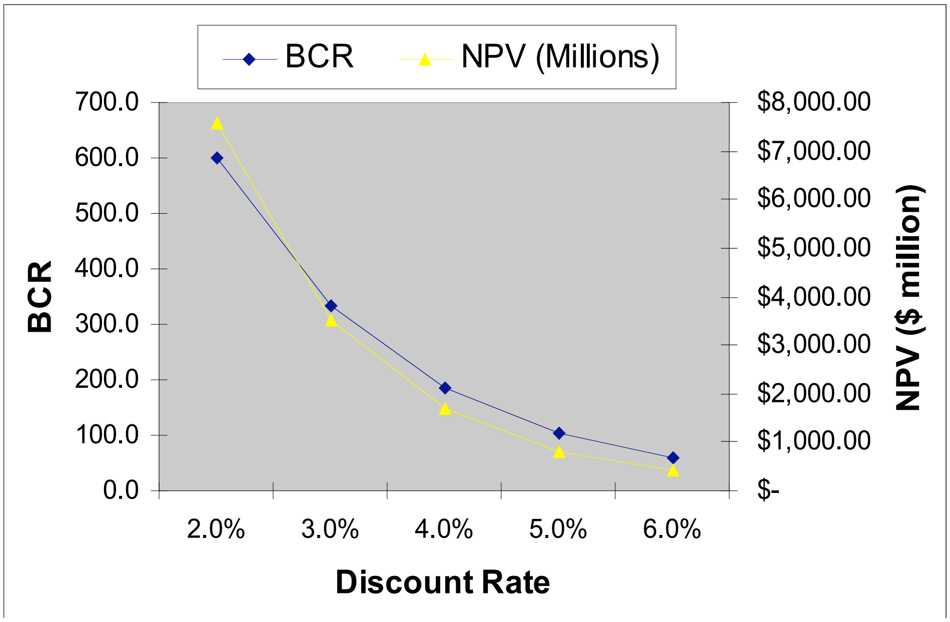


Figure 3.2.3.1. Hawkweed Biocontrol: Sensitivity Analysis for Discount Rate.

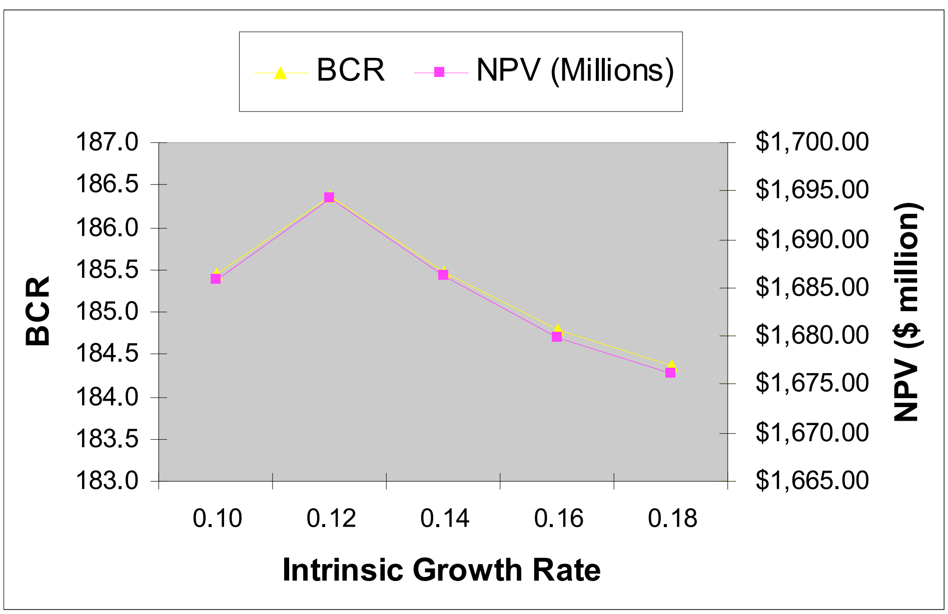


Figure 3.2.3.2. Hawkweed Biocontrol: Sensitivity Analysis for Intrinsic Rate of Growth.

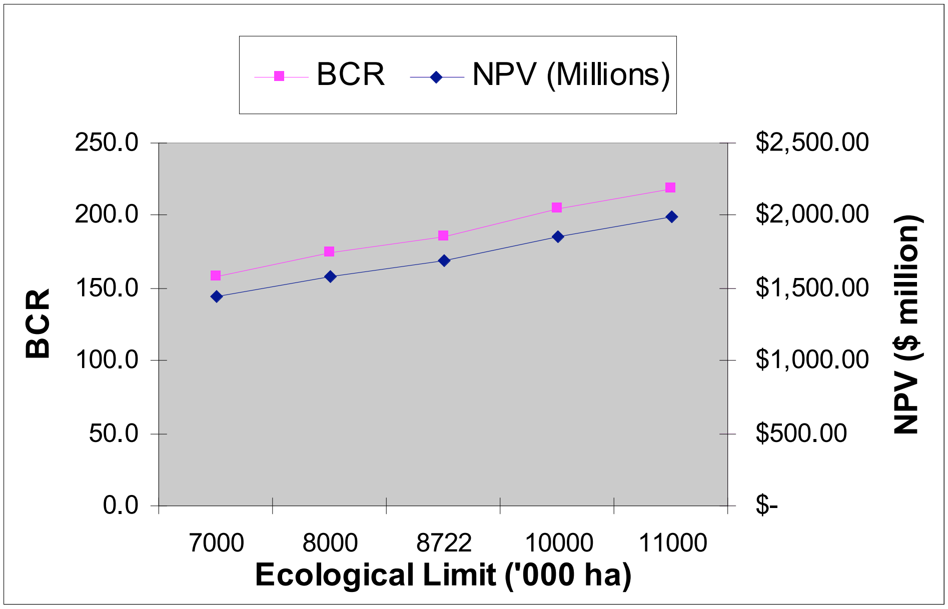


Figure 3.2.3.3. Hawkweed Biocontrol: Sensitivity Analysis for Ecological Limit.

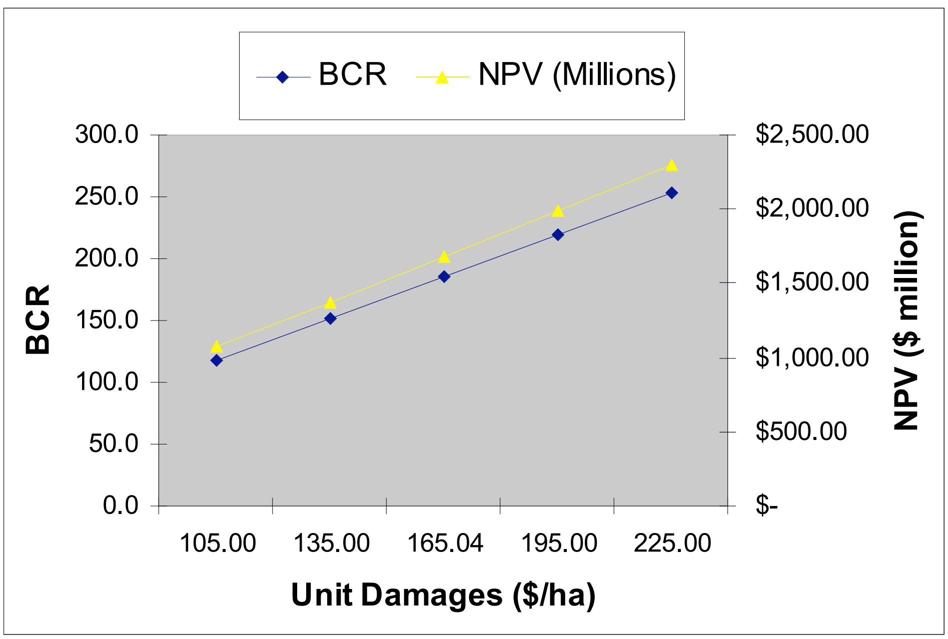


Figure 3.2.3.4. Hawkweed Biocontrol: Sensitivity Analysis for Unit Damages.

3.2.4 Economic Analysis of a Future Conventional Treatment Program for Hawkweed

Assumptions

As noted for the knapweed conventional treatment analysis, we established biological and management parameters for the conventional treatment of hawkweed that were identical to those for the equivalent biocontrol analysis assessed in the preceding section, i.e. start-up, biological (invasive), and inventory assumptions. In particular, we assumed a relatively low initial area invaded compared to the ecological limit. We also considered two treatment approaches, intended to bound the range of options that might be considered, thereby giving a better indication of potential results from chemical spraying. These two options are described below.

a) Treatment Option A – Land Management

In this option we allow for a three-year spraying program, followed by land management in the form of reseeding. Discussions with technical committee personnel (L. Wilson, pers. comm.) led to the following assumption governing re-spraying: first year (100% of area), second year (50%), and third year (25%). Costs were derived from best estimates of the contract cost of spraying (about $300/ha), which was then doubled to include administration and management costs/overheads. A $50/ha charge for reseeding was added to each year's spraying cost, yielding a three-year cost of $1137.50. To capture the intent of this option, the eradication success parameter was set high, at 90 percent.

b) Treatment Option B – No Land Management

For this option, the intent was to analyze an annual re-spraying program with very limited permanent reduction/eradication and no reseeding. Thus, there would be immediate benefits in the current year in which spraying occurs, and the costs would be reduced on a per-hectare treated basis, but the benefits are short-lived. This option mimics a reactive approach where the limited spraying budget is expended responding to current threats, with little effort aimed at longer term control as occurs with Treatment Option A. The annual spraying cost per treated hectare is $600 (see above) and there is no investment in land management (reseeding). Commensurately, the eradication parameter was set low (0.05) and the temporary damage reduction parameter was set high (0.90).

Since the collateral damages or external costs of pesticide spraying are well known (Pimentel et al. 1992), we made a crude estimate of these to add to the private spraying costs documented above. Since data on pesticide damages specific to rangelands were not available, we developed an average damage cost per hectare for planted crop area in the US. We began with an estimate of total damages on arable land from Pretty and Waibel (2005) of $1492 million in 2003 USD. We then converted this to 2006 Canadian prices using the International Financial Statistics database from the IMF. We subsequently used USDA data to estimate the total number of treatments per hectare, or "hectare-treatments" (www.ers.usda.gov/Data/ARMS/app/Crop.aspx), using crop-specific areas, proportion treated, and the number of annual treatments per hectare for each crop. This yielded an estimate of 323,887 hectare-treatments, or 2.64 hectare-treatments per hectare actually planted. Dividing this total into the value of damages yielded a cost of $6.81/ha-treatment for a single treatment (Treatment Option B) and $11.91/ha-treatment for the three year treatment option described in Treatment Option A.

Finally, for the treatment budget scenarios we analyzed, we developed an adjusted amount based on the assumption that the annual spraying budget should be approximately equivalent to the full costs of a biocontrol program for hawkweed, as described in the previous section. Thus, we took a baseline budget of $100,000/year and added to it the annualized cost of a $2 million investment program (based on the historic costs for the knapweed biocontrol program), using the 4 percent baseline discount rate for amortization. This adjustment resulted in a baseline spraying budget of $180,000/year. In addition, we considered a set of additional alternative scenarios involving a delay in the start-up of the three-year spraying program (Treatment Option A), as described above for the biocontrol program.

Results

Economic analysis of the baseline scenario with land management (treatment budget: $180,000/year and inventory budget: $100,000/year) indicates that both the NPV and the BCR for the treatment of hawkweed with land management is positive (Table 3.2.4.1). Indeed, the NPV and the BCR remains positive for all treatment budget scenarios with land management. These results imply that the treatment of hawkweed with land management is an economically viable option for controlling hawkweed. Note, however, that the benefits gained from this approach are much lower than those potentially gained from a successful biocontrol program. Although both the NPV and the BCR remain positive for all discount rate scenarios with land management, the net benefits of the treatment programs on hawkweed reduces with an increase in the discount rate.

Economic analysis of the baseline scenario without land management shows that both the NPV and the BCR for the treatment of hawkweed without land management is positive as well (Table 3.2.4.1). Nevertheless, both the NPV and the BCR values of the treatment without land management are lower compared with the NPV and the BCR values of Treatment Option A. This result indicates that the treatment of hawkweed with land management has greater potential benefits than the treatment of hawkweed without land management.

The baseline scenario for delay in start-up (escalation cost scenario) shows that both the NPV and the BCR record positive values for the land management scenario (Treatment Option A). All other scenarios, with respect to escalation cost, demonstrate similar results, but the net benefits from the hawkweed chemical treatment program decline with an increased delay in start-up.

To further examine the responsiveness of the treatment budget to the changes in economic and ecological parameters, we carried out a sensitivity analysis. Sensitivity analysis of the discount rate shows that there is a negative relationship between the discount rate and NPV as well as the BCR. According to the sensitivity analysis, there is a negative relationship between the intrinsic rate of growth and both the NPV and the BCR (Figure 3.2.4.2). This occurs because the present value of benefits of the treatment with land management declines if there is an increase in intrinsic rate of growth. The sensitivity analysis for ecological limit demonstrates that this parameter has a positive relationship with both the NPV and the BCR. This is due to the increase in the benefits of the present value of treatment with land management as a result of expansion of ecological limit (Figure 3.2.4.3). As expected, sensitivity analysis of the unit damage per hectare demonstrates that this parameter has a positive relationship with both the BCR and the NPV (Figure 3.2.4.4).

Overall, despite positive net returns to society, the conventional treatment approaches do not bring hawkweed under control permanently and, instead, only slow its progress towards eventually occupying its entire potential ecological range by about Year 75.

Table 3.2.4.1. Economic Analysis of Treatment of hawkweed: With Land Management (LM) and Without Land Management (NLM).

