Assessing the vulnerability of rare plants using climate change velocity, habitat connectivity and dispersal ability

by

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Abstract

Climate change generally requires species to migrate northward or to higher elevation to maintain constant climate conditions, but migration requirement and migration capacity of individual species can vary greatly. Individual populations of species occupy different positions in the landscape that determine their required range shift to maintain similar climate, and likewise the migration capacity depends on habitat connectivity. Here, I demonstrate an approach to quantify species vulnerabilities to climate change for 419 rare vascular plants in Alberta, Canada based on multivariate velocity of climate change, local habitat fragmentation, and migration capacity. Climate change velocities indicated that future migration requirements ranged from 1 to 5 km yr^{-1} in topographically complex landscapes, such as the Alberta Foothills and Rocky Mountains. In contrast, migration requirements to maintain constant climate in relatively flat Boreal Plains, Parkland and Grassland ranged from 4 to 8 km yr^{-1}. Habitat fragmentation was also highest in these flat regions, particularly the Parkland Natural Region. Of the 419 rare vascular plants assessed, 36 were globally threatened (G1 to G3 ranking). Three of these globally threatened species were ranked as extremely vulnerable and five as highly vulnerable to the interactions among climate change velocity, habitat fragmentation and migration capacity. Incorporating dispersal characteristics and habitat fragmentation with local patterns in climate change velocity represents a streamlined vulnerability assessment approach that may be applied to guide conservation actions, particularly where detailed species-specific data is limited.

Preface

A version of this thesis, titled "Assessing the vulnerability of rare plants using climate change velocity, habitat connectivity and dispersal ability: a case study in Alberta, Canada", has been submitted for publication in *Regional Environmental Change*. Conceptual and methodological development were aided by my co-supervisors Andreas Hamann and Scott Nielsen, who are co-authors on the submitted paper.

The multivariate climate change velocity approach used in this thesis was published in *Global Change Biology* (Hamann *et al.*, 2015), of which I am a co-author. My contributions to this paper were calculation of traditional climate change velocity for comparative purposes and a resolution sensitivity analysis. My contribution to this paper accounts for approximately 10%.

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Bad times have a scientific value. These are occasions a good learner would not miss.

- Ralph Waldo Emerson

For Elise,

the wind in my sails.

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Table of Contents

List of Tables

List of Figures

1. General introduction

1.1 Introduction

Populations of species have historically responded to climatic disequilibrium in one of three ways: they adapted to their new environment, they moved to track suitable climates, or they were locally extirpated. Unfortunately, climate change projected for the 21st century will be extreme in both rate and magnitude (IPCC, 2014; Jackson & Overpeck, 2000). Similar periods of abrupt climate change since the Last Glacial Maximum triggered large-scale population collapses and expansions (Tinner & Lotter, 2001) and modern climate change will likely cause comparable ecological shifts. Upward shift of tree lines may threaten high-elevation species (Dirnböck *et al.*, 2011), while other species may benefit from longer growing seasons and increased CO_2 concentrations (Knapp *et al.*, 2001). Boreal forests may be displaced by grasslands from the south. Climate change in the last half-century has already generated a response among diverse taxa (Parmesan & Yohe, 2003), although plant taxa have frequently lagged behind their established climatic niches (Bertrand *et al.*, 2011; Gray & Hamann, 2013).

Species' ability to persist and even thrive under climate change will be determined partly by their ability to migrate in tandem with changing climate.

Range shift is determined by the balance of range expansion through seed dispersal at the leading edge, and range contraction at the trailing edge. Seed dispersal strategies vary greatly between species, with the majority of plant species dispersing seeds through wind dispersal, water dispersal, animal-aided dispersal, self-dispersal, or some combination of these (Howe & Smallwood, 1982). Modelling studies suggest that seed dispersal capability is the main determinant of plant migration velocity (Sandel *et al.*, 2011), but forecasting migration rates is difficult even when seed dispersal is well-studied.

Ecologists have attempted to estimate plant migration rates using mechanistic models of seed dispersal (Nathan *et al.*, 2002), fossilized pollen records (Davis, 1981; Delcourt & Delcourt, 1987), palaeoclimate reconstructions (Normand *et al.*, 2011), population growth models (Clark, 1998), genetic tracking of seeds and/or pollen (Bacles *et al.*, 2006; Cain *et al.*, 2000), DNA surveys (McLachlan *et al.*, 2005), and physical seed tracking through markers or seed traps (Mack, 1995). Most studies agree that plants rarely migrate more than a few tens of meters per year through regular seed dispersal mechanisms (Howe & Smallwood, 1982; Willson, 1993), but many species could not have spread from the Last Glacial Maximum to their current distributions using these processes alone (Reid, 1899). High-latitude cryptic glacial refugia and rare long-distance seed dispersal events have been proposed to explain this discrepancy (Clark, 1998; Svenning *et al.*, 2008).

