Exotic Plant Species in the Mixedwood Section of the Southern Boreal Forest of Saskatchewan

A Thesis Submitted to the College of

Graduate Studies and Research

in Partial Fulfilment of the Requirements

for the Degree of Master of Science

in the Department of Geography

University of Saskatchewan

Saskatoon

By

Wade H. Sumners

PERMISSION TO USE

In presenting this thesis in partial fulfilment of the requirements for a Postgraduate degree from the University of Saskatchewan, I agree that the Libraries of this University may make it freely available for inspection. I further agree that permission for copying of this thesis in any manner, in whole or in part, for scholarly purposes may be granted by the professor or professors who supervised my thesis work or, in their absence, by the Head of the Department or the Dean of the College in which my thesis work was done. It is understood that any copying or publication or use of this thesis or parts thereof for financial gain shall not be allowed without my written permission. It is also understood that due recognition shall be given to me and to the University of Saskatchewan in any scholarly use which may be made of any material in my thesis.

Requests for permission to copy or to make other use of material in this thesis in whole or part should be addressed to:

Head of the Department of Geography
9 Campus Drive
University of Saskatchewan
Saskatoon, Saskatchewan
S7N 5A5

ABSTRACT

The objective of this study was to examine the distribution of exotic plants and determine the potential threats in the mixedwood section of the boreal forest. The invasion of exotic plants into natural areas is a growing concern among ecologists. Exotic species have no previous exposure to the invaded area and have been introduced, either intentionally or accidentally, by humans. These plants have the potential to suppress surrounding vegetation and acquire the majority of available resources. This dominance alters important ecosystem functions and negatively affects ecosystem structure and composition.

This study examined three types of land use (roadside right-of-way maintenance, timber harvesting and wildfire) to identify the density, frequency and cover of exotic plants within the mixedwood forest. There were also separate categories of the time since disturbance (re-current, recent and mature) for each disturbance type.

Data were collected in the summer field seasons of 2000 and 2001 in and nearby the Prince Albert Model Forest (approximately 70 km north of Prince Albert, Saskatchewan). Surveying was completed in mature forest, harvested and wildfire areas using 10 x 10 m quadrats. These quadrats were adjacent to or remote from roadside right-of-ways that were deliberately seeded with exotic species. Seeding the right-of-ways with exotic species occurred along principal and secondary highways. Surveying was also conducted within roadside right-of-ways using 1 x 1 m quadrats. These quadrats were adjacent to the recently disturbed and mature quadrats surveyed in the previous year. Each plant species observed in the quadrats had a cover value assigned, while stem counts were also conducted for exotic species.

A total of 23 exotic species were observed within the quadrats. The exotic herb species belong to the Gramineae (9 species), Leguminosae (7), Compositae (5) families

with one species each in Plantaginaceae and Boraginaceae. No exotic trees or shrubs were observed within the study sites.

Areas that were recently disturbed either by timber harvesting or wildfire had 6 different exotic species with an average density of 0.2 ± 0.1 stems/m^2 and a frequency of 72 %. Similar exotic frequencies and species in both recently harvested and burned survey sites suggests that these disturbances have a comparable affect on exotic distributions. Exotic species capable of wind dispersal had the highest frequencies in the recently disturbed survey sites. Common dandelion (*Taraxacum officinale*) had a frequency of 57 %, perennial sow thistle (*Sonchus arvensis*) was observed in 38 % of the sites and annual hawksbeard (*Crepis tectorum*) was at 25 %. Mature forest had a lower population of exotic plants, with only 2 exotic species observed. *Taraxacum officinale* and Canada bluegrass (*Poa compressa*) were observed in 13 % of the mature quadrats with an average density of 0.002 ± 0.002 stems/m^2. The right-of-way quadrats contained the highest amount of exotics with 22 observed species. The average density of exotic species in roadside right-of-ways was 117 ± 22 stems/m^2 with 94 % of the quadrats containing at least one exotic plant. The deliberate introduction, frequency of disturbance and the physical environment of roadside right-of-ways appear to influence the distribution of exotic plant species. The most frequently observed exotic species in the right-of-way areas were *Taraxacum officinale* (observed in 73 % of the quadrats at 8 stems/m^2) followed by alsike clover (*Trifolium hybridum* at 45 % and 17 stems/m^2), *Sonchus arvensis* (43 % and 4 stems/m^2), creeping red fescue (*Festuca rubra* at 36 % and 31 stems/m^2) and smooth brome grass (*Bromus inermis* at 31 % and 17 stems/m^2). These species are either common in urban areas, agricultural weeds or have been deliberately seeded into right-of-way areas. The distribution of exotic species suggests that land use contributes to increased densities and frequencies of exotic plants.

Each exotic species observed was ranked according to a system developed by Hiebert and Stubbendieck (1993). The ranking system was used to determine the current and potential threat of exotic plant species to become detrimental to ecosystem structure, composition and function. The ranking identified 14 species that were a lesser threat and easy to control, 8 species that were a lesser threat and hard to control, and one species, *Bromus inermis*, that was ranked as a serious threat and hard to control.

Additional monitoring is required as the species observed in this study may be exhibiting a "lag phase" of population expansion, which typically precedes an exponential increase. Other species (scentless chamomile (*Matricaria perforata*) and caragana (*Caragana arborescens*)) that were not observed in the study area, but are known to occur within the region, are also a concern with respect to future exotic species invasions. Anticipated climatic changes are also expected to increase the distribution of exotic species as changes to environmental attributes will produce a longer growing season and increased plant growth and productivity.

ACKNOWLEDGEMENTS

I would like to thank my committee members, Dr. O.W. Archibold, Dr. D. H. DeBoer, Dr. A.G. Thomas and Dr. R. A Wright, for their assistance and patience during the preparation of this thesis. My field assistant, Diane MacLeod, also deserves some recognition as her desire to learn and hard work aided the surveying. Staff at Weyerhaeuser Canada, Prince Albert National, Prince Albert Model Forest and Saskatchewan Environment also assisted the research presented here, which I am grateful for.

Thank you to my friends and family for your endless support. This endeavour could not have been completed without you. I would like to thank my fellow grad students, Helton, Alexis, Boyd, Shauna and Sharmalene, for making the experience memorable. Special thanks to Helton for his assistance with the maps in the thesis.

Financial support was provided by the Prince Albert Model Forest. Thank you Keith, Ott and the Ecosystem Health Working Group for your comments and assistance.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Chapter 1

Introduction

1.1 Rationale for study

The sustainability of the Saskatchewan boreal forest is a growing concern among forest users interested in the ecological, social and economic benefits of this ecosystem. The development of the Prince Albert Model Forest [PAMF] provides an opportunity for stakeholders to participate in management planning to promote the sustainable use of forest resources. The PAMF is a non-profit partnership of industry, Federal conservation and provincial resource management agencies and First Nations. This organization is a member of Canada's Model Forest Network that is committed to raising awareness about sustainability and integrated resource and ecosystem management among forest users, researchers and managers. Sustainability will ensure that the forest ecosystem is healthy, having an adequate representation of biodiversity and natural processes. A healthy ecosystem will also be resilient to disturbances, maintaining its productive capacity and containing species that are native to the area. Native plant species contribute to the structure and processes that ensure a healthy forest.

The survival of native plants is often threatened by non-native or exotic plants as they commonly have improved abilities to compete for water, light, nutrients, and space (DeFerrari and Naiman, 1994; Royer and Dickinson, 1999). The increased competitiveness of exotic species often permits the dominance of this group of plants. According to Orians (1986) and Forman (1995), exotic plants generally out-compete native plants by exhibiting:

- high reproductive outputs (more resources are allocated to reproduction);
- short life cycles;
- easily dispersed seeds;

- high individual and population growth rates;
- flexible utilization of a variety of environmental resources.

This biological dominance has been observed to reduce native plant diversity and may facilitate persistence of the exotic species (Batianoff and Franks, 1998; DeFerrari and Naiman, 1994; Higgins et al., 1999). For example, European buckthorn (*Rhamnus cathartica)* forms a dense monospecific ground cover in riparian areas in Saskatchewan (Archibold et al., 1997). This exotic shrub is an aggressive competitor and prolific seed producer whose seedlings successfully establish in the shade created by mature buckthorn shrubs. Kentucky bluegrass (*Poa pratensis*) is another vigorous competitor that forms dense mats that retain moisture in arid habitats. These mats typically allow this exotic species to become dominant over native species in Saskatchewan grasslands (Delcan, 1994).

The effects of exotic species in other habitats continue to raise concern about these plants among ecologists. Exotic plants within the United States have been implicated in altering ecosystem properties by affecting biogeochemical conditions and processes. For example, the exotic nitrogen-fixing firetree (*Myrica faya*) increases the overall biological availability of nitrogen in the soil of recently disturbed volcanic sites in Hawaii (Vitousek et al., 1987). These sites naturally have a limited amount of nitrogen and the increased supply allows this exotic species to become dominant in the area. The ice plant (*Mesembryanthemum crystallinum*) has the ability to accumulate salt (NaCl) when it is produced during decomposition in Californian grasslands. The species redistributes the salt onto the soil surface, altering the physical and chemical properties of the soil and interfering with the growth of surrounding competitors (MacDonald et al., 1989; Vitousek, 1986).

Altering ecosystem structure and function may be the most detrimental effects of exotic plants because of the potential to alter disturbance regimes and interfere with successional processes (D'Antonio and Vitousek, 1992; Hobbs and Humphries, 1995; Vitousek, 1990). For example, exotic grasses in submontane zones of Hawaii have provided the fine fuels needed to introduce and promote fire in areas where it was previously rare or absent (Hughs et al., 1991). Exotic grass species are known to recover more rapidly after fire than native plants, causing an increase in the landscape's fire frequency and susceptibility to fire (D'Antonio and Vitousek, 1992; Walker and

Smith, 1997). Changes in these processes and regimes may elevate the number of disturbances and increase the numbers of exotics, as these disturbed areas are often more susceptible to invasion (Kotanen, 1997; Orians, 1986). The elevated presence of exotic species is expected to magnify their influence on ecosystem properties. This raises concern about the group of species because of the wide recognition that exotic plants are generally detrimental to natural environments (D'Antonio and Vitousek, 1992; Hiebert and Stubbendieck, 1993; Hobbs and Humphries, 1995).

The historical introduction of exotic plants into North America involved the incidental arrival of species in the ballasts of ships, within impure crop seed, through adhesion to domestic animals and soil surrounding the roots of nursery stock and various deliberate introductions (i.e. forage, fibre, medicinal plants, ornamentals, erosion control and timber plantations) (Baker, 1986). Once exotic plant species have been introduced to a continent, the distribution can be influenced by several factors including climate, historical land use, frequency of transportation, communication and power corridors, diversity of the habitat, frequency of lakes and rivers and the historical movement of settlers (Haber, 1997).

The rate of introduction is often accelerated by human activity (Ewel et al., 1999; Westman, 1990). Deliberate use of exotic plants for landscaping and agriculture in Saskatchewan may explain the increase. Studies by McKinney (2001) and Pysek (1998b) indicate that exotic plants have elevated densities in areas that are heavily populated by humans. Once the species are established in these areas, dispersal into forested areas may follow the extensive road network, as roads often promote the dispersal of exotic species (Holl et al., 2000; Lugo and Gucinski, 2000; Parendes and Jones, 2000; Trombulak and Frissell, 2000). If the exotic species disperse into a suitable habitat, the species may establish permanently.

1.2 Objectives and Hypotheses

The objective of this study is to examine exotic plant distributions and determine the potential threats in mixedwood forest in the Prince Albert Model Forest region. This region is located in the boreal plain ecozone of the southern boreal forest of Saskatchewan. To fulfil the study objective, an inventory of exotic species in areas that are prone to the establishment of exotic species was conducted.

Surveys were carried out in roadside right-of-ways seeded with exotic plant species. A comparison of right-of-ways that had exotic species present in seed mixtures applied after road construction with right-of-ways that had not received the exotic species will identify the impact of deliberate introductions. The surveying of roadside right-of-ways was useful in determining the source of exotic plant dispersal into disturbed areas. Examining recent disturbances that were adjacent to or remote from roadside right-of-ways that were seeded with exotic species was expected to identify how the proximity to a deliberate introduction influences exotic distribution. Harvested and wildfire areas were compared to test the response of exotic species to natural and anthropogenic disturbances. A comparison of recently disturbed and mature forest stands was also completed to assess the ability of exotic species to persist throughout the successional stages of forest regrowth.

To determine the extent and possible causes of the introduction of exotic species into the mixedwood section of the southern boreal forest, the following research hypotheses were developed:

1. H_o: exotic plant incidence in roadside right-of-ways that were seeded with exotic species = exotic plant incidence in roadside right-of-ways not seeded with exotics,
2. H_o: exotic plant incidence in recently harvested forest ecosystems = exotic plant incidence in recently burned forest ecosystems,
3. H_o: exotic plant incidence in recently disturbed forest ecosystems adjacent to roadside right-of-ways deliberately seeded with exotic species = exotic plant incidence in recently disturbed forest ecosystems that are adjacent to roadside right-of-ways not seeded with exotic species,
4. H_o: exotic plant incidence in recently disturbed forest ecosystems (within 15 years since last major disturbance) = exotic plant incidence in mid-successional forest ecosystems (60 to 80 years since last major disturbance).

The study focused on aspects of the distribution, ecology and management of exotic plants and was divided into the following sections: (1) A literature review discussing the policies and issues related to exotic plant management, the principles of biological invasions and the dynamics of disturbance in the boreal forest; (2) A description of the study area and methods used to describe and analyze the vegetation surveyed; (3) A presentation of the results and analyses including exotic frequency, density, and cover data, a ranking system for exotic species and site attributes; (4) A discussion of the results; and (5) Conclusions regarding the distribution and potential threats of exotic plant species to the boreal forest ecosystem and recommendations for future management.

Chapter 2

Literature Review

2.1 Exotic Species Policy

2.1.1 Definitions

An exotic plant is a non-native species that has entered an ecosystem by a deliberate or accidental introduction by humans (Environment Canada, 2001; Richardson et al., 2000; Pysek, 1998b; Webb, 1985). Synonyms for exotic species include non-native, alien, non-indigenous, and introduced species. Other definitions describe exotic plants as species that occur outside natural ranges (Canadian Forest Service, 1999; Clinton, 1999) and before European contact (16^{th} Century) in North America (Schwartz, 1997). This particular time in history is biologically important because it marks the beginning of a considerable increase in the rate of species transportation by humans throughout North America. Although the inclusion of European settlement offers a stronger definition, this definition is restricted to use in North America and may thus have limited utility. Literature describing exotic species occurring beyond a natural range also has some limitations, as the definition often does not include a description of the range boundaries. Confirmation of exotic plant status involves several criteria including fossil evidence, historical documentation of an introduction, habitat, geographic distribution, frequency of known naturalization, and genetic diversity within the species (Webb, 1985).

Other terms that are important and often used in literature examining exotic species include 'naturalization' and 'invasive'. Richardson et al. (2000) has extensively examined definitions of these terms and concluded that naturalization is the process in which an exotic plant species reproduces consistently over many life cycles without direct human intervention. For example, the common dandelion (*Taraxacum officinale*) is often considered naturalized because of its regular occurrence in several habitats. This species is able to produce up to 200 seeds per flower head and complete an entire blooming process (flower stalk growth, bud development, flower emergence) within 48 hours (Anderson, W.P., 1999). Invasive plants are species that often produce large numbers of offspring at a considerable distance (> 100 m) from parent plants (Richardson et al., 2000). These species also reproduce regularly over numerous life cycles without direct human involvement. Invasive plants can be either native or exotic and have often been defined or referred to as organisms that have harmful ecosystem effects (Archibold et al., 1997; Clinton, 1999; Species Survival Commission, 2000). After an extensive literature review by Richardson et al. (2000) it was determined that only 50 to 80% of the plants described as invasive had harmful impacts (i.e. displacing native species, altering disturbance regime) in the habitat in which they occur. The remaining species are generally "benign invaders whose environmental or economic impacts are beyond any practical detection limits". Deliberations by other authors (Espie, 2001; Mosquin, 1997) have acknowledged this by not including damaging impacts in definitions of invasive plants. The invasive attribute of exotic species remains a concern because of the increased competitive abilities of exotic plants. The species that posses this attribute have the potential to disperse long distances and establish populations that may or may not have harmful ecosystem effects.

Another term that may be confused with exotic plant is noxious weed. The term noxious refers to the plant's ability to damage agricultural crops or animal health (Royer and Dickinson, 1999). Noxious plants are often designated by legislation to control their spread and damaging effects. Although in Canada most weedy plants are exotic, weeds are not necessarily exotic species. Weeds are species that grow in areas where they are unwanted and often have detectable economic or environmental effects.

2.1.2 Projects Examining Exotic Species

The introduction of exotic plants has received much international attention. The establishment of the Scientific Committee on the Problems of the Environment [SCOPE] by the International Council for Science [ICSU] in 1969, indicated the need to assemble, review and assess information available on man-made environmental changes. The SCOPE programme has examined the ecology of invasive plants, animals, and microbes, focussing on species that have successfully invaded non-agricultural regions and disrupted natural ecosystem processes since the mid-1980s (Usher et al., 1988).

In 1997, the SCOPE programme collaborated with its extensive group of partner organizations to develop the Global Invasive Species Programme [GISP]. In phase I of the GISP programme several experts were gathered to provide a knowledge base and new tools to address specific issues of invasive exotic species via published reports, international meetings and the construction of an international database. The issues include the ecology of invasive species, legal and institutional frameworks, assessment and best management practises, economic consequences and education. The GISP programme is currently in Phase II and is intended to foster collaboration among scientists to address key problem issues, catalyze and assist the establishment of national invasive species strategies, programmes and local initiatives. The GISP programme is an ambitious effort to build an international network of leading scientists and programmes to develop a global strategy to deal with invasive exotic species. Several partner organizations, including the World Conservation Union [IUCN], Diversitas, the Commonwealth Bureau of Agriculture [CAB International] and the United Nation's Convention on Biological Diversity [CBD] have collaborated with 42 governments, 17 intergovernmental institutions and 17 national and non-governmental organizations to establish priorities for the GISP programme (GISP, 2002).

The Invasive Species Specialist Group [ISSG] was developed by the IUCN to "reduce threats to natural ecosystems and the native species they contain by increasing awareness of invasions, and of ways to prevent, control, or eradicate them" (IUCN, 1999). This group of specialists consists of 146 scientific and policy experts on invasive species from 41 countries and has recommended that the effects of invasive plants and animals, a major cause of native species depletion, must be reduced. Several databases and communication groups in Australia, the United Kingdom, New Zealand, the United States, South Africa and Canada were also established in the early 1990s. They all have compiled lists and descriptions of exotic and invasive plants in their given countries in an ongoing attempt to document exotic plant numbers and distributions. The creation and development of the extensive international networks to educate and promote action among the scientific community acknowledges the need for exotic species research.

Initiatives within North America include an executive order issued in 1999 by the president of the United States, Bill Clinton, to “prevent the introduction of invasive species and provide for their control and to minimize the economic, ecological, and human health impacts that invasive species cause”. This action by the American government was motivated by more than 500 scientists and land and resource managers from the U.S. who wrote to the government expressing their deep concern about invasive species (National Invasive Species Council, 2001a). The order specifically instructed federal agencies to monitor invasive species populations, restore native habitats that have been invaded, conduct research on invasive species, develop technologies for control and promote public education on invasive species. The implementation of the order was overseen by the National Invasive Species Council. The council consists of several high ranking officials in the American government including the Secretary of State, Secretary of the Treasury, Secretary of Defence and the Secretary of Commerce Co-chair. Other participants of the council include the Secretary of Agriculture, the Secretary of Transportation and an administrator from the Environmental Protection Agency. The council is instructed to provide national leadership, coordinate efforts at local, state and ecosystem levels, provide international cooperation and develop a web-based information network (National Invasive Species Council, 2001b).

In 1995, the government of Canada addressed the increasing public concern about exotic species by including the issue in the development of the Canadian Biodiversity Strategy. The strategy contains objectives to implement sustainable development and conserve biodiversity with methods to prevent the introduction of detrimental exotics and eliminate or reduce their adverse effects (Environment Canada, 2001). The establishment of the Invasive Plants of Canada Project [IPCAN] by the federal government in 1995 provided several tools to monitor the impacts of invasive plants. The development of an outreach program, Invasive Plant Alert, was undertaken to promote monitoring and control activities by naturalist clubs across Canada. The project has geo-referenced historic collections, compiled recent sight records, summarized a new national survey method and produced a series of fact sheets on invasive plants. Other federal agencies that have addressed exotic species include the Canadian Park Service and the Canadian Forest Service. Literature produced by these agencies involves a series of reports (Achuff et al., 1990; Canadian Forest Service, 1999; Mosquin, 1997) that discuss the current status of exotic species, potential problem pests (plants and animals), management alternatives and guidelines for management. Both agencies recognize the harmful effects of exotic species, the need to monitor and prevent the entry of exotics and subsequent removal when necessary, but no specific directives have been written.

The issue of exotic plant species has also been gaining attention in Saskatchewan. Research and management activities occurring in grassland ecosystems include the identification, mapping and control of several invasive species by the Meewasin Valley Authority. These species include Eurasian buckthorn (*Rhamnus cathartica*), smooth brome (*Bromus inermis*) and leafy spurge (*Euphorbia esula*) in the Saskatoon area. The Saskatchewan Purple Loosestrife Eradication Project was established in 1996 to eradicate purple loosestrife (*Lythrum salicaria*) and educate the public about its ability to spread into native wetlands. Purple loosestrife is known to invade and destroy wetlands, replacing all native vegetation and disrupting ecological and hydrological processes (Ducks Unlimited, 1994). These projects and programmes in Saskatchewan often operate with several partner organizations that implement conservation initiatives. The Prince Albert Model Forest [PAMF] has also expressed interested in the effects of exotic species. Proposed actions of the PAMF include the identification and control of non-native species, and the restoration of impacted areas, if needed, in Saskatchewan forests (Prince Albert Model Forest, 2000). The efforts of the PAMF are important because of the involvement of industry, local communities, First Nations, Federal and Provincial resource management agencies. Other activities in the province include efforts by Saskatchewan Environment to document exotic species in the East Boreal Ecoregion (Kosowan and Yungwirth, 1999; Smith and Ulmer, 2001) and review the problem for the entire province (Espie, 2001). The report by Espie (2001) has made several recommendations regarding public education and management of exotic species including the need for monitoring and research initiatives, cooperation among stakeholders and an expansion of legislation to reduce the problems with exotic species. The Saskatchewan division of Weyerhaeuser Canada has addressed invasive non-native species in their 20-year management plan. The company uses native species to re-plant harvested areas and has conducted small-scale trials involving non-native species in the past. They intend to work with Saskatchewan Environment to "develop a suitable and effective vegetation mixture consisting of native and non-invasive plant species" to re-vegetate roadside right-of-ways (Golder Associates, 1999).

2.1.3 Legislation

Legislation to control exotic plants has been written for both provincial and federal jurisdictions in Canada. A review of federal and Saskatchewan legislation is provided by Espie (2001) whereas Achuff et al. (1990) present federal, provincial and state legislation from British Columbia, Alberta and Montana. Federal legislation is generally limited to the prevention of the introduction of exotic species while provincial legislation focuses on the control of species on a regional scale. Both types of legislation contribute to the limitation of harmful exotic species within the PAMF and permit the instigation of additional control measures, if necessary.

The Wild Animal and Plant Protection and Regulation of International and Interprovincial Trade Act [WAPPRIITA] protects Canada from the introduction of exotic species that may be harmful to indigenous species (Government of Canada, 1996). After negotiating a Memorandum of Understanding in 1997, Saskatchewan is obligated to enforce WAPPRIITA for the interprovincial transport of exotic species that are harmful to Canadian ecosystems. The draft National Policy on the Introductions and Transfers of Aquatic Organisms by the Department of Fisheries and Oceans focuses on minimizing the negative impacts of deliberate introductions of exotic species and the transfer outside their normal ranges (Government of Canada, 2000). It involves a risk assessment process where species introductions or transfers are only permitted when the potential risk is clearly determined to be low or minimal. The policy is applicable to all live aquatic organisms including fresh water plants which provide habitat for fish. The Seeds Act focuses on seed quality for agricultural activities and ensures the reduction of spread for species that are designed as noxious under the Act (Government of Canada, 1985). The Plant Protection Act provides a means to protect plant life by preventing the importation, exportation and spread of injurious pest species (Government of Canada, 1990). This Act focuses on the agricultural and forestry sectors and allows officials to prohibit or restrict movement into areas that are infested. It also provides financial or technical assistance for the control and eradication of the pest. Federal legislation is a useful tool to prevent the spread of harmful exotics species but appears to have little impact on established populations (Espie, 2001).

Provincial legislation provides methods to control exotic species on a regional scale and allows for early detection of exotic populations. In Saskatchewan, the Noxious Weeds Act provides a list of species, predominately exotic, for provincial weed inspectors to investigate and control, if needed (Government of Saskatchewan, 1984). The legislation identifies a means to delegate powers to weed inspectors and assign offences and penalties if municipalities, landowners or occupants do not comply with the Act. Similar lists of noxious plants that are detrimental to agriculture and animal health have been compiled for each province. These lists highlight species that are a problem in each particular region. The Forest Resources Management Act also recognizes the need to limit the introduction of noxious weeds or any exotic plant as designated by the minister (Government of Saskatchewan, 1996). The Act does not entirely restrict the use of exotic species permitting research involving the introduction and propagation of exotic plants on provincial forest land.

2.2 Invasion Theory

A biological invasion is the "first time" exposure of any organism to an area not previously occupied by that species. The species must be able to disperse to the new area and establish a population that can reproduce successfully. It is a common belief among ecologists that plant invasions are a natural biological process that have always been part of evolutionary history (Huenneke, 1997; Vitousek et al., 1996). Once the initial introduction is accidentally or deliberately caused by humans the species becomes exotic in its new habitat.

Successful invasion presents a complex series of obstacles to the invading species. Johnstone (1986) provides a brief explanation of the possible causes of plant invasions and comments about the shortcomings of each hypothesis (Table 2.1). A discussion by Mack et al. (2000) provides an additional review of these well known theories in an attempt to identify future invaders and vulnerable communities. The theories in Table 2.1 are attempting to tackle an issue that is increasingly complex and several exceptions have been identified in subsequent literature (Blaney and Kotanen 2001; Levine and D'Antonio, 1999; Mack, 1996; Rejmanek and Richardson, 1996;

Stohlgren et al., 1999; Wiser et al., 1998). The theories also refer to specific ecological situations and Davis et al. (2000) reminds the reader that the invasion process is not a static or permanent attribute, but a condition that can fluctuate over time. Rejmanek (1996) adds that generalizations about biological invasions are mostly speculative because the number of failed introductions are usually unknown.

Table 2.1 A summary of a review by Johnstone (1986) of concepts relating to plant invasions.

Theory name	Explanation	Described further by:
Simple Community Hypothesis	Rapid invasion occurs where ecological diversity is low	Elton 1958
Freedom from Predators Hypothesis	Invasion occurs in the absence of a species' natural predator	Harper 1969, 1977
Superior Reproductive Hypothesis	The invading species has a greater reproductive potential	Sculthorpe 1967
Poorly-adapted (native) Species Hypothesis	Invasions are possible when resident species are not "well adapted"	Griggs 1940
Chemical Change Hypothesis	Invasion occurs after the chemical characteristics of a habitat have been altered	Pearsall 1918, 1921
Disturbance Generated Gaps Hypothesis	Competitor-free gaps in existing vegetation provides for an invasion	Watt 1947
Distance/Area/Time Controlling Success Hypothesis	An invasion is a function of distance, area, or time	Carlquist 1965
Empty Niche Hypothesis	An unfilled niche is waiting to be stocked by an appropriate species	Doing 1985

Plant invasions are currently receiving more attention because of the increased rates of invasion (Lodge, 1993; Usher, 1988). Recent introductions of exotic plants have been facilitated by the ease and expansion of international travel and commerce (Huenneke, 1997; Mack. 1996; Mooney and Cleland, 2001). With increased transportation of people and goods, the potential for an introduction of a plant species into a foreign land increases. Mack et al. (2000) states that recent increases in the rate of introductions have undoubtedly exceeded the extent, frequency and impact of the movements of organisms by any natural forces in any 500 year period in Earth's history.

Once a plant species migrates into an area, successful establishment is not guaranteed. According to Johnstone (1986) there are several barriers to a plant invasion:

- botanical barrier- which is caused by the presence of a plant;
- non-botanical barrier- an abiotic or biotic hazard not caused by the presence of a plant;

- selective barrier- an obstacle to the particular species being excluded;
- non-selective barrier- an impediment to all species that are excluded.

These barriers are based on the type and selectivity of an obstruction that prohibits an invasion. An example of a botanical barrier involves the release of substances by *Agropyron repens* that repels surrounding plants (Mack et al., 2000). A non-botanical barrier could include a series of factors in combination or acting alone. These factors may be the region's climate, disturbance regime, dispersal distance, lack of mutualist species, presence of herbivores and pathogens or community structure. A selective barrier may involve increased amounts of pollution (salinization, heavy metals, dust), water, or fluctuations in temperature a plant is incapable of tolerating while completing its life cycle. Non-selective barriers may include the mixing of soil horizons which occurs during the construction of roadways or a lack of resources (sunlight, water, nutrients) to sustain plant life.

Lonsdale (1999) states that an invasion will be influenced by a species' ability to migrate to an area (natural or anthropogenic), the attributes of the species and the susceptibility of the environment to the establishment of the new species. Invasions may be altered by elements that are apparent (i.e. effective pollinators) or obscured (i.e. multiple-combinations of factors) to the investigator. The invasion process may also exhibit a lag phase in which species population levels remain low for extended periods of time and then rapidly expand at a later date. Mooney and Cleland (2001) describe the behaviour using *Bromus tectorum* as an example while other authors (Booth et al., 2003; Ewel et al., 1999; Mack et al., 2000) provide explanations of the phenomenon. According to Crooks and Soule (1999), the length of a lag phase may depend upon the species' physical and biotic environment and any genetic changes that may occur. Additional reasons for a lag phase have been described by Ewel et al. (1999) and include difficulty of detection, local adaptation, exponential growth, climate change and delayed introductions of mutualists.