Treatment Scenarios	Parameter Assumption	NPV ($ M)	BCR	IRR (%)
Conventional (LM)-inventory budget				
Low	50000	39.2	7.9	25.1
Medium-base line	100000	38.4	6.6	22.1
High	150000	37.6	5.6	20.0
Conventional (LM) –treatment budget				
Low	50000	18.3	4.9	18.3
Medium-base line	100000	38.4	6.6	22.1
High	150000	58.7	7.4	24.0
Conventional (LM)-escalation cost				
Base line	0	38.4	6.6	22.1
Low	5	18.1	4.2	20.9
Medium	10	8.1	2.7	19.0
High	20	0.9	1.3	11.9
Conventional (LM)-discount rate				
Low	2	75.3	7.2	NA
Medium-baseline	4	38.4	6.6	NA
High	6	20.4	5.4	NA
Conventional (NLM) - inventory budget				
Low	50000	0.9	1.2	4.2
Medium-base line	100000	0.1	1.0	4.2
High	150000	-0.7	0.9	NA
Conventional (NLM) – treatment budget				
Low	50000	-0.8	0.8	NA
Medium-base line	100000	0.1	1.0	4.2
High	150000	0.9	1.1	5.3
Conventional (NLM) -discount rate				
Low	2	1.2	1.1	NA
Medium-baseline	4	0.1	1.0	NA
High	6	-0.7	1.9	NA

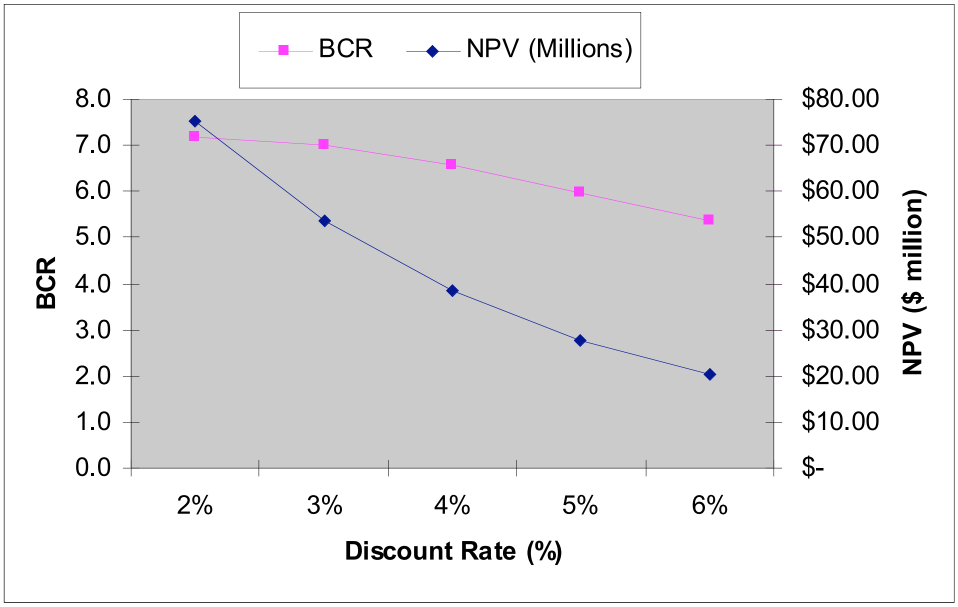


Figure 3.2.4.1. Hawkweed Conventional Treatment (with land management): Sensitivity Analysis for Discount Rate.

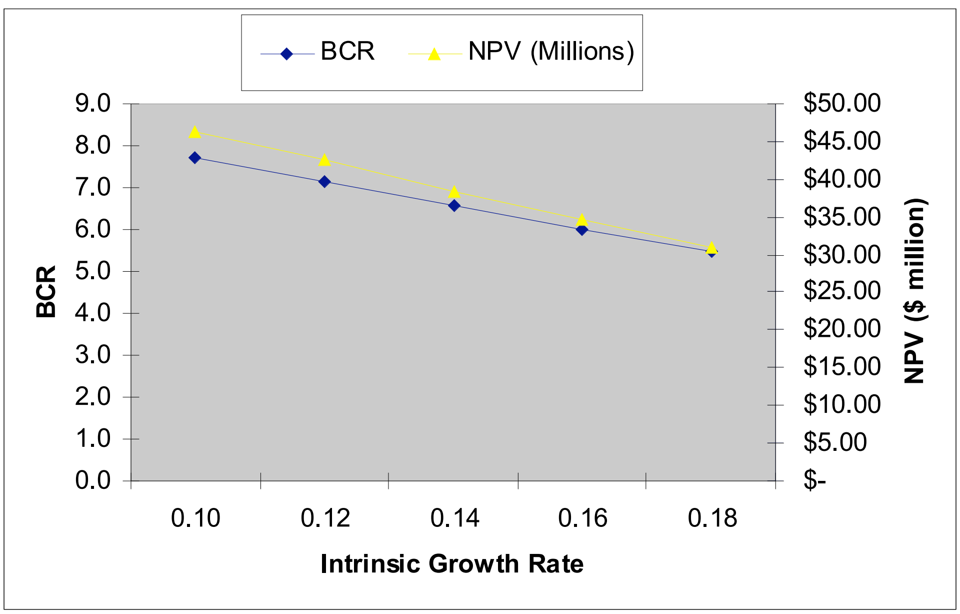


Figure 3.2.4.2. Hawkweed Conventional Treatment (with land management): Sensitivity Analysis for Intrinsic Growth Rate.

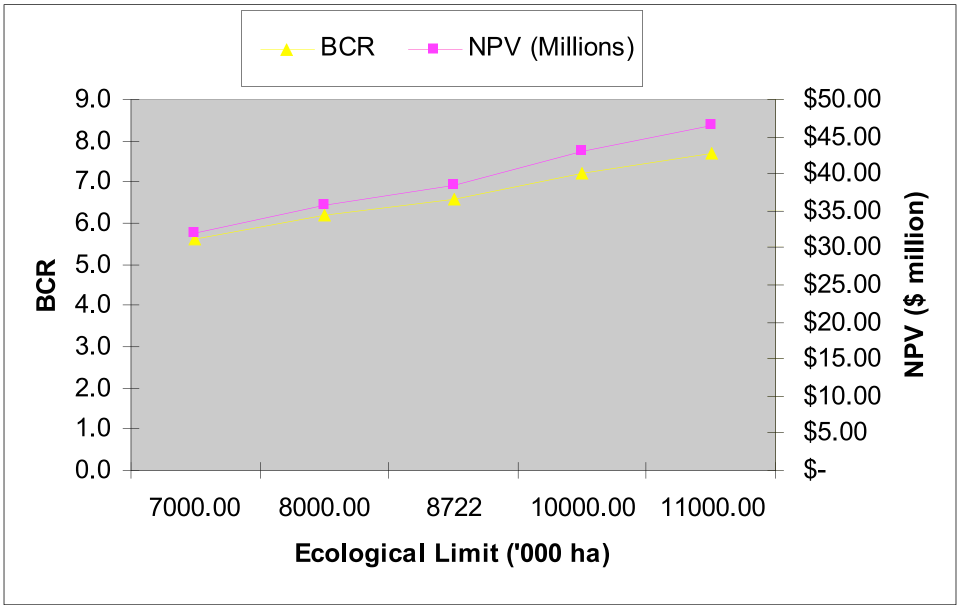


Figure 3.2.4.3. Hawkweed Conventional Treatment (with land management): Sensitivity Analysis for Ecological Limit.

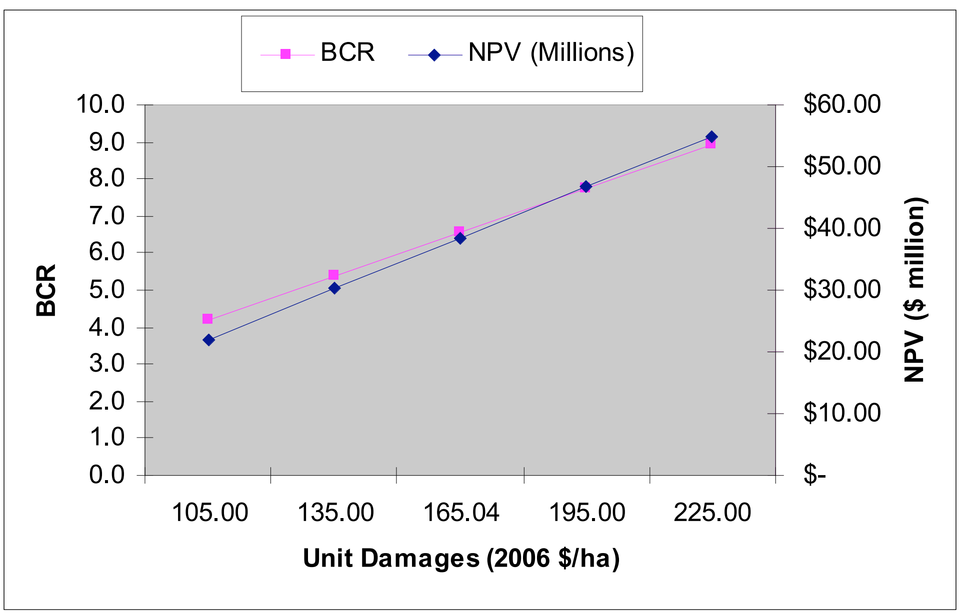


Figure 3.2.4.4. Hawkweed Conventional Treatment (with land management): Sensitivity Analysis for Damage Cost from Invasive.

3.2.5 Scotch Broom

Assumptions

This analysis differs from the others we undertook because it was concerned with a localized problem: invasion by Scotch broom along a representative highway corridor on Vancouver Island. Scotch broom treatment includes cutting mature plants to within 5-10 cm of the ground and chipping the plant waste, then spreading it on the site as mulch; immature plants are hand pulled and chipped as with mature plants (B. Brown, pers. comm.). Our biological assumptions reflect this case study in that the proportion of the known invaded area is quite large and the ecological limit is relatively small (150 ha). Treatment is carried out using inmate labour, which is substantially less expensive than contract crews, and so this cost is adopted for our analysis. The following assumptions were made:

- Based on the area data for 2008, it takes about 308 person-hours to remove one hectare of Scotch broom.
- The Corrections Program spends approximately $41.54 in wages per hour for an 8-person inmate crew and one Corrections Officer, totalling 9 person hours.

These assumptions result in a cost of $1,421/ha of Scotch broom removed, including time spent on site and driving to and from the site. In contrast, a contract crew costs approximately $192/hour for wages (8 labourers and 1 supervisor), resulting in a cost of $6,571 per ha removed.

We assumed that the average Scotch broom infestation was 4m wide, on either side of the road. Each treated site was assumed to require re-treatment annually and, therefore, our treatment assumptions reflect this situation (e.g., zero eradication success but high damage reduction). The primary reason for re-treatment is not re-growth of cut plants but removal of new plants, since depleting the seedbank requires a long-term commitment: each plant produces up to 32,000 seeds annually and each seed can remain viable for up to 70 years. On newly established sites with no seedbank in the soil, retreatment may only be required for a few years if the plants are removed before they set seed; this is captured in our "Early" cohort portion of the treated area (see the model description).

Finally, we considered only a limited baseline budget scenario, given the small-scale treatment program. Based on discussions with the Technical Committee (A. Planiden, pers. comm.), we set the baseline budget at $20,000/year and considered two options of $10,000 and $40,000/year.

Results

Economic analysis of the baseline scenario (treatment budget: $20,000/year) indicates that the NPV for mechanical treatment of Scotch broom is negative (Table 3.2.5.1). Indeed, the NPV remains negative and the BCR never exceeds 0.20 for all treatment budget scenarios. These results imply that the treatment of Scotch broom is not economically viable at a local site level when only the benefits we have captured are considered. To understand the responsiveness of the treatment budget to the changes in economic and ecological parameters, we carried out a sensitivity analysis. Sensitivity analysis of the discount rate shows that there is a positive relationship between NPV and the discount rate, but this is reversed for the BCR and the discount rate (Figure 3.2.5.1). This occurs because the proportionate change in the present value of the cost of the treatment program is greater than the proportionate change in the present value of the benefits of the treatment program. Sensitivity analysis for Scotch broom also shows that the intrinsic rate of growth initially has a positive relationship with both the NPV and the BCR. However, this reaches a peak at the baseline estimate for the intrinsic growth rate and then flattens or slightly declines at higher values (Figure 3.2.5.2). The ecological limit for Scotch broom has a positive relationship with both the NPV and the BCR. Although the NPV initially declines as the ecological limit increases, it quickly reverses and then rises with the increase in ecological limit (Figure 3.2.5.3). As expected, sensitivity analysis of the unit damage per hectare demonstrates that this parameter has a positive relationship with both the BCR and the NPV (Figure 3.2.5.4).

Although we did not analyze the treatment option using full-cost contract crews, it is clear that if the unit cost of mechanical removal were to rise by 4 to 5 times, the negative results we found would only worsen. For a discussion of alternative ways of approaching this analysis that incorporate the elements missing here, and which provide quite different results, see the discussion of Scotch broom in Section 4.

Table 3.2.5.1. Economic Analysis of Treatment of Scotch broom – Mechanical.

Treatment Scenarios	Parameter Assumption	NPV ($ M)	BCR	IRR (%)
Conventional – treatment budget ($/yr)				
Low	10000	-0.2	0.1	NA
Medium-base line	20000	-0.5	0.1	NA
High	40000	-0.9	0.1	NA
Conventional – discount rate (%)				
Low	2%	-0.8	0.1	NA
Medium-base line	4%	-0.5	0.1	NA
High	6%	-0.3	0.1	NA

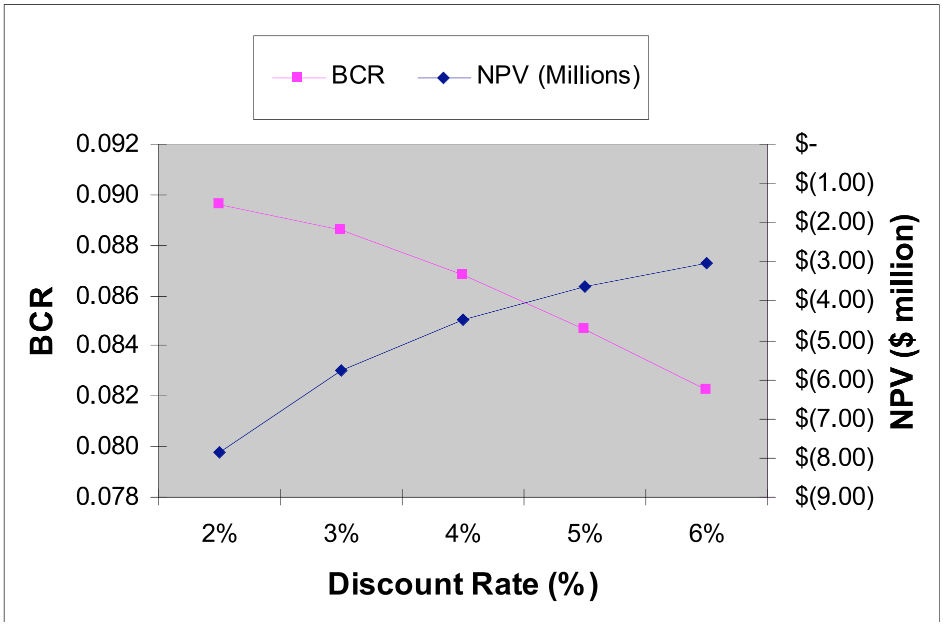


Figure 3.2.5.1. Treatment of Scotch broom (Mechanical): Sensitivity Analysis for Discount Rate.

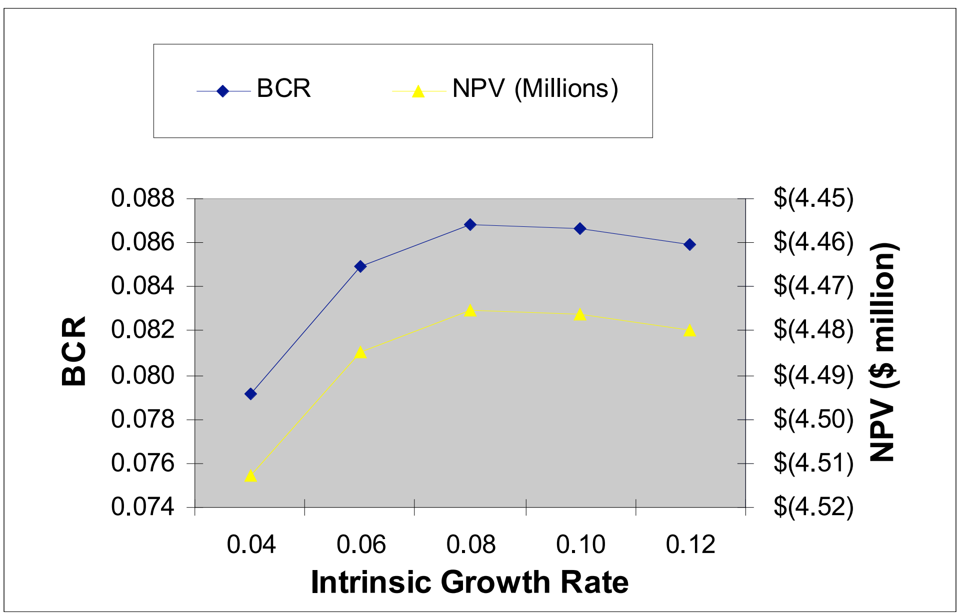


Figure 3.2.5.2. Treatment of Scotch broom (Mechanical): Sensitivity Analysis for Intrinsic Rate of Growth.

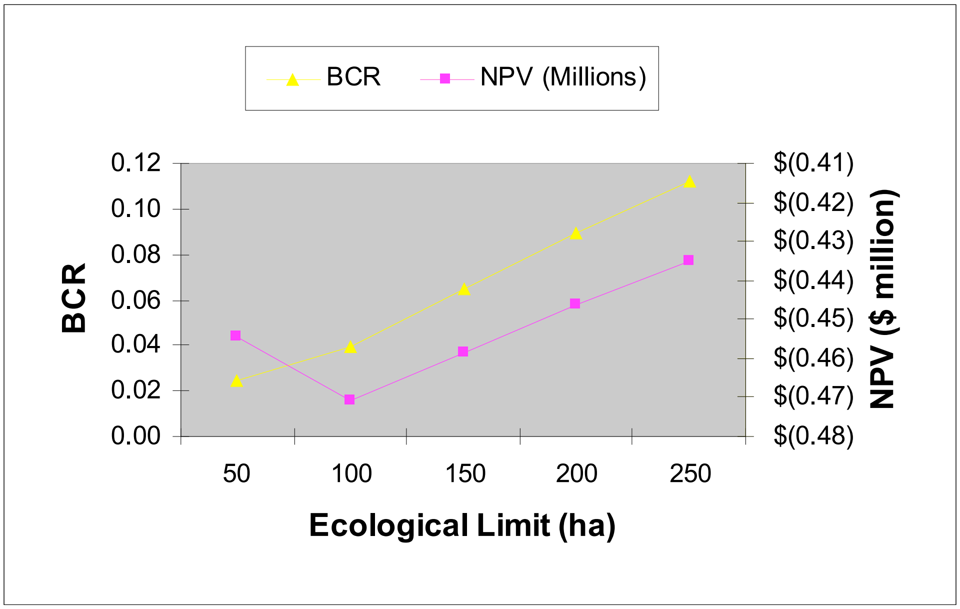


Figure 3.2.5.3. Treatment of Scotch broom (Mechanical): Sensitivity Analysis for Ecological Limit.

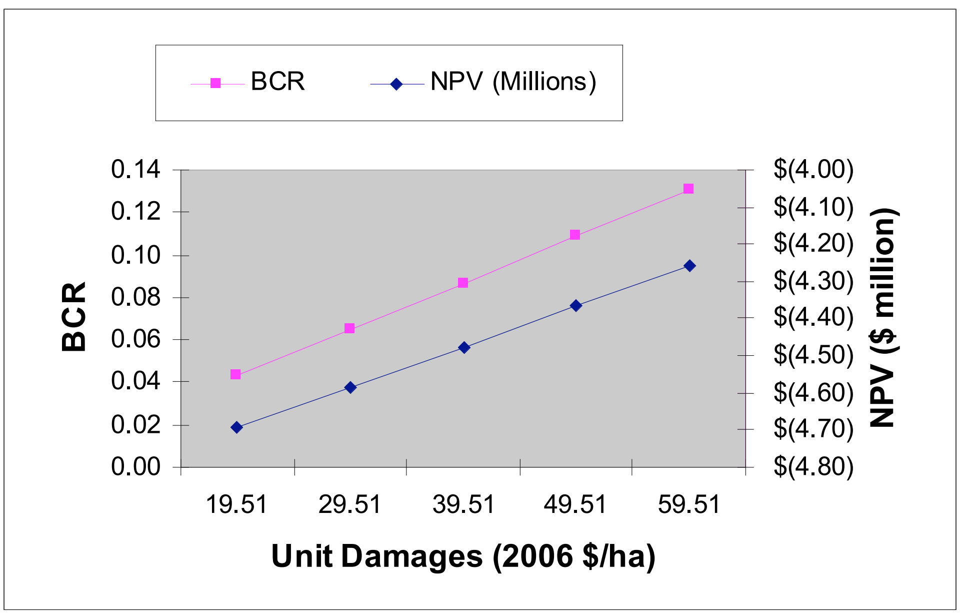


Figure 3.2.5.4. Treatment of Scotch broom (Mechanical): Sensitivity Analysis for Unit Damages.

3.2.6 Eurasian Watermilfoil

Assumptions

At present, control of watermilfoil is concentrated in the Okanagan Basin and has been carried out there for several decades. We established our analysis as a hypothetical mechanical treatment program at the provincial level, but used the parameters from the Okanagan treatment program as a basis. Since watermilfoil may be spread from one lake to another, this approach makes intuitive sense. Nonetheless, given the very high recreational use values associated with the Okanagan lakes, we adopted a relatively limited provincial treatment program for our analysis that was more or less consistent with the amounts expended in the Okanagan Basin at present. The area invaded by Eurasian watermilfoil in the Okanagan lakes is thought to be close to its ecological limit for that area of 1100 ha, and perhaps even to have suffered some dieback (A. Warwick Sear and I. Horner, pers. comm.), so we based our start-up conditions on this information. The analysis is made more complex by the use of two treatment approaches in the Okanagan lakes, one involving harvesting or cutting below the water's surface and another that entails roto-tilling the lakebed in infested areas during the winter to prevent re-growth.[4] The latter approach is much more successful at reducing damages but more costly and limited to areas with no wharves or water intakes. As a result, roto-tilling is chiefly used near public beaches and high use areas, while harvesting is done in other areas. To maintain the desired level of control in either situation, we assumed that treatment is carried out annually, which is almost always the case in reality.

Establishing the per-hectare treatment cost in BC required some assumptions, which we developed in consultation with officials at the Okanagan Basin Water Board (OBWB). First, we checked the recent budgets for treatment at the OBWB and determined these to be about $500,000, including management and administrative overheads and an allowance for funds to be set aside to replace worn-out machinery. As this budget can be considered relatively complete, in terms of both fixed and variable costs, we set this as our baseline. Next, we had to establish the area treated each year, so we could estimate an annual unit cost (per ha) for control. Unfortunately, this information is not known for the current period matching our baseline budget. Aerial photo information suggests up to 430 ha have been subject to treatment in recent years (310 ha roto-tilling, 120 ha harvesting), so this would represent an extreme upper bound for the area treated annually. Earlier documents and analyses suggest that the area treated annually ranged from 54–108 ha in the late 1980s and as much as 150 ha in the early 1990s. However, it is not known whether these amounts include some area treated twice in a single year, since this practice was quite common during the initial expansion of watermilfoil in the Okanagan. Thus, we decided to use 100 ha as the historic baseline area treated; a budget of $500,000 implies a unit cost for treatment of about $5000/ha.

[4] In contrast, an alternative to mechanical treatment is the use of herbicide pellets, a chemical approach that is the preferred treatment south of the border, and far less costly. However, the potential impacts from the use of these chemicals was not available but may not be negligible.

Note that for this estimate we do not differentiate between the two types of control methods, so this value represents a pooled or average cost across both methods used. We believe this simplified approach is acceptable for two reasons. Since the more expensive method is applied where use values are higher, the "net returns", or difference between benefits and costs, may well be quite similar under either treatment alternative. In any event, we use a pooled approach in assessing the damages from watermilfoil in Section 2 and it only makes sense to do so again here. We maintained the pooled approach to establish the damage reduction parameter, using the aerial photo information describing areas treated with each method cited above as the weights. We assumed the roto-tilling approach achieves a 90 percent reduction in damages, and harvesting only 50 percent. These assumptions yield an estimate of 0.79 for the damage reduction parameter.

Results

Economic analysis of the baseline scenario (treatment budget: CDN $500,000 and inventory budget: CDN $200,000) indicates that the NPV for the mechanical treatment of watermilfoil is positive (Table 3.2.6.1). NPV can be negative when the inventory budget is low or the treatment budget high. The rate of BCR varies between 0.9 and 1.5 for all scenarios. This result indicates that in general both the conventional inventory and treatment budgets generate net benefits to society. We also carried out a sensitivity analysis to understand the responsiveness of the inventory and treatment budgets with respect to economic and ecological parameters. Sensitivity analysis of the discount rate shows that there is a negative relationship between the NPV and the discount rate. The BCR appears to be less sensitive to the discount rate than the NPV (Figure 3.2.6.1). Sensitivity analysis for watermilfoil shows that the intrinsic rate of growth initially has a positive relationship with both the NPV and BCR. Nevertheless, this reaches a peak at the point of 13 percent and then flattens or slightly declines at higher values (Figure 3.2.6.2). This indicates that the proposed budgets are not sufficient to generate benefits when there is a rapid rate of growth.

According to the sensitivity analysis for ecological limit, this parameter has a positive relationship with both the NPV and the BCR (Figure 3.2.6.3). This occurs because the proportionate increase in the present value of benefits is greater than the proportionate increase in present value of cost. Sensitivity analysis for unit damages demonstrates that this parameter has positive relationship with both the NPV and the BCR (Figure 3.2.6.4). This implies that when the unit damages increase, the present value of benefits of avoiding the future damages also increase compared to the present value of cost of treatment program.

Table 3.2.6.1. Economic Analysis of Treatment of Watermilfoil: Mechanical.