2.2.1 Dispersal

Once introduced, a species' ability to effectively disperse its propagules throughout the landscape is essential for its success and subsequent invasion. Mack (1996) observes that species with high migration rates were among the most damaging exotic plants. Exotic species by definition are introduced into habitats by anthropogenic sources but dispersal beyond the initial introduction is determined by the plant's ability to disperse, the habitat and the environmental conditions at the time of dispersal. Work completed by Bazzaz (1996) explains several capabilities of plants to disperse to 'choice habitats'. These dispersal capabilities include relatively wide propagule dispersion, targeted migrations to suitable habitats, dispersal with a supply of some required resource, locating areas where root systems are less congested by neighbours and changing life histories to coincide with resource availability. The 'choice habitats' would provide the species with a supply of resources (i.e., light, water, nutrients) for growth and reproduction and an adequate source of pollinators, dispersers, and symbionts. The area would also be relatively free of herbivores, predators and pathogens, except those that attack their competitors. The conditions during dispersal (i.e. slow or fast winds, upwind or downwind direction, presence of updrafts, squirrel vs. bird dispersal) and the structure of the vegetation in the area can also affect colonization patterns (Willson, 1993). For example, a clearcut forest can have increased wind speeds compared to a wildfire area which may have several standing live or dead trees. Johnson et al. (1998) explains that seed distribution can partly be explained by the mortality patterns within wildfire areas allowing for a relatively short distance to the nearest surviving seed source.

Patterns of dispersal have a major effect upon the rate at which a species will spread. A species which spreads as an advancing front will spread more slowly and can be more easily managed than one which spreads as scattered, isolated infestations (Panetta and Hopkins, 1991). Plants have morphological adaptations (i.e. winged fruits) to promote the dispersal of their propagules. For example, dispersal into a wildfire area is generally characterized by seed that is blown in from adjacent stands that were undisturbed (Archibold, 1979).

Propagules that are able to retain their viability for extended periods of time also increase their chances of establishing during favourable conditions. Longer viabilities will produce "a greater propagule pool and an increased probability of establishing an isolated focus to start an invasion" (Richardson et al., 2000). An example of an exotic species with increased viability and quantities of propagules is white sweet-clover (*Melilotus alba*). This clover is capable of producing up to 350,000 seeds/plant in a single growing season, that have the potential to retain their viability for up to 80 years (Royer and Dickinson, 1999). If the seeds are located in a disturbed area, the type of disturbance may influence the amount of seeds present and ultimately their viability. Archibold (1979) noted that soil seed banks in wildfire areas may be reduced as the organic soil, which contains most of the viable seed, is generally destroyed by severe fire. Although harvesting operations have been shown to alter the distribution of seeds in the soil profile, this rearrangement appears to have little effect on the total number of species present in post-harvest seed banks or understory vegetation (Qi and Scarratt, 1998).

The distances propagules travel to reach suitable sites also affect exotic plant invasions. A larger maximum distance may give an advantage for finding appropriate sites for colonizing new habitats (Green, 1983; Howe and Smallwood, 1982). Willson (1993) states that two ways for a plant to increase the probability of some of its propagules dispersing relatively far are to produce many seeds and use some form of long-distance dispersal mechanism. A study by Archibold (1980) in Saskatchewan mixedwood stands confirms that species which produce large numbers of seeds (i.e. fireweed (*Epilobium angustifolium*), paper birch (*Betula papyrifera*) and trembling aspen (*Populus tremuloides*)) dominate seed input during the first year following a fire. Seed dispersal distance is generally larger for wind-dispersed seeds compared to seeds that are propelled by ballistic methods or seeds that have no obvious dispersal adaptation. The distance wind-dispersed seeds travel is influenced by the height of seed release, the size, shape, and aerodynamic properties of the seed, as well as the speed and turbulence of the wind (Hughs et al., 1994; Willson, 1993).

The potential for propagule movement increases throughout the roadside right-of-way as this narrow corridor permits the movement of species. Roads assist dispersal for exotics in three ways: by providing habitat for an invasion to continue, by stressing or removing native species and by allowing easier movement by natural or human vectors (Parendes and Jones, 2000; Trombulak and Frissell, 2000). Schmidt (1989) has observed that the majority of plants growing alongside roads can have their propagules transported by vehicles. This dispersal of seeds by vehicles is promoted by the connection between exotic plants and humans. Articles by Forman and Alexander (1998) and Forman and Deblinger (2000) examined the major ecological effects of roads and the extent of those effects. They concluded that the effects extend beyond 100 m from the road edge and estimate that 15 to 20% of the United States is ecologically impacted by roads. It is well known in the literature that roads influence habitat beyond their immediate vicinity affecting the surrounding vegetation (Anderson, R., 1999; Angold, 1997; Amor and Stevens, 1975; Motto et al., 1970; Viskari and Karenlampi, 2000). Several authors (Fytianos et al., 1985; Motto et al., 1970; Panetta and Hopkins, 1991) have also noted that the effects of roads diminish as the distance from the road increases.

2.2.2 Species Attributes

Once species propagules reach a suitable area, successful establishment is dependent upon the plants' germination and growth requirements. A well-known list of optimal traits was produced by Baker (1974) describing the "ideal weed". The traits for this plant have since been included in discussions regarding successful plant invasions. In a paper by Mack (1996) it is agreed that the world's most successful invaders do possess many of Baker's traits but no one species has all the features of the "ideal weed". The author continues to state that combinations of traits are unrecognized by such lists, and as barriers for introductions are removed by international commerce, invasions are likely to occur. A conclusion from the article explains that a restricted predictive power exists for lists of traits. Work involving transgenic plant release and predicting invasiveness has prompted Williamson (1993, 1994) to examine the value of Baker's traits. The author had several scientists categorize British plants for weediness and concluded that Baker's traits should not be used to predict a species' ability to become a pest. Lodge (1993) confirmed the general position about the subject by asserting that generalizations about biological invasions and species attributes are often prone to exceptions and a lack of statistical testing. Despite this it is possible to compile a list of widely cited generalizations about 'successful' traits including:

- r-selected species;
- single-parent reproduction;
- phenotypic plasticity;
- human commensalism;
- high dispersal rates;
- vegetative reproduction;
- large native range.

These species attribute lists are common, appearing in much of the literature (see Rejmanek and Richardson, 1996; Orians, 1986; Forman, 1995), despite criticism of the reliability of such lists (Mack, 1996). Work completed by Pysek (1998a) addresses these misgivings while attempting to determine if a taxonomic pattern to plant invasions exists. Criticisms (regarding unreliability of available data, misidentification of species, and difficulties assessing immigration status) are provided about the exercise from peers and himself, while reviewing plant families throughout the world. A conclusion by the author regarding the largest families (Gramineae, Compositae, Leguminosae, and Cruciferae) contributing the most to the total number of exotic species of local floras concurs with an examination by Mack et al. (2000). These authors suggest that the attributes that have allowed these families to expand within their native ranges have also permitted the spread of exotic individuals throughout the world. These attributes vary among the largest families and include a highly evolved inflorescence, high reproductive rates, an ability to fix nitrogen and specialized dispersal structures. In a review of seed plant invasiveness, Rejmanek (1996) discusses generalizations about species attributes and suggests that exotic species belonging to exotic genera are more likely to flourish than exotics from genera represented in the native flora.

The attributes of successful exotic species are influenced by the native environment of the species. Reiners (1983) maintained that the European origin of the majority of weeds in the USA is a result of the longer evolution of disturbance-adapted species in Europe. Work completed by Pysek (1998a) and di Castri (1989) support this claim as European species tend to have an enhanced invasive potential. The occurrence of humans and thus exotics in North America can be thought as relatively recent, compared to the years of intensive land use in Europe. The work by Reiners (1983) continues to explain that "the more frequent the disturbance, the stronger the representation of better adapted species and theoretically, the stronger the selection pressure for adaptive traits". Huenneke (1997) adds that forests dominated by early successional species often have faster invasions than species migrating into a "closed" forest of late-successional species. Exotic species will thus have an advantage as the forest and its environment in Saskatchewan are linked in an irregular "pulse" strategy of alternating disturbance and regrowth (Dix and Swan, 1971).

Once an invading species has entered a new environment, the traits that allowed the species to persist in its native habitat are often enhanced. In an article examining the evolutionary impact of invasive species, Mooney and Cleland (2001) provide evidence that in populations of invasive species the individuals are often larger in their new territory than in their native land. Blaney and Kotanen (2001) describe a similar situation where larger seeds banks of *Mimosa pigra* and various *Acacia* species were found in non-native habitats in Australia and South Africa. These new traits are a result of the species reacting to different environmental attributes.

Climate change is also expected to influence the growth characteristics of plant species and accelerate species invasions (Davis et al., 2000). The anticipated climatic changes are a result of predictions by various computer models as atmospheric carbon dioxide doubles. A review by Wheaton (1997) stated that the elevated amount of carbon dioxide is expected to increase the mean annual temperature producing a longer growing season and an increase in plant growth and productivity. A summary of the general assertion among models in the mid-latitude Northern Hemisphere includes:

- night-time temperatures rising faster than day-time temperatures;
- soil moisture increasing in the winter and decreasing in summer;
- precipitation increasing in the winter;
- snow cover area and duration decreasing;
- elevated ground temperatures.

According to Davis et al. (2000), the changes in water availability are likely to promote more efficient water use by resident vegetation. This may result in an initial increase in the availability of water and favour invasions by certain species. A northward shift in vegetation is expected in the Northern Hemisphere in response to the increased temperatures (Bazzaz, 1996; Singh and Wheaton, 1991; Stocks, 1993). Singh and Wheaton (1991) have noted that when plant communities change, the introduction of species from warmer regions may create unanticipated problems. Huenneke (1997) contends that species with good dispersal abilities will be favoured when climate change alters habitats. It has also been suggested that species with greater abilities to shift resources to root systems will increase their total growth and plants with a “high evolutionary potential” (rapid reproduction, high levels of diversity, a short generation time and large populations) will have the greatest advantage. Bazzaz (1996) adds that early successional plants are expected to evolve CO_2-responsive genotypes faster than late successional species and will increase their distributions throughout the world as mean annual temperatures increase.

2.2.3 Invasibility

Most habitats receive propagules from invading plants. The environment often has barriers that obstruct invading species from completing their life cycle. A review by Richardson et al. (2000) identifies the major barriers limiting plant dispersal as unique to each habitat. As each barrier is removed, different species will be allowed to spread beyond the barrier according to the attributes of that species. The removal of barriers by disturbance events often will facilitate the establishment of species that were previously unable to establish.

Davis et al. (2000) provide a theory to describe the susceptibility of an environment to invasion by non-resident plant species (invasibility). The theory simply states that “a plant community becomes more susceptible to invasion whenever there is an increase in the amount of unused resources”. Resource availability increases when resource use by resident vegetation declines (i.e. anthropogenic disturbance, herbivory, disease) or supply increases faster than resident vegetation can sequester it (i.e. increased precipitation, light levels, eutrophication). The authors are aware that a community’s susceptibility to an invasion will fluctuate over time and is dependent upon an adequate propagule supply. The theory is intended to explain differences and changes in invasibility and provides ample explanations of how changes in resource supply affect species invasions. It also attempts to integrate existing hypotheses regarding community invasibility and resolve conflicting and ambiguous results from other studies.

Several authors (Pysek, 1998a; Rejmanek and Richardson, 1996; Williamson, 1993) indicate that exotic plants are often successful when the invaded climate is similar to their native climate. This homo-climatic theory was originally formulated at the species level by Chicoine et al. (1986) and was not addressed, at the time, at the family level. Work conducted by Pysek (1998a) investigated the issue at the family level and concluded that families have not overcome their evolutionary and ecological limitations. Plant families tend to invade climates that are similar to their native area, thus confirming the homo-climatic theory is applicable at the family level. According to Huenneke (1997), paleoecological literature demonstrates that invasions into areas of similar climate appear to be rapid. Mack (1996) also agrees that predicting future invaders based on the climate in their home range can be effective, although it is subject to exceptions. The native ranges of many potential invaders are diverse and may not react similarly when climate change occurs.

2.2.4 Predicting invasions

Predicting the ability of a species to invade a particular area is often a difficult and inexact science. As stated by the National Research Council (2002), "no known broad scientific principles govern 'invasive potential' for all plant pests in all environmental circumstances". Attention about the issue is critical when one considers the impacts of species invasions are often irreversible (Ewel et al., 1999). Predictions should identify species that cannot be allowed to invade and spread and species that will have no detrimental effects and require no additional effort (Booth et al., 2003).

Predicting invasion success is dependent upon plant community and species level properties that change throughout the invasion process (Case, 1990; Lodge, 1993; Wiser et al., 1998). Mack (1996) suggested a series of experiments to effectively predict the invasibility of a particular species. Methods that provide varying levels of predictive power include comparing performances of species in a new range belonging to the same genus (congeners), deliberately sowing an exotic species beyond its current range and habitat manipulation of the deliberately introduced species. These experiments appear to be useful when time is not a constraint, requiring the experimenter to possibly commit large amounts of effort. Work completed by Rejmanek and Richardson (1996) uses species attributes to predict the invasiveness of pine trees (*Pinus* spp.) in the Northern and Southern Hemispheres. Although the authors are aware of the limitations of developing useful generalizations, they identify three attributes to predict invasiveness: a short juvenile period, short interval between large seed crops and small seed mass. The article is limited to a discussion about woody species of seed plants with remarks about the invasiveness of herbaceous species focusing on similar climates and latitudinal ranges in their native and newly invaded environments. Work by Williamson (1993) has attempted to describe the percentage of species that will have harmful impacts. The author is aware that according to geography or habitat, a plant may be a serious pest in one place and benign in another. Reference is made to the well-known 10:10 rule for plant invaders in Britain. The rule states that 10% of introduced species become naturalized and that 10% of those become pests. Hence, only 1% of the introduced species become pests. This rule appears to work adequately for species in Britain but has not been tested in an adequate number of areas throughout the world, thereby limiting its use.

Ranking a species on its ability to become detrimental can provide valuable insights into the likelihood of an invasion and its consequences. An evaluation of this risk assessment process has been conducted by the National Research Council (2002). The authors contend that risk assessment reduces the dependence on the biological information needed in a predictive system. The documentation and transparent logic used in risk assessment permits public scrutiny and independent evaluation. Work completed by Hiebert and Stubbendieck (1993) provides a ranking system to sort exotic plants according to the level of species impact and its innate ability to become a pest. Successful implementation of this system has occurred at Pipestone National Monument in Minnesota. Various species attributes are addressed including abundance, distribution relative to disturbance regime, mode of reproduction, germination requirements, and competitive ability. The feasibility of control examines abundances within areas (number of populations, aerial extent of population), ease of control (nature of seed banks, level of effort required, side effects of chemical/mechanical control) and urgency of control. This ranking system provides a logical and analytical approach to support decisions made by researchers and land managers.

2.3 Boreal Forest

The boreal forest stretches across Canada from Newfoundland in the east to northern British Columbia in the west. It is the dominant forest region in Canada comprising 82% of the total forested area (Boreal Forest Research Centre, 2003). This forest is bounded by tundra to the north and grassland, montane and the Great Lakes regions to the south, west and east. It belongs to the Northern Hemisphere circumpolar boreal forest continuum that is also present throughout Eurasia.

The boreal forest of Saskatchewan is composed of various mixtures of broad-leaved and coniferous tree species. Information collected by the Council of Saskatchewan Forest Industries (2001) describes Saskatchewan forests with regards to area totals, ownership, species mixtures, amounts of harvesting and wildfire and economic information. Tree species cover is dominated by coniferous species (39%) while deciduous (36%) and mixedwood (25%) stands comprise the remaining forest cover. The broad-leaved tree species are largely represented by trembling aspen (*Populus tremuloides*), balsam popular (*Populus balsamifera*) and white birch (*Betula papyrifera*) while the coniferous trees are dominated by white spruce (*Picea glauca*), black spruce (*P. mariana*) and jack pine (*Pinus banksiana*) with lesser amounts of balsam fir (*Abies balsamea*) and tamarack (*Larix laricina*). Work completed by Acton et al. (1998) provides a description of forested ecosystems in Saskatchewan. This hierarchical system of classification incorporates several aspects of the Saskatchewan environment. A field guide produced by Beckingham et al. (1996) focuses on the mid-boreal ecoregions of Saskatchewan. The guide provides information about the vegetation, soil, site and forest productivity in the area while providing management interpretations at the ecosite level.

Soils that can be found in the boreal forest include Luvisols, Brunisols and Organics. Luvisols develop in well-drained areas and have a high nutrient content. Brunisols typically develop on sandy deposits and mainly occur in drier regions. Organic soils are located in low lying areas where water accumulates. The anaerobic conditions in organic soils produce slower rates of decomposition resulting in peat accumulation. This soil supports slow growing black spruce and tamarack forest stands.

2.3.1. Disturbance

Disturbance plays a major role in Canada's boreal forest, defining its ecological character, productivity, and landscape pattern (Kimmins 1996). Disturbances that remove the resident vegetation will increase the availability of resources (water, light, and nutrients) for plant growth (Davis et al., 2000). Chapin (1983) confirms that deforested areas generally have more light and higher nutrient availability than unmodified vegetation. The increase in resources contributes to an increased susceptibility to plant invasion in the recently disturbed area (Davis et al., 2000). As the area is exposed to anthropogenic influences, the opportunities for exotic plant species to disperse to the location increases. The matter raises concern because the disturbances examined in this study are all exposed to human contact.

A review of the historical background of timber harvesting in Saskatchewan is provided by Goode et al. (1996). Harvesting has occurred, to some degree, in Saskatchewan since the 1800s while large scale harvesting began in 1968. Modern timber harvesting and associated activities (i.e. road construction) have been documented to influence surface temperatures and watershed attributes (Forman and Alexander, 1998; Northern Forestry Centre, 1986; Swanson and Hillman, 1977; Trombulak and Frissell, 2000). The average yearly amount of harvested areas in Saskatchewan on provincially owned crown land is 24,740 hectares (Council of Saskatchewan Forest Industries, 2001). The cutting rotation is dependent on the species harvested and generally occurs on a 70 to 90 year cycle (Christensen, pers. comm.). Once stands are harvested, the area is left to regenerate naturally or undergoes post-harvest treatments, if required. These treatments generally involve some type of site preparation (disc trencher, drum chopper, v-blade, Bracke scarifier), that is used alone or in combination with other treatments. Site preparation increases the amount of microsites, exposing mineral soil and providing resources (i.e. nutrients and water) for planted tree seedlings. Stand maintenance involves the removal of undesirable tree species with a brush saw, decreasing the competition among seedlings and other vegetation. These maintenance activities promote the growth of the planted seedlings, although they may not be required depending on the characteristics of the regenerating forest. All of these activities of timber harvesting and renewal require the use of temporary and permanent roadways.

Roadways in the Saskatchewan boreal forest have either paved or gravel surfaces. Several private and commercial types of vehicles (passenger cars, small to very large trucks, buses) use these roadways to transport people, manufactured goods, agricultural and forestry products, fuel and earth materials on short to very long distance hauls. The roadway provides a permanent structure that has a distinct effect on the surrounding forest whether it is used temporarily or for extended periods of time. Forman and Deblinger (2000) noted that the major ecological effects of roads include:

- habitat loss by road construction;

- altered water routing and downstream peak flows;
- soil erosion and sedimentation impact on streams;
- altered species patterns;
- human access and disturbance into remote areas.

Roads produce long-term legacies on the landscape and involve four major phases of development including building, operating, maintaining and abandonment. Road building is often the most harmful to adjacent ecosystems because earth movement and other activities may disturb watershed attributes. Road construction removes the forest canopy and may require the placement of additional materials to obtain the proper slopes for vehicle safety. A study conducted by Grayson et al. (1993) corroborates that large increases in sediment production occur immediately after road construction. Road operations involve a variety of materials that are deposited into the surrounding soils and absorbed by the adjacent plants.

Saskatchewan highways are managed with a "bare pavement" policy that ensures the safety of vehicle travel (Hanson, pers. comm.). A mixture of sand and de-icing salts (i.e. sodium chloride [NaCl], calcium chloride [$CaCl_2$], potassium chloride [KCl] and magnesium chloride [$MgCl_2$]) is used to reduce packed snow and ice. The application of this mixture is dependent upon environmental conditions and has a reduced effect when exposed to extreme temperatures (i.e.- 40° C). Environment Canada (2003) is currently developing a code of practice for the management of road salts. The code is intended for municipalities and provincial highway departments that use more than 500 tonnes of road salt on public roads. The recommendations of the code are to develop and implement road salt management plans using best management practises, establish procedures to monitor progress and review the plans after five years to determine if other steps are needed to reduce the negative impacts of road salt on the environment. Saskatchewan Highways and Transportation is currently developing a road salt management plan (Hanson, pers. comm.).

The operation of roadways also contributes to the deposition of materials on the surface of surrounding vegetation. Settled dust can block photosynthesis, respiration and transpiration causing physical injury to plants (Loney and Hobbs, 1991; Trombulak and Frissell, 2000). The dust may also provide nutrients for plant growth, alter soil pH and supply fine sediments and contaminants to aquatic ecosystems (Forman and Alexander, 1998; Trombulak and Frissell, 2000). In areas of heavy vehicle use, pollutants from vehicle exhaust can have varied effects on roadside plants including altering photosynthesis, causing phytotoxic damage or even promoting plant growth (Angold, 1997).

Roadways and adjacent right-of-ways are also important vectors for water transport. Forman and Alexander (1998) state that water runs rapidly off relatively impervious road surfaces, which can increase peak discharge rates in adjacent stream networks. Trombulak and Frissell (2000) confirm that the hydrologic effects of roads are likely to persist for as long as the road remains a physical feature on the landscape. Increased amounts of water, especially during storm and snowmelt events, has been observed to promote the growth of colonizing plants and increase plant productivity in roadside habitats (Loney and Hobbs, 1991; Milton and Dean, 1998).

Maintenance of a roadway will vary according to the season and the type of road surface. Road maintenance in Saskatchewan during the summer months consists of grading operations and $CaCl_2$ applications (to reduce dust) on gravel roads with minimal maintenance of paved surfaces. In winter months, paved roadways often have snow removal equipment deposit additional snow into the roadside right-of-way while gravel roads have minimal maintenance.

Roadside right-of-way maintenance can involve the application of herbicides, mowing, haying and burning of vegetation at regular intervals. Consistent maintenance of roadside vegetation has been noted by Lugo and Gucinski (2000) to favour the establishment and survival of exotic and weedy species. It appears that the selective and repetitive nature of regular maintenance provides a suitable environment that particular roadside plants require. The elimination of regular maintenance activities and a reduction in motorized vehicle use occurs once the roadway is closed. The physical structure of the roadway will continue to have an influence on the surrounding vegetation even though a reduction in disturbance effects occurs.

Wildfire is another disturbance that affects the structure, function and composition of the flora in the boreal forest. It has long been recognized as the primary rejuvenating disturbance in the area (Heinselman 1973; Rowe 1961). This disturbance creates an increase in resources that may have once been unavailable to the plant species present. A burned forest floor provides a blackened and porous substrate that has highly fluctuating temperature and moisture regimes (Nguyen-Xuan et al., 2000). These variable attributes contribute to an environment that may have shifting nutrient budgets. Additional investigations by Cale and Hobbs (1991) and D'Antonio and Vitousek (1992) reveal that the effects of a fire may be strong enough to create a nutrient imbalance as phosphorous is released in the form of ash and nitrogen and carbon losses occur. Kimmins (1996) adds that by removing the excessive accumulation of woody material, which is lignin-rich and has a high carbon/nitrogen ratio, fire may promote nutrient cycling. Chapin (1983) states that the release of most organically bound nutrients occurs immediately after a fire.

The frequency of disturbances associated with wildfire, logging, and roadway operations and maintenance varies among these types of events. Wildfire is the "most important non-anthropogenic agent of change in the boreal forest" (Armstrong, 1999). The pattern of stand ages in the mixedwood boreal forest consists of smaller patches of older forest embedded within a matrix of larger patches of younger forest (Weir et al., 2000). The recurrence of fire or fire cycle in western Canada occurs on average every 50 to 100 years (Heinselman, 1981; Johnson and Rowe, 1975). The fire cycle is capable of significant changes as a result of land use in the surrounding area and climatic changes (Weir et al., 2000). Land use can lengthen the fire cycle by fragmenting the forest that was once continuous, decreasing the ability of wildfire to spread. Since the behaviour of a fire is mainly controlled by climatic conditions (Bessie and Johnson, 1995; Johnson et al., 1998; Stocks, 1993; Turner and Romme, 1994), climate change may have a larger affect on wildfires in the boreal forest. Past changes in climate at the end of the Little Ice Age coincided with the shift in the fire cycle from a short (15 year) cycle to a longer (75 year) cycle (Weir et al., 2000). In an examination of landscape dynamics in crown fire ecosystems, Turner and Romme (1994) state that future climate change will alter crown fire frequency and severity. While summarizing the possible scenarios associated with climate change, Wheaton (1997) confirms that disturbances are expected to increase. Elevated frequencies and intensities of drought are expected to result in a net loss of the total area of forest in the Western Canadian boreal forest.

Literature regarding the frequency of timber harvesting is relatively scarce as the yield among forest types is highly variable and specific information is retained by private forest companies. Management of the forest traditionally has focussed on the economic aspects of maintaining and enhancing the fibre supply for mills. The obligation to incorporate all forest stakeholders has prompted organizations to adopt ecological forest management as a guiding principle for harvesting operations in Canada (McRae et al., 2001). The largest timber harvesting company in Saskatchewan, Weyerhaeuser Canada, has incorporated ecosystem based management into its twenty year management plan (Golder Associates, 1999). This approach attempts to accommodate social and economic needs within a framework that emulates natural disturbance regimes.

The concept that timber harvesting can produce ecological results similar to wildfire has been addressed in the literature. Comparisons of harvesting and wildfire disturbances by the Alberta Research Council (1999) and Thrasher-Haug (1997) confirm that the greatest differences among species richness and forest structure occur immediately after the disturbance. Support for these findings is provided by Nguyen-Xuan et al. (2000) having observed that pioneer and lichen species were more abundant after wildfire and that residual species had greater abundances after timber harvesting. Carleton and MacLellan (1994) examined woody vegetation responses to wildfire and timber harvesting and concluded that disturbance type and intensity lead to different recovery patterns of woody vegetation. The work completed by the Alberta Research Council (1999) in aspen stands determines that early fire communities are unique and cannot be replicated by any practical harvest plan. The authors explain that this uniqueness appears to diminish as the forest structure and biotic communities develop. The vegetation becomes more comparable among disturbance types during the first 28 years with differences being restricted to relative abundances rather than presence or absence of species after 28 years.

Although the vegetation appears to converge over time in harvested and wildfire areas, particular features will not coincide with one another. The removal of carbon from harvested areas may eventual lead to a reduction in carbon content within the soil organic layer (Alberta Research Council, 1999). Harvested areas will also have a road network and its associated effects that have no equivalency in a forest disturbed by fire (McRae et al. 2001). This network will reduce the areas available for reforestation, fragment the landscape for some species and ecological functions and allow easier access by humans. The scale of disturbance will also be difficult to emulate because the patch size created by harvesting is a subset of the range created by wildfire. The pattern of disturbance is another feature that will be challenging to imitate as the heterogeneity of fire patterns has several ecological influences that form a complicated mosaic of post-fire conditions (McCullough et al., 1998; Turner and Romme, 1994).

2.3.2 Maturation

As the time since disturbance lengthens, the maturation of a forest will also affect species composition, function and structure. The development of the stand is dependent on the attributes of the disturbance. Thorpe (1996) provides a review of literature regarding the post-fire development of mixedwood forest. The author discusses the main pathways of succession relevant to the southern boreal forest of Saskatchewan. As the stand develops, dense sprouting of hardwood tree species will occur after a wildfire that causes high mortality. Work completed by Barbour and Billings (1988) summarize the environmental changes associated with canopy closure: 1) an increase in thickness of the organic layer, 2) a decrease in nutrient availability, 3) a decrease in summer soil temperature, 4) a decrease in soil drainage capabilities and 5) decreased exposure of sunlight. In an article by Miller (1983) the effects of increased amounts of shade are described as the vegetation develops. The study reveals stabilizing soil temperatures and changes to the amount of water available for plant growth in shaded areas. Precipitation will also be increasingly intercepted in the canopy and annual mean soil temperature and soil evaporation will decrease. Stabilizing temperature and moisture levels in soils will affect the growth of plant species and interactions among those species. A closed canopy will have a reduction in the levels of soil nutrients as the competition among trees and shade-tolerant herbs is elevated (Mooney and Gulmon, 1983; Wiser et al., 1998). Exotic species that occur in disturbance prone landscapes may thus be at a disadvantage in these more stable environments. The late-successional species in a closed canopy are able to capture resources that are unavailable to shallow rooted plants because of their ability to produce deep roots.

Chapter 3

Study Area

3.1 Location

The study area is located within the Boreal Plain Ecozone which extends across Manitoba, Saskatchewan and Alberta and extends to the Northwest Territories in the north and British Columbia in the west. This ecozone occupies more than 17 million hectares or 27 % of the area in Saskatchewan (Acton et al., 1998). Forestry, hunting and trapping, outdoor recreation and tourism are the primary land-uses in this ecoregion. Other land uses include oil and gas exploration and production, mining and agriculture. This ecozone lies south of the Precambrian shield.

The survey sites are located either within the Prince Albert Model Forest [PAMF] or close to its boundaries. The PAMF is centrally located in the province of Saskatchewan, and includes the communities of Waskesiu, Montreal Lake and Candle Lake. The area contains portions of the Nisbet Provincial Forest, Prince Albert National Park [PANP] and Weyerhaeuser Canada's Prince Albert Forest Management Agreement [FMA] area. The PAMF is 70 km north of Prince Albert, Saskatchewan, and is located in the Mid-boreal Upland Ecoregion (Figure 3.1). This ecoregion is a part of the Boreal Plain Ecozone and occupies over 10 million hectares in Saskatchewan. It is characterized by well-drained uplands with a mixture of coniferous and deciduous trees and extensive wetlands dominated by black spruce (*Picea mariana*), sedges (*Carex* spp.) and mosses (Acton et al., 1998).