Treatment Scenarios	Parameter Assumption	NPV ($ M)	BCR	IRR (%)
Conventional - inventory budget ($/yr)				
Low	100000	-1.0	0.9	NA
Med –base line	200000	2.8	1.2	8.4
High	300000	8.6	1.5	10.5
Conventional – treatment budget ($/yr)				
Low	350000	6.5	1.5	12.9
Med-base line	500000	2.8	1.2	8.4
High	650000	-0.9	1.0	NA
Conventional – discount rate (%)				
Low	2%	4.8	1.1	8.4
Med-base line	4%	2.8	1.2	8.4
High	6%	1.3	1.1	8.4

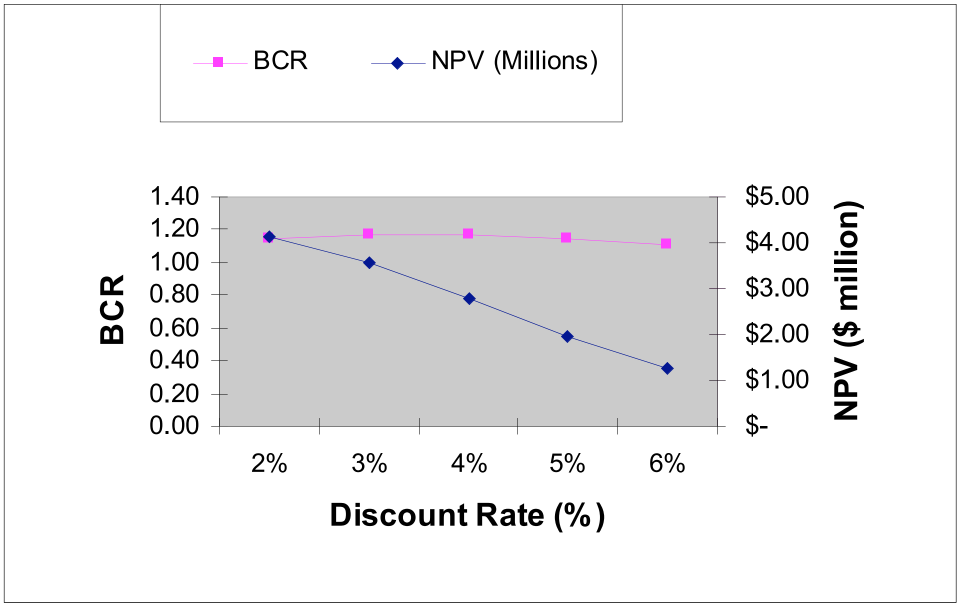


Figure 3.2.6.1. Treatment of Watermilfoil (Mechanical): Sensitivity Analysis for Discount Rate.

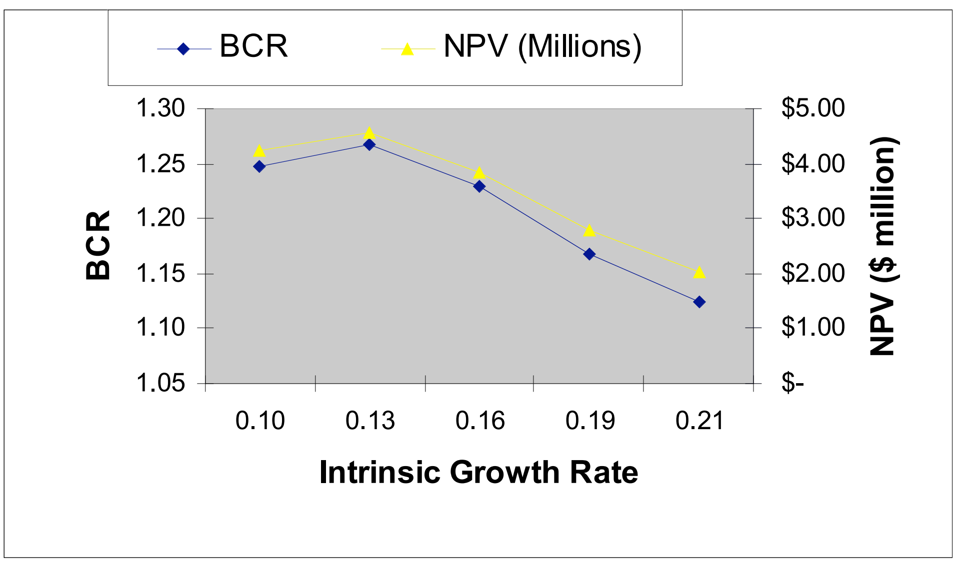


Figure 3.2.6.2. Treatment of Watermilfoil (Mechanical): Sensitivity Analysis for Intrinsic Growth Rate

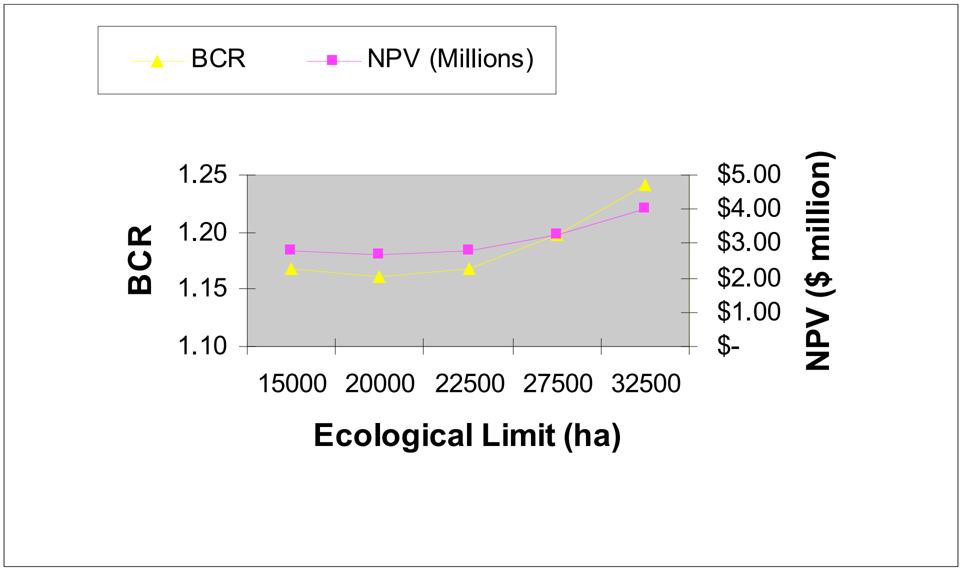


Figure 3.2.6.3. Treatment of Watermilfoil (Mechanical): Sensitivity Analysis for Ecological Limit.

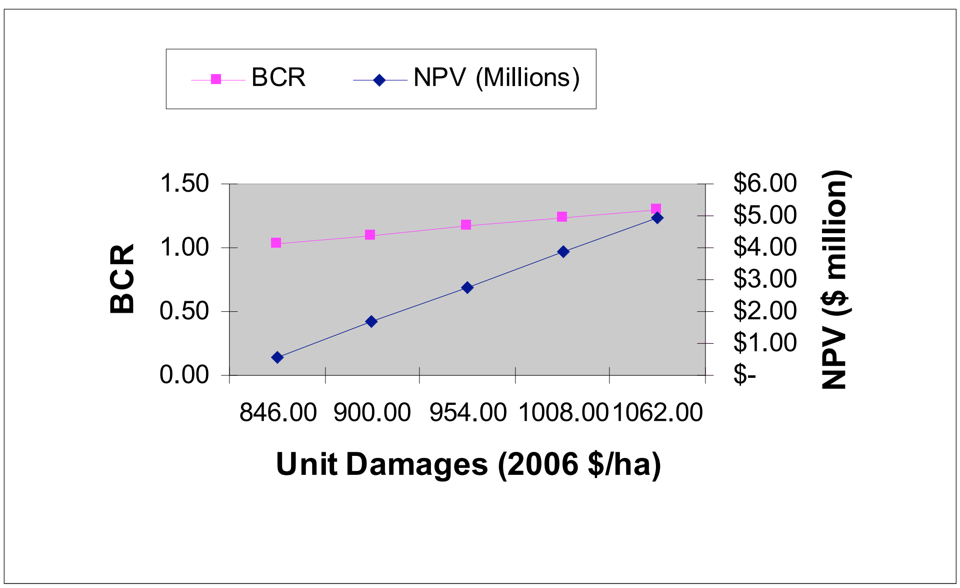


Figure 3.2.6.4. Treatment of Watermilfoil (Mechanical): Sensitivity Analysis for Unit Damages.

3.2.7 Summary of Results

Tables 3.2.7.1 and 3.2.7.2 show summaries of the results of our analyses for the net present value (NPV) and benefit cost ratio (BCR) under Phase 2.

Table 3.2.7.1. Results of the Analyses of Management Scenarios for Four Species – Net Present Value (NPV, 2006 Millions).

	Units	Diffuse Knap -weed Biocont.	Diffuse Knap -weed Chem.	Hawk -weed Biocont.	Hawk -weed (LM) Chem.	Scotch Broom Mech.	Water milfoil Mech.
A. Inventory Budget							
Low	$ M	-	-	1687.0	39.2	-	-1.0
Medium – Baseline	$ M	-	-	1686.1	38.4	-	2.8
High	$ M	-	-	1685.3	37.6	-	8.6
B. Treatment Budget – Conventional							
Low	$ M	-	-0.7	-	18.3	-0.2	6.5
Medium – Baseline	$ M	-	-0.4	-	38.4	-0.5	2.8
High	$ M	-	0.4	-	58.7	-0.9	-0.9
C. Treatment Budget – Biocontrol							
Low	$ M	-	-	1373.9	-	-	-
Medium – Baseline	$ M	-	-	1686.1	-	-	-
High	$ M	-	-	1893.1	-	-	-
D. Escalation in Costs Assumptions							
Baseline	$ M	-	-	1681.1	38.4	-	-
Low	$ M	-	-	1284.3	18.1	-	-
Medium	$ M	-	-	957.8	8.1	-	-
High	$ M	-	-	475.4	0.9	-	-
E. Discount Rates							
Low	$ M	86.7	-0.2	7578.5	75.3	-0.8	4.8
Medium – Baseline	$ M	16.0	-0.4	1681.1	38.4	-0.5	2.8
High	$ M	2.4	-0.8	409.8	20.4	-0.3	1.3

Table 3.2.7.2. Results of the Analyses of Management Scenarios for Four Species – Benefit Cost Ratio (BCR, 2006 prices).

	Units	Diffuse Knap -weed Biocont.	Diffuse Knap -weed Chem.	Hawk -weed Biocont.	Hawk -weed Chem.	Scotch Broom Mech.	Water milfoil Mech.
A. Inventory Budget							
Low		-	-	214.1	7.9	-	0.9
Medium – Baseline		-	-	185.5	6.6	-	1.2
High		-	-	163.6	5.6	-	1.5
B. Treatment Budget – Conventional							
Low		-	0.8	-	4.9	0.1	1.5
Medium – Baseline		-	0.9	-	6.6	0.1	1.2
High		-	1.1	-	7.4	0.1	1.0
C. Treatment Budget – Biocontrol							
Low		-	-	164.2	-	-	-
Medium – Baseline		-	-	185.5	-	-	-
High		-	-	193.0	-	-	-
D. Escalation in Costs Assumptions							
Baseline		-	-	185.5	6.6	-	-
Low		-	-	172.2	4.2	-	-
Medium		-	-	156.6	2.7	-	-
High		-	-	115.9	1.3	-	-
E. Discount Rates							
Low		25.5	1.0	600.5	7.2	0.1	1.1
Medium – Baseline		7.6	0.9	185.5	6.6	0.1	1.2
High		2.4	0.8	57.8	5.4	0.1	1.1

4. Recommendations for Invasive Plant Managers and Decision Makers

4.1 Diffuse Knapweed

Our analysis of diffuse knapweed consisted of a retrospective cost-benefit analysis of the historical biocontrol program and a counterfactual economic analysis of a conventional spraying program hypothetically implemented over the same time period. The latter analysis was budgeted to be roughly equivalent in scope to the biocontrol program. Under baseline assumptions, the historical analysis of the biocontrol program showed a benefit-cost ratio (BCR) of 7.6. Relative to other biocontrol programs, this results falls within the lower range of documented BCRs. For example, Hill and Greathead (2000) analyzed 27 different biocontrol programs against invasive plant and insect pests and found that the BCR varied from 0.99 to 7405. However, the biocontrol program performed much better than the counterfactual conventional spraying program, which demonstrated a maximum BCR of 1.1.

Based on our comparison of the two programs, it seems clear that the biocontrol program is a success. One caveat to this result is that we assume a high level of damage reduction by biocontrol agents. Some observers have questioned this by pointing out that even though densities of the target invasive plant have been reduced, lack of adequate land management has resulted in non-target invasive plants filling the empty niche; from a rangeland perspective, this equates to no increase in forage. To explore the implications of a lower damage reduction parameter for the biocontrol agents, we carried out a further sensitivity analysis to determine the value of this parameter at which the BCR would fall to 1.0. This appears to happen at about 10 percent damage reduction versus the baseline assumption of 80 percent damage reduction. This result suggests that even if the damage reduction as a result of biocontrol was modest relative to our baseline assumption, the diffuse knapweed biocontrol project would have been a worthwhile endeavour. A logical question that arises from this result is whether it is better to spend the time and resources finding a highly effective agent or whether resources should be expended on releasing less effective agents earlier on in a program. Such a trade-off could be explored further using the diffuse knapweed biocontrol model.

If we consider only investments made by the province of BC towards the biocontrol program, then the BCR for the baseline assumptions of the diffuse knapweed program increases to 17.0. This result shows that by working as part of a consortium on the research and development of a biocontrol program, BC was able to increase the value of its investment. By cooperating and sharing the responsibility of research, individual agencies and levels of government are far more likely to be successful in the development of biocontrol than if they attempt to tackle the problem alone.

One important aspect of our analysis of diffuse knapweed biocontrol is that our model projects net positive returns are not experienced until the program is well underway (2005 under baseline assumptions). This is because during the first two decades of the program, releases were conducted over a long period with minimal budgets. One lesson learned for future biocontrol programs may be the importance of having a large enough release budget early on in a program to ensure that source populations of agents become well-established. This, however, may be counterbalanced by the need to ensure that early releases are effective.