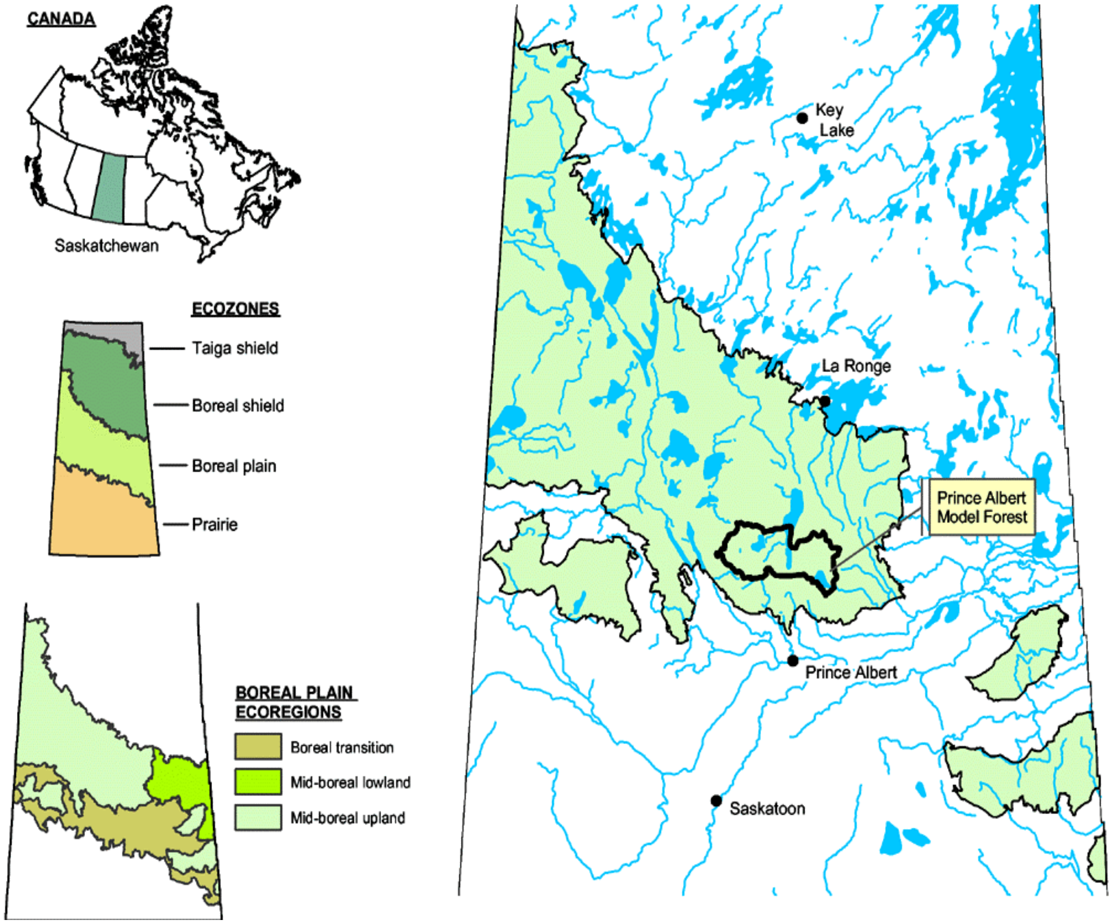

Figure 3.1 The location of Prince Albert Model Forest within the Mid-Boreal Upland Ecoregion of Saskatchewan.

3.2 Vegetation

The surveying was conducted in and immediately adjacent to mixedwood stands that were characterized by *Populus tremuloides-Picea glauca* and *Picea glauca-Populus tremuloides* (HS and SH) stand classifications (Table 3.1). These mixedwood stands occupy approximately 25% of the 35.5 million hectares of forested land in the province (CSFI, 2001). These types of forest were selected for this study because they are common in the area and commercially important (Acton et al., 1998). According to provincial inventories, the stands in the survey had year of origins ranging from 1890 to 1990 (Saskatchewan Environment, 2004). Figure 3.2 illustrates the amount of mixedwood stands within Weyerhaeuser's Prince Albert FMA area. Mixedwood stands occupy 160,671 ha within this area. ArcView 3.3 (2002) was used to construct the maps describing the study area throughout the Prince Albert National Park [PANP] and

Weyerhaeuser Canada's Prince Albert Forest Management Agreement [FMA] area. All maps were located in zone 13 of datum NAD 83.

Table 3.1 A description of the mixedwood survey sites prior to disturbance events (Prince Albert National Park, 1994; Saskatchewan Environment, 2004).

Attribute	Range
Stand classification	HS (25 to 50 % softwoods by volume)
	SH (50 to 75 % softwoods by volume)
Height class	17.5 to 22.5 m and greater than 22.5 m
Year of origin	1890 to 1940
Crown closure	50 to 80 % and 80 to 100 %

Other tree species in mixedwood stands include paper birch (*Betula papyrifera*), balsam popular (*Populus balsamifera*) and balsam fir (*Abies balsamea*). Tall shrub species located in mixedwood study sites include alder (*Alnus* spp.), beaked hazelnut (*Corylus cornuta*), buffalo-berry (*Shepherdia canadensis*), dogwood (*Cornus stolonifera*), chokecherry (*Prunus virginiana*), pincherry (*Prunus pensylvanica*), Saskatoon (*Amelanchier alnifolia*) and various willow species (*Salix* spp.). Lower shrub species include blueberry (*Vaccinium* spp.), cranberry (*Viburnum* spp.), currant (*Ribes* spp.), honeysuckle (*Lonicera* spp.), raspberry (*Rubus idaeus*), rose (*Rosa acicularis*) and snowberry (*Symphoricarpos* spp.). Herb species observed in the study sites belong to the following plant families: fern (Polypodiaceae), clubmoss (Lycopodiaceae), horsetail (Equisetaceae), sedge (Cyperaceae), grass (Gramineae), composite (Compositae), dogwood (Cornaceae), evening-primrose (Onagraceae), ginseng (Araliaceae), heath (Ericaceae), honeysuckle (Caprifoliaceae), lily (Liliaceae), madder (Rubiaceae), orchid (Orchidaceae), pea (Leguminosae), rose (Rosaceae), violet (Violaceae) and wintergreen (Pyrolaceae). The plant nomenclature was taken from Moss (1994).

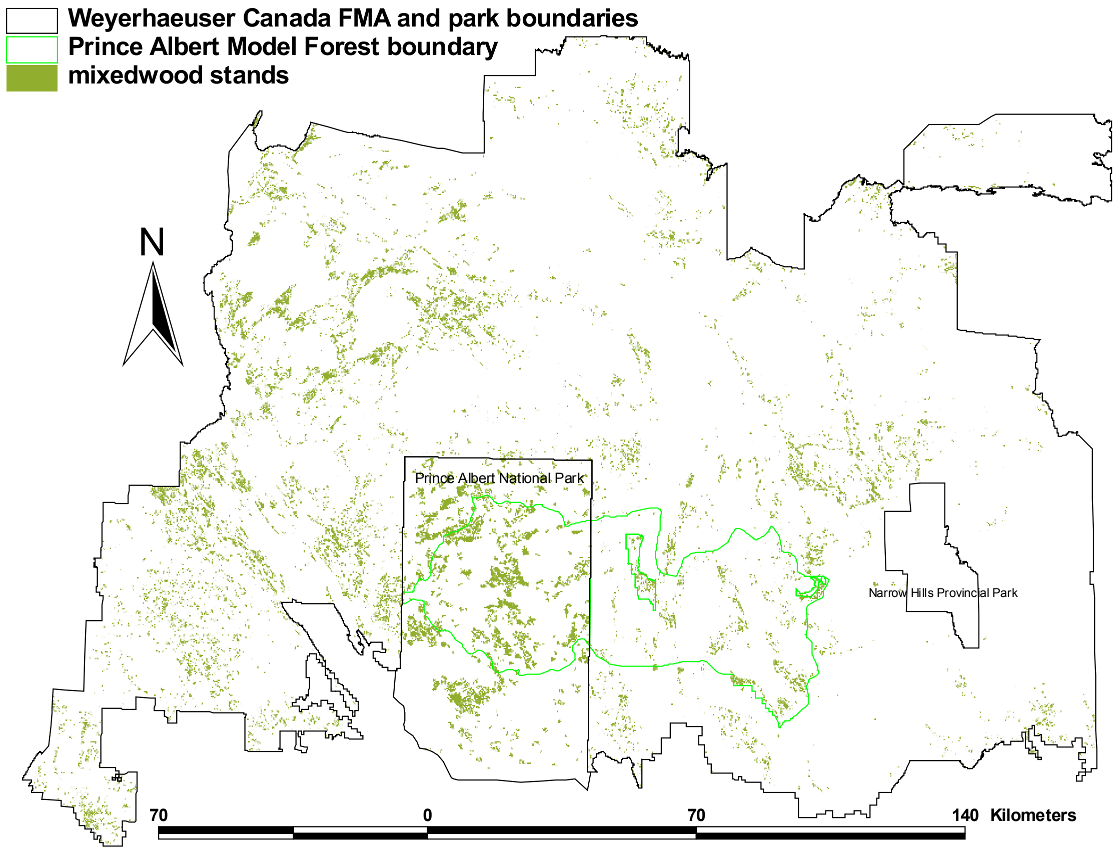

Figure 3.2 The mixedwood stands within Weyerhaeuser Canada's Prince Albert FMA area and Prince Albert National Park.

3.3 Timber Harvesting

Increased harvesting in Saskatchewan forests has raised concerns about the impact of exotic plants on the boreal forest (Figure 3.3). The potential for exotic species to establish in disturbed areas is well known (Anderson, W.P., 1999; Dean et al., 1994; Holland and Olson, 1989; Hobbs and Huenneke, 1992; Kotanen, 1997). This issue is important as forestry operations in Saskatchewan are proposed to continue to expand. Since 1999, more than 8,000 new direct and indirect jobs were created in the forestry industry (Saskatchewan Industry and Resources, 2002). Approximately $900 million has been invested into new and expanded facilities with industry and government considering an additional $800 million in new projects. The Government of Saskatchewan appears to have met the majority of its objectives, set in 1999, to double the forestry industry in Saskatchewan, and expects continued growth in the future. The lumber production is expected to triple in Saskatchewan by the addition and expansion of oriented strand board (OSB), finger-jointer, planer and sawmill projects. Current harvest levels in all merchantable stand types are approximately 24,470 hectares per year of crown (provincially owned) land throughout the province (CSFI, 2001). Table 3.2 summarizes the sizes of cutblocks examined in the study. Harvest levels in mixedwood stands within Weyerhaeuser's Prince Albert FMA area are approximately 5,863 ha per year (Weyerhaeuser, 2004). Figure 3.4 illustrates harvesting on mixedwood stands from 1989 to 1998. A total 58,633 ha of crown land was harvested during this 10 year time period.

Table 3.2 A comparison of survey sites that were recently disturbed by harvesting and wildfire events (Weyerhaeuser Canada, 2004; Saskatchewan Environment, 2004).

site ID	disturbance type	name of site	size (ha)	year of disturbance
9	timber harvesting	Timber Cove	96	1990
32	wildfire	Monday fire South	15,600	1995
43	timber harvesting	Nikik	42	1991
44	wildfire	Waskesiu fire South	17,000	1998
46	wildfire	Muskeg fire	13,050	1989
50	wildfire	Monday fire North	15,600	1995
51	timber harvesting	Highway 120	9	1989
53	timber harvesting	Snowfield	93	1992
54	timber harvesting	Mino west	31	1995
55	timber harvesting	Mino east	51	1995
57	wildfire	Cobra fire	42,000	1998
58	wildfire	Waskesiu fire North	17,000	1998

Figure 3.3 A recently harvested survey site examined in the study. Surveyed cutblocks were clearcut between 1989 and 1995 and re-planted with white spruce (*Picea glauca*) seedlings.

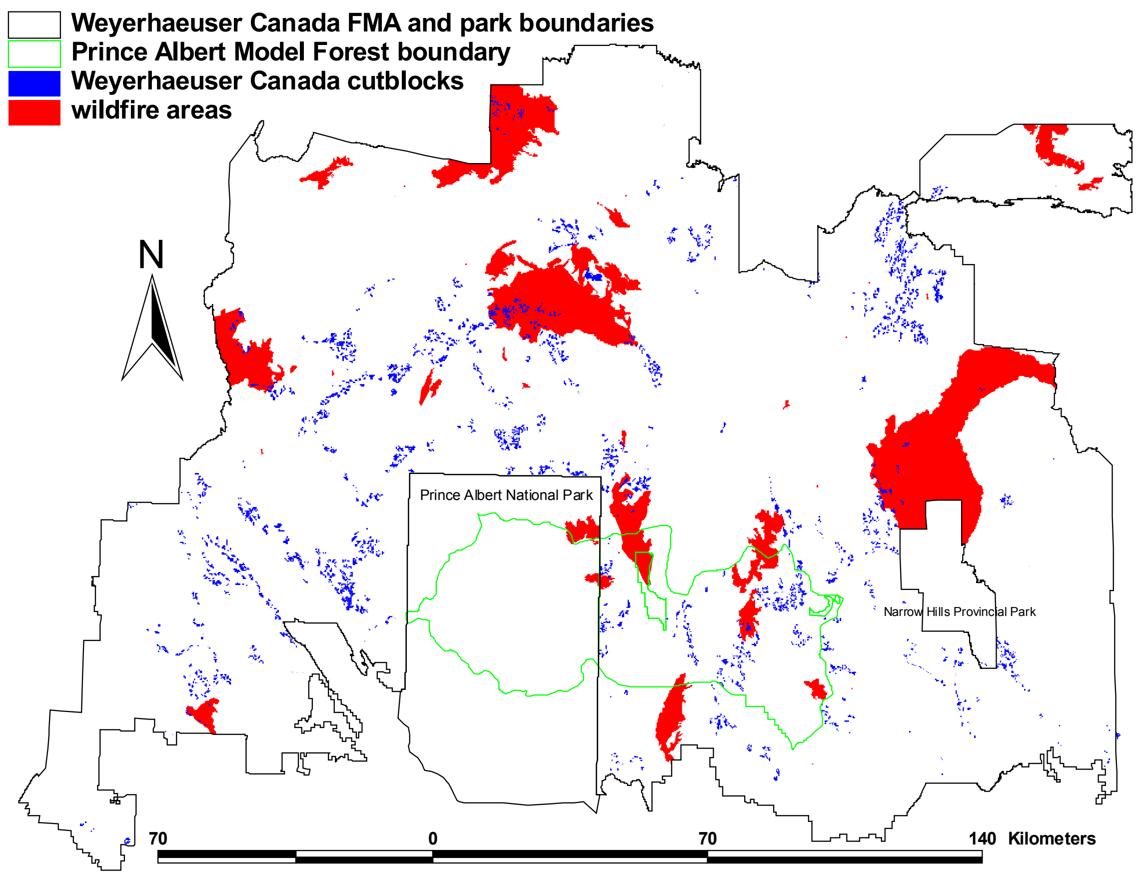

Figure 3.4 The harvested and wildfire mixedwood stands within Weyerhaeuser Canada's Prince Albert FMA and Prince Albert National Park. These sites were disturbed between 1989 and 1998.

3.4 Wildfires

The yearly average (over the past 25 years) of burned hectares on Saskatchewan crown land is 304,000 (Council of Saskatchewan Forest Industries, 2001). In 1995, extreme weather conditions allowed wildfires to burn 1.39 million hectares of crown land in Saskatchewan. Wildfires within Weyerhaeuser's Prince Albert FMA area and Prince Albert National Park over a 10 year period (1989-1998) are shown in Figure 3.4. Wildfires within this area occupied 265,435 ha (Saskatchewan Environment, 2004; PANP, 1994). The size and year of disturbance for each wildfire examined are summarized in Table 3.2.

The examination of the wildfire sites (Figure 3.5) will contribute to an understanding of the influence of this natural disturbance on exotic species distribution. Natural disturbances are anticipated to increase as global circulation models forecast rapid changes in climatic patterns that are associated with increased atmospheric carbon dioxide levels (Wheaton, 1997). As this increase is expected to promote exotic plant species, wildfires will also promote an expanding exotic distribution. Warmer temperatures are projected to produce longer fire seasons and drier fuel moisture conditions (Stocks, 1993). The anticipated climate change is also expected to increase the quantity, severity and area affected by wildfire (Flannigan and Van Wagner, 1991; Overpeck et al., 1990). The increased disturbance is expected to promote exotic plant invasions (Hobbs, 2000).

Figure 3.5 A recently burned survey site. A crown fire has promoted dense resprouting of hardwood tree species.

3.5 Roadways

Areas for these right-of-way were calculated according to the average width and total length of each class of roadway. The highway system within the PAMF is approximately 360 km in length and only right-of-ways that were maintained by Saskatchewan Highways and Transportation and Prince Albert National Park were examined. Clearing the area adjacent to roadway permits the construction of the right-of-way and associated maintenance activities (Figure 3.6). Regular maintenance provides a re-occurring disturbance to road and right-of-way areas. The immediate roadside edge is mowed every year while the entire right-of-way is mowed every two years with a full cut (mowing and brush cutting) occurring every three years. The frequency and type of mower (grass or brush) used within the right-of-way is dependent on the condition of vegetation. Northern bush roads that are primarily used to transport timber receive the same right-of-way maintenance.

Figure 3.6 Roadside right-of-way construction. The removal of adjacent forest permits regular maintenance activities.

In the winter, sand and salt are applied to road surfaces with snow removal completed by heavy equipment. The removal of snow deposits additional precipitation into the right-of-way which is carried away by a ditch during the spring snowmelt. Additional disturbances to the immediate edge of gravel roads includes burial of vegetation from summer grading operations and the application of $CaCl_2$ on roadways to reduce dust particles. No direct chemical applications to control vegetation occurred within the right-of-ways study sites.

A summary of the roadway network in Weyerhaeuser Canada's FMA is illustrated in Figures 3.7 and 3.8. Map files for the provincial highway system were collected from the University of Saskatchewan Library Services (2004). The width of the right-of-way for each class was measured and an average width was used to calculate the total area of right-of-way. Principal or class 2 highways (i.e. highways 2, 120, 263, 264 and 265) had right-of-ways occupying an area of 1362 ha. Secondary or class 3 highways (i.e. highways 926, 930, 969) had 1014 ha of roadside right-of-way while local or class 5 roadways (i.e. roads within Candle Lake village) had right-of-ways occupying 3688 ha. Abandoned or maintained roadways (class 6) within the Prince Albert National Park had the smallest amount of right-of-way among the road classes, occupying 378 ha. The total area of maintained roadside right-of-ways within Weyerhaeuser Canada's FMA is 6442 ha.

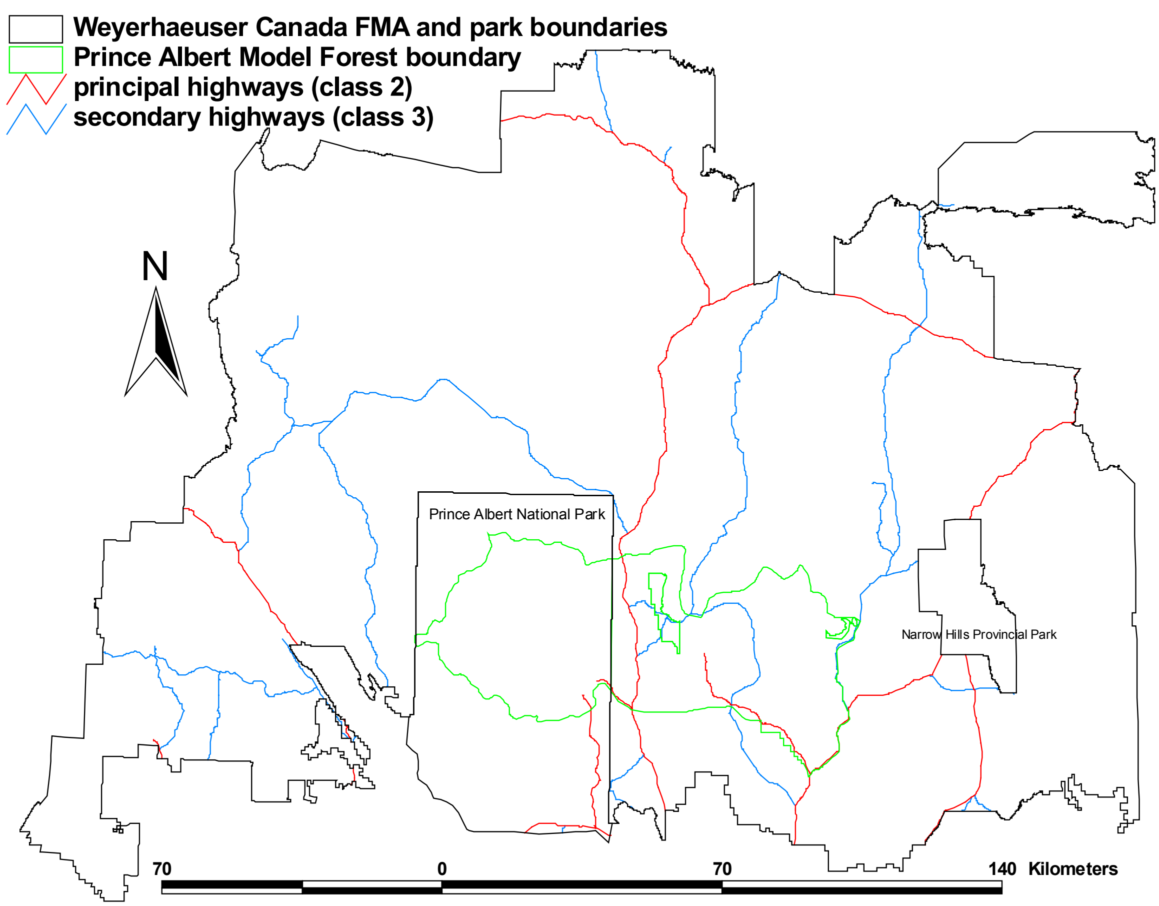

Figure 3.7 Principal and secondary highways within Weyerhaeuser Canada's FMA area and Prince Albert National Park.

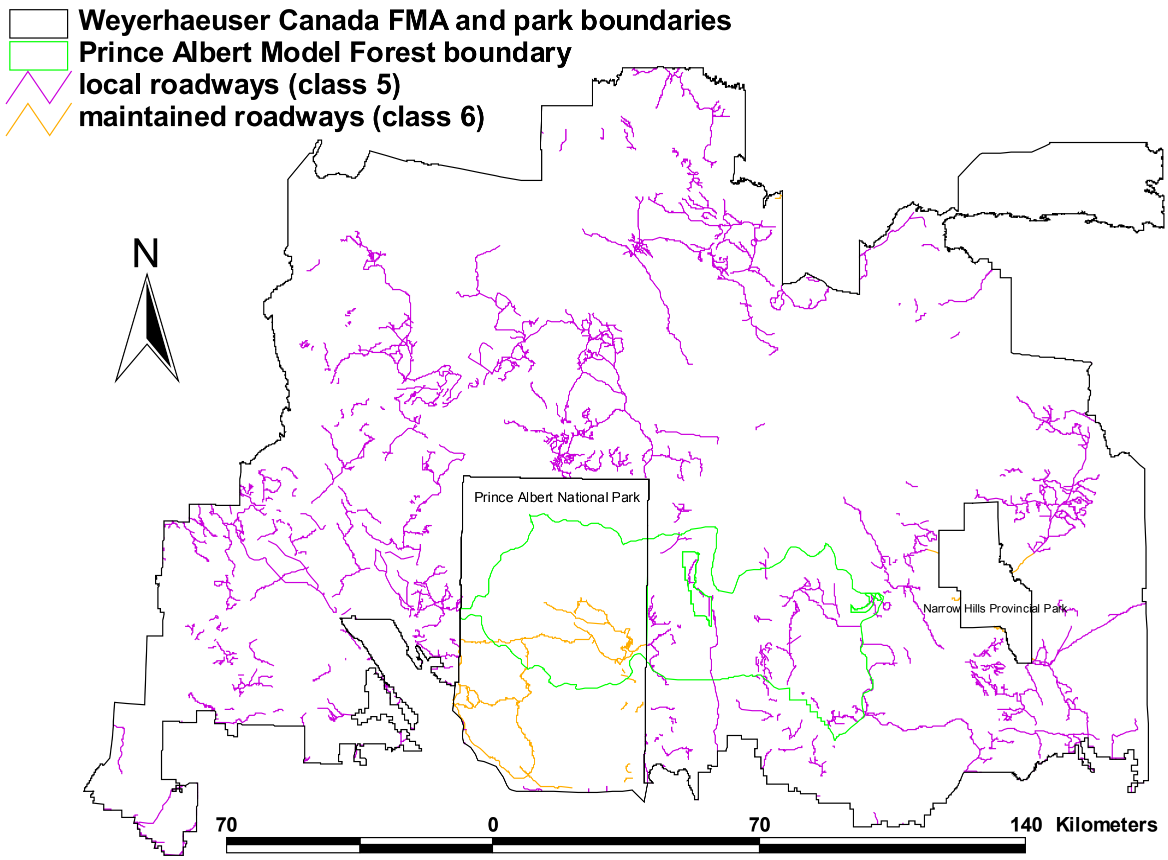

Figure 3.8 Local and maintained roadways within Weyerhaeuser Canada's FMA area and Prince Albert National Park.

Seeding of exotic species occurred along certain primary and secondary highways throughout the study area. The plant species used by Saskatchewan Highways and Transportation and the Prince Albert National Park to revegetate the right-of-way study areas are listed in Table 3.3. The mixture contained perennial grass and legume species except for one plant, *Lolium multiflorum*, which has an annual life-form. Various mixtures of these species were applied shortly after construction of Highways 2 and 120 in 1967-1969, while Highways 263 and 264 were seeded when constructed in 1949 and 1967, respectively. Other roadsides in the PANP that were seeded shortly after road construction include the Northshore and Narrows Roads constructed in 1963 and 1931, respectively. Additional seeding of these roadways occurred after reconstruction in 1982 and 1983. Seed mixtures containing exotic species were generally applied to class 2 and 3 highways while the remainder of the right-of-ways (classes 5 and 6) generally had no seed mixtures applied.

Table 3.3 Plant species used to revegetate roadside right-of-ways in Prince Albert National Park and along provincial roadways (Saskatchewan Highways and Transportation, 1993; Weir pers.comm., 2000). The status of these species are as listed in Loonman (1983), McGregor et al. (1986) and Moss (1994).

Species	Common name	Status
Agropyron cristatum (L.) Gaertn.	created wheat grass	exotic
Agropyron riparium (Scribn. and Smith) Bowden	stream bank wheat grass	native
Bromus inermis Leyss.	smooth brome grass	exotic
Bromus biebersteinii Roemer and J.A. Schultes	meadow brome grass	exotic
Elymus junceus Fisch.	Russian wild rye	exotic
Festuca rubra L.	creeping red fescue	exotic
Lolium multiflorum Lam.	annual rye grass	exotic
Medicago sativa L.	alfalfa	exotic
Onobrychis viciifolia Scop.	sainfoin	exotic
Poa pratensis L.	Kentucky blue grass	exotic
Trifolium hybridum L.	alsike clover	exotic
Trifolium pratense L.	red clover	exotic

3.6 Soils

Luvisolic (Gray Wooded) soils are the most common in the boreal plain ecozone representing 44 % (6.4 million hectares) of the soils in the region (Agriculture Canada, 1992). Luvisols are well-drained forest soils that usually develop on glacial till and lacustrine parent materials. Soil associations and map units that were examined include the Bittern Lake (map unit Bt1, Bt2, Bt3,), Loon River (Ln5) and Waitville (Wv1) associations as described by Anderson and Ellis (1976). Luvisols usually have loam or clay textures and show signs of significant weathering. The downward movement of clay minerals is a characteristic feature of these soils. Trembling aspen and white spruce mixedwood forest also commonly occurs on this soil type.

3.7 Climate

The Mid-boreal Upland Ecoregion has a mean annual temperature of 0.3°C and a mean annual precipitation of 452 mm. The winters are long, cold to very cold with a continuous snow cover normally lasting five to six months. The summers are short, warm and moist with the majority of the annual precipitation falling in May to September. The mean July temperature is 16.3°C and the mean January temperature is –18.9°C in the Mid-boreal Upland Ecoregion (Acton et al., 1998).

Data from the Waskesiu Lake Climate Station (53° 55’ N 106° 4’ W) indicate that in the year 2000 the mean annual temperature and precipitation were 1.0°C and 400 mm, respectively. Data for 2001 show that the mean annual temperature and precipitation were 3.0°C and 235 mm, respectively (Environment Canada, 2003).

Chapter 4

Methods

4.1 Experimental Design

Three types of land use were chosen in areas that were expected to be more susceptible to an exotic plant invasion. Since exotic species are more likely to become established in disturbed areas, wildfire and harvested sites were examined because of the anticipated increase in occurrences within the study area. The treatments have fixed factors involving combinations of the disturbance type, time since disturbance and the proximity to a seeded right-of-way. Additional surveying occurred within roadside right-of-ways where mowing activities provide a consistent disturbance.

The time since disturbance was separated into three categories. Sites that were recently disturbed had experienced timber harvesting or wildfire events between 1989 and 1998. These sites were examined within 15 years of the disturbance to ensure early successional habitats were surveyed. Thorpe (1996) has noted that the regrowth of hardwoods (*Populus tremuloides*) can obtain heights of 9 to 11 m at 15 years of age in a mixedwood canopy. The survey sites that were disturbed annually and/or biennially were located in roadside right-of-ways. Regular maintenance activities provide disturbances in the right-of-ways when the vegetation is cut by mowers to a specified height. Mature sites occurred on harvesting or wildfire disturbances which occurred more than 60 years ago. A review of forest inventory and historical maps assisted in the identification of mature sites. The mature sites were examined during this stage of successional development to ensure that mid-successional habitats were examined.

To examine whether exotic plant invasions into forest disturbances were influenced by the proximity to a site with a deliberate introduction of exotic species, study sites were chosen from both “seeded” and “unseeded” right-of-ways. Survey sites were considered "seeded" if the area was planted with the propagules of the species listed in Table 3.3. Sites were also considered "seeded" if the site was contiguous or connected by a short (< 1 km) corridor to a seeded right-of-way. A short corridor provides exposure to a deliberate introduction. According to Forman and Alexander (1998) few documented cases exist of species spreading more than 1 km from roadway right-of-ways. A total of 132 km of roadway within the Prince Albert Model Forest [PAMF] have had exotic seed mixtures applied to roadside right-of-ways. Sites were considered "unseeded" if they were adjacent to a right-of-way that had not been planted with an exotic seed mixture or were greater than 1 km from a right-of-way seeded with exotic species. There are approximately 225 km of unseeded right-of-ways within the PAMF.

Ten treatments were constructed by combining the fixed factors. These treatments were used to address the hypotheses (Section 1.2) and are as follows:

Seeded right-of-way	Unseeded right-of-way
Mature forest-fire-seeded	Mature forest-fire-unseeded
Mature forest-harvested-seeded	Mature forest-harvested-unseeded
Recently disturbed forest-fire-seeded	Recently disturbed forest-fire-unseeded
Recently disturbed forest -harvested-seeded	Recently disturbed forest -harvested-unseeded

The sampling design had treatments with randomly chosen quadrats organized into sites. The sites had 5 to 12 quadrats depending on the location. The sampling had a nested design comparable to the description by Sokal and Rohlf (1981). Figure 4.1 represents the organization of the design that was used to examine the hypotheses of the study.