One final recommendation with respect to diffuse knapweed biocontrol is that there are actually very little data on evaluating the effectiveness of the program at a provincial scale and better evaluation is required. There are some site-specific studies (Myers et al. 2009) but only anecdotal accounts of the overall success of the program. Our analysis therefore relies heavily on modeling. It is often the case with resource management programs that once the implementation of a project is complete there is little or no funding left for evaluation. However, in the long term this is not a viable approach because it prevents us from ever clearly learning from our actions. Future biocontrol programs should explicitly include a project evaluation plan upfront and should ensure that sufficient funding is left to complete it post-implementation.

4.2 Hawkweed

For hawkweed we carried out a comparative analysis similar to the knapweed analysis described above. However, the analysis of hawkweed differs in several respects. First, the hawkweed analysis is for a biocontrol program that has been initiated only recently, and therefore the analysis is *ex ante* rather than *ex post* as discussed earlier. Second, we perform an additional analysis that involves two different chemical control programs: the first is the same type of program as used in the diffuse knapweed analysis which involves repeated visits to a site for spraying and seeding for a period of three years; for the second alternative conventional program, no seeding is used and spraying must be carried out every year to maintain a level of control because there is no complementary investment in land management. The trade-off we test in this comparison is whether it is better to allocate a high level of restoration resources per unit area, as required for seeding and repeated site visits, and apply them to a smaller area overall (land management scenario), or alternatively to use fewer resources per unit area but attempt restoration over a larger proportion of the landscape (no land management scenario).

The baseline BCR for a hawkweed biocontrol program is 185.5, which is much higher than the value for the historical diffuse knapweed program, but still well within the range of BCRs reported by Hill and Greathead (2000). Under baseline assumptions the BCR for conventional spraying is significantly greater than 1.0 as long as land management is carried out. However, with a BCR of 6.6, the net returns from conventional spraying are much lower than from an equivalently budgeted biocontrol program, even if land management is included in the spraying program. If no land management is carried out, then a conventional spraying program is only marginally viable, if at all. This highlights the importance of implementing land management actions, like seeding, that ensure long-term benefits from spraying of a target invasive plant species. In the case of hawkweed, it appears that allocating resources to effective control over a smaller area is a better approach than trying to spread limited restoration resources across a larger portion of the landscape.

For biocontrol of hawkweed, the more attractive returns compared to knapweed reflect the higher damages per hectare and ecological limit we have estimated for the former. Another aspect of this program that is different from diffuse knapweed is that we assume a consistent and sufficiently large budget to carry out ample releases at the outset of the program. This leads to net returns being experienced within 18 years of the program start for hawkweed, whereas for diffuse knapweed it takes 38 years to reach this point. This highlights the importance of allocating sufficient resources for field releases early on in the program in order to maximize the benefits gained from the resources allocated to the development of the agents themselves.

Another difference between the biocontrol and conventional control approaches for hawkweed is that conventional approaches are much more sensitive to delays in the beginning of the program implementation. With a 20-year delay in the hawkweed program, the BCR only decreased by a factor of 38 percent for biocontrol whereas for conventional management it decreased by 80 percent. In both cases there is a cost of delaying, but the cost is much greater in the case of conventional management. This is due to the fact that biocontrol is self-maintaining and eventually can spread across the landscape, whereas sites controlled by conventional means may become re-infested and management must be actively applied across the entire landscape. A general recommendation from this result is that the cost of delay is high, and since it is uncertain how long it will take to develop successful agents for biocontrol, it is prudent to implement conventional management strategies in the short term if this can be done in tandem with activities aimed at developing biocontrol agents for future use. Finally, the lower sensitivity to delay in the implementation of biocontrol suggest that another trade-off that would be worth exploring with the model is delaying the release of agents and waiting until more effective agents are found, rather than releasing less effective agents early on.

4.3 Scotch Broom

Our analysis for the control of Scotch broom along the Island Highway showed that this approach at the current scale is likely not economically viable. One important caveat, however, is that we only considered the impacts of management actions on the corridor itself. It is likely that benefits from the control program could also be experienced outside of the corridor, particularly if the surrounding area is vulnerable to infestation by propagules spreading from the corridor outwards.

To explore the effect of corridor context we conducted a sensitivity analysis to the overall study area size by increasing the ecological limit. It appears that if we consider the analysis area to be beyond 0.5 km on either side of the corridor (4800 ha) with a relatively un-invaded surrounding area vulnerable to invasion, then the program is beneficial in that it maintains the surrounding area in a less invaded state than if there was no control within the corridor. This suggests that it may be useful to managers to consider the areas surrounding the corridor and their vulnerability to invasion from plants within the corridor. This could be used as a criterion for prioritizing which transportation and utility corridors should receive limited resources for invasive species management. It is important not to consider the corridor alone but how the impacts of the invader may extend beyond it.

4.4 Eurasian Watermilfoil

Results from our analysis suggest that there is a potentially modest benefit to society from the conventional management of Eurasian watermilfoil at a provincial level (BCR = 1.2 under baseline assumptions). One of the key results of the analysis is that the modest gains can drop to a loss if too few resources are allocated to inventory versus management. This result can be explained by the fact that once established, watermilfoil populations persist and must be managed in perpetuity. Establishing an inventory program that allows for early detection and rapid response to potential introductions into new lake systems is therefore very important. However, the practical implementation of such an approach is unclear. There are various approaches that could be used for keeping watermilfoil from infesting currently un-infested lakes, such as public education and mandatory boat cleaning stations and inspections. Evaluating the effectiveness of the first approach is difficult, and implementation of the second at a broad-enough scale may not be possible. Further research into how to keep watermilfoil out of pristine aquatic systems is clearly necessary. In addition, because of its rapid spread rate and persistence in the environment, watermilfoil may be another invasive plant that warrants the development of biocontrol.

4.5 Recommendations

The following recommendations were derived from the results of our study across all species:

1. Our study suggests that biocontrol of diffuse knapweed has been successful and could potentially be very successful to manage hawkweed. We recommend continuing efforts to develop a set of successful bio-agents for hawkweed.

2. Future biocontrol programs should include a plan for evaluation at multiple spatial scales: individual plants, release sites, and both regional and provincial. Evaluation criteria should not be based solely on the establishment of the agents but should include an assessment of the benefits, such as increases in forage, biodiversity, and ecosystem services. The implementation of such a plan would allow for a more accurate assessment of the economic and ecological benefits derived from the biocontrol program.

3. Further analysis is needed to evaluate the trade-off between releasing less-effective agents sooner and delaying releases until more effective biocontrol agents are discovered.

4. Future participation by the province in the research and development of biocontrol agents undertaken by similar consortia is a worthwhile investment. This is based on past investment in the diffuse knapweed biocontrol program undertaken by a consortium of government agencies that generated a significant return, which would not have been possible if a single entity attempted to develop the program alone.

5. Ensure that sufficient resources are available for conducting field releases as early as possible in a biocontrol program, without compromising the prevention of non-target effects. Sufficient resources early on in the program can greatly increase the net economic benefits generated from the investment in the development of the agents released.

6. Economic evaluation of the cost of invasive plant species should be made prior to the release of biological control agents as a baseline on which success can be evaluated.

7. Standardized monitoring procedures should be developed to track changes in the densities of the target invasive plant, the biological control agents, and the plant community. Quadrat sampling along fixed transects is a repeatable and efficient procedure for collecting temporal data on plant densities and plant community structure. These data should be made available through regular reports or on websites so that they can be publicly accessible. Sites with and without agents should be monitored to provide evidence of the effectiveness of the introduced agents (Carson et al. 2008).

8. Efficacy testing should be part of the development of biological control agents to improve the success rate of introduced agents in reducing plant density and to reduce the number of exotic species being introduced. Seed predators should only be used in cases in which the target invasive plant has been shown to be seed limited.

9. Land management actions, such as grazing management and seeding, are an important component of an invasive plant control program. Even though these actions may be costly in the short term, their long-term benefits far outweigh the cost of implementation.

10. The management of invasive plants along utility and transportation corridors requires prioritization of corridors that have the potential to impact the surrounding area. Corridors surrounded by vulnerable, un-invaded habitat should be prioritized over those that are not.

11. A key aspect of a control program against Eurasian watermilfoil is the allocation of resources towards inventory and education aimed at preventing the infestation of currently un-invaded, but vulnerable, lake systems.

12. The present study probably represents the limits of what can be done with the current level of damage information. More primary research is required into the valuation of damages from invasive plants in BC. As an example, a small research program could be sponsored that would fund student research at the Masters and PhD levels.

13. The impacts of climate change on the distribution of the important invasive plant species should be considered for future analysis.

5. Recommendations for Future Work

5.1 Future Data Collection

5.1.1 Biocontrol Effectiveness

Biological control success can be evaluated in a variety of ways: (1) scientific success – quantified decline of the target invasive plant, (2) ecological success – restoration of the vegetation community to a desired state, (3) economic success – reduced cost of control, (4) political success – support for future funding, (5) social success – change in perception of the invasive plant, and (6) legal success – development of laws to prevent future introductions. A failure of many biological control programs is that they do not quantify these successes (Myers and Bazely 2003). However, approximately 30 to 50 percent of biological control programs worldwide have been considered to be effective in some way, which is usually judged by a decline in the target invasive plant.

The first successful biological control program in British Columbia was against St. John's wort (*Hypericum perforatum*) in the 1960s. The density of this pasture species was reduced in many areas following the establishment of two introduced *Chrysolina* beetles. The next success was the control of another pasture invasive plant, tansy ragwort (*Senecio jacobaea*) in the Lower Mainland of BC, primarily by the flea beetle, *Longitarsus jacobaeae*. The strain of the flea beetle that has been successful in coastal areas does not survive in Interior areas and thus a new strain must be found for these areas. Some success is reported with a cold-adapted strain in the USA (Littlefield et al. 2007) that might also be effective in BC. More recently, populations of the wetland invasive plant, purple loosestrife, have declined due to the attack by the introduced leaf beetle, *Galarucella calmariensis* (Denoth and Myers 2005).

Hound's-tongue (*Cynoglossum officinale*) is a serious rangeland invasive plant in British Columbia, because the seeds become attached to the faces and hides of cattle and the foliage is toxic to large mammals. The root-boring weevil *Mogluones cruciger* has effectively established at many sites in BC where it attacks over 90 percent of flowering plants, kills over half the rosettes, and has reduced the density of hound's-tongue at most of the release sites (De Clerck-Floate and Schwarzländer, 2002; De Clerck-Floate et al. 2005). Dalmatian toadflax is another rangeland invader in BC. The stem-boring weevil, *Mecinus janthinus* Germar, has been highly successful in reducing densities of this invasive plant (De Clerk-Floate and Harris 2002). And finally, after 30 years, the biological control of diffuse knapweed appears to be successful at many sites following attack by the introduced weevil *Larinus minutus* (Myers et al. 2009). This program involved the release of 12 agents over approximately 25 years. Initially, the emphasis was on the release of seed predators. As recently shown by Garren and Straus (2009) and previously by Myers and Risley (2000), seed predators are not successful in situations in which seedling success is site limited rather than seed limited. This is the situation for diffuse knapweed, and only after *L. minutus*, which attacks many parts of the plant, became established did plant densities decline.

The successes mentioned above have been scientific successes in that the measure has been the reduction of invasive plant densities. Only for knapweed, however, have changes in rangeland plant communities been evaluated after the decline of the target invasive plant (Stephens and Myers, in review). Both introduced and native grasses increased in percent cover with the decline of knapweed, particularly by introduced cheatgrass (*Bromus tectorum*). Unfortunately, the decline of knapweed has not resulted in a measurable increase in the capacity of the rangeland to be grazed (Anne Skinner, pers. comm.). Another potentially successful biological control agent is the root-boring beetle (*Cyphocleonus achates*) on spotted knapweed (*Centaurea stoebe micranthos*). This weevil is reported to reduce the density of spotted knapweed in Montana (Story et al. 2006) but no comparable data have been published for British Columbia.

Shea et al. (2002) suggest that biocontrol programs are ideally suited to active adaptive management experiments where uncertainties are identified, experimental management strategies are designed, predictions are made across alternative hypotheses for identified uncertainties, management actions are implemented, the results are evaluated, and the knowledge of the system and subsequent management actions are modified. This kind of management experimentation could in particular be applied to agent selection, release strategies, and land management actions. However, the reality is that funding for any type of evaluation has rarely been available in past programs.

Until the level of support for effectiveness evaluation increases, it will be difficult to predict the potential benefits of future biological control programs or even to properly evaluate the benefits of past biocontrol programs. For example, our evaluation of the diffuse knapweed biocontrol program relies heavily on the damage reduction parameter for biocontrol. Based on the evidence that diffuse knapweed densities have been greatly reduced, we use a high value for this parameter. However, a better estimate of this parameter would be based on the ecological and economic evaluation of the program success; in other words, has the vegetation community moved towards a more desirable state or has the removal of the target invasive plant merely resulted in other undesirable species, such as cheatgrass, moving into the niches opened by the removal of knapweed? As well, has the removal of the knapweed resulted in increased forage production and a reduction in soil erosion? Until these questions have been answered with certainty we must interpret the results of our analysis with caution. Therefore, what must be established is a consistent, efficient, and cost-effective program to follow the impact of introduced agents and to provide information that allows evaluation of the program in at least a comparative manner.

5.1.2 Species Distributions under Current and Future Climate

One key aspect of our analysis involved the development of an estimate of the ecological limit for each species that we analyzed. Our estimates were mainly derived from expert opinion and a gap analysis conducted by Miller and Wikeem (2005). Better estimates of the potential distributions of invasive plants would not only help refine the accuracy of our economic analysis, but would also help direct inventory efforts to vulnerable regions, prioritize restoration efforts to habitats where success may be more likely and anticipate the effects of climate change on the potential distribution of invaders. Various techniques are available for modeling the potential distribution of invasive plants which generally involve matching occurrence and absence data with various potential biogeoclimatic environmental variables and deriving a statistical model that predicts habitat suitability (Evangelista et al. 2008). To do this, accurate occurrence data are required, ideally both in the native range and in the invaded range of the species (Broennimann and Guisan 2008). These kinds of modeling techniques should be applied to important invaders and potential invaders in BC using both current climate data and projected future climate data.

5.1.3 Economic Data

One of the challenges we faced in preparing this report was the very limited secondary information on economic damages arising from invasive plants in BC. There has been very little investigation into this subject by economists. As a result, the research we present in the report makes use of "borrowed" information using the technique referred to as benefits transfer. In addition, only a subset of the full set of damages from each invasive species, as described by the impact diagrams and total economic value framework we presented earlier, could be valued because of the limited information. For example, we did not include the potential external costs associated with introducing biocontrol agents into the host environment. While the externalities involved in chemical treatment are better known and documented – and were included in our study – relatively little is known about the secondary and unintended economic damages from biocontrol introductions (for an exception see Sinden et al. 2004). On a related note, the non-use value associated with losses of biodiversity and other impacts on pristine environments invaded by non-indigenous species may not be trivial. Yet we were not able to include these lost values. Again, further investigation by social scientists of the values held by people towards pristine versus invaded ecosystems are needed.

Thus, there is a strong argument for more research into the valuation of damages from invasive plants in BC, taking a more rigorous approach involving primary data gathering. This would involve both a scientific component (e.g., field and experiment station trials) and a social science component (e.g., stakeholder interviews and surveys). A substantive research program should be directed in this area before further provincial scale analyses be undertaken. This suggestion recognizes that the present study has provided some initial estimates and directions for extending research but probably represents the limits of what can be done with the current level of damage information. As an example of what might be undertaken, the IPCBC or its collaborators could sponsor a small research program to fund student research at the Masters and PhD level with the intent of improving the information on damages. The project team would be please to provide more detailed suggestions of what such a program might require.

5.2 First Nations Values

All of the species assessed for this project pose a potential risk to First Nations in BC. Two main impact pathways stand out from impacts likely to be equally felt by First Nations and non-First Nations people in the affected area. The first pertains to a reduced availability of biotic traditional resources for food, social, and ceremonial purposes, which could have cultural, economic, and health impacts on the First Nations communities that rely on these resources. The second (and related) pathway pertains to a reduction in well-being. There is a disparity in well-being (life expectancy, income, and educational attainment) between Registered Indians and other Canadians (Cook et al. 2004); therefore, health or economic impacts from invasive plants may be greater for First Nations communities than for other residents of BC. For example, disparities in income, combined with historical reliance on traditional foods, are likely to result in greater economic and/or nutritional impacts for First Nations from a reduced availability of plants and animals harvested for food. These impacts are illustrated in Figure 5.2.1.

Further investigation of the potential economic impacts of these (and other) invasive plants on First Nations in BC would require detailed information on traditional uses across geographic areas and First Nations communities. Traditional knowledge must be a primary component to any studies on the economic impacts of invasive plants on First Nations. A description of the “six faces” of traditional ecological knowledge by Houde (2007) describes the nature and breadth of this type of knowledge, providing insights into the inter-connection between ecology and culture, values, and world view for First Nations which is often poorly understood by non-aboriginal natural resource scientists and managers.

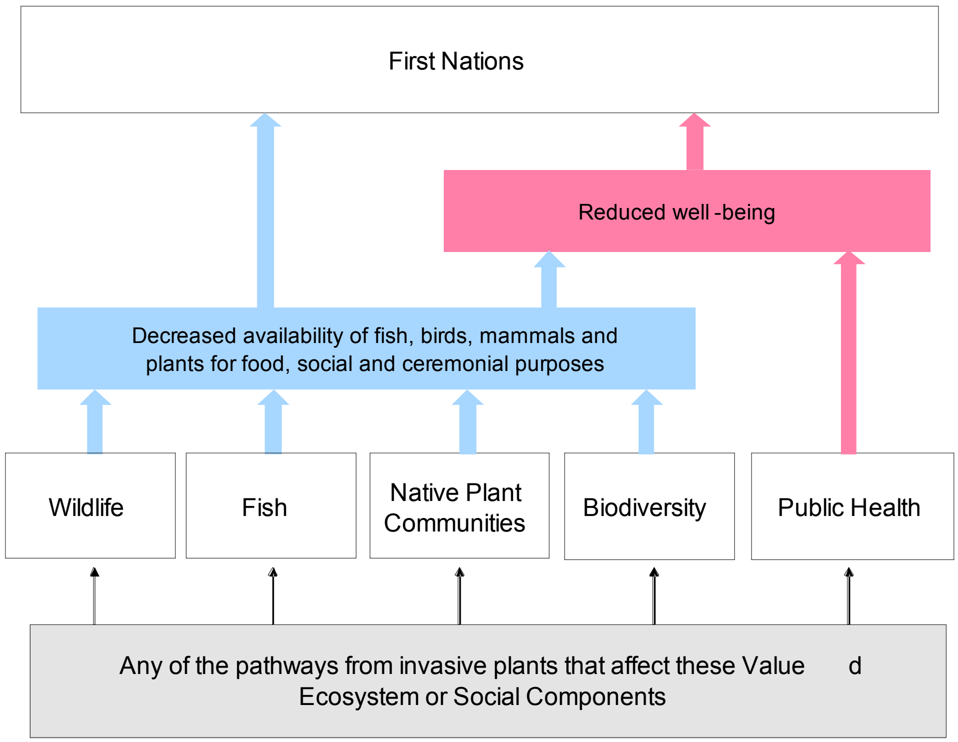


Figure 5.2.1. Impact Diagram for Invasive Plants on First Nations Communities in BC.

5.3 Additional Species

5.3.1 Yellow Starthistle

Yellow starthistle, *Centaurea solstitialis* (L.), is a Eurasian native introduced to California sometime after 1849 as a contaminant of alfalfa seed from Chile, where it was originally introduced as a seed contaminant from Eurasia (Maddox 1981). It is a winter annual; seeds germinate with the first autumn rains and plants flower the next spring (Young et al. 2005). Yellow starthistle can grow in deep soils and in well-drained, shallow, rocky soils from sea level to 2500m (Maddox et al. 1985). Although native to the Mediterranean and most common in southern Europe, yellow starthistle now occurs in all temperate areas of the world.

The spread of yellow starthistle was monitored in California from an estimated 405,000 ha in 1958, to 3.24 million ha by 1985, and 6.7 million ha by 2006 (Pitcairn et al. 2006). More recently, yellow starthistle was introduced to Washington, again most likely as a contaminant of alfalfa seeds and, although its distribution is primarily in the southeastern part of the state, it does occur in counties adjacent to the Canadian border. In Washington the rate of spread has also been rapid, from 60–4000 ha between 1954 and 1964. In 1994 this aggressive invader was estimated to be spreading at a rate of 6,070–20,234 ha per year (Sheley and Larson 1994).

Yellow starthistle is dispersed mostly by human activities, such as road building, movement of hay, and through uncertified, contaminated seed. On a local scale, it is moved by humans and animals and little wind dispersal of seeds occurs. Its seedbank lasts 3 to 4 years. Yellow starthistle occurs as far east as New York but does best in Mediterranean climates in the west and in the Great Basin.

In California, yellow starthistle is ranked as the most important of 29 weeds (Maddox and Mayfield 1985). It spreads rapidly into rangeland and can reduce dryland wheat production. It reduces the grazing capacity of rangelands because it is poor quality forage in the spring and then becomes unpalatable in the summer. It causes a neurological disorder if ingested by horses. In addition, yellow starthistle invades orchards, vineyards, roadsides, and wastelands in California. The spines associated with the capitula of the plant make it very unpleasant to walk through infested areas and thus it also has an impact on recreational lands. Yellow starthistle also can deplete soil moisture and has been estimated to cost 16 to 56 million dollars a year in the Sacramento watershed in terms of water conservation (http://wric.ucdavis.edu/yst). From a survey of ranchers in California, the estimated cost of yellow starthistle to forage losses was $7.65 million with expenditures to ranchers of $9.45 million a year. These losses make up 6 to 7 percent of the total annual harvested value (Eagle et al. 2007). One benefit of yellow starthistle, however, is that it is a good nectar source, and beekeepers in infested areas value it for honey production.

Management of yellow starthistle depends on a multi-pronged approach involving pulling, mowing, burning, herbicide treatment, tillage, grazing, and biological control. Short-term intensive grazing can select against yellow starthistle and favor the recovery of grass in the spring. Mowing and herbicide treatment were effective on roadside infestations of it (Young and Claassen 2008). Burning in three consecutive years can reduce both the seedbank and vegetation cover (Di Tomaso 1999). Reseeding with native bunchgrasses after fire can be beneficial in reducing the return of yellow starthistle (Di Tomaso et al. 2000). Details of various management strategies, including biological control, can be found at http://wric.ucdavis.edu/yst.

Thus far, six insects have been introduced for the biological control of yellow starthistle in California, all of which attack seed heads. These have not been effective at reducing plant densities (Pitcairn 2006). It has been shown experimentally that reducing seed production will not be sufficient for reducing plant densities (Garren and Strauss 2009). Currently, two beetles and a lace bug that feed on immature plants and the early flowering stage are being studied and tested for host specificity in the USA (Paolini et. al. 2008).

Threat to BC

In some ways, it is surprising that yellow starthistle has not yet spread to British Columbia. Given that the primary mechanism for spread is contaminated seed or hay, it has the potential to jump to rangelands throughout the province, and introductions will not be limited to spread by diffusion from sites in Washington to adjacent locations in BC along the border. Yellow starthistle flowers are quite distinctive because of their bright yellow colour and spines, and they should be easily identified by landowners, naturalists, and others. As a legislated provincially noxious weed, yellow starthistle has received increased attention at a provincial level. It has also been one of the target species of an international education campaign for the past six years, as part of the Weeds Cross Borders project[5]. Yellow starthistle should continue to be the focus of a publicity campaign, expanding to regions beyond those located along the international border, to increase the awareness of this threat so that initial introductions can be rapidly identified. A procedure should be developed so that an eradication program can be rapidly mounted following the probable introduction in the future (Simberloff 2009). Whether yellow starthistle is as well-adapted to northern environments as it is to California and southeastern Washington is not known, but this is a species that should be guarded against with a rapid, aggressive removal program if it is found. Consequently, BC needs to fully implement a formal Early Detection Rapid Response (EDRR) system.

5.4 Ecological Context

5.4.1 Single Species versus Community Management Approach

The present study addresses invasive plants as single species and is not a community model. It is recognized that this is a simplistic view of non-native plant invasions, however this approach successfully establishes baseline data from which more complex analyses and models may be developed.

The traditional single species approach to invasive plant management ignores the complexity inherent in natural systems and the interactions of multiple invasive plant species across a landscape. Another difficulty of a single-species model is that one invader may have large effects in some areas and negligible effects in others (Byers *et al.* 2002). Although some invasive species are capable of altering the normal functioning of ecosystems or the interactions of organisms even in relatively small numbers, calculating economic impacts for a single species may not have much utility when most landscapes have mixed species present. Furthermore, species that affect system-level processes may in turn facilitate the invasion of additional species (Simberloff and Von Holle 1999), and this may not be fully recognized until a certain level of control is achieved. In BC's Southern Interior there is anecdotal evidence to suggest that the reduction of certain invasive plants as a consequence of successful biological control appears to have paved the way for invasion by other non-native species (A. Skinner, pers. comm). This is supported by analyses of plant inventories conducted over a five-year period on a select rangeland site in the Okanagan Valley, which indicate that decline of diffuse knapweed is followed by an increase in grass cover, with the non-native *Bromus* species predominating (Stephens and Myers, *in review*). This is also supported by Thomas (1986), who cites examples of management practices that were used to combat one invasive species resulting in stand disturbance sufficient to allow the establishment of a second non-native species that had a greater negative impact on the habitat than that caused by the initial infestation.

Many natural areas contain far more non-native species than their managers can control, so priorities must be set for the control, prevention, or containment of only a fraction of the invasive species they face. Future research that distinguishes species with negligible effects from those

[5] The project is a partnership of land managers, agencies and regional committees in the greater Okanagan region of Canada and the United States, and provides an integrated and coordinated approach to invasive plant management, sharing resources for education, training, inventory and control.

that cause significant damage to native biodiversity would allow land managers to direct attention and resources to the most important concerns, thereby maximizing protection of natural systems (Byers et al. 2002). However, future studies should also explore the potential for modeling at a community level, which may more accurately reflect the reality of what is actually occurring on the landscape, and additionally assist with providing direction for invasive plant management for a particular plant community, not an individual species.

5.4.2 Land Management Actions and their Impacts on Invasive Species

Natural areas that are most affected by invasive plants are often under stress from disturbances, such as air and water pollution, and habitat fragmentation (White and Haber 2003). Programs that reduce these disturbances might be more effective in the long term to re-establish natural conditions in an area than attempting to remove invasive species that are more of a symptom than the basic problem. White and Haber (2003) view southern BC as one of two areas in Canada where this would be particularly challenging. As noted in Section 3.1.2, land management actions that focus on reducing the likelihood of invasion through improved range and development practices are a recognized tool. However, modeling land management actions is complex and challenging because there are so many different types of approaches to land management, and the economic responses are equally variable.

Invasive plant management needs to be based on a fine-tuning of managed ecosystems, in which land uses must be comprehensively adjusted to confront species with a wide array of control measures. Land managers must have the primary role in this process because of their holistic knowledge of the ecosystems they manage (Jordan et al. 2003). Changes in land use may retard or facilitate invasion, and control might therefore be possible through indirect pathways (Vitousek et al. 1997). An improved knowledge of these pathways should be the focus of future research, so that land managers are better informed. If managers are aware of side effects of their land management activities, they can seek alternative actions or at least develop ways to mitigate the effects of actions that cannot be avoided (Byers et al. 2002). The analysis of hawkweed in BC (Section 4.2) illustrates the importance of undertaking land management actions that complement selected treatments, in order to achieve effective, long-term results.