A nested analysis of variance (ANOVA) was used, when appropriate, to test for differences among treatments. The nested analysis examined the differences among the treatments (fixed factor) without the consideration of the differences among the sites (nested random factor). The analysis will use the site as the error term when examining differences among the treatments. The equation for the two-level nested (Model II) ANOVA was

$$Y_{ijk} = \mu + A_i + B_{ij} + \varepsilon_{ijk} \qquad (4.1)$$

where Y_{ijk} is the *k*th observation in the *j*th subgroup of the *i*th group, μ is the parametric mean of the population, A_i is the random contribution for the ith group of the treatment, B_{ij} is the random contribution for the *j*th subgroup (site) of the *i*th group; ε_{ijk} is the error term of the *k*th item in the *j*th subgroup of the *i*th group. The ANOVA requires that the data have a normal distribution and equal variances in order for the technique to be valid. A normal distribution among the data was examined with a probability plot (P-P plot) which graphs the expected cumulative probabilities vs. the observed cumulative probabilities. If the selected variable exhibits a normal distribution the points in the graph will cluster around a straight line. Equality of variances among the variables was tested with a Levene's test (Dytham, 1999). If the critical value was less than 0.05, the variances among the data were declared unequal and the data were heteroscedastic. A non-parametric test (Mann-Whitney U tests) was used if the assumptions of the ANOVA were seriously violated. A nested design was also used during the non-parametric testing. All statistical analyses were performed with SPSS (1999) using a critical level (p-value) of 0.05 for the rejection of the null hypothesis being tested. The confidence interval (95%) accompanied the reporting of all mean values calculated. According to Dytham (1999) the confidence interval is preferred over standard errors when several sets of observations are being compared.

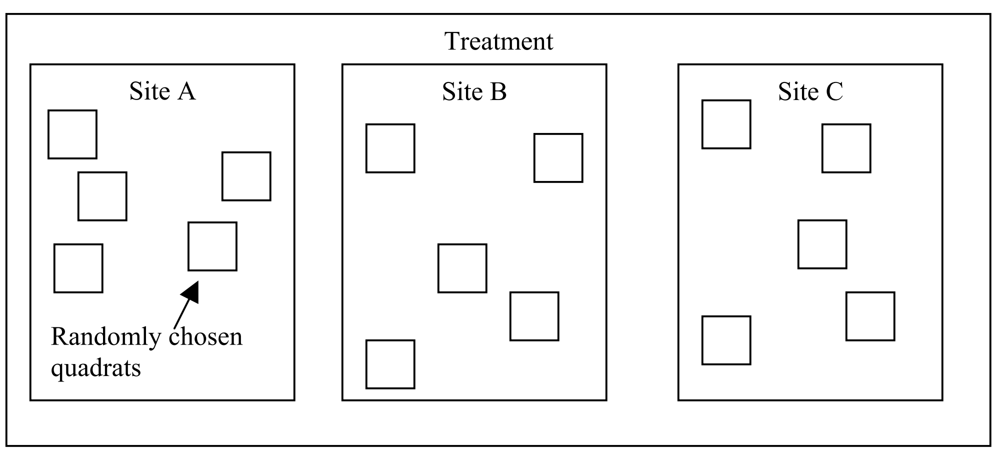

Figure 4.1 A representation of the nested design used to test the study hypotheses. Each treatment had a series of sites that contained 5 to 12 quadrats.

Regression and correlation analyses were also performed to examine relationships among variables. Regressions were used when a cause and effect relationship between the variables was supported. It was conducted during the examination of the relationship between the distance from the right-of-way and exotic quantities. This approach was also used to determine the strength of relationship between native and exotic plant cover in the study area. The null hypothesis for the regression analyses was that no relationship existed between the variables and therefore the slope of the regression line was zero. Correlation analysis was used to determine the degree to which one variable varied with another. This technique does not permit predictions of one variable from another. Correlation analysis was used to examine the relationship between the soil attributes (thickness of the organic layer (LFH) and depth to the C horizon) and exotic densities and frequencies.

4.2 Field Methods

4.2.1 Vegetation Surveying

A total of 312 quadrats were surveyed in July and August of 2000 and 2001. The surveying was completed during these months to ensure the majority of the mixedwood flora was identified in the survey. The roadside right-of-ways had surveying conducted at 207 quadrats (20 sites) while mature and recently disturbed sites had 45 (9 sites) and 60 quadrats (12 sites), respectively. Table 4.1 shows the survey sites and the fixed treatments examined in 2000. The numbers are not in chronological order as the site selection criteria did not allow every site visited to be surveyed. A total of 30 sites were visited from an initial selection of 89 sites. Surveying occurred in 21 forest sites with five quadrats surveyed per site. Figure 4.2 illustrates the approximate locations of each survey site. Mature sites were disturbed 60 to 80 years ago while recently disturbed sites were disturbed between 1989 and 1998. Surveying in 2001 occurred in roadside right-of-ways that were adjacent to forest sites visited in 2000.

Table 4.1 Sites and treatments examined in 2000. Each number represents a survey site and its associated treatment. Twenty-one sites were surveyed with five 10 x 10 m quadrats surveyed at each site.

Disturbance type	Adjacent to seeded right-of-way	Time since disturbance	
		Mature	Recently disturbed
Fire	No (unseeded)	14, 15, 17	32, 44, 50, 57, 58
	Yes (seeded)	8, 13	46
Logged	No (unseeded)	2, 19, 21, 10	53, 54, 55
	Yes (seeded)	n/a	9, 43, 51

The design is incomplete, as no sites were available for the mature-harvested-seeded treatment. The inability to locate these sites may have been a result of past forestry policy that did not allow harvesting next to roadways.

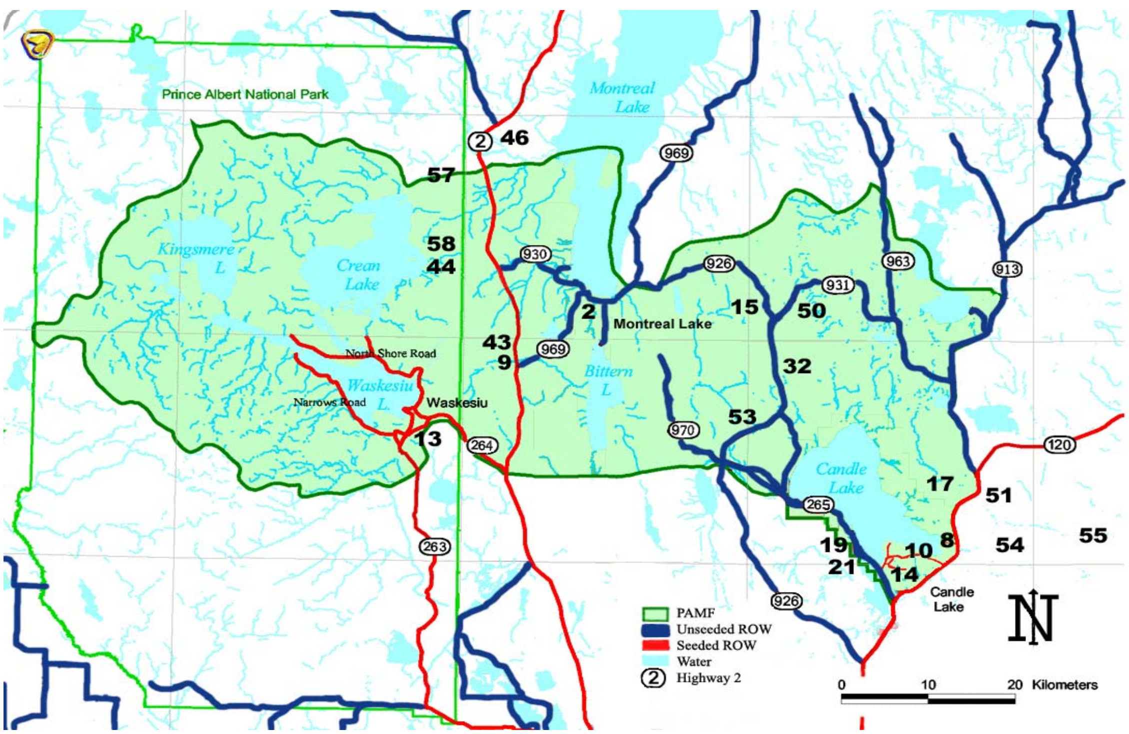

Figure 4.2 The approximate location of survey sites (bold numbers) within and near the Prince Albert Model Forest.

At each site in the 2000 field season, five 10 x 10 metre quadrats were randomly placed within a mature or recently disturbed forest (Figure 4.3A). Random placement of a quadrat was determined by placing a numbered grid on a map of the stand being surveyed. A random number generator (i.e. calculator) was then used to select the coordinates of each quadrat within the stand. To ensure quadrats did not overlap each other and edge boundaries did not influence plant distributions, the quadrats were located more than 10 m from each other and 30 m from any trail/road edges or forest stand boundaries. The quadrats were split into subplots (NW, SE) with one 2 x 2 m area located in both the northwest and southeast corners (Figure 4.4).

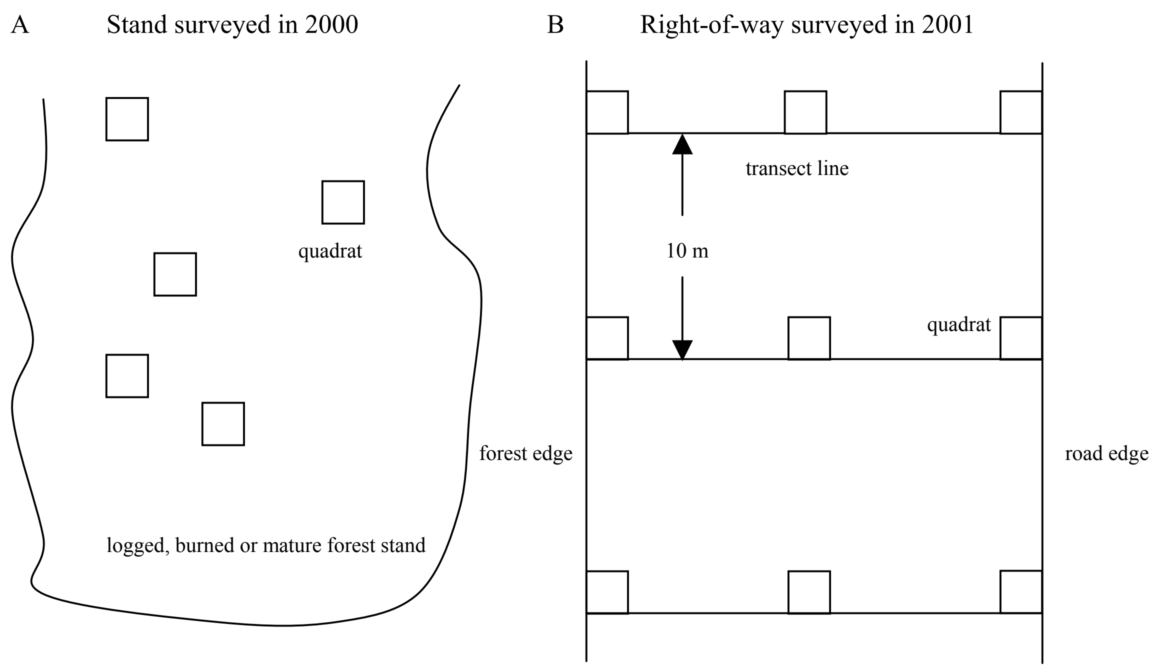

Figure 4.3 Representative quadrat locations within a) harvested, wildfire or mature forest stands and b) the roadside right-of-way.

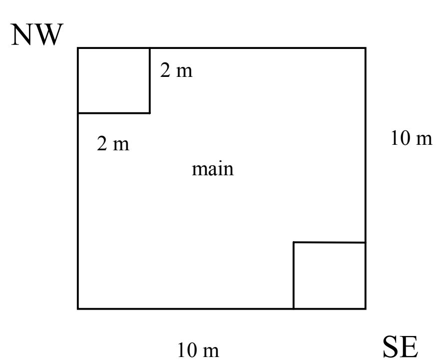

Figure 4.4 A diagram of the quadrat and subplot shape and dimensions surveyed in 2000.

The roadside right-of-way surveys were conducted along three parallel transects that were 10 m apart containing a total of nine to twelve 1 x 1 m quadrats (Figure 4.3B). The total number of quadrats for each transect differed because of varying right-of-way widths. Quadrats were located a minimum of 2 m apart from each other. If the right-of-way was more than 15 m in width twelve quadrats were surveyed. Each transect had one quadrat located at the roadside and forest edges while the remainder of the quadrats were positioned at equal distances from each other. Distances were measured to ensure quadrats were properly placed. Edge quadrats were placed with the nearest quadrat boundary, depending on the right-of-way orientation, contacting the road or forest edge.

Sites surveyed in 2001 were located in roadside right-of-ways and were positioned and labelled according to sites surveyed in 2000. Sites were positioned this way to ensure an accurate measurement of the distance to the road edge. The first right-of-way transect was placed in a parallel position to the southern edge of its associated site surveyed in 2000 (Appendix A). The remainder of the transects were positioned along the right-of-way at 10 m intervals in a manner that coincided with the previously surveyed stand. For example, if the stand had a north/south orientation the first transect would be positioned parallel to the southern edge of the stand and the remainder of the transects would be positioned northward of the first transect. Transect locations were selected in this manner to ensure the surveying was conducted in areas that were likely to be exposed to a similar environment and dispersal vectors. This procedure of transect placement remained consistent for all right-of-way sites, even though some sites had an associated site surveyed in 2000 that was greater than 2 km from the road edge.

Surveying in the right-of-way adjacent to site 54 was not completed because this site coincided with the right-of-way survey location for site 55. Thus roadside right-of-way survey sites were reduced to 20 sites.

Data collected from each quadrat in the roadside right-of-way, recently disturbed and mature forest included the cover and stratum layer for every plant species. Stem counts were completed for exotic species. Plant cover was recorded using a percentage cover class that is based on the cover-abundance scale in Table 4.2. This scale is a modified Braun-Blanquet scale used by the Saskatchewan Forest Ecosystem Classification program (Wright, pers. comm.). The plant stratum was recorded according to the codes and height ranges in Table 4.3. Plants were identified to the species level while certain plant groups (i.e., willows and sedges) were documented at the genus level. Plant species were collected for identification by the W.P. Fraser Herbarium staff at the University of Saskatchewan. Plant names and exotic species status are designated by Moss (1994).

Table 4.2 Cover-abundance scale used to estimate plant cover and quantify cover codes.

Cover code	percentage cover	mid-point value
6	> 75%	87.5
5	50-75%	62.5
4	25-50%	37.5
3	15-25%	20
2	5-15%	10
1	1-5%	2.5
+	< 1% - more than one	0.5
R	< 1%- one stem	0.25

Table 4.3 The stratum (trees, shrubs and herbs) layer assigned to each species and associated codes.

Vegetation	Height range	Layer code
trees	> 10 m (dominant in canopy)	1
trees	> 10 m (subdominant in canopy)	2
tree & shrubs	> 2m to <10 m	3
tree & shrubs	.5 m to 2 m	4
tree & shrubs	< 0.5 m	5
herbs & graminoids	Variable	6

The treatment of the data collected in 2000 differed from the surveying in roadside right-of-ways because of the presence of subplots. Every species within the 2 x 2 m subplot had a percentage cover and stratum layer assigned. Plants that were within the main plot were assigned a percentage cover and stratum layer only if they did not occur in either subplot. Quadrats that were in the roadside right-of-way did not have a subplot so that cover was estimated using the entire 1 x 1 m quadrat. The stem counts of exotic species were completed using the entire area for both 10 x 10 m and 1 x 1 m quadrats. Additional surveying of the canopy species occurred at sites that were not recently disturbed or within the right-of-way. Three representative trees were selected, based on the average diameter at breast height [dbh], to typify the canopy species in the quadrat. Height was measured for these trees using a clinometer and dbh was recorded with a diameter tape. Increment cores were taken to determine tree age.

4.2.2 Site Attributes

The physical attributes of the quadrats and site were recorded to further describe sites surveyed in 2000. The attributes recorded included the topographic position, aspect, seepage and primary water source as described in Beckingham et al. (1996). Additional data measured at every quadrat included depth of the soil organic layer, soil texture of the B horizon and depth to the C horizon. These data were recorded at three random locations along a diagonal line splitting the quadrat into equal halves. The soil texture of the quadrat was observed at the B horizon (approximately 20 to 40 cm below the soil surface) to assess the relative proportion of sand (2.0 to 0.05 mm), silt (0.05 to 0.002 mm) and clay (< 0.002 mm) at the effective rooting zone for most plants. These site attributes were used to determine the quadrat drainage and moisture classes as described by Denholm and Schut (1993). This moisture and drainage classification was chosen because it considered soil depth and effectively represents the soil moisture regime. Additional data collected at the logged sites only included the depth of the disc trenches (post-harvest planting trench) and whether stand maintenance occurred. Site attributes were not recorded for the roadside right-of-way surveys as a result of road and ditch construction. The right-of-ways have an altered soil profile providing an unsuitable comparison with sites surveyed in 2000. Insufficient resources and time also contributed to the lack of site surveying in these quadrats.

Each quadrat also had a GPS location (WGS 84 datum) recorded in the northwest corner of the quadrat (Appendix B). The readings included the latitude, longitude, estimated positional error (EPE), date and time of the recording. Quadrat locations were recorded in degrees, minutes and decimal seconds. The GPS data were not corrected to a base station as the GPS unit (Garmin 12XL) was unable to do so.

Light readings (W/m^2) were taken approximately one metre from the ground at ten random locations inside each quadrat surveyed in 2000. The readings were recorded with a Li-Cor light meter (Model LI-250) that measured the photosynthetically active radiation (PAR). These readings were compared to a measurement taken in the nearest clearing immediately following the recordings within the quadrat. The mean quadrat light reading was compared to its corresponding clearing value and a percentage of full light exposure was calculated. The amount of light in a right-of-way was not recorded as consistent maintenance has removed all overhead tree or shrub species greater than 1 m in height.

To examine the influence of the proximity of the quadrat to a source of exotic species, the distance from the roadside edge to each quadrat was determined. The distance was measured on forest inventory maps using the shortest corridor (i.e. bush road, well-used trail or waterway) connecting the survey site and right-of-way. The values obtained were used to determine the relationship between the distance from a maintained road and right-of-way and exotic plant distribution.

4.3 Describing the Understory Vegetation

Density, frequency and mean cover were used to describe the vegetation once surveying was completed. Density (number of stems per m^2) was determined by dividing the exotic stem count for a quadrat by the total area of the quadrat.

Frequency was determined to assess the probability of finding an exotic species in a quadrat placed within the sample area. The frequency values were created by dividing the number of quadrats that contain exotic species by the total number of quadrats surveyed at the site. Frequencies of exotic species were determined by adding together the total number of quadrats that contained the species and dividing by the total number of quadrats in the treatment or the study.

Midpoints of the cover classes were used to examine the relationship between the area occupied by native plants and that occupied by exotic plants. The quadrats surveyed in 2000 had two subplots and mean cover was calculated if a species was observed in both subplots. If a species was observed in one subplot only it was treated in a similar manner with the mean value calculated using a zero cover value for the subplot that did not have the species. Therefore plant cover for each species had one value in the subplots. To obtain a cover for each species for the entire quadrat, the mean value from the subplots was added together with the cover from the main plot. The exotic and native cover values for the entire quadrat were obtained by adding together the calculated cover values for each group of species observed within the quadrat. If the species was observed in the main plot only, no mean cover was calculated. The native and exotic cover values within the roadside right-of-way were calculated in a similar manner but did not include a mean subplot value as there were no subplots in these quadrats. Once plant cover was assigned for each quadrat, a mean value was used to describe the native and exotic plant cover for each quadrat, site and treatment examined in the survey. Confidence intervals (95%) are included with each mean value calculated.

4.4 Species Ranking

Exotic species observed during the survey (Appendix C) were assessed according to their potential to become detrimental to ecosystem composition, structure and function. The assessment was completed with the ranking system developed by Hiebert and Stubbendieck (1993). The system assigns values to species on the basis of their significance of impact and feasibility of control. The significance of impact is the current level of impact and the innate ability of the species to become a pest. Various attributes are addressed including species abundance, distribution relative to disturbance regime, mode of reproduction, germination requirements and competitive ability. The feasibility of control examines abundance of plant populations within the area (number and size of population), ease of control (nature of seed banks, level of effort required and side effects of chemical/mechanical control measures) and urgency of control. The assigned values were added together to obtain totals for the significance of impact and feasibility of control. The totals for each observed exotic plant species are located in Appendix D. Species were then plotted on a graph to categorize each species according to its probable threat and amount of control required to reduce or eliminate it.

Chapter 5
Results

5.1 Exotic Plant Distribution

A total of 23 exotic species (Appendix C) were observed in this study with 12 perennial species, 4 annuals and 7 species possessing the ability to live as winter annuals, biennials or short lived perennials. No exotic trees or shrubs were observed during this study. The majority of the species belong to the Gramineae (9 species), Leguminosae (7) and Compositae (5) families. One species each was also observed in Plantaginaceae and Boraginaceae. The families with the majority of exotic species in this study are among the largest in the world and substantially contribute to the total number of exotic species in local floras (Pysek, 1998a).

5.1.1. Seeded and unseeded roadside right-of-ways

Twenty-two exotic plant species were observed in the roadside right-of-ways (Table 5.1). The majority of the species are common along roadsides while a small number are common in waste grounds and fields (Moss, 1994). A perennial life-form (12 species) is dominant among the right-of-way species while annual (4) or biennial (6) life cycles were less common. Surveying was unbalanced as a result of an unequal number of quadrats surveyed per treatment. The surveying within the right-of-way had six seeded sites while fourteen sites were not seeded with exotic species.

Seeding right-of-ways after road construction is done to promote the re-vegetation of the right-of-way. Six out of eleven exotic species persisted since the initial seeding over 30 years ago within the roadside right-of-ways. The exotic species present in the seed mixtures that persisted within the right-of-way quadrats included *Bromus inermis*, *Festuca rubra*, *Medicago sativa*, *Poa pratensis*, *Trifolium hybridum* and *T. pratense*. These species are commonly used in various agricultural and landscaping applications and were observed in both seeded and unseeded right-of-ways.

Table 5.1 Exotic species observed in roadside right-of-way quadrats (n = 207). Confidence intervals (± 95%) accompany the mean values.

Species name	Common name	Density (stems/m^2)	Frequency (%)	Average Cover (%)
Agropyron repens	quack grass	4	5	13 ± 9
Bromus inermis	smooth brome	17	31	9 ± 3
Cirsium arvense	Canada thistle	0.2	4	3 ± 3
Crepis tectorum	annual hawksbeard	0.03	2	0.3 ± 0.1
Festuca rubra	creeping red fescue	31	36	12 ± 3
Lappula squarrosa	blue bur	0.02	1	0.4 ± 0.2
Lolium persicum	Persian darnel	0.02	0.5	0.5 ± 0
Matricaria matricarioides	pine-apple weed	0.2	4	0.5 ± 0.1
Medicago lupulina	black medick	0.6	9	2 ± 1
Medicago sativa	alfalfa	3	7	14 ± 9
Melilotus alba	white sweet clover	3	23	2 ± 1
Melilotus officinalis	yellow sweet clover	0.1	2	0.9 ± 0.8
Panicum capillare	witch grass	0.04	0.5	0.5 ± 0
Phleum pratense	timothy	0.2	3	1.1 ± 0.8
Plantago major	common plantain	0.6	15	1.1 ± 0.8
Poa compressa	Canada bluegrass	0.4	3	2 ± 3
Poa pratensis	Kentucky bluegrass	23	10	19 ± 8
Sonchus arvensis	perennial sow thistle	4	43	2.4 ± 0.8
Taraxacum officinale	common dandelion	8	73	2.2 ± 0.6
Trifolium hybridum	alsike clover	17	45	5 ± 1
Trifolium pratense	red clover	2	16	1.9 ± 0.8
Trifolium repens	white clover	0.6	4	2 ± 2

The seeded right-of-ways had the largest number of exotic species. Right-of-ways that were seeded had an average exotic density (± 95 % confidence interval) of 123 ± 35 stems/m^2, a frequency of 97% and an average cover of 19 ± 6 %. Unseeded right-of-way quadrats contained an exotic density of 113 ± 27 stems/m^2, a frequency of 91% and a cover of 13 ± 3 %. Native cover in seeded and unseeded right-of-ways were similar with an average cover of 19 ± 5 % in seeded right-of-ways and 19 ± 3 % in unseeded right-of-ways.

Comparisons of average exotic plant densities among seeded and unseeded right-of-way quadrats failed to show significant differences (Mann-Whitney U-test, n = 207, $p > 0.05$). The exotic cover in the seeded and unseeded right-of-way also showed no statistically significant differences (Mann-Whitney U-test, n = 207, $p > 0.05$). As similar values were observed for exotic frequencies and native cover and no significant differences were observed among exotic densities and cover, the differences in exotic plant distributions between seeded and unseeded right-of-ways appear to be negligible.

Further examination of the individual seeded species present in the right-of-way (*Bromus inermis*, *Festuca rubra*, *Medicago sativa*, *Poa pratensis*, *Trifolium hybridum*, and *T. pratense*) also failed to show statistically significant differences. Comparisons of mean exotic densities among seeded and unseeded areas for the seeded species observed did not detect significant differences ($p > 0.05$). An examination of the exotic frequencies of these seeded species in seeded and unseeded right-of-ways was highly significant (Mann-Whitney U-test, $n = 20$, U statistic = 3976.5, $p < 0.001$) for *Medicago sativa* only. This species was observed in 2 % of the unseeded right-of-ways and 17 % of the seeded right-of-ways.

Exotic species in the roadside right-of-ways had the highest densities and frequencies within the study areas. *Taraxacum officinale* had the highest frequency (present in 73% of the right-of-way quadrats) followed by *Trifolium hybridum* (45%), *Sonchus arvensis* (43%), *Festuca rubra* (36%) and *Bromus inermis* (31%). The stem counts of exotic species ranged from 1 to 800 stems/m^2 in the right-of-ways. The highest exotic densities in the right-of-ways were *Festuca rubra* (31 stems/m^2), *Poa pratensis* (23 stems/m^2), *Trifolium hybridum* (17 stems/m^2), *Bromus inermis* (17 stems/m^2) and *Taraxacum officinale* (8 stems/m^2).

The quadrats within the right-of-way were examined to determine the influence of the roadside and forest edge on the vegetation. The quadrats within the roadside right-of-ways were 1 to 30 m from the road edge. The roadside quadrats were < 2 m from the roadside while the forest edge quadrats were the farthest from the road edge being < 2 m from the forest edge. Quadrats that were located in the "middle" of the transect had an average distance of 10 ± 1 m and a range of 4 to 24 m from the roadside. A decreasing frequency of exotic species was observed as the distance from the road edge increases (Figure 5.1). Similar trends were observed with the exotic density. For example, the average exotic density was 150.1 ± 42.8 stems/m^2 at the road edge while quadrats that were mid-way along the transect had 125.3 ± 33.6 stems/m^2 and quadrats that were closest to the forest edge had an exotic density of 70.1 ± 34.4 stems/m^2.

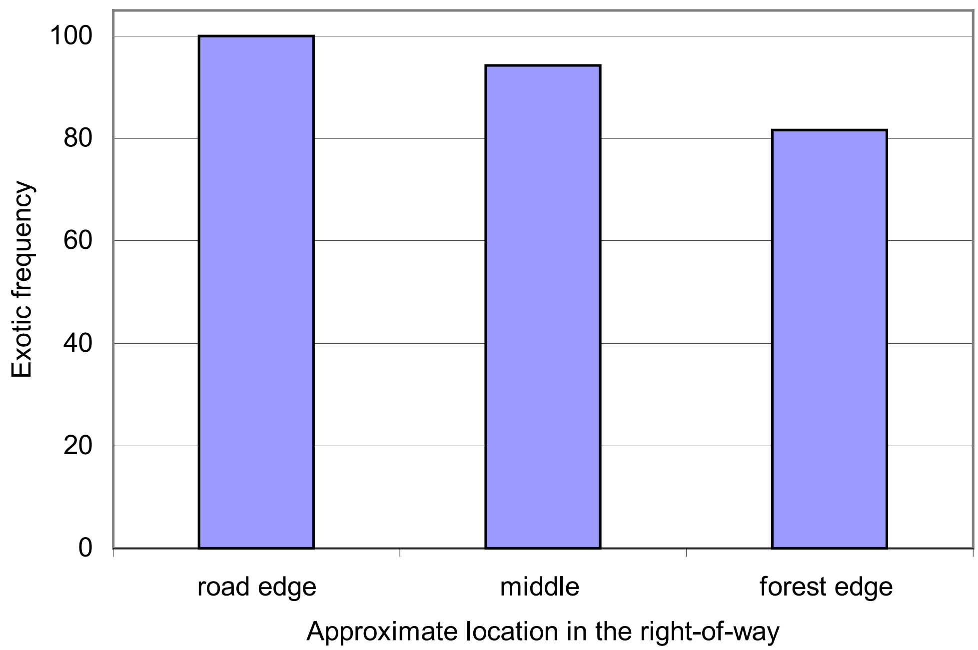

Figure 5.1 Exotic plant frequency and the approximate location in the roadside right-of-way.

The data indicate that an increase in distance from the road edge causes a corresponding decrease in the amount of exotic species. Highly significant ($p < 0.001$) regressions were observed between the distance from the roadside edge and the mean exotic density and frequency within right-of-way areas (Table 5.2). The distance from the roadside edge was used as the independent variable in the analyses. Average native species have an increase in mean cover (5.7 ± 2.2 % to 25.7 ± 5.1 %) as surveying extends towards the forest edge.

The average native plant cover (18.6 ± 2.6 %) in the right-of-way quadrats was similar to the average exotic plant cover (17.5 ± 2.8 %). An ANOVA test confirms that no significant differences were observed between the native and exotic cover in the roadside right-of-way ($n = 414$, $p < 0.552$). Native species frequencies were also similar to exotic species frequencies in the right-of-ways. *Fragaria virginiana* was observed in 47% of quadrats, *Achillea millefolium* in 43%, and *Agropyron trachycaulum* in 34%. Native species were not as prevalent in roadside right-of-ways when compared to other survey sites. A regression examining the relationship between native and exotic cover within the right-of-ways was highly significant ($r^2 = 0.16$, $n = 207$, $p < 0.001$). It appears that an increased exotic cover causes a decrease in the native cover within the roadside right-of-way. Appendix E contains a list of the native species observed in the study sites.