A clear understanding of how best to alter land management practices to reduce the invasibility of the landscape is lacking, however. Knowledge of factors that fortify the resistance of plant communities to invasion could lead to more effective land management techniques (Byers et al. 2002).

5.4.3 Integrated Management Approaches

Invasive plant management requires a strategic approach that integrates available control techniques (i.e., use of herbicides, physical removal, cultural control, and biocontrol agents) with preventative measures. This concept is referred to as ecological or integrated pest management, or simply integrated management. Prevention is a pillar of integrated pest management (Norris et al. 2003) and arguably the most cost-effective approach that land managers can take. The most effective and efficient combination depends on factors such as the biology of the particular invasive species and the circumstances under which it is growing. It means dealing with invasive plant problems not simply at the level of the individual infestation but also at other scales, such as the management unit, watershed, and landscape and regional scales. The regional scale is typically used for natural resource management planning, and yet few credible models can be found which can be used for invasive plant planning at this scale (Lindsay and Herpich 2008).

The complexities and scales inherent with integrated management precluded such analyses in the present study, with the exception of hawkweeds for which a combination of two tools was modeled (Section 4.2). Interestingly, the net returns from the combination of a conventional spray program and land management (i.e., seeding) for hawkweed were determined to be much lower than from an equivalently budgeted biocontrol program. One needs to interpret these results cautiously, however, and not conclude that an integrated approach is less effective for hawkweed, particularly as the actual success of a biological control program for hawkweeds is unknown at this time. Ultimately, site-specific conditions will dictate the most suitable techniques to be employed. For any one species, the degree of control that is possible or appropriate for at a particular site may be highly variable.

Removal of an invasive plant from a community requires removal of the disturbance factor that allowed the non-native plant to originally invade, and restoration of the habitat as near as possible to its original condition (Thomas 1986). The most successful endeavours follow an adaptive management strategy whereby land managers select and implement the combination of most suitable control measures, focus on the vegetation or community desired in place of the invasive plants, and periodically re-evaluate whether their programs are moving them toward this objective (Randall 1996).

Three types of studies are recommended by Byers et al. (2002) that could be applied to the invasive plant situation in BC to assist in determining what combination of control strategies is most effective for a particular species or suite of species:

1. The life history and demographic models and key factor analyses to identify the most influential aspects of a species' population increase;
2. Field trials of prevention and control techniques; and
3. Testing of site designs and placement that most reduce invasibility.

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Appendix 1: Description of the Management Model

Our model begins with the simple assumption of a logistic growth model in discrete time. In the absence of management the area invaded by a particular plant over time can be described by the following logistic growth equation:

$$N_{t+1} = N_t + r \quad 1 \quad \frac{N_t}{K} \quad N_t \quad \textbf{(1)}$$

Where N_t is the area invaded at time t, r is the intrinsic growth rate in discrete time and K is the area of the ecological limit for the species in the area of analysis[6].

To include management in this system we break the population into four components: area that is invaded and known to managers *Nk*, area that is invaded and unknown to managers *Nu*, area that is being contained by a biological control agent *Nb* and *Ne* area that is in the first year of invasion and is unknown to managers.

The area that is known to managers over time will decrease as a function of eradication:

$$E_t = se T_t \quad \textbf{(2)}$$

Where *se* is the rate at which treatments eradicate areas treated and T_t the area treated at time *t*.

The area known to managers will increase as a function of detection of unknown areas:

$$D_t = Nu_t \quad \frac{si\left(I_t + sk\left(K \quad Nk_t\right)\right)}{K \quad Nk_t} \quad \textbf{(3)}$$

Where Nu_t is the area that is invaded but unknown to managers at time *t*, *sk* is the proportion of the landscape that has a known state prior to inventory due to factors such as past knowledge, research, education and outreach. *K* is the ecological limit and I_t is the area inventoried at time *t*. This term represents the proportion of the unknown landscape that receives successful inventory, either directly or through outreach and education, multiplied by the absolute amount of unknown area invaded. Our assumption is that inventory is allocated randomly across the landscape and that the likelihood that an area that is invaded will be inventoried is in direct proportion to the proportion of the total of the unknown area that is inventoried.

Note that the following condition applies:

$$I_t \quad \left[\left(1 \quad sk\right)\left(K \quad Nk_t\right)\right] \quad \textbf{(4)}$$

[6] Note that the area of analysis and consequently the ecological limit we apply will vary and can range from the Island Highway corridor for Scotch Broom to the entire province for hawkweed and Diffuse Knapweed.

This is because the area inventoried must be less than the area within the ecological limit that is in an unknown to managers. This constraint ensures that the new detections will at most equal the total area invaded that is unknown to managers.

The area that is known to managers will also decrease as a function of successful biological control releases and subsequent spread from established biological control populations:

$$B_{kt} = Nb_t r_b \left(1 - \frac{Nb_t}{N_t}\right) \frac{Nk_t}{(N_t - Nb_t)} + srR_t \quad \textbf{(5)}$$

Where R_t is the area of releases at time t, sr is the release success rate for biological control and r_b is the intrinsic spread rate of biocontrol. Here we assume that the amount of biocontrol spread to areas that are invaded and known to managers occurs in proportion to their occurrence relative to the total weed population.

So in summary the area that is known to managers can be described as:

$$Nk_{t+1} = Nk_t - E_t + D_t - B_{kt} \quad \textbf{(6)}$$

New invasions will be affected by eradication (see above), the area of early detection and rapid response, and the area where treatments did not result in eradication but did result in preventing the spread of the areas treated.

The area of early detection is similar to the area of new detections and can be described as:

$$ED_t = Ne_t \frac{si(I_t + sk(K - Nk_t))}{K - Nk_t} \quad \textbf{(7)}$$

This represents the proportion of the weed population that was eliminated because it was detected early enough (in the first year) to make this easily accomplished. We assumed that easy elimination can only occur within one year of initial invasion. This allows us to create a simple model that does not require an age structure. As with the detection of unknown areas, our assumption is that inventory is allocated randomly across the landscape and that the likelihood that an area that is in an early infestation will be inventoried is in direct proportion to the proportion of the total of the unknown area that is inventoried.

The area where spread is prevented can be described as:

$$P_t = spT_t + bsp\,Nb_t \quad \textbf{(8)}$$

Where *sp* is the success rate for preventing the spread of areas that are treated and *bsp* is the success rate for preventing spread from invaded areas that are being contained by biological control agents. Note that the sum of *se* and *sp* must be less than or equal to one.

New invasions can then be described as:

$$Inv_t = r\ 1\ \ \frac{N_t\ \ E_t\ \ ED_t}{K}\ (N_t\ \ E_t\ \ ED_t\ \ P_t) \qquad \textbf{(9)}$$

The numerator of the fraction represents the effective population size that is using up a portion of the carrying capacity after eradication and early detection. The last term in brackets represents the effective reproductive population size after eradication, early detection and spread prevention.

The area of early invasion will also decrease as a function transitions to the biocontrol state from spread of the biocontrol agents. This can be described as:

$$B_{et} = Nb_t r_b\ 1\ \ \frac{Nb_t}{N_t}\ \frac{Ne_t}{N_t} \qquad \textbf{(10)}$$

This term is similar to B_{kt} but note that in this case there is no release term because populations of initial infestations are unknown to managers and therefore biocontrol agents can't be released there.

The last factor by which the proportion of the population that is in the first year of invasion will change is the growth and succession of the population into a full blown invasion after the passage of one year, in other words a transition from the early invasion stage *Ne* to the unknown invasion stage *Nu*. This succession can be described as:

$$S_t = Inv_{t\ 1}\ \ ED_{t\ 1} \qquad \textbf{(11)}$$

In other words the area that had new invasions last year moves on to become a full blown invasion this year after we remove the area that was successfully detected early, last year. So the change in the early invasion portion of the population can be described as:

$$Ne_{t+1} = Ne_t + Inv_t\ \ S_t\ \ ED_t\ \ B_{et} \qquad \textbf{(12)}$$

Finally, the area that is invaded but unknown to managers decreases as a function of new detections (see above), increases as a function of succession from the early infestation stage (see above) and decreases as a function of the spread of biological control agents into unknown areas:

$$B_{ut} = Nb_t r_b \; 1 \; \frac{Nb_t}{N_t} \; \frac{Nu_t}{N_t} \qquad \textbf{(13)}$$

The area that is invaded but unknown to managers can be described as:

$$Nu_{t+1} = Nu_t \quad D_t + S_t \quad B_{ut} \qquad \textbf{(14)}$$

Finally, aside from increasing due to releases and spread, the area of biocontrol (Nb) decreases as a function of the extinction of the host plant due to the attack of the biocontrol agents. This can be described as:

$$E_{bt} = {}^{sbe} Nb_t \qquad \textbf{(15)}$$

Where *sbe* is the eradication success of the biocontrol agents. Note that in our analysis we assume *sbe* to be zero and that biocontrol agents never cause the extinction of the host plant from a site, only a reduction in density.

So the area of weeds being contained by biocontrol over time can be described as:

$$Nb_{t+1} = Nb_t \quad E_{bt} + B_t \qquad \textbf{(16)}$$

In summary, the four portions of our population can be described as:

$$\begin{gathered} Ne_{t+1} = Ne_t + Inv_t \quad S_t \quad ED_t \quad B_{et} \\ Nu_{t+1} = Nu_t \quad D_t + S_t \quad B_{ut} \\ Nk_{t+1} = Nk_t \quad E_t + D_t \quad B_{kt} \\ Nb_{t+1} = Nb_t \quad E_{bt} + B_t \end{gathered}$$

The equations above represent area invaded over time. Damages are the product of area invaded and damages per unit area (calculated in Phase 1). Note that treatments can reduce damages in the model which includes a damage reduction term that is multiplied against the area treated. The area contained by biocontrol agents is also subject to a damage reduction term associated with the average density reduction in the plant that is caused by the biocontrol agents.

Appendix 2: White Lake Ranch Study: Responses of the rangeland plant community to successful biological control

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Introduction

White Lake Ranch in the White Lake Basin (Regional District Okanagan-Similkameen) (49°19.18N, 119°37.82W) was purchased by The Nature Trust of British Columbia in the late 1990s. This purchase included fee simple land, as well as long-term provincial Crown grazing leases and federal grazing leases. When Nature Trust acquired the grazing lands, studies were conducted to obtain baseline information on the plant communities of the pastures and on the distribution of several of the most severe invasive plants. Eco-Matters Consulting was contracted in 2001 to complete this baseline study. From 2001 to 2005 studies of the plant communities in 6 pastures were monitored in a project led by Dr. Pam Krannitz, Canadian Wildlife Service. Here we summarize the findings of these studies in light of the successful biological control programs for diffuse knapweed, *Centaurea diffusa*, and Dalmatian toad flax, *Linaria dalmatica.* We also report an interview with the rancher who uses this site for cattle grazing to determine if these biological control programs have translated into improved rangeland.

Plant Inventories

Diffuse knapweed, was determined to be widely distributed and therefore was not inventoried and mapped in the initial study by Eco-Matters (2001). However, patches of several other invasive plants that were increasing in density, including Dalmatian toadflax, sulphur cinquefoil and hound's-tongue, were sufficiently discrete that they could be mapped. Polygons of the patches of weed distribution were mapped and the distributions of the plants were scored using Luttmerding's (1990) ranking scale from 1 (single plant) to 9 (dense, homogeneous distribution of the target plant). These distribution codes are the standards used today by the Ministry of Forests and Range (http://www.for.gov.bc.ca/hra/Plants/application.htm).

Starting in 2001 and continuing to 2005, the Krannitz study estimated the number and cover of plant species in six pastures of 100-200ha on an annual basis using 3 to 9 haphazardly placed transects of a total of 422 small (20 × 50 cm; $0.1m^2$) permanent plots. The plots were spaced approximately 10 m apart with 50-90 plots per pasture. Percent cover was visually estimated to the nearest 1% of all plant species in each plot. Surveys were conducted in late May to early June of 2001, 2002, 2003 and 2005. In 2004 only plots that had had diffuse knapweed were surveyed. These data have been analyzed by Andrea Stephens (Ph.D. student UBC) to determine changes the species richness in pastures and of the cover of species based on four categories; native grasses, native forbs, introduced grasses and introduced forbs.

Transect Data

The density of diffuse knapweed began to decline in the White Lake Basin in approximately 1999 and continued to be low from 2003 to 2008 in the White Lake Basin (Figure A2.1).

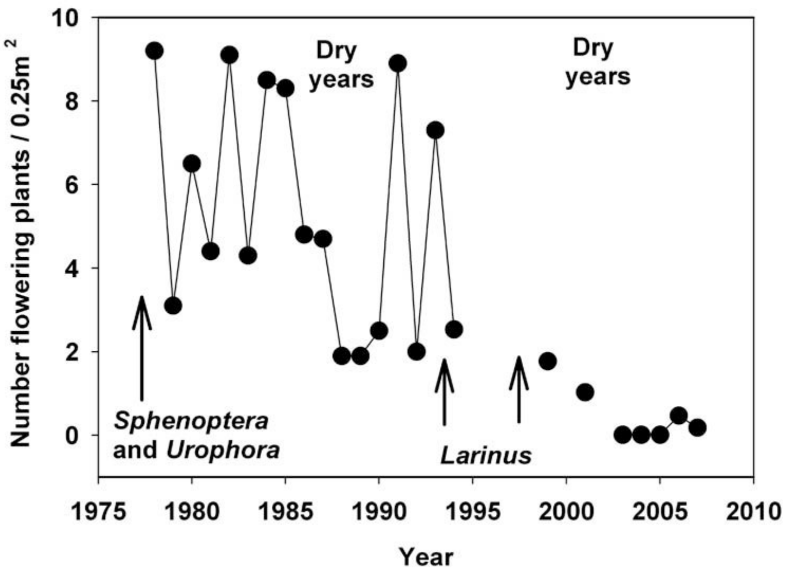

Figure A2.1. Changes in the Density of Flowering diffuse knapweed Plants at the White Lake Observatory.
Arrows indicate when biological control agents were introduced, the root boring beetle Sphenoptera jugoslavica, two gall forming flies, Urophora affinis and U. quadrifasciata, and most recently the seed weevil, Larinus minutus. (Myers et al. in Press).