Table 5.2 Highly significant results from regression analyses examining the distance from the roadside edge and various measures of vegetation (n = 207).

	Linear regression	
	r^2	F statistic
average exotic density	0.07	16.05 **
exotic cover	0.12	28.66 **
native cover	0.15	37.14 **

** ($p < 0.001$)

5.1.2. Recently harvested and wildfire areas

Exotic plants were more abundant in harvested sites compared to wildfire sites. Mann-Whitney U- tests on the exotic densities (n = 60, $p < 0.01$) and cover (n = 60, $p < 0.02$) show a significant difference among the treatments (Table 5.3). Logged quadrats had a mean exotic density of 0.4 stems/m^2 ± 0.3 and mean cover of 1.7 ± 0.9 % while wildfire quadrats had a mean density of 0.07 ± 0.04 stems/m^2 and mean cover of 0.6 ± 0.3 %. The exotic frequency among harvested and wildfire areas shows a different trend. ANOVA indicated that the exotic frequencies were not significantly different between harvested and wildfire areas. Exotic species occur in 80% of the recently harvested quadrats while 63% of the recently disturbed wildfire quadrats contained at least one exotic species.

Table 5.3 A comparison of the quantities of exotic and native plants in the recently harvested and wildfire treatments. Mean values are shown with 95 % confidence intervals.

	harvested areas	wildfire areas	Test statistic	P value
mean exotic density (n = 60)	0.4 ± 0.3 stems/m^2	0.07 ± 0.04 stems/m^2	U = 279	0.010
exotic frequency (n = 21)	80 %	63 %	F = 0.228	0.638
mean exotic cover (n = 60)	1.7 ± 0.9 %	0.6 ± 0.3 %	U = 295	0.020
mean native cover (n = 60)	86 ± 21 %.	113 ± 24 %	U = 338	0.098

The composition of the exotic plant community indicates that similar species are invading both these disturbances. Both types of disturbance had six exotic species with species differences only occurring among the grass plants. Table 5.4 summarizes the densities and frequencies for each exotic species present in recently disturbed wildfire and harvested treatments. *Sonchus arvensis* had the highest density with 0.16 stems/m^2 and second highest frequency occupying 53 % of the harvested plots. The most widespread species in cutovers was *Taraxacum officinale*, occurring in 63 % of the harvested quadrats. All of the exotic species observed, except for the *Poa* species, are wind-dispersed. These species appeared to have a patchy distribution with lower frequencies of *Poa pratensis*, occurring in 10 % of the logged areas, and *Poa annua* occurring in 3 % of the wildfire sites. *Poa pratensis* showed higher densities (0.14 stems/m^2) than *Poa annua* (0.007 stems/m^2) suggesting the species has better abilities to establish within the study area.

Table 5.4 The exotic species and densities and frequencies in recently disturbed harvested and wildfire quadrats.

Species	Common name	Density (stems/m^2)		% Frequency	
		harvested	wildfire	harvested	wildfire
Cirsium arvensis	Canada thistle	0.018	0.008	27	17
Crepis tectorum	annual hawksbeard	0.021	0.012	30	20
Poa annua	annual bluegrass	0	0.007	0	3
Poa pratensis	Kentucky bluegrass	0.14	0	10	0
Sonchus arvensis	perennial sow thistle	0.16	0.013	53	20
Taraxacum officinale	common dandelion	0.06	0.026	63	50

Sonchus arvensis was the only species that showed a significant difference between harvested and wildfire sites ($n = 60$, U statistic = 780, $p < 0.018$). Analyses examining the differences between exotic species in harvested and wildfire quadrats were limited to the exotic species that were wind dispersed as these species were observed in both treatments.

The native plant cover in recent wildfire and harvested sites was greater than in right-of-way sites. Wildfire sites had the highest native cover among all treatments with an average of 113 ± 24 %. An examination of the influence of exotic cover on native cover produced a highly significant regression ($r^2 = 0.25$, $n = 30$, $p < 0.001$) within recently disturbed wildfire treatments. An increased native cover appears to limit the cover of exotic species. Recently harvested sites had an average native cover of 86 ± 21 %. Statistically significant differences between native cover in recent wildfire and harvested sites were not observed (Table 5.5). A regression examining exotic and native plant cover in harvested quadrats did not have significant results ($r^2 = 0.009$, n = 30, p = 0.61).

Site attributes of each disturbance type may have also contributed to the differences in exotic densities between recently harvested and wildfire sites. The season of timber removal, planting and site preparation varied among the harvested sites from summer, fall and winter months (Table 5.5). Only two sites received stand maintenance which involved the removal of vegetation above 1 m in height. Recently harvested sites in this study were clearcut, removing all merchantable timber. This harvesting occurred between 1989 and 1995 and had a mean size of 53 ha (Table 3.2). A roadway had been built into each site during harvesting activities. All harvested areas had disc-trenching completed as a post-harvest treatment and were planted with white spruce (*Picea glauca*) seedlings. Recently disturbed wildfire study sites were characterized by high intensity crown fires. Two wildfires, Waskesiu and Cobra, occurred inside Prince Albert National Park, while the Monday and Muskeg fires were located within the Provincial Forest. The average size of the wildfire disturbances was approximately 20,042 ha (Table 3.2). The wildfire areas had various densities of standing dead trees with patches of blow down. All of the fires were accessible by a trail or roadway, while the Waskesiu and Muskeg fires were contiguous with the seeded right-of-way adjacent to highway 2. All of the wildfire sites also had suppression activities to extinguish the wildfire.

Table 5.5 Attributes of recently harvested sites. Stand maintenance involved removing shrubs and small trees with thinning saws and occurred as needed. Site preparations were completed by a disc-trencher and sites were re-planted with white spruce (*Picea glauca*) seedlings after harvesting operations.

site ID	season and year of harvest	planting season	season of site preparation	stand maintenance	average trench depth ± C.I. (cm)
55	winter 1995	spring	summer	absent	19.9 ± 6.3
9	winter 1990	fall	spring	present	29.3 ± 6.3
43	winter 1991	fall	fall	absent	24.6 ± 4.4
51	summer 1989	spring	fall	present	20.7 ± 3.8
53	winter 1992	spring	fall	absent	17.7 ± 10
54	winter 1995	spring	summer	absent	27.8 ± 5.2

Light levels among recently harvested and wildfire areas had significant differences (Mann-Whitney U-test, $n = 60$, U statistic = 290, $p < 0.018$). The average light levels in recent cutblocks were 73 ± 8 % while recent wildfires had 59 ± 9 % of full sunlight exposure. This increased light exposure may contribute to additional resource availability, although a regression failed to indicate a significant relationship between light and exotic density ($r^2 = 0.002$, $n = 60$, $p = 0.762$,). The depth of the organic layer (LFH horizons) was also compared in recently harvested and wildfire sites. The average depth of the organic layer in harvested quadrats was 6.0 ± 0.9 cm while the wildfire quadrats had an average depth of 5.4 ± 0.8 cm. An ANOVA comparing the depth of the organic layer among these treatments failed to identify any significant differences ($n = 60$, F statistic = 0.912, $p < 0.344$).

The distances from an exotic source were considerably different for recently harvested and wildfire areas. Logged areas were, on average, 1,093 ± 233 m from a provincial highway while wildfire areas were 3,363 ± 1012 m from these maintained roadside right-of-ways. A Mann-Whitney U-test confirms that the differences in distance from a maintained right-of-way between the recently harvested and wildfire quadrats were highly significant ($n = 60$, U statistic = 191, $p < 0.001$). The results suggest that exotic species have a further distance to disperse to wildfire disturbances in this study.

Although differences were detected among exotic densities in harvested and wildfire treatments, these disturbances have similarities with regards to exotic plant frequencies, depth of the organic layer and native plant cover. Additional comparisons can be made among the exotic plants present with regards to species densities and frequencies. Differences among timber harvesting and wildfire are inherent in these disturbance types but the occurrences of similar exotic species suggest that these disturbances have similar vectors of dispersal.

5.1.3. Recently disturbed sites adjacent to seeded and unseeded right-of-ways

To determine if exotic species are dispersing into recent disturbances from seeded right-of-ways, the proximity of recent disturbances to this source of exotic plants was examined. Recently disturbed areas exhibited a scattered distribution of exotic plant species occurring in 72 % of the quadrats and having an average density of 0.2 ± 0.1 stems/m^2. The exotic species observed are common in urban and rural areas and belong to the Compositae and Gramineae families. The Compositae species are capable of producing large amounts of wind-dispersed seeds and were more prevalent. *Taraxacum officinale* was observed in 57% of the recently disturbed quadrats while *Sonchus arvensis* occurred in 38% (Table 5.6). The highest densities in recently disturbed areas were observed with *Sonchus arvensis* with 0.09 stems/m^2, *Poa pratensis* with 0.07 stems/m^2 and *Taraxacum officinale* at 0.05 stems/m^2 .

Table 5.6 Exotic species observed in recently disturbed quadrats (n = 60).

Species name	Common name	Density (stem/m^2)	Frequency (%)
Cirsium arvense	Canada thistle	0.01	22
Crepis tectorum	annual hawksbeard	0.02	25
Poa annua	annual bluegrass	0.003	2
Poa pratensis	Kentucky bluegrass	0.07	5
Sonchus arvensis	perennial sow thistle	0.09	38
Taraxacum officinale	common dandelion	0.05	57

The seed mixture applied (Table 3.3) to the right-of-way contains species that have the potential to spread into disturbance areas. These areas are connected to right-of-ways via roadways, trails and waterways and were disturbed by either wildfire or timber harvesting between 1989 and 1998. Although the potential exists for all species in the seeded mixture to spread throughout the area, only one species from the mixture was observed beyond the right-of-way. *Poa pratensis* was observed in recently harvested areas that were adjacent to both seeded and unseeded right-of-ways. The frequency (5 %) and density (0.07 stems/m^2) of this species was reduced in the recently disturbed quadrats as compared to right-of-ways (frequency of 10 % and 23 stems/m^2). There were no significant differences in exotics between recently disturbed sites that were adjacent to seeded right-of-ways and sites that were not adjacent to seeded right-of-ways. Table 5.7 shows the lack of significant ($p > 0.05$) results between the treatments with regards to exotic density, cover and frequency. Recently disturbed quadrats that were adjacent to seeded areas (n = 20) had an exotic frequency of 55% and an average exotic density of 0.2 ± 0.1 stem/m^2. Quadrats that were "remote" from seeded right-of-ways (n = 40) had an exotic frequency of 80% and average density of 0.3 ± 0.2 stems/m^2.

Table 5.7 A comparison of exotic plants in quadrats that were adjacent to seeded right-of-ways to quadrats that were adjacent to unseeded right-of-way. Results from statistical analyses examining the differences among the treatments are also included.

	Seeded	Unseeded	Test statistic	P-value
average exotic density	0.2 ± 0.1	0.3 ± 0.2	U = 331	0.270
exotic frequency	55 %	80 %	F = 1.684	0.224
average exotic cover	0.9 ± 0.5 %	1.3 ± 0.7 %	U = 324	0.227
average native cover	96 ± 31 %	102 ± 19 %	n/a	n/a

Although the exotic densities, frequencies and cover were similar, the length of the corridor connecting the site to the roadside right-of-way was different for seeded and unseeded treatments. The average distance of recently disturbed quadrats that were adjacent to seeded right-of-ways was 466 ± 169 m. Recently disturbed quadrats that were "remote" from an exotic source had an average distance of 3,089 ± 670 m from the roadside right-of-way. Highly significant differences were observed (Mann-Whitney n = 60, U statistic = 20, $p < 0.001$) among the distances from recently disturbed quadrats that were adjacent to seeded right-of-ways and quadrats that were "remote" from a source of exotic species. Further analyses involving a linear regression failed to provide significant results between the distance from right-of-way (independent variable) and the exotic plant frequencies (dependent) in recently disturbed quadrats (n = 21, F statistic = 0.221, $p < 0.648$,). Proximity of recently harvested or wildfire areas to a seeded right-of-way does not appear to influence exotic plant distributions.

Further examination of the disturbance type indicates that no exotic species were observed in recently disturbed wildfire areas that were adjacent to seeded right-of-ways. Wildfire quadrats that were "remote" from an exotic source had a frequency of 76 ± 29

% and density of 0.08 ± 0.04 stems/m^2 (Table 5.8). Recently harvested areas that were "remote" from an exotic source had larger amounts of exotics as exotic frequency was 87 ± 19 % with 0.6 ± 0.6 stems/m^2. Harvested quadrats adjacent to seeded right-of-ways had less exotic species with a frequency of 73 ± 26 % and 0.25 ± 0.19 stems/m^2. No significant differences among exotic densities (nested ANOVA n = 30, F statistic = 0.832, $p < 0.370$,) and frequencies (ANOVA n = 6, F statistic = 0.80, $p < 0.422$) were observed in recently harvested areas that were either seeded with a mixture containing exotic species or were distant from this source of exotic species.

Table 5.8 The quantities of exotic plants in harvested and wildfire quadrats that were adjacent to seeded and unseeded right-of-ways.

	timber harvesting (n = 30)		Wildfire (n = 30)	
	seeded	unseeded	seeded	unseeded
exotic density	0.25 ± 0.19	0.6 ± 0.6	0	0.08 ± 0.04
exotic frequency	73 ± 26 %	87 ± 19 %	0	76 ± 29 %
exotic cover	1.4 ± 0.67 %	2.3 ± 1.7 %	0	0.7 ± 0.4 %

5.1.4. Recently disturbed and mature forest

Considerable differences were observed among the recently disturbed and mature sites in this study. The recently disturbed areas had exotics in 72 % of the quadrats while the mature sites had a frequency of 13%. Average exotic densities were larger in recently disturbed quadrats at 0.2 ± 0.1 stems/m^2 compared to 0.002 ± 0.002 stems/m^2 observed in mature quadrats. And average exotic cover was higher at 1.1 ± 0.5 % in recently disturbed quadrats compared to mature quadrats at 0.05 ± 0.04 %. Statistical analyses using Mann-Whitney U-tests confirm that differences among exotic frequencies and densities were highly significant ($p < 0.001$) when comparing recently disturbed and mature areas (Table 5.9).

Table 5.9 The Mann-Whitney U-test results comparing the recently disturbed and mature treatments.

Factor	Recently disturbed	Mature	Sample size	U statistic
average exotic density (stems/m^2)	0.2 ± .02	0.002 ± 0.002	105	474.5 **
exotic frequency (%)	72	13	21	8.5 **
average distance (km)	2.2 ± 0.5	0.5 ± 0.2	105	457.0 **
average light (% of full exposure)	66 ± 6	19 ± 4	105	130.0 **

** ($p < 0.001$)

Mature sites contained the smallest density and frequency of exotic species while native plants occupied the majority of survey quadrats. The average cover for native species in mature sites was 89.7 ± 12.2% whereas the average exotic cover was 0.05 ± 0.04 %. The frequency of native species in mature sites was also increased as *Cornus canadensis* and *Linnaea borealis* occupied 100% of the quadrats followed by *Maianthemum canadensis* and *Rubus pubescens* (96%) and various moss species (93%). The mature sites had only two exotic species present compared to 100 native plant species. *Taraxacum officinale* had a frequency of 9% and *Poa compressa* was observed in 4% of the mature quadrats. This treatment also had the lowest exotic densities. *Taraxacum officinale* had 0.0013 stems/m^2 (a total of 6 stems) and *Poa compressa* had 0.0011 stems/m^2 (5 stems). Minimal exotic species cover and elevated native species cover in mature sites suggests that exotics are unlikely to become dominant in areas disturbed more than 60 years ago.

Although larger densities and frequencies were observed in recently disturbed quadrats, these sites were largely occupied by native species. Average native cover was 99.5 ± 16.3% whereas exotic cover was 1.1 ± 0.5%. These large differences between natives and exotic cover are coupled with higher frequencies of native species. *Epilobium angustifolium* was observed in 100% of the quadrats followed by *Populus tremuloides* (95%), *Cornus canadensis* (93%), various *Carex* species (85%), *Calamagrostis canadensis* and various moss species (83%). Exotic frequencies did not approach these levels as *Taraxacum officinale* had the highest frequency at 57% followed by *Sonchus arvensis* at 38%. Table 5.8 shows the density and frequency of exotic species in recently disturbed areas. The amount of native species greatly surpasses exotic species in recently disturbed areas as 6 different exotic species were observed compared to 137 native species.

Recently disturbed and mature treatments also had highly significant differences ($p < 0.001$) between the distance from the roadside right-of-way to the treatment quadrats (Table 5.9). The recently disturbed quadrats had an average distance of 2,215 ± 548 m from the roadside right-of-way while the mature quadrats were closer to the right-of-way having an average distance of 548 ± 279 m.

The site attributes did not appear to influence the distribution of exotic species. Each survey site was selected to occur, either presently or previously, in mixedwood stands and luvisolic soils. Spearman correlations examining the relationship between exotic densities ($n = 105$) and the depth of the organic layer ($r = - 0.138$, $p < 0.161$) and to the C horizon ($r = - 0.026$, $p < 0.791$) were not significant. Significant Spearman correlations were also lacking when the exotic frequency ($n = 21$) was examined with the depth of the organic layer ($r = - 0.370$, $p < 0.099$) and to the C horizon ($r = 0.031$, $p < 0.893$).

In September, 2000 site attributes were measured for 105 recently disturbed and mature quadrats that had floristic surveys completed that year. The average depth of the organic layer (LFH) was 6.6 ± 0.5 cm with ranges of 1.8 to 10.8 cm. The average depth to the C horizon was 58 ± 4 cm with a range of 25 to 125 cm. The texture at the B horizon varied from loamy sand to silty clay loam. These soil attributes were needed to determine the quadrat drainage and moisture classification. The classes of drainage varied from moderately well to well drained while the moisture regime varied from dry to fresh. The drainage classes observed indicate that the soils can be deficient or contain an excess of soil moisture for short periods of time. The moisture classes indicate that water is either very rapidly to readily removed in relation to supply and the soil is moist for either brief, short or moderately short periods of time following precipitation (Saskatchewan Environment, 2004). Precipitation was the primary water source for all quadrats surveyed.

The amount of light the quadrat received may have influenced exotic species distributions. The average light level in recently disturbed forest was 66 ± 6 % of full exposure while average values for mature forests were 19 ± 4 %. Differences detected by a Mann-Whitney U-test were highly significant ($p < 0.001$) among the light levels of recent disturbed and mature forests (Table 5.9). Linear regressions examining the exotic density and light exposure did not produce significant results ($p > 0.05$) in recently disturbed ($n = 60$) or mature quadrats ($n = 45$). It appears that light exposure did not cause an increase in exotic densities in recently disturbed quadrats.

Although the decreased light exposure in mature quadrats did not appear to cause a reduction in exotic plant density, the intact canopy has lowered the amount of light that reaches the quadrat. The canopy trees were characterized by *Populus tremuloides* (81 samples) and *Picea glauca* (33). Table 5.10 indicates that the average age of the canopy species was 60 ± 4 years with an average height and diameter at breast height (dbh) of 18.1 ± 0.7 m and 22 ± 1 cm, respectively. Three representative canopy trees were sampled from each quadrat with a total of 135 samples.

Table 5.10 Average measurements (± 95 % confidence interval) from the canopy tree species of the mature quadrats.

Canopy species	dbh (cm)	height (m)	age (years)	trees sampled
Abies balsamea	25.9 ± 7.9	18.9 ± 6.4	62 ± 20.8	3
Betula papyrifera	21.7 ± 1.7	14.3 ± 2.3	75.3 ± 11.6	9
Picea glauca	26.5 ± 4.2	17.5 ± 1.7	64.5 ± 9.9	33
Populus balsamifera	24.3 ± 3.6	17.6 ± 2.3	56.2 ± 8.8	9
Populus tremuloides	20.7 ± 1.2	18.8 ± 0.8	56.6 ± 3.6	81

Increased native plant cover appears to have inhibited the cover of exotic species. Native plants have a reduced cover in roadside right-of-ways whereas the amounts in recently disturbed and mature forests were larger. Average cover of native plants in the right-of-way was 19 ± 3 % while exotic cover was 18 ± 3 % (Table 5.11). Recently disturbed and mature areas had the largest amount of native cover (100 ± 16 % and 90 ± 12 %, respectively) and the lowest exotic cover (1.1 ± 0.5 % and 0.05 ± 0.04 %, respectively). A linear regression has indicated a highly significant ($r^2 = 0.174$, $n = 312$, $p < 0.001$) inverse relationship between the native (independent) and exotic plant cover (dependent) within the study area (Figure 5.2). Native plants appear to increase in cover as the cover of exotic species decreases.

Table 5.11 Summaries of exotic density, frequency and cover in three land uses examined in this study. Mean values are shown with 95% confidence intervals.

	recently disturbed	mature	right-of-way
average exotic density (stems/m^2)	0.2 ± 0.2	0.002 ± 0.002	117 ± 22
exotic frequency (%)	72	13	94
average exotic cover (%)	1.1 ± 0.5	0.05 ± 0.04	18 ± 3
average native cover (%)	100 ± 16	90 ± 12	19 ± 3

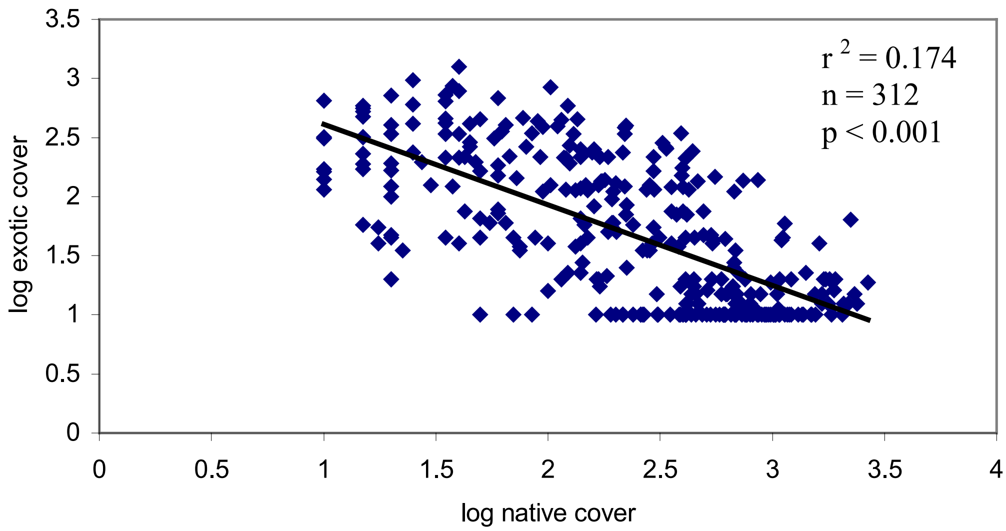

Figure 5.2 A highly significant regression illustrating a decrease in exotic cover as the native cover increases. The values in the graph have transformed into log10 to better visualize the relationship.

5.2 Exotic Species Ranking

Figure 5.3 shows the twenty-three observed exotic plant species that were ranked according to their potential to become detrimental to ecosystem compostion, structure and function. Appendix C contains a list of these exotic species and associated abbreviations. The ranking identified 14 species that were a lesser threat and easy to control, 8 species that were a lesser threat and hard to control and one species, *Bromus inermis*, that was ranked as a serious threat and hard to control. Appendix D contains a description and assigned value for each attribute used in the ranking. Characteristics common to all the observed exotic species include: seeds that remain viable in the soil for at least 3 years, susceptibility to management practices (biological, chemical, and cultural methods of control) and the ability to reproduce sexually one or more times a year. The exotic plants observed are native to European, Asian and African habitats being introduced into North America throughout a 300 year time period (the 1600s to 1900s) (Fairey, 1985 and 1986; Goplen and Gross, 1984; Looman, 1982 and 1983; Royer and Dickinson, 1999).

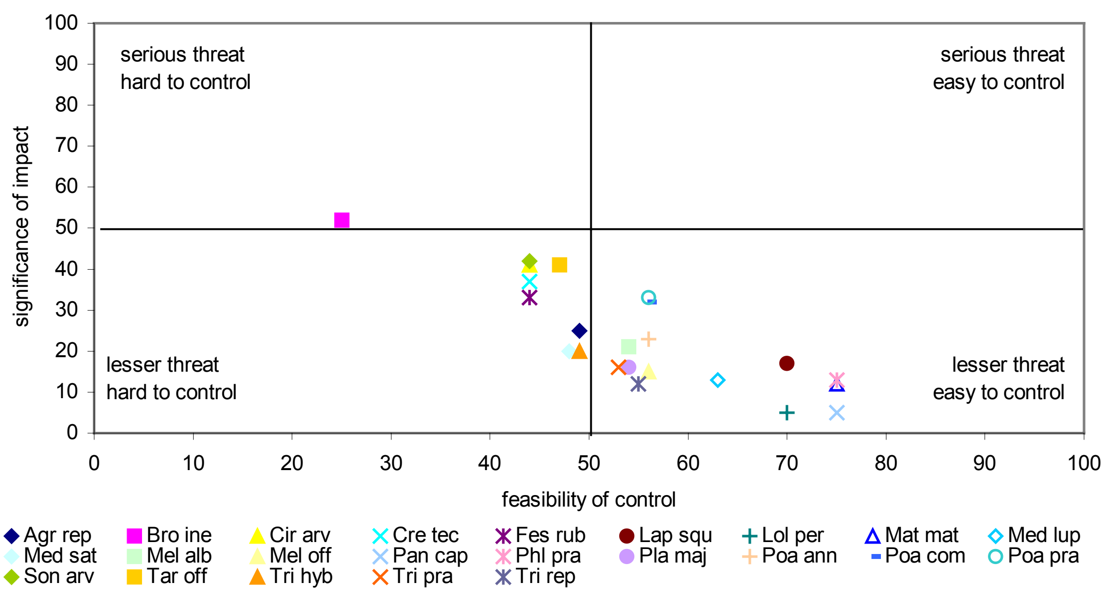

Figure 5.3 Ranking of observed exotic species. A key to the abbreviated species names is given in Appendix C.

5.2.1 Serious threat, hard to control

Bromus inermis or smooth brome grass was ranked as a serious threat and hard to control because of an increased density in the area and its ability to spread aggressively. It was directly seeded into the roadside right-of-ways and was ranked at 25 for its significance of impact and 52 for its feasibility of control or management.

This species was ranked into this category because it is capable of spreading by rhizomatous growth and can flourish in a variety of habitats. It is a perennial cool-season grass that performs extremely well in cultivation in the Dark Brown and Black soil zones of the Saskatchewan prairie ecosystem. Smooth brome grass was observed in roadside right-of-ways survey sites only with a frequency of 21% and density at 0.3 stems/m^2.

This species was also assigned a medium urgency ranking indicating that a delay in action will result in a moderate increase in effort required for successful control. It may become a 'problem' because of its ability to reproduce aggressively, through self-seeding or vegetative spread and potential to invade and modify native plant communities (Butterfield et al., 1996). Removal of the species is relatively difficult because of its strong rhizomatous growth and its ability to withstand chemical control (i.e. glyphosate applications) (Kruger, 1997).

5.2.2 Lesser threat, hard to control

The species in this group were ranked as a lesser threat and hard to control because they were present at low densities, require repeated chemical or mechanical control measures and possess the potential to produce large amounts of seed that may be highly mobile (wind-dispersed). Species in this group include *Taraxacum officinale*, *Sonchus arvensis*, *Trifolium hybridum*, *Festuca rubra*, and *Cirsium arvense* (Table 5.12). The average density (0.2 ± 0.1 stems/m^2) and frequency (21 ± 14 %) were greater than the species ranked as a lesser threat and easy to control. The majority of species in this group are native to Europe while *Taraxacum officinale* and *Sonchus arvensis* originated in Eurasia (Royer and Dickinson, 1999). One species in this group (*Festuca rubra*) has origins in North Africa, Eurasia and Iceland (Looman, 1983). This species is also native in parts of North America including the Allegheny Mountains, Atlantic marsh areas and the coastal ranges of the Rocky Mountains. Moss (1994) has designated *Festuca rubra* as native and exotic but the more specific information provided by Looman (1983) endorses its exotic status in Saskatchewan forests, where it was deliberately introduced.

Table 5.12 Exotic species that were ranked as a lesser threat and hard to control. The impact and control values represent the ranking of the species according the system described in Appendix D.

Species	impact	control	Frequency (%)	density (stems/m^2)
Agropyron repens	25	49	3	0.07
Cirsium arvense	41	44	7	0.01
Crepis tectorum	37	44	6	0.01
Festuca rubra	33	44	24	0.59
Medicago sativa	20	48	4	0.05
Sonchus arvensis	42	44	36	0.12
Taraxacum officinale	39	47	61	0.18
Trifolium hybridum	20	49	27	0.34

Three species (*Festuca rubra*, *Medicago sativa*, *Trifolium hybridum*) in this group were deliberately introduced into the roadside right-of-ways after roadway construction. Other species (*Agropyron repens*, *Taraxacum officinale*, *Sonchus arvensis*, and *Cirsium arvense*) are common in the area and considered weeds in urban and agricultural regions of Saskatchewan. The species surveyed in this group were primarily located in recently disturbed and roadside right-of-ways, except for a reduced presence of *Taraxacum officinale* in mature sites. The presence of these similar species in both right-of-ways and recent disturbances suggests that wind dispersal is a common method of dispersal among these disturbance types. The majority of species from this group that do not have this ability were deliberately introduced. A low urgency level was assigned to these species indicating that a delay in action will result in a small increase in the level of effort required for successful control.

5.2.3 Lesser threat, easy to control

The species recorded as a lesser threat and easy to control were ranked this way because their seeds are typically unable to travel great distances, there are low quantities of individuals and the species are generally not a problem in other ecosystems. Table 5.13 shows the species densities and frequencies present in this category. These species include *Melilotus alba*, *Trifolium pratense*, *Plantago major*, *Poa pratensis* and *Medicago lupulina*. Native habitats of the exotic species in this group can be found in Europe and Asia. The mean density is 0.05 ± 0.07 stems/m^2 while the average frequency of exotic species in this group is 4.5 ± 2.5 %.

Table 5.13 Exotic species that were ranked as a lesser threat and easy to control.