Data from the White Lake Ranch show that the decline of knapweed from 5% absolute cover in 2001 to almost 0% in 2003-2005 was associated with a 4% increase in the cover of introduced *Bromus* species, annual grasses, and an approximate 7% increase in cover of native grass species, mostly perennial bunch grasses) (Figures A2.2 and A2.3).

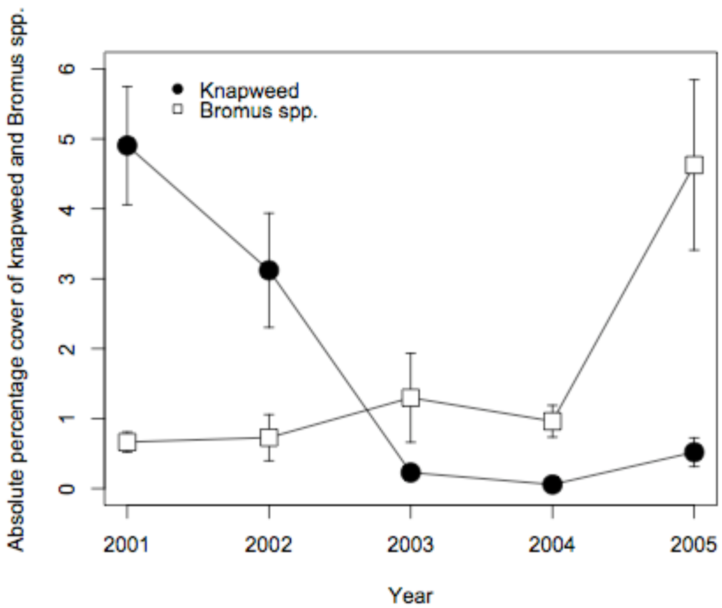

Figure A2.2. Change in Average Percent Cover of diffuse knapweed and Bromus sp. for Six Pastures at the White Lake Ranch.
Bars are standard error.

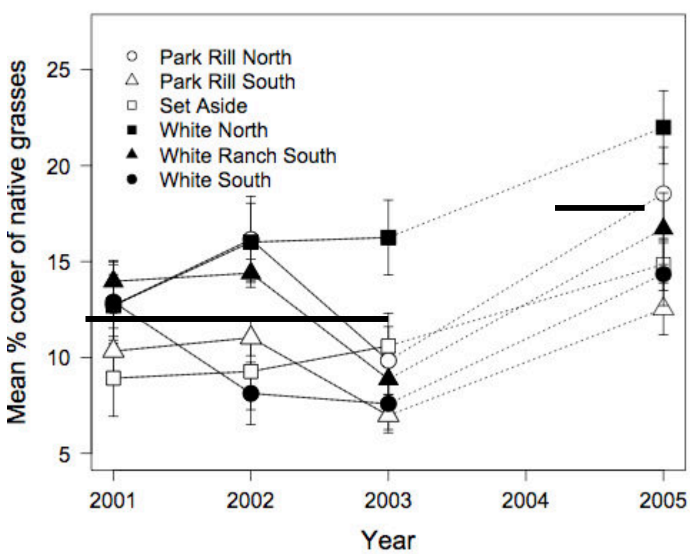

Figure A2.3. Mean Percent Cover (se) of Native Grasses in Six Pastures at the White Lake Ranch Following the Decline of diffuse knapweed.
Horizontal lines are approximate mean values for 2001 -2003, and 2005.

These data could be interpreted to indicate a 7% increase in forage yield from native grasses following the removal of knapweed.

As can be seen in Figure A2.4, although native grasses increased in absolute cover following knapweed decline, they did not increase in relative cover compared to introduced grasses. Neither introduced nor native forbs changed with the reduction in knapweed. This is encouraging as it indicates that introduced forbs are not replacing the knapweed in the White Lake basin.

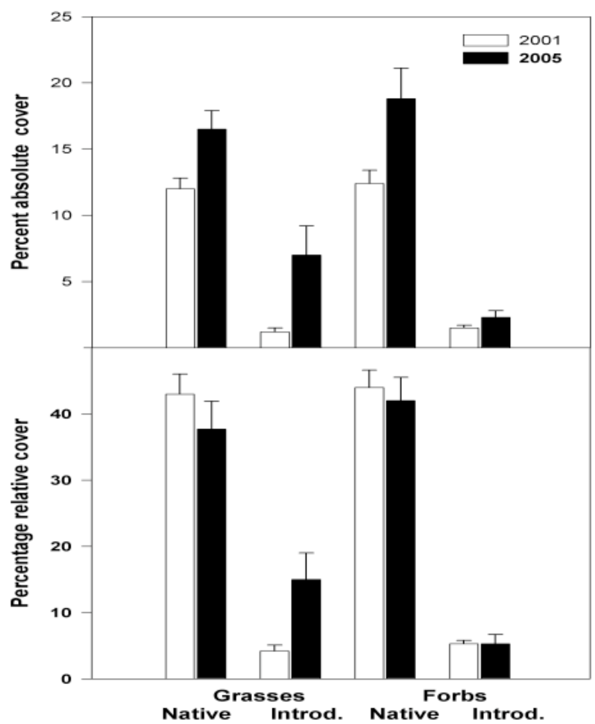

Figure A2.4. Average Change in Absolute and Relative Cover of Native and Introduced Grasses and Forbs for Six Ranges in the White Lake Ranch.

Polygon data – Dalmatian Toadflax

On 16 and 17 September 2008 Lisa Scott and Judy Myers visited the Parker Mountain and Parker North pastures on the White Lake Ranch where the distribution of Dalmatian Toadflax had been mapped in 2001. With the aid of GIS coordinates and orthographic photos, the mapped polygons were located and the density and distribution assessed according to the scale used in the initial study. During the field assessment it was also determined that it should be possible to remap the size of patches using a GPS, although this was not trialed during the present study. Because of time constraints only six easily accessible patches were revisited with the following results (note: site numbers correspond to those indicated in the Eco-Matters 2001 report):

Site #105
- in 2001, this site was coded as DT -7 and covered an area of ~ 600 m^2
- in 2008, 3 small plants were observed which could potentially be one plant

Site #107

- in 2001, this site was coded as DT-9 and covered an area of ~ 900 m^2
- in 2008, this site had 2 distinct sites with a total coverage of 200 - 300 m^2 and a much lower distribution code

These observations indicate a marked decline in the density of Dalmatian Toadflax associated with the release of the biological control agent, *Mecinus janthinus*, and possibly *Rhinusa* spp. Both species of biocontrol agents were detected on toadflax plants at White Lake during the 2008 field season (L. Scott, pers. obs.).

Rancher Interview

On 17 September 2008, Myers and Scott interviewed Wade Clifton whose family run cattle on the White Lake Ranch. Specifically we were interested in the question of whether the reduction in weeds associated with the biological control of diffuse knapweed and Dalmatian toadflax could be translated into changes in the number of cattle grazed on the ranch. Clifton's view was that too many other changes such as variation in rainfall from year to year, made it difficult to assess the impact of weed reduction. He felt that prior to biological control, Dalmatian toadflax was increasing in distribution by approximately 20% a year. He was very enthusiastic about this biological control program. When asked about the value of *Bromus* he felt that it was beneficial for early spring grazing but that the dry seeds in the late summer could be detrimental to the cows if they became stuck in their mouths. He mentioned that he hoped to increase crested wheatgrass in the Parker Mountain pasture by grazing in the autumn so that cows would knock the seeds down and trample them into the soil for germination in the spring.

Of major concern to Clifton at the moment were packs of dogs that were killing his cows. It is clear that invasive weeds are only a small element of the many impacts such as drought, changing fuel prices, the strong Canadian dollar, and other mortality factors that determine the economic viability of ranching. Thus measuring the economic value of what might seem to be increased forage from *Bromus* sp. and native grasses following successful biological control of Dalmatian toadflax and diffuse knapweed is difficult to impossible.

A subsequent interview with Anne Skinner, agrologist with the Ministry of Forests and Range, indicated the grazing levels have not increase since the decline in knapweed.

Future work:

To understand the prolonged impact of an invasive plant, it is necessary to know how the habitat will respond when the weed declines. Following the decline of diffuse knapweed in the White Lake Ranch area over five years, a small increase in the cover of native bunch grass and introduced cheat grass occurred, but this was not sufficient to be translated into an apparent increase in usable forage. Myers and Berube (1983) showed that a knapweed density of approximately 15 flowering plants / m^2 reduced the growth of grass in the summer by about 66% and biomass of grass by 50%. Although we have no estimates of the forage available on the White Lake Ranch before the invasion of diffuse knapweed, Harris and Cranston (1979) estimated that knapweed caused a 43% reduction in forage availability in invaded rangelands.

By converting this to a number of animal units per month they calculated the loss of 0.22 AUM per ha in knapweed invaded rangelands. It would be interesting to know what the forage production and AUM per ha for the White Lake Ranch is.

Several hypotheses for the lack of a strong recovery of grasslands following knapweed reduction can be envisioned. To understand how successful the biological control program for diffuse knapweed actually was in terms of improving rangelands, and to make recommendations for rangeland restoration, several hypotheses should be tested.

1. The forage available in the White Lake Pastures is similar following the decline of knapweed to that measured or estimated by Harris and Cranston 1979 for knapweed infested pastures and forage has not recovered following knapweed reduction.

Forage availability should be estimated for the 6 White Lake Pastures evaluated for the plant community study (Stephens, Myers and Krannitz. In review) taking into consideration the estimates of the amount of diffuse knapweed measured in 2001 in this location. If estimates of forage production are the same as that reported in Cranston and Harris, and if they are the same regardless of knapweed density in 2001, the reduction in knapweed achieved by biological control would not appear to be successful in achieving an improvement in range conditions.

2. Rangelands in BC have not recovered following the reduction of diffuse knapweed because the long-term grazing pressures have resulted in an overall deterioration of rangeland quality.

Over the years exclosures have been established to prevent grazing in small areas of rangeland. To determine if grazing pressure has reduced the resilience of rangelands such that they do not respond to the reduction of diffuse knapweed, these long-term exclosures in grassland areas in BC should be used to determine if the species and forage production inside and outside of the exclosures are similar. If long-term data sets can be found and analyzed, hypotheses about the impacts of knapweed infestation and grazing can be tested.

3. Knapweed infestation causes an overall deterioration in the rangeland through the depletion of soil nutrients, and thus rangelands are slow to recover following the decline in knapweed.

To test this hypothesis soil samples should be obtained from pastures with a history of more or less knapweed to determine if they differ in major nutrients; nitrogen, phosphorous, potassium, or other potentially important elements. In addition, comparisons can also be made to areas of the White Lake Ranch where diffuse knapweed is still dense because of heavy cattle grazing in the vicinity of water sources.

4. Interactions between perennial bunchgrasses and the annual *Bromus* spp. determine the invasibility of *Bromus* spp. into rangelands following knapweed control.

Cover of both native and introduced grasses increased in the White Lake Ranch between 2001 and 2005. Of particular interest is crested wheatgrass, an introduced perennial grass that was only common in one pasture where bluebunch wheatgrass was also relatively common and *Bromus* spp. less common than in the other pastures. This suggests that crested wheatgrass might be particularly effective at reducing the invasion of *Bromus* spp. An experimental analysis of the interactions among these three species would show if it were advantageous to promote reseeding of the bunchgrasses as a management strategy in rangeland areas. In addition a more detailed analysis of the Krannitz data set and other data collected by contractors hired by The Nature Trust could be done to look for the relationship between bunchgrasses and *B. tectorum* or other *Bromus* species in pastures on the White Lake Ranch.

5. The frequency of grassland fires in the Okanagan Valley has increased with the increased invasion of *Bromus tectorum* following the decline of diffuse knapweed and other invasive plants due to biological control.

Bromus tectorum is reported to increase the frequency of rangeland fires in the Great Basin of North America and in this way it is considered to be an environmental engineer. If this is the case in BC, the decline of diffuse knapweed that has been associated with the increase of cheatgrass might be considered to present a new threat. To test if cheatgrass is associated with increased frequency of fires in the Okanagan Valley, fire start records could be analyzed to determine if fires have been more likely to become established in areas with high *B. tectorum* density than areas with little *B. tectorum*.

6. Biological control of Dalmatian toadflax and hounds tongue have reduced the densities of these two species while sulphur cinquefoil, that has no biological control agents, has expanded.

Remeasurement of the polygons of these three species originally plotted on the White Lake Ranch in 2001 would provide evidence for the success of the biological control programs and evaluation of the potential of sulphur cinquefoil to continue to spread.

Budget

The research outlined above could be the basis of a Ph.D. project if it were properly developed.

Student stipend - $27,000 per year for 5 years	$135,000
Summer assistant $10,000 per year for 5 years	$ 50,000
Research expenses – travel, materials etc.	$ 25,000
Total for 5 years	$210,000

Remonitoring of the patches of Dalmatian toadflax, hounds tongue and sulphur cinquefoil

Contractor – 14 days at $500/day	$ 7,000

Citations

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