Species	impact	control	frequency (%)
Lappula squarrosa	17	70	0.6
Lolium persicum	5	70	0.3
Matricaria matricarioides	12	75	3
Medicago lupulina	13	63	6
Melilotus alba	21	54	15
Melilotus officinalis	15	56	2
Panicum capillare	5	75	0.3
Phleum pratense	13	60	2
Plantago major	16	54	10
Poa annua	21	56	0.3
Poa compressa	32	56	3
Poa pratensis	33	56	8
Trifolium pratense	16	53	11
Trifolium repens	12	55	3

The majority of the species in this group are weeds of waste grounds, roadsides and fields while a few species are known to have escaped cultivation (Moss, 1994). Although *Trifolium pratense* and *Poa pratensis* were directly seeded into roadside right-of-ways, they have relatively low frequencies and densities. *Trifolium pratense* was observed in 11 % of the quadrats with 0.04 stems/m^2, while *Poa pratensis* had a frequency of 8 % and a density of 0.5 stems/m^2. Although the Poa species had a higher impact on the ecosystem, these species were ranked in to this group because of an increased ease of control, reduced frequency and capability to disperse propagules long distances. *Poa pratensis* is another species with native and introduced elements. Looman (1983) identifies that the species was introduced into America from Europe before 1700 and its spread has been so extensive that it has been considered a native plant. In this study, *Poa pratensis* was deliberately introduced into the roadside right-of-way and will thus maintain its exotic status.

Exotic species in this group were primarily observed in roadside right-of-way survey sites. A low urgency level was also assigned to these species.

CHAPTER 6

DISCUSSION

The central question of this research addressed aspects of the distribution of exotic plants and the threat these species have to become detrimental to the boreal forest. The majority of exotic species observed in this study belong to the largest plant families in the world. Mack et al. (2000) identified the Gramineae, Leguminosae and Compositae families to contribute the most to the total number of exotic species in local floras. The interaction of the exotic species characteristics and invasibility of landscape features has permitted the establishment of 23 exotic plant species. Land use in the study area appears to have contributed to an increased density and frequency of exotic plant species.

6.1 Seeded and unseeded roadside right-of-ways

Although the surveying had unbalanced treatments, the deliberate seeding of exotic species into roadside right-of-ways has contributed to an expanding exotic distribution. Similar amounts of exotic species in unseeded and seeded right-of-ways demonstrates the increased density and frequency of exotic plants throughout the entire road network. The hypothesis that exotic plant species have comparable occurrences in roadside right-of-ways that were seeded and right-of-ways that were not seeded should be accepted.

Six exotic plants have persisted since the deliberate seeding approximately 30 years. These species generally had no differences among their frequencies and densities in seeded and unseeded right-of-ways. Three of these species (*Festuca rubra*, *Medicago* *sativa* and *Trifolium hybridum*) have been ranked as a lesser threat and hard to control, two species (*Poa pratensis* and *Trifolium pratense*) are regarded as a lesser threat and easy to control and one species, *Bromus inermis*, has been identified as a serious threat and hard to control. The occurrence of these seeded species are among the highest for exotic plants that rely on non-wind dispersion (Table 3.3). The seeding and persistence of these species in right-of-way areas indicates that a deliberate introduction has prompted a successful exotic invasion. The continuous habitat created by the consistent maintenance within the right-of-way has promoted this expanding distribution. This maintenance exposes organisms to selective and repetitive forces that favour the establishment and survival of exotic and weedy species (Lugo and Gucinski, 2000).

Activities within both types of right-of-way also appear to reduce dispersal barriers. The right-of-way mowing in both seeded and unseeded right-of-ways surveyed is conducted by the same equipment providing a common dispersal vector. According to Forman and Alexander (1998) roadside mowing also tends to reduce plant species richness and favour exotic plants. The use of vehicles on the adjacent roadway and within the right-of-way will also contribute to the dispersal of exotic plants. Several authors (Milton and Dean, 1998; Parendes and Jones, 2000; Schmidt, 1989; Lonsdale and Lane, 1994) have noted that passenger vehicles will transport propagules on tires and in mud that collects on the vehicle. The use of all-terrain-vehicles (ATV) and snowmobiles contributes to the increased quantity of dispersal vectors as these recreational vehicles pass directly through the right-of-way. These inadvertent vectors of transportation will increase the dispersal distance of the exotic species present in the right-of-way and expose more species to this habitat. Richardson et al. (2000) have suggested that an increased supply of propagules will increase the chance of recruiting offspring thus creating a greater probability of a successful invasion.

The structure and function of the right-of-way contributes to the environment that favours exotic plant species. The linear shape of the right-of-way creates a large boundary-to-area ratio with its dominant features determined by its relationship with the surrounding area (Loney and Hobbs, 1991). Although right-of-ways within the study area were generally contiguous with undisturbed forest, the extensive road network provides a connection to other habitats. The common border of the roadside right-of-way with farmland in the boreal transition forest permits exotic species to disperse northward into the study area. Work completed by Thomas et al. (1996) has identified 10 of the 23 exotic plants observed in this study to occur as weeds in cereal, oilseed and pulse crops in regions south of the study area.

The increase of exotic species in right-of-ways may also be a function of the creation of bare patches within the right-of-way. Although the cover of bare ground was not measured directly, personal observations within the right-of-way identified the increased presence of bare soil compared to the recent disturbances and mature forest surveyed. A reduced total plant cover in the right-of-ways also suggests an increased exposure of bare soil and additional sites for plant establishment. Loney and Hobbs (1991) note that bare areas in right-of-ways provide an appropriate environment for the establishment of opportunistic plants. Maintenance of gravel roadways will contribute to the increase in bare patches as vegetation is consistently buried at the road edge. The increased frequency of exotic species at the road edge (Figure 5.1) illustrates the effectiveness of this increase in disturbed habitat. Increased exposure of bare ground was also observed by Smith and Ulmer (2001) in the right-of-ways located in boreal transition forest of Saskatchewan.

The environment created in the right-of-way by the land use practises will have an increased amount of resources for plant growth and establishment. Regular maintenance removes vegetation greater than 30 cm allowing a full exposure to sunlight. The presence of a ditch within the right-of-way directs precipitation and associated runoff away from the area but not before it is intercepted by the right-of-way vegetation. Winter road maintenance also contributes to increased water availability and concentrations of salt and sand as materials are deposited into right-of-ways.

The increase in resources may have a role in influencing the exotic distributions in this study. Work by Cale and Hobbs (1991) show corresponding increases in soil nutrients (i.e. phosphorous) and the cover and diversity of exotic species in western Australia. Davis et al. (2000) have developed a theory that identifies fluctuations in resource availability as the controlling factor for invasibility by non-resident biota. This theory is well suited to explain the observed increases of exotic species in right-of-way sites in this study. An increase in resources will ultimately promote species that are better adapted to utilize the resources. Work completed by Reiners (1983) and Davis et al. (2000) identifies that exotic plant species are adapted to take advantage of disturbance prone landscapes and an increase in resource availability. The consistent disturbances and inputs from the adjacent roadway contribute to an increase in resource availability promoting the distribution of exotic plant species. The environment of the roadside edge appears to promote exotic species, while the forest edge promotes native species. A highly significant regression ($p < 0.001$) indicates that an inverse relationship exists among the exotic density and frequency and the distance from roadside edge.

The road network has provided a habitat that permits the establishment and dispersal of exotic plants. The roadside right-of-way thus appears to be the initial and main source of introduction for exotic plants into the survey area. Whether the right-of-ways received a deliberate introduction of exotic species is irrelevant as the features of the right-of-way have promoted the distribution of exotic plants throughout the road network.

Although no invasive plant species were observed in the study area, concern about smooth brome grass (*Bromus inermis*) should be raised as this species has the potential to become invasive. A similar native climate and extensive use as a forage crop have contributed to its persistence since the initial introduction. Its native habitat includes the meadows, ungrazed steppes, parklands and open woods of Europe and Siberia (Looman, 1983). Smooth brome has a close association with humans as its been cultivated in Canada for over 100 years and is often a principle component of grass-legume mixtures for both irrigated and non-irrigated pastures. The anticipated climate change may promote increased distributions as a northward shift in vegetation that has been described by Wheaton (1997) will provide more habitat similar to its native environment. This species is invasive in grassland and aspen parkland ecozones in southern Saskatchewan and has successfully invaded closed canopy forests in these areas. The potential exists for this species to spread in a similar manner within Saskatchewan mixedwood stands when climatic change shifts vegetation northward. *Bromus inermis* was not considered invasive at this time because it has not successfully established beyond the right-of-way. It currently does not produce large numbers of offspring at a considerable distance (> 100 m) from the parent plants.

6.2 Recently harvested and wildfire areas

The examination of recently disturbed areas determined that the density of exotic plants was greater in harvested quadrats than in wildfire quadrats. Work by Buckley et al. (2003) observed that haul roads used to transport timber in Michigan were primary conduits for the dispersal of exotic plant species. These exotics entered into the interior of harvested stands and contributed to a significant shift in plant richness and composition. Although a shift in plant communities has not occurred in harvested survey sites at this time, the presence of exotic species common in urban areas should raise some concern among stakeholders.

Increased vehicle traffic into forested areas increases the vectors of dispersal as humans have been linked to increasing quantities of exotic species. Examples of a direct human involvement have been noted to occur in nature reserves. Macdonald et al. (1989) identified a causal relationship between the number of humans and exotic plant richness in Northern American nature reserves. The authors identified a positive regression among the number exotic plant species and number of visitor per week in temperate woodlands. Work completed by Lonsdale and Lane (1994) also indicated that movement of seeds by vehicles may be responsible for weed infestations into Kakadu National Park in Australia. The authors suggest that the type of vehicle is the best predictor of the amount of transported seeds, as vehicles that are capable of exploratory off-roading will carry more seeds.

The use of vehicles in harvested areas may have contributed to the increased density of exotic species. Smith and Ulmer (2001) observed a similar mean exotic cover in cutblocks in boreal transition forests. The landing areas of these cutblocks had an increased cover of exotic plants as vehicles (i.e. skidders and haul trucks) and roadways converge to transport timber to processing mills. Harvesting equipment used to cut, forward and load timber will spread propagules throughout the disturbance as these vehicles move over the entire disturbance area. Additional exposure occurs with the movement of vehicles associated with work shift changes and the use of maintenance vehicles to refuel and repair harvesting equipment. *Poa pratensis* may have been transported this way to recently harvested sites as Lonsdale and Lane (1994) have noted that grass species are most likely to be transport by vehicles. This species is commonly used in residential lawn mixtures and is considered an important forage species in the boreal transition ecozone (Looman, 1983). The equipment used to harvest timber in the study sites and throughout the Prince Albert Model Forest is often used to harvest areas in the boreal transition ecozone suggesting the two ecozones are exposed to similar vectors of transport. Four of the six exotic species in the recent disturbances examined in this study were also observed in Smith and Ulmer's (2001) examination of Saskatchewan Environment's east boreal ecodistricts. Their work identified 31 exotic plant species in provincial forests of the boreal transition ecozone.

The amount of light each treatment receives did not appear to contribute to the differences between exotic densities in harvested and wildfire sites. Although increased sunlight has been recognized by Amor and Stevens (1975) to contribute to increased levels of exotic plant species, the results obtained in these two treatments do not appear to support their findings. Significant differences were observed in light exposure between recently harvested and wildfire quadrats but a significant regression examining light and exotic density was absent.

The attributes of the disturbance in harvested and wildfire areas will also influence the exotic densities in each disturbance type. Wildfire that consumes the accumulated litter and humus will reduce the number of shrub and herb seeds (Archibold, 1980). Wildfire is thus capable of destroying exotic propagules that may be present in the soil, possibly reducing exotic densities. Johnson et al. (1998) have determined that the dispersal of plants into a wildfire area often occurs from plants that survived the fire or were at margins of the fire. The dispersal of exotic species from the adjacent undisturbed forest in this study is unlikely as the amount of exotic plants in these areas is greatly reduced. Dispersal into these recent disturbances most likely occurred naturally as the majority of exotic species (*Cirsium arvense*, *Crepis tectorum*, *Sonchus arvensis* and *Taraxacum offininale*) present in recent cutblocks and wildfires are able to disperse by wind. These species most likely established and dispersed along corridors that were used to suppress fire and/or harvest timber. The source of exotic species in recently harvested and wildfire disturbances appears to be the roadside right-of-ways. The logging and post-harvest activities may also have contributed to the increased exotic densities in harvested sites. These activities have been noted to encourage seed germination and promote exotic species as propagules are redistributed within the soil profile (Qi and Scarratt, 1998).

The exotics observed in recently disturbed cutblocks and wildfire treatments were generally ranked as a lesser threat and hard to control. Their adaptation for wind dispersal suggests that the corridors leading to the recent disturbance are facilitating this invasion. No invasive species were observed in this group because the exotic species that have successfully established away from the roadside right-of-way have reduced densities and frequencies. This group is considered hard to control because of the increased dispersal of propagules and need for repeated control measures. Climate change can be expected to promote the distribution of this group of species because of its high dispersal rates, abilities to establish in a variety of habitats and common occurrence in urban areas.

The similarities in exotic frequencies between cutblocks and wildfires may be the result of comparable vectors of dispersal. Wildfire sites use vehicles during the construction of fire guards and corridors to supply water and equipment for ground suppression activities. Although this machinery requires similar maintenance services as machinery used at harvested sites, the use of vehicles is different for each disturbance type. Harvesting equipment passes over the entire area whereas vehicles associated with wildfires create linear corridors that assist suppression activities. The management of wildfires in the area also incorporates the use of aerial suppression and hand tools to extinguish the wildfire. These activities generally have a reduced reliance on vehicles compared to harvesting activities.

Although densities of exotic species were significantly higher in harvested quadrats, the exotic frequencies, native plant cover and the exotic species present in harvested and wildfire sites are similar. The hypothesis stating that the occurrence of exotic species in harvested and wildfire areas is comparable should be accepted. The environment created by these disturbances appears to have a similar increase in available resources that has contributed to similar exotic species and frequencies in these disturbances. The removal of tree and shrub species in both types of disturbances increases levels of light reaching the ground and nutrient availability. Fire improves soil seedbeds and increases nutrient availability by increasing soil pH and temperature, removing vegetation and possibly reducing microbial competition (Kimmins, 1996). Post-harvest treatments conducted by a disc-trencher exposes mineral soil increasing the availability of nutrients to establishing vegetation (British Columbia Ministry of Forests, 1999). The disturbances in recently disturbed sites may also have "residual" events as site preparation typically occurs the year following harvest and high wind events may cause standing dead trees to be up-rooted in wildfire areas. These activities provide increased numbers of microsites for the continued establishment of early successional plant species. Although the potential exists for exotics to flourish in these disturbed areas, the lack of a deliberate introduction and the increased cover of native species (Table 5.3) appears to limit the quantity of exotic species. Native plants in recently harvested and wildfire sites continue to be the characteristic species in these disturbances.

6.3 Recently disturbed areas adjacent to seeded and unseeded right-of-ways

The proximity of recently disturbed sites to the seeded and unseeded right-of-ways appeared to have no influence on exotic plant distributions in this study. An increased quantity of exotics were observed in sites adjacent to unseeded right-of-ways even though these sites were significantly further away from the roadside right-of-way than sites that were adjacent to seeded areas. The hypothesis that the occurrence of exotic plant species in sites adjacent to the right-of-way and sites distant from the right-of-way is similar should be accepted.

The reduced cover of exotic species in sites adjacent to seeded right-of-ways appears to contradict literature that describes decreasing amounts of exotic species as the distance from the roadside right-of-way increases. Amor and Stevens (1975) observed this trend during an examination of the spread of exotic plants from an old roadside into forest communities in Victoria, Australia. The authors identified a decreasing frequency of exotic plants as the distance from a road increases. Tyser and Worley (1992) observed a similar decrease in exotic species richness in Montana. Transects were parallel to roadways and were positioned at increasing distances from the roadside extending into grassland areas. These studies were conducted in areas extending into natural habitat that were adjacent to the right-of-way. They provide sufficient evidence to support the notion that roadside right-of-ways are a source of exotic plant species, but do not examine how the proximity of a recent disturbance to the right-of-way influences the subsequent exotic plant distributions. Work by Deferrari and Naiman (1994) addresses this issue but does not identify the distance from roadway as the most important factor influencing exotic numbers. The distance from a major highway was related to the number and cover of exotic species but was still influenced by the landscape patch type and patch area. The landscape patch (riparian or upland forest) and plant community type were the most important indicators of exotic plant invasion within the Dungeness and Hoh river watersheds on the Olympic Peninsula, Washington.

6.4 Recently disturbed and mature forest

Recently disturbed areas had significantly larger quantities of exotic plant species than mature forests. Surveys in recently disturbed sites occurred in forest stands that previously had the same site attributes, forest and soil classifications as sites in mature forests. The only difference occurs between the land use in these treatments. The hypothesis that recently disturbed and mature forests will have equal occurrences of exotic plant species should be rejected.

The attributes of the mature forest appear to provide a selective barrier to the establishment of exotic species. *Poa compressa* and *Taraxacum officinale* were the only exotic species observed in mature sites. These species occurred in sites that had more human contact than other mature survey sites. These exotics were located in quadrats adjacent to roadways in the vicinity of Candle Lake resort village. These roadways lead to the Montreal Lake town site (32 km away) and garbage and recreational facilities (i.e. stocked trout pond) for the residents of Candle Lake.

The reduced presence of exotic species in undisturbed forest in this study is supported by literature examining the issue. A study by Wiser et al. (1998) examined the invasion of a mountain beech forest by an exotic perennial herb, *Hieracium lepidulum*, in New Zealand. The research identifies that natural forests do not have to be disturbed to be invaded by exotic species. Rejmanek (1989) has complied a list of exotic species that have invaded undisturbed plant communities. The author identifies that the list is very short and many questions remain regarding the direct or indirect effects of human disturbance. The short list is not surprising as it is indicated that the probability of survival from a seed to a reproducing plant is in the range of 10^{-4} to 10^{-6} for trees. Although no probabilities were given for herbaceous species, the author continues to state that there is no reason to expect a lower mortality rate for exotic species in temperate regions.

The physical attributes of mature stands in this study appear to influence the distribution of exotic plants. Although a significant regression regarding light exposure and exotic density was absent, the intact tree canopy of a mature forest significantly reduced the amount of light the quadrat receives. Recently disturbed forest had 66 % of full sunlight exposure while mature forest had 19%. Work completed by Panetta and Hopkins (1991) concluded that light availability appeared to be a major controlling factor in the frequency of exotic species. Reduced light levels in closed forest may partially control the recruitment of annual, biennial, and perennial weeds. Parendes and Jones (2000) corroborate these findings observing a decline in exotic species as the canopy closed and light levels reduced.

Dispersal of exotic species throughout mature sites appears to have been limited by a reduced exposure to humans. Although mature forests were closer to roadside right-of-ways, there have not been similar amounts of anthropogenic exposure as compared to recently disturbed study sites. The recently disturbed areas examined in this study have increased levels of human activities and increased incidences of exotic plant species. Exposure of the mature forest is limited to trails or roads that leave much of the forest block unexposed to human contact. Several authors (Lonsdale and Lane, 1994; MacDonald et al.; 1989) have identified an increase in exotic plants as the exposure to humans increase. Mature forest stands in this study are less visited by humans and thus have a limited dispersal of exotic plants.

As the canopy closes, a decrease also occurs in the amount of water and nutrients available. Miller (1983) describes the changes associated with canopy closure including a decrease in the amount of surface water and a reduction of annual grass and forb species. Trees and shade-tolerant herbs are able to tolerate lowered water levels even though increased competition among these types of vegetation will reduce the levels of soil nutrients (Wiser et al., 1998). These conditions in mature study sites appear to have inhibited exotic species. Work by Mooney and Gulmon (1983) describes a slow decline in productivity as the canopy closes. This decline appears to have limited early successional exotic species from becoming the dominant plant group. Observations from the mature forest surveyed corroborate these findings as the two exotic species observed in mature forests were perennial and had modest appearances. Observations of the stunted growth of *Poa compressa* and an absence of flowering stalks in *Taraxacum officinale* appears to demonstrate this reduction in plant vigor.

Chapter 7

Conclusion

7.1 Summary

The objectives of this study were to examine the distribution of exotic plant species in the mixedwood section of boreal forest in Saskatchewan and to assess the species according to their potential threats to ecosystem structure, composition and function. Studies indicate that disturbance areas are more prone to be invaded by exotic plant species. The initial introduction into the study area occurred in roadside right-of-ways with an application of seed mixtures containing exotic plant species. Six exotic species have persisted within the right-of-way habitat since the deliberate introduction more than 30 years ago.

The examination of seeded and unseeded roadside right-of-ways demonstrates that these species have moved beyond the initial point of introduction and into unseeded portions of the right-of-way. The densities and frequencies of exotic species in right-of-ways appear to have been promoted by the deliberate introduction into the roadside habitat.

Although the proximity of recent disturbances to the right-of-way did not influence exotic plant distributions, the type of disturbance had an effect. The highest exotic frequencies and density were observed in study sites that had an increased frequency of disturbances. Roadside right-of-ways had regular maintenance activities with increased anthropogenic contact. The maintenance equipment used in the right-of-way is capable of spreading exotic propagules and creating an environment favourable to early successional plant species. The other study sites had reduced numbers of exotics typically being disturbed every 50 to 100 years (Christensen pers.com.;

Heinselman, 1981; Johnson and Rowe, 1975). Noble and Slatyer (1980) have identified recurrent disturbances as an important selective force in plant populations. The species that occupy these communities are adapted to these disturbances. They have an advantage as increased exposure to disturbances has increased the selection pressure for adaptive traits (Reiners, 1983). The extended land use in the European and Asian habitats of the observed exotic species has given these species a longer time to adapt to a disturbance landscape.

Although right-of-ways had the largest exotic frequencies and densities, the majority of species located in the roadside right-of-way were ranked as a lesser threat and easy to control. The quantities of all the exotic species in this group may be related to the reduced dispersal capabilities of the majority of these species. Dispersal to these areas appears to have occurred naturally by the common boundary of roadside right-of-ways and southern farmlands or by anthropogenic vectors including transportation on maintenance vehicles.

The lack of a deliberate introduction appears to have limited the distribution of exotic species in harvested and wildfire sites. The differences among exotic species in cut and wildfire areas were negligible as similar frequencies and species were observed in these disturbance types. The majority of exotic species were dispersed by the wind suggesting a conduit for dispersal exists. All four of the exotic species observed within the right-of-way, that are capable of wind dispersal, were also observed within the recently harvested and wildfire sites. The reduced presence of the bluegrass (*Poa*) species in recently harvested and wildfire sites indicates a limited method of introduction. The tendency of these species to be transported by vehicles and common occurrence in seed mixtures indicates a close anthropogenic association. Therefore these species may have been introduced into the disturbances by anthropogenic contact suggesting that humans are effectively introducing exotic plant species.

Native plants in the recent wildfire and harvesting disturbances continue to be the characteristic species in these early successional communities. Reduced cover of exotic plants in mature forests suggests limited dispersal capabilities and resource availability are barriers to an exotic plant invasion. The increased presence of native plant species also appears to diminish the quantities of exotic plant species. A significant inverse relationship was identified among the native and exotic plant cover contributing to the decrease in the susceptibility of these environments to an exotic invasion.

The environment of the undisturbed forest appears to have excluded exotic plants as reduced light exposure, water and nutrient availability have increased the competiveness of native species. The dominant plants in these sites were late successional native species.

7.2 Recommendations

Management of exotic species should focus on the attributes that have allowed this group to become established. Deliberate introductions of exotic plants should cease and the examination of areas with increased human contact should continue. Work by Smith and Ulmer (2001) has highlighted exotic distributions in areas with increased frequencies of human contact. Humans often influence exotic plant distributions and increased amounts of disturbance, resource availability and adequate supplies of propagules will promote the establishment of exotic species. The ranking of each species observed in the study area identifies that most species do not require immediate action at this time. Although exotic species generally do not persist in disturbed areas that are allowed to re-grow into mature forest, the presence of exotic plants throughout the study area should raise concern among stakeholders. These populations may be in a "lag phase" of development and could experience an exponential growth with the anticipated changes to the climate and increase in harvesting and wildfire disturbances in Saskatchewan.

Other exotic species that were not observed within the quadrats but are known to occur within the area should raise concern about an expanding distribution of exotic plants. Scentless chamomile (*Matricaria perforata*) and caragana (*Caragana arborescens*) have been identified by staff at Saskatchewan Highways and Prince Albert National Park to occur within and adjacent to the study area. These species are invasive in other natural habitats in Saskatchewan and methods to control these species have been initiated. Continued monitoring of these species by these agencies is required. Additional attention to areas that are prone to the establishment of exotic plants should also be given. Surveying riparian forest and areas adjacent to heavy grazing may identify additional exotic plant species in Saskatchewan, as only upland sites were examined in this study. Floristic surveys in these areas would also contribute to an increased understanding of the invasion process.

If additional monitoring identifies expanding exotic distributions, control of vegetation should be based on methods to minimize costs and increase the effectiveness of control. The first step would be to prevent areas from infestations so that weedy exotic plants will never be exposed to an area (Grice 1998). This prevention would decrease the susceptibility to an invasion and could possibly impose limits on the use of known invasive plants. It would target sources of dispersal (i.e. hunter's bait stations, seeding right-of-ways, exotic ornamentals in cottage gardens). The next procedure would involve treating areas that are heavily infested. Treatments would vary depending on the plant species and the equipment and manpower available. Possible treatment methods may include chemical, mechanical (cutting or pulling), fire, or biological control methods that could be used independently or in combination. Repeated treatments should be expected as these species generally produce large amounts of seeds and sufficient underground reserves allow these plants to re-establish after a disturbance.

Literature Cited

Achuff, P.L., A.W. Bailey and L.M. Brusnyk. 1990. Non-native plant management in western region national and historic parks: issue analysis and recommendations. D.A. Westworth and Associates Ltd., Edmonton.

Acton, D. F., G. A. Padbury and C. T. Stushnoff. 1998. The Ecoregions of Saskatchewan. Hignell Printing Ltd., Winnipeg.

Agriculture Canada. 1992. Soil landscapes of Canada, Saskatchewan. Publication 5243/B. Canada Soil Inventory, Ottawa.

Alberta Research Council. 1999. Fire and harvest residual (FAHR) project: the impact of wildfire and harvest residuals on forest structure and biodiversity in aspen-dominated boreal forests of Alberta, final summary report. Retrieved November 2003 from http://www3.gov.ab.ca/env/resedu/biodiversity/FaHRSummaryRpt.pdf

Amor, R.L. and P.L. Stevens. 1975. Spread of weeds from a roadside forest at Dartmouth, Australia. Weed Research. 16:111-118.

Anderson, D.W. and J.G. Ellis. 1976. The soils of the provincial forest in the Prince Albert map area. Publ. SF3. Saskatchewan Institute of Pedology, University of Saskatchewan, Saskatoon.

Anderson, R. 1999. Disturbance as a factor in the distribution of sugar maple and the invasion of Norway maple into a modified woodland. Rhodora. 101: 264- 273.

Anderson, W.P. 1999. Perennial weeds: characteristics and identification of selected herbaceous species. Iowa State University Press, Ames, U.S.A.

Angold, P.G. 1997. The impact of a road upon adjacent heathland vegetation: effects on plant species composition. Journal of Applied Ecology. 34: 409-417.

Archibold, O.W. 1979. Buried viable propagules as a factor in post fire regeneration in northern Saskatchewan. Canadian Journal of Botany. 57: 54-58.

Archibold, O.W. 1980. Seed input into a post fire forest site in northern Saskatchewan. Canadian Journal of Forest Research. 10: 129-134.

Archibold, O.W., D. Brooks and L. Delanoy. 1997. An investigation of the invasive shrub European buckthorn, *Rhamnus cathartica L.*, near Saskatoon, Saskatchewan. The Canadian Field-Naturalist. 111: 617-621.

ArcView 3.3. 2002. Computer software for GIS applications. Product ID 855861101561.

Armstrong, G.W. 1999. A stochastic characterization of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. Canadian Journal of Forest Research. 29:424-433.

Baker, H.G. 1974. The evolution of weeds. Annual Review of Ecology and Systematics. 5: 1-24.

Baker, W.L. 1986. Patterns of plant invasion in North America. In: Mooney H.A. and J.A. Drake (Ed.). Ecology of Biological Invasions of North America and Hawaii (pp. 44-57). Springer-Verlag, New York.

Barbour, M. G. and W. D. Billings (Ed.) 1988. North American Terrestrial Vegetation. Cambridge University Press, New York.

Batianoff, G. N. and A. J. Franks. 1998. Weed invasion of the tropical Mackay Coast, Queensland, Australia. Plant Protection Quarterly. 13: 123-130.

Bazzaz, F.A. 1996. Plants in changing environments: linking physiological, population, and community ecology. Cambridge University Press, United Kingdom.

Beckingham, J.D., D.G. Nielson and V.A. Futoransky. 1996. Field guide to ecosites of the mid-boreal ecoregions of Saskatchewan. Canadian Forest Service, Northwest Region, Northern Forestry Centre. Special Report 6. UBC Press, Vancouver.

Bessie, W.C. and E.A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology. 76: 747-762.

Blaney, C.S. and P.M. Kotanen. 2001. Post-dispersal losses to seed predators: an experimental comparison of native and exotic old field plants. Canadian Journal of Botany. 79:284-292.

Booth, B.D., S.D. Murphy and C.J. Swanton. 2003. Weed ecology in natural and agricultural systems. CAB International, New York

Boreal Forest Research Centre. 2003. The boreal forest. Retrieved December 2003 from www.borealcentre.com/the_boreal_forest/

British Columbia Ministry of Forest. 1999. Silviculture manual, site preparation. Retrieved September 2003 from http://www.for.gov.bc.ca/hfp/pubs/silvman/

Buckley, D.S., T.R. Crow, E.A. Nauertz and K.E. Sculz. 2003. Influence of skid trails and haul roads on understory plant richness and composition in managed forest landscapes in upper Michigan, USA. Forest Ecology and Management. 175: 509-520.

Butterfield, C., J. Stubbendieck and J. Stumpf. 1996. Species abstracts of highly disruptive exotic plants. Retrieved November 2002 from http://www.npwrc.usgs.gov/resource/othrdata/exoticab/exoticab.

Cale, P. and R. Hobbs. 1991. Condition of roadside vegetation in relation to nutrient status. In D.A. Saunders and R.J. Hobbs (Ed.), Nature Conservation 2: the Role of Corridors (pp. 353-362). Beatty and Sons Surrey, Australia.

Canadian Forest Service. 1999. Alien forest pests: context for the Canadian forest Services Science Program, Science Branch, Ottawa.

Carleton, T.J. and P. MacLellan. 1994. Woody vegetation responses to fire versus clear-cutting logging: a comparative survey in the central Canadian boreal forest. Ecoscience. 1: 141-152.

Carlquist, S. 1965. Island Life. Natural History Press, New York.

Case, T.J. 1990. Invasion resistance arises in strongly interacting species-rich model competition communities. Ecology. 87: 9610-9614.

Chapin, F.S. 1983. Patterns of nutrient absorption and use by plants from natural and man-modified environments. In H.A. Mooney and M. Gordon (Ed.), Disturbance and Ecosystems, Components of Response (pp. 175-187). Springer-Verlag, New York.

Chicoine, T.K., P.K. Fay and G.A. Nielson. 1986. Predicting weed migration from soil and climate maps. Weed Science. 34:57-61.

Christensen, B. Personal communication. Weyerhaeuser Canada, Saskatchewan Division. Prince Albert, Saskatchewan.

Clinton, W.J. 1999. Executive order 13112, invasive species. Retrieved June 1999 from http://www.invasivespecies.gov/laws/eo13112.pdf

[CSFI] Council of Saskatchewan Forest Industries. 2001. Saskatchewan forest facts. Pamphlet, Saskatoon.

Crooks, J.A. and M.E. Soule. 1999. Lag times in population explosions of invasive species: causes and implications. In: Sandlund, O.T., P.J. Schei, and A. Vilken (Ed.), Invasive species and biodiversity management (pp. 103-125). Kluwer Academic, Boston.

D'Antonio, C.M. and P.M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics. 23: 63-87.

Davis, M.A., J.P. Grime and K. Thompson. 2000. Fluctuating resources in plant communities: a general theory of invasibility. Journal of Ecology. 88: 528-534.

Dean, W.R.J., S.J. Milton, P.G. Ryan and C.L. Moloney. 1994. The role of disturbance in the establishment of indigenous and alien plants at Inaccessible and Nightingale Islands in the South Atlantic Ocean. Vegetatio. 113: 13-23.

Deferrari, C.M and R.J. Naiman. 1994. A multi-scale assessment of the occurrence of exotic plants on the Olympic Peninsula, Washington. Journal of Vegetation Science. 5: 247-258.

Delcan. 1994. Saskatoon natural grasslands resource management plan, final report. Saskatoon, unpublished.

Denholm, K.A. and L.W. Schut (Ed.). 1993. Field manual for describing soils in Ontario. 4th Ed. Publication No.93-1. Ontario Center for Soil Resource Evaluation, Guelph.

di Castri, F. 1989. History of biological invasions with special emphasis on the Old World. In J.A. Drake, H.A. Mooney, F. di Castri, R.H. Groves, F.J. Kruger, M. Rejmanek et al. (Ed.), Biological Invasions, a Global Perspective (pp. 1-26). John Wiley and Sons, New York.

Dix, R.L. and J.M.A. Swan. 1971. The roles of disturbance and succession in upland forest at Candle Lake, Saskatchewan. Canadian Journal of Botany. 49:657-676.

Doing, H. 1985. Coastal fore-dune zonation and succession in various parts of the world. Vegetatio 61: 65-75.

Ducks Unlimited. 1994. Purple loosestrife in western Canada. Pamphlet, Saskatoon.

Dytham, Calvin. 1999. Choosing and using statistics: a biologist's guide. Blackwell Science Ltd., United Kingdom.

Elton, C.S. 1958. The ecology of invasions by animals and plants. Chapman and Hall, London.

Environment Canada. 2001. Canadian biodiversity strategy, executive summary. Retrieved April 2001 from http://eqb-dqe.cciw.ca/eman/reports/publications/rt_biostrat/cbs16.htm

Environment Canada. 2003. Climate Station Catalogue. Retrieved November 2003 from http://www.msc-smc.ec.gc.ca/climate/station_catalogue/index_e.cfm.

Espie, R. 2001 Strangers in a new world: the problem of exotic organisms in Saskatchewan. Report 01-3. Saskatchewan Environment and Resource Management, Fish and Wildlife, Regina.

Ewel, J.J., D.J. O'Dowd, J. Bergelson, C.C. Daehler, C.M. D'Antonio, L. D. Gomez, D.R. Gordon, R.J. Hobbs, A. Holt, K.R. Hopper, C.E. Hughs, M. LaHart, R.R.B. Leakey, W.G. Lee, L.L. Loope, D.H. Lorence, S.M.Louda, A.E. Lugo, P.B. McEvoy, D.M. Richardson, P.M. Vitousek. 1999. Deliberate introductions of species: research needs, benefits can be reaped, but risks are high. BioScience. 4: 619-630.

Fairey, D.T. 1985. Red clover. Publication 1614/E. Agriculture Canada, Ottawa.

Fairey, D.T. 1986. Alsike clover. Publication 1264/E. Agriculture Canada, Ottawa.

Flannigan, M.D. and C. E. Van Wagner. 1991. Climate change and wildfire in Canada. Can. Journal of Forest Research. 21: 66-72.

Forman, R.T.T. 1995. Some general principles of landscape and regional ecology. Landscape Ecology. 10: 133-142.

Forman, R.T.T. and L.E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics. 29:207-231.

Forman, R.T.T. and R.D. Deblinger. 2000. The ecological road-effect zone of a Massachusetts (USA) suburban highway. Conservation Biology. 14: 36-46.

Fytianos, K., G. Vasilikiotis and V. Samanidou. 1985. Contamination of roadside vegetation with lead cadmium and zinc. Chemosphere. 14: 271-277.

Global Invasive Species Programme. 2002. GISP website. Retrieved February 2002 from http://www.gisp.org/

Golder Associates. 1999. Twenty-year forest management plan and environmental impact statement, summary. Saskatoon.

Goode, P., J. Champ and L. Amundson. 1996. The Montreal Lake region: its history and geography. Sentar Consultants, Saskatoon.

Goplen, B.P and A.T.H. Gross. 1984. Sweetclover production in Western Canada. Publication 1613/E. Agriculture Canada, Ottawa.

Government of Canada. 1985. Seeds Act. Department of Justice, Ottawa, Ontario. Retrieved May 2003 from http://laws.justice.gc.ca./en/S-8/index.html

Government of Canada. 1990. Plant Protection Act. Department of Justice, Ottawa. Retrieved May 2003 from http://laws.justice.gc.ca/en/P-14.8/text.html

Government of Canada. 1996. Wild Animal and Plant Protection and Regulation of International and Interprovincial Trade Act [WAPPRIITA]. Department of Justice, Ottawa. Retrieved May 2003 from http://www.cws-scf.ec.gc.ca/publications/wappa/index_e.cfm

Government of Canada. 2000. National Policy on Introductions and Transfers of Aquatic Organisms-Draft. Department of Fisheries and Oceans, Ottawa.

Government of Saskatchewan. 1984. Noxious Weeds Act. Queen's printer, Regina. Retrieved May 2003 http://www.qp.gov.sk.ca/index.cfm?fuseaction=publications.details&p=742

Government of Saskatchewan. 1996. Forest Resources Management Act. Queen's printer, Regina. Retrieved May 2003 from http://www.qp.gov.sk.ca/documents/english/statutes/statutes/f19-1.pdf

Grayson, R.B., S.R. Haydon, M.D.A. Jayasuriya, and B.L. Finlayson. 1993. Water quality in mountain ash forests-separating the impacts of roads from those of logging operations. Journal of Hydrology. 150: 459-480.

Green, D.S. 1983. The efficacy of dispersal in relation to safe site density. Oecologia. 56: 356-358.

Grice, Anthony C. 1998. Ecology in the management of Indian jujube *Ziziphus mauritiana*. Weed Science. 46: 467-474.

Griggs, R.F. 1940 The ecology of rare plants. Bulletin of the Torrey Botanical Club. 67: 575-594.

Haber, Erich. 1997. Guide to Monitoring Exotic and Invasive Plants. National Botanical Services. Retrieved October, 1999 from http://www.eman-rese.ca/eman/ecotools/protocols/terrestrial/exotics/intro.html

Harper, J.L. 1969 The role of predation in vegetational diversity. Diversity and Stability in Ecological Systems. Brookhaven Symposia in Biology. 22: 48-61. In Johnstone, I.M. 1986. Plant invasion windows: a time-based classification of invasion potential. Biological Review. 61: 369-394

Harper, J.L. 1977. Population biology of plants. Academic Press, London.

Harvey, B.D., A. Leduc, and Y. Bergeron. 1995. Early post harvest succession in relation to site type in the southern boreal forest of Quebec. Canadian Journal of Forest Research. 25: 1658-1672.

Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Research. 3: 329-382.

Heinselman, M.L. 1981. Fire and succession in the conifer forests of northern North America. In D.C. West, H.H. Shugart, and D.B. Botkin (Ed.), Forest succession: Concepts and Application (pp. 374-405). Springer-Verlag, New York.

Hiebert, R.D. and J. Stubbendieck. 1993. Handbook for ranking exotic plants for management and control. Natural Resources Report NPS/NRMWRO/NRR-93/08. Retrieved April, 2000 from http://www.nature.nps.gov/publications/ranking/ranking.pdf

Higgins, S.I., D.M. Richardson, R.M. Cowling and T.H. Trinder-Smith. 1999. Predicting the landscape-scale distribution of alien plants and their threat to plant diversity. Conservation Biology. 3: 303-313.

Hobbs, R.J. 2000. Land-use changes and invasion. In: Mooney, H.A. and R.J. Hobbs (Ed.). Invasive Species in a Changing World (pp. 55-64). Island Press, Washington D.C.

Hobbs, R.J. and L.F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology. 6: 324-335.

Hobbs, R.J. and S.E. Humphries. 1995. An integrated approach to the ecology and management of plant invasions. Conservation Biology. 9:761-770.

Holl, K.D., H.N. Steele, M.H. Fusari and L.R. Fox. 2000. Seed banks of maritime chaparral and abandoned roads: potential for vegetation recovery. Journal of the Torrey Botanical Society. 127: 207-220.

Holland, P. and S. Olson. 1989. Introduced versus native plants in austral forests. Progress in Physical Geography. 13: 260-290.

Howe, H.F. and J. Smallwood. 1982. Ecology of seed dispersal. Annual Review of Ecology and Systematics. 13: 201-228.

Huenneke, L.F. 1997. Outlook for plant invasions: interactions with other agents of global change. In: J.O. Luken and J.W. Thieret (Ed.), Assessment and Management of Plant Invasions (pp. 95-103). Springer-Verlag, New York.

Hughs, F., P.M. Vitousek, and T. Tunison. 1991. Alien grass invasion and fire in the seasonal submontane zone of Hawaii. Ecology. 72: 743-746.

Hughs, L., M. Dunlop, K. French, M.R. Leishman, B. Rice, L. Rodgerson and M. Westoby. 1994. Predicting dispersal spectra: a minimal set of hypotheses based on plant attributes. Journal of Ecology. 82:933-950.

IUCN. 1999. "World Conservation Union (IUCN) Mobilizes International Team of Experts to Save Plant Species". Press Release. Retrieved November, 1999 from http://www.iucn.org/info_and_news/press/ssc.html

Johnson, E.A. and J.S. Rowe. 1975. Fire in the sub-arctic wintering ground of the Beverly Caribou Herd. American Midland Naturalist. 94: 1-14.

Johnson, E.A., K. Miyanishi and J.M.H. Weir. 1998. Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. Journal of Vegetation Science. 9: 603-610.

Johnstone, I.M. 1986. Plant invasion windows: a time-based classification of invasion potential. Biological Review. 61: 369-394.

Kimmins, J.P. 1996. Importance of soil and role of ecosystem disturbance for sustained productivity of cool temperate and boreal forests. Soil Science Society of America Journal. 60: 1643-1654.

Kosowan, A. and F. Yungwirth. 1999. Canada thistle survey summary, 1999, east boreal ecoregion. Saskatchewan Environment, Prince Albert.

Kotanen, P. M. 1997. Effects of experimental soil disturbance on revegetation by native and exotics in coastal Californian meadows. Journal of Applied Ecology. 34: 631-644.

Kruger, G. 1997. Smooth bromegrass, seed production. Saskatchewan Forage Council, Saskatoon.

Levine, J.M. and C.M. D'Antonio. 1999. Elton revisited: a review of evidence linking diversity and invasibility. Oikos. 87: 15-26.

Lodge, D.M. 1993. Biological invasions: lessons for ecology. Trends in Ecology and Evolution. 8: 133-137.

Loney, B. and R.J. Hobbs. 1991. Management of vegetation corridors: maintenance, rehabilitation and establishment. In D.A. Saunders and R.J. Hobbs (Ed.), Nature Conservation 2: the Role of Corridors (pp. 299-311). Beatty and Sons, Surrey, Australia.

Lonsdale, W.M. 1999. Global patterns of plant invasions and the concept of invasibility. Ecology. 80: 1522-1536.

Lonsdale, W.M. and A.M. Lane. 1994. Tourist vehicles as vectors of weed seeds in Kakadu National Park, Northern Australia. Biological Conservation. 69: 277-283.

Looman, J. 1982. Prairie grasses, identified and described by vegetative characters. Publication 1413. Agriculture Canada, Ottawa.

Looman, J. 1983. 111 range and forage plants of the Canadian prairies. Publication 1751. Agriculture Canada, Ottawa.

Lugo, A.E. and H. Gucinski. 2000. Function, effects, and management of forest roads. Forest Ecology and Management. 133: 249-262.

Lutz, H. J. 1956. Ecological effects of forest fires in the interior of Alaska, Washington. Technical Bulletin No. 1133. U.S. Dept of Agriculture, Washington D.C.

Macdonald, I.A.W., L.L. Loope, M.B. Usher and O. Hamann. 1989. Wildlife conservation and the invasion of nature reserves by introduced species: a global perspective. In Drake J.A. et al. (Ed.). Biological Invasions: a Global Perspective (pp. 215-245). John Wiley and Sons, New York.

Mack, R.N. 1996. Predicting the identity and fate of plant invaders: emergent and emerging approaches. Biological Conservation. 78: 107-121.

Mack, R.N., D. Simberloff, W.M. Lonsdale, H. Evans, M. Clout and F.A. Bazzaz. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. Ecological Applications. 10: 689-710.

McCullough, D.G., R.A. Werner and D. Neumann. 1998. Fire and insects in northern and boreal forest ecosystems of North America. Annual Review of Entomology. 43: 107-127.

McKinney, M.L. 2001. Effects of human population, area, and time on non-native plant and fish diversity in the United States. Biological Conservation. 100: 243-252.

McRae, D.J., L.C. Duchesne, B. Freedman, T.J. Lynham and S. Woodley. 2001. Comparisons between wildfire and forest harvesting and their implications in forest management. Environmental Reviews. 9: 223-260.

Miller, P.C. 1983. Comparison of water balance characteristics of plant species in “natural” versus modified ecosystems. In H.A. Mooney and M. Gordon (Ed.), Disturbance and Ecosystems, Components of Response (pp. 186-212). Springer-Verlag, New York.

Milton, S.J. and R.J. Dean. 1998. Alien plant assemblages near roads in arid and semi-arid South Africa. Diversity and Distributions. 4: 175-187.

Mooney, H.A. and E.E. Cleland. 2001. The evolutionary impact of invasive species. Proceedings of the National Academy of Sciences of the United States of America. 98: 5446-5451.

Mooney, H.A. and S.L. Gulmon. 1983. The determinants of plant productivity-natural versus man-modified communities. In H.A. Mooney and M. Gordon (Ed.), Disturbance and Ecosystems, Components of Response (pp. 146-158). Springer-Verlag, New York.

Mosquin, T. 1997. Management guidelines for invasive alien species in Canada's National Parks. Parks Canada, Ottawa.

Moss, E.H. 1994. Flora of Alberta. 2nd Ed. University of Toronto Press, Toronto.

Motto, H.L., R.H. Daines, D.M. Chilko, and C.K. Motto. 1970. Lead in soils and plants: its relationship to traffic volume and proximity to highways. Environmental Science and Technology. 4:231-238.

National Invasive Species Council. 2001a. National invasive species council information. Retrieved October 2002 from http://www.invasivespecies.gov/council/niscinfo.shtml

National Invasive Species Council. 2001b. Meeting the invasive species challenge, management plan. Retrieved October 2002 from http://www.invasivespecies.gov/council/mpfinal.pdf

National Research Council (U.S). 2002. Predicting invasions of nonindigenous plants and plant pests. National Academy Press, Washington, D.C.

Nguyen-Xuan, T., Y Bergeron, D. Simard, J.W. Fyles and D. Pare. 2000. The importance of forest floor disturbance in the early regeneration patterns of the boreal forest of western and central Quebec: a wildfire versus logging comparison. Canadian Journal of Forest Research. 30: 1353-1364.

Noble, I.R. and R.O. Slatyer. 1980. The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. Vegetatio. 43:5-21.

Northern Forestry Centre. 1986. Hydrologic effects of clear-cutting at Marmot Creek and Streeter watersheds, Alberta. Info Report NOR-x-278. Northern Forestry Centre, Edmonton.

Orians, G.H. 1986. Site characteristics favouring invasions. In Mooney and Drake (Ed.), Ecology of biological invasions of North America and Hawaii (pp. 133-145). Springer-Verlag, New York.

Overpeck, J.T., D. Rind and R. Goldberg. 1990. Climate-induced changes in forest disturbance and vegetation. Nature. 343:51-53.

Panetta, F.D. and A.J.M. Hopkins. 1991. Weeds in corridors: invasion and management. In D.A. Saunders and R.J. Hobbs (Ed.), Nature Conservation 2: the Role of Corridors (pp. 341-351). Beatty and Sons, Surrey, Australia.

Parendes, L.A. 1998. Spatial patterns of invasion by exotic plants in a forested landscape. Ph.D. dissertation, Oregon State University, Corvallis, Oregon, U.S.

Parendes, L.A. and J.A. Jones. 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews experimental forest, Oregon. Conservation Biology. 14: 64-75.

Pearsall, W.H. 1918. On the classification of aquatic plant communities. Journal of Ecology. 6: 75-84.

Pearsall, W.H. 1921. The development of vegetation in the English lakes, considered in relation to the general evolution of glacial lakes and rock basins. Proceedings of the Royal Society. B92: 259-284.

Prince Albert Model Forest. 2000. Ecosystem based integrated resource management plan. Retrieved September 2001 from http://www.pamodelforest.sk.ca/pubs/PAMFPLAN.pdf

[PANP] Prince Albert National Park. 1994. Biophysical data. Waskesiu Lake, Saskatchewan. Contact Dave Wieder, unpublished.

Pysek, P. 1998a. Is there a taxonomic pattern to plant invasions? Oikos. 82: 282-294.

Pysek, P. 1998b. Alien and native species in Central European urban floras: a quantitive comparison. Journal of Biogeography. 25: 155-163.

Qi, M. and J.B. Scarratt. 1998. Effect of harvesting method on seed bank dynamics in a boreal mixedwood forest in northwestern Ontario. Canadian Journal of Botany. 76: 872-883.

Reiners, W.A. 1983. Disturbance and basic properties of ecosystem energetics. In H.A. Mooney and M. Gordon (Ed.), Disturbance and Ecosystems, Components of Response (pp. 83-98). Springer-Verlag, New York.

Rejmanek, M. 1989. Invasibility of plant communities. In J.A. Drake, H.A. Mooney, F. di Castri, R.H. Groves, F.J. Kruger, M. Rejmanek et al. (Ed.), Biological Invasions, a Global Perspective (pp. 369-388). John Wiley and Sons, New York.

Rejmanek, M. 1996. Species richness and resistance to invasions. In Orians, Dirzo, and Cushman (Ed.), Biodiversity and Ecosystem Processes in Tropical Forests (pp. 153-172). Springer-Verlag, Berlin.

Rejmanek, M. and D.M. Richardson. 1996. What attributes make some plant species more invasive. Ecology. 77:1655-1660.

Richardson, D.M., P. Pysek, M. Rejmanek, M.G. Barbour, F.D. Panetta and C.J. West. 2000. Naturalization and invasion of alien plants: concepts and definitions. Diversity and Distributions. 6:93-107.

Rowe, J.S. 1961. Critique of some vegetational concepts as applied to forests of Northwestern Alberta. Canadian Journal of Botany. 39: 1007-1017.

Royer, F. and R. Dickinson. 1999. Weeds of Canada and the Northern United States. Lone Pine Publishing and the University of Alberta, Edmonton.

Saskatchewan Environment. 2004. Forest inventory and digital map information. Government of Saskatchewan, Regina. Contact Doug Campbell, unpublished.

Saskatchewan Highways and Transportation. 1993. Purchase order for seed to re-vegetate roadside right-of-ways. Prince Albert, Saskatchewan.

Saskatchewan Industry and Resources. 2002. Forestry status report. Retrieved January 2005 from http://www.ir.gov.sk.ca/Default.aspx?DN=3357,3071,3070,3038, 2936,Documents.

Schmidt, W. 1989. Plant dispersal by motor cars. Vegetatio. 80:147-152.

Schwartz, Mark W. 1997. Defining indigenous species: an introduction. In J.O. Luken and J.W. Thieret (Ed.), Assessment and Management of Plant Invasions (pp. 7-17). Springer-Verlag, New York.

Sculthorpe, C.D. 1967. The biology of aquatic vascular plants. Edward Arnold, London. As seen in Johnstone, I.M. 1986. Plant invasion windows: a time-based classification of invasion potential. Biological Review. 61: 369-394.

Singh, T. and E.E. Wheaton. 1991. Boreal forest sensitivity to global warming: implications for forest management in western interior Canada. The Forestry Chronicle. 67: 342-348.

Smith, J.R. and J.A. Ulmer. 2001. Abundances of exotic plant species in disturbances of Saskatchewan's east boreal transition forests. Saskatchewan Environment, Prince Albert, Saskatchewan.

Sokal, R.R. and F.J. Rohlf. 1981. Biometry. 2nd Edition. W.H. Freeman and Company, San Francisco.

Species Survival Commission. 2000. IUCN guidelines for the prevention of biodiversity loss caused by alien invasive species. Retrieved February 2002 from http://www.iucn.org/themes/ssc/pubs/policy/invasivesEng.htm

SPSS for Windows. 1999. Computer software for statistically analysis: Release 10.0.5. Site license 181902.

Stocks, B. J. 1993. Global warming and forest fires in Canada. The Forestry Chronicle. 69: 290-293.

Stohlgren, T.J., D. Binkley, G.W. Chong, M.A. Kalkhan, L.D. Schell, K.A. Bull, Y. Otsuki, G. Newman, M. Bashkin and Y. Son. 1999. Exotic plant species invade hot spots of native plant diversity. Ecological Monographs. 69: 25-46.

Swanson, R.H. and G.R. Hillman. 1977. Predicted increased water yield after clear-cutting verified in west-central Alberta. Canadian Forest Service, Northern Forest Research Centre, Edmonton.

R.L. McGregor, T.M. Barkley, R.E. Brooks and E.K. Schofield (Ed.). 1986. Flora of the Great Plains. University Press of Kansas, Lawerence, Kansas.

Thomas, A.G., R.F. Wise, B.L. Frick and L.T. Juras. 1996. Saskatchewan weed survey, cereal, oilseed, and pulse crops. Weed Survey Series publication 96-1. Agriculture and Agri-Food Canada, Saskatoon.

Thorpe, J. 1996. Models of succession in the southern boreal forest. Prince Albert Model Forest. Prince Albert, Saskatchewan.

Thrasher-Haug, J. 1997. Plant species diversity patterns after fire and clearcutting disturbances in the southern boreal forest of Saskatchewan. Master's thesis, University of Saskatchewan, Saskatoon.

Trombulak, S.C. and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14: 18-30.

Turner, M.G. and W.H. Romme. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology. 9: 59-77.

Tyser, R.W. and C.A. Worley. 1992. Alien flora in grasslands adjacent to road and trail corridors in Glacier National Park, Montana (U.S.A.). Conservation Biology. 6: 253-262.

University of Saskatchewan Library Services. 2004. Map files of Saskatchewan highways. Saskatoon, Saskatchewan.

Usher, M.B. 1988. Biological invasions of nature reserves: a search for generalizations. Biological Conservation. 44: 119-135.

Usher, M.B., F.J. Kruger, A.W. Macdonald, L. Loope and R.E. Brockie. 1988. The ecology of biological invasions into nature reserves: an introduction. Biological Conservation. 44: 1-8.

Viskari, E. and L. Karenlampi. 2000. Roadside scots pine as an indicator of deicing salt use, a comparative study from two consecutive winters. Water, Air, and Soil Pollution. 122: 405-419.

Vitousek, P.M. 1986. Biological invasions and ecosystem properties: can species make a difference? In Mooney H.A. and J.A. Drake (Ed.), Ecology of Biological Invasions of North America and Hawaii (pp. 44-57). Springer-Verlag, New York.

Vitousek, P.M. 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. Oikos. 57: 7-13.

Vitousek, P.M., C.M. D'Antonio, L.L. Loope, and R. Westbrooks. 1996. Biological invasions as global environmental change. American Scientist. 84: 468-478.

Vitousek, P.M., L.R. Walker, L.D. Whiteaker, D. Mueller-Dombois and P.A. Matson. 1987. Biological invasion by *Myrica faya* alters ecosystem development in Hawaii. Science. 238: 802-804.

Walker, L.R. and S.D. Smith. 1997. Impacts of invasive plants on community and ecosystem properties. In J.O. Luken and J.W. Thieret (Ed.), Assessment and management of Plant Invasions. Springer-Verlag, New York.

Watt, A.S. 1947. Pattern and process in the plant community. Journal of Ecology. 35:1-22.

Webb, D.A. 1985. What are the criteria for presuming native status? Watsonia. 15:231-236.

Weir, J.M.H. Personal communication. Prince Albert National Park. Waskesiu Lake, Saskatchewan.

Weir, J.M.H., E.A. Johnson and K. Miyanishi. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. Ecological Applications. 10: 1162-1177.

Weyerhaeuser Canada. 2004. Management data for GIS map analysis. Prince Albert, Saskatchewan. Contact Brian Christensen, unpublished .

Westman, W.E. 1990. Park management of exotic plant species: problems and issues. Conservation Biology. 4: 251-260.

Wheaton, E. 1997. Forest ecosystem and climate. Appendix B. In R. Herrington, B. Johnson, and F. Hunter. Responding to global change in the prairies. Environment Canada, Ottawa.

Williamson, M. 1993. Invaders, weeds and the risk from genetically manipulated organisms. Experientia. 49: 219-224.

Williamson, M. 1994. Community response to transgenic plant release: predictions from British experience of invasive plants and feral crop plants. Molecular Ecology. 3: 75-79.

Willson, M.F. 1993. Dispersal mode, seed shadows, and colonization pattern. Vegetatio. 107/108: 261-280.

Wiser, S.K., R.B. Allen, P.W. Clinton, K.H. Platt. 1998. Community structure and forest invasion by an exotic herb over 23 years. Ecology. 79: 2071-2081.

Appendix A

Locations of the right-of-way transects

Specific locations of the right-of-way transects. The placement of the first transect occurred at the southern edge of a site surveyed in 2000. In cases where previously surveyed sites were distant from a roadside right-of-way, the first transect was placed in a parallel position to the southern boundary of a previously surveyed stand. The remainder of the transects were located north of the first transect at 10 m intervals. Each right-of-way surveyed had regular maintenance activities to remove overhead (>1 m) vegetation.

Site ID	Site name	Highway right-of-way to be surveyed	Side of highway	Location
2	Montreal Lake	969	East	11.71 km North from highway 2 and 969 junction
8	Minowakuw	Minowakuw Road	North	0.5 km West of Minowakuw and highway 120 junction
9	Timber Cove	2	West	1.94 km North of junction of highway 969 and 2
10	Tawow Camp	Candle Lake North Access Road	North	2.010 km West of highway 120 and Candle Lake North access road junction
13	Narrows	Narrows Road-Prince Albert National Park	North	1.7 km south of highway 264 and Narrows road junction
14	Garbage Road	Landfill Road	East	656 m south of Landfill Road and Candle Lake North Access road junction
15	N Meeyomoot	926 (Snowfield Road)	West	2.045 km North of Meeyomoot Road
17	Elk Corral	120	West	Site distant from main road, surveying occurred at trailhead 3.8 km North of Minowakuw turnoff
19	Trout Pond	265	North	834 m north of trout pond turnoff
21	Trout Pond	265	North	280 m south of trout pond turnoff
32	Monday South	926	East	17.7 km north of highway 970 and 926 junction or 8.05 km North of Round Hill fire tower Road
43	Nikik	2	West	4.68 km North of highway 2 and 969 junction
44	Waskesiu Fire	2	West	distant from main road, will survey 13.65 km North of highway 2 and 969 junction
46	Muskeg Fire	2	East	30 m South of Halfway House parking lot
50	Monday North	921 (Meeyomoot Road)	South	site distant from main road, will survey 1.03 km from highway 926 and 921 junction
51	Hwy 120	120	East	3.8 km North of Minowakuw turnoff
53	Snowfield	926	West	Site located away from main road, surveying will begin 7.035 km N of Bittern Lake Road and 926 junction
54	Mino West	Torch River Road	North	1.885 km East of Minowakuw turnoff, near active gravel pit, ditch contiguous with open meadow
55	Mino East	120	East	distant from main road way, surveying will begin at Minowakuw turnoff
57	Cobra Fire	2	West	distant from main road, surveyed 1.75 km North of Crean River
58	Waskesiu Fire	2	West	distant from main road, will survey 14.65 km from highway 2 and 969 junction

Appendix B

GPS locations of each quadrat surveyed in 2000

Each quadrat surveyed in 2000 had a GPS (global positioning system) location recorded in the northwest corner of the quadrat. The readings included the latitude, longitude, estimated positional error (EPE), date and the time of the recording. The GPS data was not corrected to a base station as the GPS unit (Garmin 12XL) was unable to do so. The data was recorded using the WGS 84 datum.

site ID	quadrat ID	latitude	longitude	EPE	Date	Time
2	1	54' 02" 36.7°	105' 47" 56.0°	9	14-Sep-00	11:21 AM
2	2	54' 02" 33.7°	105' 47" 55.7°	8	14-Sep-00	12:52 PM
2	3	54' 02" 29.5°	105' 47" 50.0°	12	14-Sep-00	3:09 PM
2	4	54' 02" 15.0°	105' 48" 08.7°	7	14-Sep-00	4:52 PM
2	5	54' 02" 10.0°	105' 48" 04.4°	11	15-Sep-00	7:45 AM
8	1	53' 46" 31.3°	105' 08" 31.3°	7	28-Sep-00	7:30 AM
8	2	53' 46" 34.7°	105' 08" 35.7°	10	28-Sep-00	9:30 AM
8	3	53' 46" 34.5°	105' 08" 37.4°	5	28-Sep-00	12:00 PM
8	4	53' 46" 38.0°	105' 08" 38.5°	6	28-Sep-00	1:44 PM
8	5	53' 46" 37.7°	105' 08" 41.5°	7	31-Aug-00	8:17 AM
9	1	53' 59" 11.0°	105' 54" 55.9°	6	29-Aug-00	8:50 AM
9	2	53' 59" 33.8°	105' 54" 47.4°	8	29-Aug-00	9:21 AM
9	3	53' 59" 20.7°	105' 55" 23.2°	6	29-Aug-00	10:04 AM
9	4	53' 59" 12.2°	105' 55" 34.2°	10	29-Aug-00	10:34 AM
9	5	53' 59" 01.5º	105' 55" 44.7°	4	29-Aug-00	11:06 AM
10	1	53' 45" 40.2°	105' 11" 00.6°	8	22-Sep-00	8:30 AM
10	2	53' 45" 40.3°	105' 10" 54.3°	4	21-Sep-00	9:24 AM
10	3	53' 45" 40.4°	105' 11" 04.5°	9	22-Sep-00	10:35 AM
10	4	53' 45" 43.9°	105' 11" 07.7°	7	22-Sep-00	11:25 AM
10	5	53' 45" 43.4°	105' 11" 02.2°	14	22-Sep-00	12:25 PM
13	1	53' 53" 59.9°	106' 04" 32.7°	15	28-Aug-00	3:39 PM
13	2	53' 54" 03.2°	106' 04" 14.4°	25	28-Aug-00	11:21 AM
13	3	53' 54" 03.1°	106' 04" 22.5°	20	28-Aug-00	11:54 AM
13	4	53' 54" 13.5°	106' 04" 39.7°	9	28-Aug-00	12:37 PM
13	5	53' 54" 02.0°	106' 04" 48.8°	7	23-Aug-00	5:44 PM
14	1	53' 44" 44.8°	105' 14" 39.0°	10	26-Sep-00	2:30 PM
14	2	53' 44" 46.6°	105' 14" 39.6°	7	26-Sep-00	4:15 PM
14	3	53' 44" 49.8°	105' 14" 39.2°	8	26-Sep-00	4:45 PM
14	4	53' 44" 49.5°	105' 14" 33.3°	5	07-Oct-00	10:11 AM
14	5	53' 44" 53.1°	105' 14" 33.9°	8	07-Oct-00	11:28 AM
15	1	54' 02" 23.5°	105' 28" 25.8°	12	05-Oct-00	10:40 AM
15	2	54' 02" 36.7°	105' 28" 20.9°	6	05-Oct-00	11:00 AM
15	3	54' 02" 45.5°	105' 28" 35.4°	8	05-Oct-00	2:05 PM

15	4	54' 02" 48.9°	105' 28" 33.2°	9	05-Oct-00	3:42 PM
15	5	54' 02" 50.5°	105' 28" 23.3°	6	05-Oct-00	5:11 AM
17	1	53' 48" 18.3°	105' 10" 47.3°	5	27-Sep-00	9:30 AM
17	2	53' 48" 14.6°	105' 10" 46.6°	17	27-Sep-00	10:24 AM
17	3	53' 48" 11.4°	105' 10" 58.5°	43	27-Sep-00	11:30 AM
17	4	53' 48" 11.4°	105' 11" 02.7°	6	27-Sep-00	12:47 PM
17	5	53' 48" 06.9°	105' 11" 04.1°	12	27-Sep-00	2:24 PM
19	1	53' 48" 45.7°	105' 21" 47.1°	7	02-Oct-00	10:28 AM
19	2	53' 48" 48.8°	105' 21" 53.3°	7	06-Oct-00	4:02 PM
19	3	53' 48" 47.6°	105' 22" 02.9°	9	02-Oct-00	1:41 PM
19	4	53' 48" 52.0°	105' 21" 59.8°	7	02-Oct-00	10:45 AM
19	5	53' 48" 51.5°	105' 22" 01.2°	6	02-Oct-00	11:30 AM
21	1	53' 48" 28.7°	105' 20" 47.7°	9	06-Oct-00	9:45 AM
21	2	53' 48" 32.3°	105' 21" 00.4°	12	06-Oct-00	10:30 AM
21	3	53' 48" 35.5°	105' 21" 00.6°	9	06-Oct-00	1:04 PM
21	4	53' 48" 39.7°	105' 20" 59.8°	7	06-Oct-00	1:15 PM
21	5	53' 48" 36.7°	105' 21" 11.0°	5	06-Oct-00	2:34 PM
32	1	53' 59" 47.7°	105' 25" 47.8°	10	02-Oct-00	2:38 PM
32	2	53' 59" 51.9°	105' 25" 58.8°	6	03-Oct-00	3:27 PM
32	3	53' 59" 54.2°	105' 25" 59.4°	6	03-Oct-00	4:14 PM
32	4	53' 59" 58.0°	105' 25" 59.0°	5	03-Oct-00	4:51 PM
32	5	54' 00" 01.2°	105' 25" 53.4°	5	03-Oct-00	5:35 PM
43	1	54' 00" 24.4°	105' 54" 29.8°	7	20-Sep-00	8:48 AM
43	2	54' 00" 24.6°	105' 54" 35.7°	5	20-Sep-00	9:30 AM
43	3	54' 00" 43.8°	105' 54" 26.7°	5	20-Sep-00	10:09 AM
43	4	54' 00" 43.7°	105' 54" 23.5°	16	20-Sep-00	12:22 PM
43	5	54' 00" 40.6°	105' 54" 25.2°	5	20-Sep-00	1:12 PM
44	1	54' 05" 10.1°	106' 01" 18.5°	5	30-Aug-00	12:25 PM
44	2	54' 05" 10.8°	106' 01" 25.6°	13	30-Aug-00	12:40 PM
44	3	54' 05" 23.5°	106' 01" 20.8°	10	30-Aug-00	1:10 PM
44	4	54' 05" 17.7°	106' 01" 05.4°	6	30-Aug-00	1:37 PM
44	5	54' 05" 03.2°	106' 00" 41.4°	7	30-Aug-00	3:53 PM
46	1	54' 14" 15.8°	105' 56" 49.7°	10	29-Aug-00	12:16 PM
46	2	54' 14" 16.5°	105' 56" 44.1°	5	29-Aug-00	12:40 PM
46	3	54' 14" 15.5°	105' 56" 46.5°	22	29-Aug-00	1:03 PM
46	4	54' 14" 11.5°	105' 56" 53.2°	10	29-Aug-00	1:38 PM
46	5	54' 14" 09.9°	105' 56" 53.6°	7	29-Aug-00	1:56 PM
50	1	54' 01" 27.1°	105' 25" 07.2°	7	04-Oct-00	9:34 AM
50	2	54' 01" 37.7°	105' 25" 05.2°	7	04-Oct-00	10:48 AM
50	3	54' 01" 40.0°	105' 25" 07.1°	8	04-Oct-00	11:45 AM
50	4	54' 01" 10.8°	105' 24" 53.8°	9	04-Oct-00	1:02 PM
50	5	54' 01" 07.9°	105' 25" 00.4°	5	04-Oct-00	1:52 PM

51	1	53' 48" 23.0°	105' 08" 03.9°	5	31-Aug-00	10:59 AM
51	2	53' 48" 23.0°	105' 08" 09.5°	5	31-Aug-00	11:18 AM
51	3	53' 48" 18.7°	105' 08" 08.9°	4	31-Aug-00	11:31 AM
51	4	53' 48" 18.9°	105' 08" 03.0°	4	31-Aug-00	11:41 AM
51	5	53' 48" 12.2°	105' 07" 59.1°	5	31-Aug-00	12:10 PM
53	1	53' 55" 16.3°	105' 29" 09.3°	4	03-Oct-00	9:00 AM
53	2	53' 55" 14.1°	105' 29" 03.9°	5	03-Oct-00	10:14 AM
53	3	53' 55" 10.6°	105' 28" 59.3°	5	03-Oct-00	11:09 AM
53	4	53' 55" 10.5°	105' 28" 58.0°	7	03-Oct-00	11:18 AM
53	5	53' 55" 09.1°	105' 28" 57.8°	4	03-Oct-00	12:57 PM
54	1	53' 46" 37.9°	105' 06" 12.9°	4	26-Sep-00	7:30 AM
54	2	53' 46" 38.0°	105' 06" 08.4°	6	26-Sep-00	9:30 AM
54	3	53' 46" 41.2°	105' 06" 13.1°	5	26-Sep-00	10:25 AM
54	4	53' 46" 39.4°	105' 06" 34.3°	4	25-Sep-00	3:30 PM
54	5	53' 46" 35.7°	105' 06" 33.7°	7	25-Sep-00	4:30 PM
55	1	53' 46" 54.6°	105' 01" 38.1°	4	31-Aug-00	8:55 AM
55	2	53' 46" 52.6°	105' 01" 28.8°	6	31-Aug-00	9:17 AM
55	3	53' 46" 46.1°	105' 01" 28.3°	6	31-Aug-00	9:30 AM
55	4	53' 46" 49.0°	105' 01" 43.5°	5	31-Aug-00	9:59 AM
55	5	53' 46" 49.0°	105' 01" 49.1°	8	31-Aug-00	10:11 AM
57	1	54' 11" 28.5°	106' 04" 19.2°	7	24-Aug-00	3:14 PM
57	2	54' 11" 15.3°	106' 04" 17.4°	5	12-Sep-00	2:40 PM
57	3	54' 11" 01.8°	106' 04" 15.5°	5	12-Sep-00	3:19 PM
57	4	54' 11" 00.7°	106' 04" 37.8°	5	12-Sep-00	4:40 PM
57	5	54' 11" 14.4°	106' 04" 38.5°	5	12-Sep-00	6:08 AM
58	1	54' 05" 47.8°	106' 00" 23.0°	5	07-Sep-00	8:36 AM
58	2	54' 05" 46.0°	106' 00" 37.2°	7	07-Sep-00	9:40 AM
58	3	54' 05" 44.7°	106' 00" 42.3°	7	07-Sep-00	9:52 AM
58	4	54' 05" 44.2°	106' 00" 48.1°	5	07-Sep-00	10:12 AM
58	5	54' 05" 43.6°	106' 00" 57.5°	10	07-Sep-00	11:10 AM

Appendix C

Exotic species observed in the study sites

Exotic species observed in the study along with the associated common names and species abbreviations used in Figure 5.4 and Appendix D.

Species	**Common name**	**Species Abbreviation**
Agropyron repens (L.) Beauv.	quack grass	Agr rep
Bromus inermis Leyss.	smooth brome	Bro ine
Cirsium arvense (L.) Scop.	Canada thistle	Cir arv
Crepis tectorum L.	annual hawksbeard	Cre tec
Festuca rubra L.	creeping red fescue	Fes rub
Lappula squarrosa (Retz.) Dumort.	blue bur	Lap squ
Lolium persicum Boiss. & Hohen.	Persian darnel	Lol per
Matricaria matricarioides (Less.) Porter	pine-apple weed	Mat mat
Medicago lupulina L.	black medick	Med lup
Medicago sativa L.	alfalfa	Med sat
Melilotus alba Descr.	white sweet clover	Mel alb
Melilotus officinalis (L.) Lam.	yellow sweet clover	Mel off
Panicum capillare L.	witch grass	Pan cap
Phleum pratense L.	timothy	Phl pra
Plantago major L.	common plantain	Pla maj
Poa annua L.	annual bluegrass	Poa ann
Poa compressa L.	Canada bluegrass	Poa com
Poa pratensis L.	Kentucky bluegrass	Poa pra
Sonchus arvensis L.	perennial sow thistle	Son arv
Taraxacum officinale Weber	common dandelion	Tar off
Trifolium hybridum L.	alsike clover	Tri hyb
Trifolium pratense L.	red clover	Tri pra
Trifolium repens L.	white clover	Tri rep

Appendix D

The ranking of each exotic plant species observed

The following ranking system was developed by Hiebert and Stubbendieck (1993) and involves plant species and population attributes. The ranking of exotic plants will provide an assessment that predicts the potential of a species to become a pest in the future. Once the species has been ranked, a plot of the significance of impact vs. the feasibility of control will determine which species need an immediate response (Figure D.1). Further descriptions of ranking procedures and criteria are provided by Hiebert and Stubbendieck (1993). The ranking of the exotic species observed in the study sites is based on the following criteria;

I. Significance of Impact

A. Current Level of Impact

1. Distribution relative to disturbance regime	
a. found only within sites disturbed within the last 3 years of sites regularly disturbed	-10
b. found in sites disturbed within the last 10 years	1
c. found in mid-successional sites disturbed 11-50 years before present (BP)	2
d. found in late-successional sites disturbed 51-100 years BP	5
e. found in high quality natural areas with no known major disturbance for 100 years	10
2. Abundance	
a. number of populations (stands)	
(1) few; scattered (<5)	1
(2) intermediate number; patchy (6-10)	3
(3) several; widespread and dense (>10)	5
b. aerial extent of populations	
(1) <5 ha	0
(2) 5-10 ha	2
(3) 11-50 ha	3
(4) >50 ha	5
3. Effect on natural processes and character	
a. plant species having little or no effect	0
b. delays establishment of native species in disturbed sites up to 10 years	3
c. long-term (more than 10 years) modification or retardation of succession	7
d. invades and modifies existing native communities	10
e. invades and replaces native communities	15
4. Significance of threat to park resources	
a. threat to secondary resources negligible	0
b. threat to areas' secondary (successional) resources	2
c. endangerment to areas' secondary (successional) resources	4
d. threat to areas' primary resources	8
e. endangerment to areas' primary resources	10

5. Level of visual impact to an ecologist

a. little or no visual impact on landscape	0
b. minor visual impact on natural landscape	2
c. significant visual impact on natural landscape	4
d. major visual impact on natural landscape	5

Possible Total = 50

B. Innate Ability of Species to Become a Pest

1. Ability to complete reproductive cycle in area of concern

a. not observed to complete reproductive cycle	0
b. observed to complete reproductive cycle	5

2. Mode of reproduction

a. reproduces almost entirely by vegetative means	1
b. reproduces only by seeds	3
c. reproduces vegetatively and by seed	5

3. Vegetative reproduction

a. no vegetative reproduction	0
b. vegetative reproduction rate maintains population	1
c. vegetative reproduction rate results in moderate increase in population size	3
d. vegetative reproduction rate results in rapid increase in population size	5

4. Frequency of sexual reproduction for mature plant

a. almost never reproduces sexually in area	0
b. once every five or more years	1
c. every other year	3
d. one or more times a year	5

5. Number of seeds per plant

a. few (0-10)	1
b. moderate (11-1,000)	3
c. many seeded (>1,000)	5

6. Dispersal ability

a. little potential for long distance dispersal	0
b. great potential for long distance dispersal	5

7. Germination requirements

a. requires open soil and disturbance to germinate	0
b. can germinate in vegetated areas but in a narrow range or in special conditions	3
c. can germinate in existing vegetation in a wide range of conditions	5

8. Competitive ability

a. poor competitor for limiting factors	0
b. moderately competitive for limiting factors	3
c. highly competitive for limiting factors	5

9. Known level of impact in natural areas

a. not known to cause impacts in any other natural area	0
b. known to cause impacts in natural areas, but in other habitats and different climate zones	1
c. known to cause low impact in natural areas in similar habitats and climate zones	3
d. known to cause moderate impact in natural areas in similar habitats and climate zones	5
e. known to cause high impact in natural areas in similar habitats and climate zones	10

Possible Total = 50

II. Feasibility of Control or Management

A. Abundance Within Area

1. Number of populations (stands)

a. several; widespread and dense	1
b. intermediate number; patchy	3
c. few; scattered	5

2. Aerial extent of populations

a. > 50	1
b. 11-50 ha	2
c. 5-10	3
d. < 5ha	5

B. Ease of Control

1. Seed banks

a. seeds remain viable in the soil for at least 3 years	0
b. seeds remain viable in the soil for 2-3 years	5
c. seeds viable in the soil for 1 year or less	15

2. Vegetative regeneration

a. any plant part is a viable propagule	0
b. sprouts from roots or stumps	5
c. no resprouting following removal of aboveground growth	10

3. Level of effort required

a. repeated chemical or mechanical control measures required	1
b. one or two chemical or mechanical treatments required	5
c. can be controlled with one chemical treatment	10
d. effective control can be achieved with mechanical treatment	15

4. Abundance and proximity of propagules near study area

a. many sources of propagules near park	0
b. few sources of propagules near park, but these are readily dispersed	5
c. few sources of propagules near park, but these are not readily dispersed	10
d. no sources of propagules are in close proximity	15

C. Side Effects of Chemical/Mechanical Control Measures

1. control measures will cause major impacts to community	0
2. control measures will cause moderate impacts to community	5
3. control measures will have little or no impact on community	15

D. Effectiveness of Community Management

1. the following options are not effective 0
2. cultural techniques (burning, flooding) can be used to control target species 5
3. routine management of community or restoration or preservation practices (e.g., prescribed burning, flooding, controlled disturbance) effectively controls target species 10

E. Biological Control

1. biological control not feasible (not practical possible, or probable) 0
2. potential may exist for biological control 5
3. biological control feasible 10

Total Possible = 100

Urgency
1. Delay in action will result in large increase in effort required for successful control. High
2. Delay in action will result in moderate increase in effort required for successful control. Medium
3. Delay in action will result in little increase in effort required for successful control Low

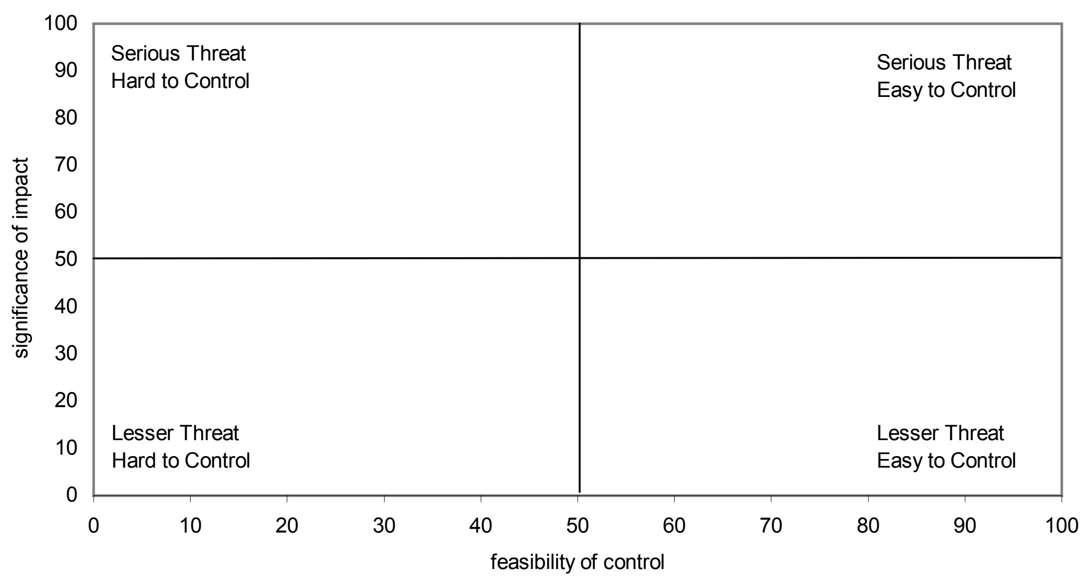

Figure D.1 The plot used to determine the threat and amount of control needed for each exotic species

Table D1. The results of the ranking of each exotic species observed in the study sites. Species abbreviations can be deciphered on page 112.

species	Agr rep	Bro ine	Cir arv	Cre tec	Fes rub	Lap squ	Lol per	Mat mat
I. Significance of Impact								
A. Current Level of Impact								
Question 1	-10	-10	1	1	-10	-10	-10	-10
2a	3	5	3	3	5	1	1	1
2b	0	2	2	2	2	0	0	0
3	0	15	0	0	7	0	0	0
4	0	2	0	0	0	0	0	0
5	0	4	0	0	2	0	0	0
B. Innate Ability of Species to Become a Pest								
1	5	5	0	5	5	5	0	5
2	5	5	5	3	5	3	3	3
3	3	5	5	0	1	0	0	0
4	5	5	5	5	5	5	5	5
5	3	5	5	5	3	5	3	5
6	0	0	5	5	0	5	0	0
7	5	3	0	5	3	3	3	3
8	5	5	5	3	5	0	0	0
9	1	1	5	0	0	0	0	0
total	25	52	41	37	33	17	5	12
II. Feasibility of Control								
A. Abundance within Area								
1	3	1	3	3	1	5	5	5
2	5	3	5	5	3	5	5	5
B. Ease of Control								
1	0	0	0	0	5	0	5	0
2	5	5	5	10	5	10	5	10
3	1	1	1	1	5	15	10	15
4	10	0	0	5	10	10	15	15

species	Agr rep	Bro ine	Cir arv	Cre tec	Fes rub	Lap squ	Lol per	Mat mat
C. Side Effects of Chemical/Mechanical Control Measures								
	15	5	15	15	5	15	15	15
D. Effectiveness of Community Management								
	10	10	10	5	10	10	10	10
E. Biological Control								
	0	0	5	0	0	0	0	0
Total	49	25	44	44	44	70	70	75
Urgency	low	med	low	low	low	low	low	low

Table D1 cont'd. The results of the ranking of each exotic species observed in the study sites.

species	Med lup	Med sat	Mel alb	Mel off	Pan cap	Pla maj	Phl pra	Poa ann
I. Significance of Impact								
A. Current Level of Impact								
Question 1	-10	-10	-10	-10	-10	-10	-10	1
2a	3	3	3	1	1	3	1	1
2b	0	0	0	0	0	0	0	2
3	0	0	0	0	0	0	0	0
4	0	0	0	0	0	0	0	0
5	2	2	4	0	0	0	0	0
B. Innate Ability of Species to Become a Pest								
1	5	5	5	5	0	5	5	0
2	3	5	3	3	3	5	3	3
3	0	1	0	0	0	0	0	0
4	5	5	5	5	5	5	5	5
5	5	5	5	5	3	5	3	3
6	0	0	0	0	0	0	0	0
7	0	0	0	0	3	0	3	3
8	0	3	5	5	0	3	3	3
9	0	1	1	1	0	0	0	0
total	13	20	21	15	5	16	13	21
II. Feasibility of Control								
A. Abundance within Area								
1	3	3	3	5	5	3	5	5
2	5	5	5	5	5	5	5	5
B. Ease of Control								
1	0	0	0	0	5	0	5	5
2	10	5	5	5	5	5	5	5
3	5	10	1	1	15	1	5	1
4	15	0	10	10	15	15	10	15

species	Med lup	Med sat	Mel alb	Mel off	Pan cap	Pla maj	Phl pra	Poa ann
C. Side Effects of Chemical/Mechanical Control Measures								
	15	15	15	15	15	15	15	15
D. Effectiveness of Community Management								
	10	10	10	10	10	10	10	5
E. Biological Control								
	0	0	5	5	0	0	0	0
Total	63	48	54	56	75	54	60	56
Urgency	low	low	low	low	low	low	low	low

Table D1 cont'd. The results of the ranking of each exotic species observed in the study sites.

species	Poa com	Poa pra	Son arv	Tar off	Tri hyb	Tri pra	Tri rep
I. Significance of Impact							
A. Current Level of Impact							
Question 1	5	1	1	1	-10	-10	-10
2a	1	1	3	5	5	3	1
2b	2	2	2	2	2	0	0
3	0	0	0	0	3	3	0
4	0	0	0	0	0	0	0
5	0	0	0	0	2	2	0
B. Innate Ability of Species to Become a Pest							
1	0	0	5	0	5	5	5
2	5	5	5	5	3	3	5
3	3	5	0	0	0	0	1
4	5	5	5	5	5	5	5
5	3	3	5	5	5	5	5
6	0	0	5	5	0	0	0
7	5	5	5	5	0	0	0
8	3	5	5	5	0	0	0
9	0	1	1	3	0	0	0
total	32	33	42	41	20	16	12
II. Feasibility of Control							
A. Abundance within Area							
1	5	5	3	1	1	3	5
2	5	5	5	5	3	5	5
B. Ease of Control							
1	5	5	0	0	0	0	0
2	5	5	5	5	5	5	5
3	1	1	1	1	5	5	5
4	15	10	0	5	10	10	10

species	Poa com	Poa pra	Son arv	Tar off	Tri hyb	Tri pra	Tri rep
C. Side Effects of Chemical/Mechanical Control Measures							
	15	15	15	15	15	15	15
D. Effectiveness of Community Management							
	5	10	10	10	10	10	10
E. Biological Control							
	0	0	5	5	0	0	0
Total	56	56	44	47	49	53	55
Urgency	low	low	low	low	low	low	low

Appendix E

A list of the native plants observed in the study sites

Table E1 A list of the native species observed in the study sites.

Observed Native Species
Abies balsamea (L.) Mill.
Achillea millefolium L.
Achillea sibirica Ledeb.
Actaea rubra (Ait.) Willd.
Adoxa moschatellina L.
Agoseris glauca (Pursh) Raf.
Agropyon dasystachyum (Hook.) Scribn.
Agrostis scabra Willd.
Agropyon smithii Rydb
Agrimonia striata Michx.
Agropyon subsecundum (Link) A.S. Hitche
Agropyon trachycaulum (Link)
Alnus crispa (Ait.) Pursh
Alnus rugosa (DuRoi) Spreng.
Amelanchier alnifolia Nutt.
Antennaria microphylla Rydb
Antennaria neglecta Greene
Apocynum androsaemifolium L.
Aquilegia brevistyla Hook.
Aralia nudicaulis L.
Arctostaphylos uva-ursi (L.) Spreng.
Arenaria lateriflora (L.) Fenzl.
Aster borealis (T.G.) Prov.
Aster ciliolatus Lindl.
Aster conspicuus Lindl.
Aster hesperius A. Gray
Aster laevis L.
Aster puniceus L.
Betula occidentalis Hook.
Betula papyrifera Marsh.
Betula pumila L.
Bromus ciliatus L.
Calamagrostis canadensis (Michx.) Beauv.
Calamagrostis inexpansa A. Gray
Caltha palustris L.
Calamagrostis purpurascens R. Br.
Calamagrostis rubescens Buckl.
Campanula rotundifola L.
Chenopodium capitatum (L.) Aschers.
Cicuta maculata L.
Circaea alpina L.

Observed Native Species
Collomia linearis Nutt.
Coptis trifolia (L.) Salisb.
Cordalis aurea Willd.
Cornus canadensis L.
Corylus cornuta Marsh.
Corallorhiza maculata Raf.
Cordalis sempervirens (L.) Pers.
Cornus stolonifera Michx.
Deschampsia cespitosa (L.) Beauv.
Disporum trachycarpum (S. Wats.) B. & H.
Dracocephalum parviflorum Nutt.
Elymus innovatus Beal
Elymus virginicus L.
Epilobium angustifolium L.
Epilobium ciliatum Raf.
Epilobium glandulosum Lehm.
Epilobium watsonii Barbey
Equisetum arvense L.
Equisetum hyemale L.
Equisetum pratense Ehrh.
Equisetum scirpoides Michx.
Equisetum slyvaticum L.
Erigeron canadensis L.
Erigeron glabellus Nutt.
Fragaria vesca L.
Fragaria virginiana Duchesne
Fungi spp.
Galium boreale L.
Galium triflorum Michx.
Gentiana amarella (L.) Borner
Geocaulon lividum (Richards.) Fern.
Geranium bicknellii Britt.
Geum aleppicum Jacq.
Geum rivale L.
Glyceria grandis S. Wats. ex A. Gray
Goodyera repens (L.) R.Br.
Gymnocarpium dryopteris (L.) Newm.
Habenaria hyperborea (L.) R.Br.
Habenaria obtusata (Pursh) Richards.
Habenaria viridis (L.) R.Br.
Halenia deflexa (Sm.) Griseb
Hedysarum alpinum L.
Hieracium umbellatum L.
Hordeum jubatum L.

Observed Native Species
Lappula squarrosa (Retz.) Dumort
Larix laricina (DuRoi) K.Koch
Lathyrus ochroleucus Hook.
Lathyrus venosus Muhl.
Ledum groenlandicum Oeder
Lepidium densiflorum Schrad.
Lilium philadelphicum L.
Linnaea borealis L.
Lonicera dioica L.
Lonicera involucrata (Richards.) Banks
Lycopodium annotinum L.
Lycopodium clavatum L.
Lycopodium complanatum L.
Lycopodium obscurum L.
Maianthemum canadense Desf.
Marchantia polymorpha L.
Melampyrum lineare Descr.
Mentha arvensis L.
Mertensia paniculata (Ait.) G. Don.
Mitella nuda L.
Moss spp.
Oryzopsis asperifolia Michx.
Oryzopsis hymenoides (R. & S.) Ricker
Oryzopsis pungens (Torr.) A.S. Hitchc.
Osmorhiza longistylis (Torr.) DC.
Parnassia palustris L.
Petasites palmatus (Ait.) A. Gray
Phacelia franklinii (R.Br.) A. Gray
Picea glauca (Moench) Voss
Picea mariana (Mill.) BSP.
Picea pungens Engelm.
Pinus banksiana Lamb.
Poa palustris L.
Populus balsamifera L.
Populus tremuloides Michx.
Potentilla fruticosa L.
Potentilla gracilis Dougl. ex Hook.
Potentilla norvegica L.
Potentilla tridentata Ait.
Prunus pensylvanica Lif.
Prunus virginiana L.
Pyrola asarifolia Michx.
Pyrola secunda (L.) House
Pyrola virens Schweig.

Observed Native Species
Ranunculus abortivus L.
Ribes americanum Mill.
Ribes glandulosum Grauer
Ribes lacustre (Pers.) Poir.
Ribes oxyacanthoides L.
Ribes triste Pall.
Rosa acicularis Lindl.
Rubus idaeus L.
Rubus pubescens Raf.
Rush spp.
Salix spp.
Schizachne purpurascens (Torr.) Swallen
Sedge spp.
Senecio congestus (R.Br.) DC.
Senecio eremophilus Richards.
Senecio pauperculus Pursh
Shepherdia canadensis (L.) Nutt.
Smilacina stellata (L.) Desf.
Solidago canadensis L.
Solidago missouriensis Nutt.
Solidago multiradiata Ait.
Solidago rigida L.
Solidago spathulata DC
Sorbus scopulina Greene
Spiranthes romanozoffiana Cham. & Schlecht
Stellaria longifolia Muhl.
Symphoricarpos albus (L.) Blake
Symphoricarpos occidentalis Hook.
Trientalis borealis Raf.
Triglochin maritima L.
Urtica dioica L.
Vaccinium angustifolium Ait.
Vaccinium myrtilloides Michx.
Vaccinium vitis-idaea L.
Viburnum edule (Michx.) Raf.
Vicia americana Muhl.
Viola adunca J.E. Smith
Viola canadensis L.
Viola nephrophylla Greene
Viola renifolia A. Gray