Plant migration since the Last Glacial Maximum may provide an indication of how plants will respond to 21st-century climate change, which will also challenge plants' ability to track their climatic niche through normal dispersal processes (Corlett & Westcott, 2013; Loarie *et al.*, 2009). While the rapidity of modern climate change is not unique in the geologic record, several novel factors exacerbate the risk it poses to plants. Human activities have caused land use changes and habitat fragmentation across most of the Earth's surface, with direct impacts on approximately 83% of the Earth's surface, excluding

Antarctica (Sanderson *et al.*, 2002). This fragmentation will limit colonization at northern range boundaries, slowing migration rates significantly (Honnay *et al.*, 2002; Meier *et al.*, 2012).

As a consequence, human interventions through managed relocation (i.e. assisted migration) may be necessary to ensure the survival of species that are threatened by loss of climatically-suitable habitat. However, assisted migration is a controversial tool for conservation among ecologists (McLachlan *et al.*, 2007; Ricciardi & Simberloff, 2009). Also, the rate and magnitude of projected climate change have significant error margins and will vary greatly depending on emission scenario among other factors (IPCC, 2013), forcing conservation managers to make assumptions about future climates. Thomas *et al.* (2004) used species distribution modelling to predict the shifting range of 1,103 animal and plant species under projections of climate change. Depending on species dispersal ability and emission scenario, they found that 15% to 37% are committed to extinction.

Rare or endemic plant species are among the most vulnerable of taxa with regards to global climate change (IPCC, 2014; Thomas *et al.*, 2004). First, endemic and rare species have very little room for range contraction, yet a narrow range of environmental tolerances reduces the availability of suitable habitats for colonization (Parmesan, 2006). Secondly, endemic species often have limited intraspecific genetic variation, preventing adaptation to climate change that that may occur faster in common tree species (Rehfeldt *et al.*, 2002; Savolainen *et al.*, 2007). Furthermore, it is difficult to characterize rarely-studied species' climate vulnerability, since often factors such as ecological niche, seed dispersal ability, or biotic interactions remain unknown.

The estimation of climate change vulnerability requires quantifying a species' sensitivity to climate change, its adaptive capacity, and its climate change exposure (Williams *et al.*, 2008). Various climate change metrics have been used to describe climate change exposure, including the absolute value of temperature or precipitation change, ratio of climate change to interannual variation, climate change velocity, and change in distance to baseline-analogous climate, among many others (Garcia *et al.*, 2014). While several climate change vulnerability assessments suggest climate anomaly as a measure of climate change exposure (Beardmore & Winder, 2011), climate change velocity (Loarie *et al.*, 2009) is more biologically-relevant since it provides an estimate of the distance required for plants to migrate in order to maintain their historical climate niche. Climate change velocity is improved further when adjusted to account for habitat fragmentation (Meier *et al.*, 2012) and seed dispersal ability (Sandel *et al.*, 2011). Climate change velocity can also be used to estimate migration vectors, which allows definition of migration corridors and identification of migration sources and sinks (Burrows *et al.*, 2014).

1.2 Thesis Objectives

In this thesis I develop an approach to assess climate change exposure of plant populations with a focus on rare and endangered species in Alberta. I aim to integrate multiple aspects of climate change vulnerability, including climate change exposure, landscape fragmentation and seed dispersal ability. The approach must be straightforward enough to be used for generally understudied, rare species. Species distribution modelling techniques, for example, are not suitable for assessment of poorly-sampled, range-restricted species. Here, I use a combination of climate change velocity, habitat fragmentation, and migration capacity inferred from seed dispersal ability. I use Alberta, Canada as a case study, where 419 vascular plants are locally ranked as critically imperiled (S1), imperiled (S2), or vulnerable (S3) (Young *et al.*, 2010). My vulnerability assessment may be used to guide resources toward intervention or monitoring of species populations that have been identified as likely threatened by climate change.

2. Literature Review

2.1 Global and regional climate change

Anthropogenic climate change is already producing a biological response on a global scale including range shifts across a variety of taxa (Parmesan *et al.*, 1999; Parmesan & Yohe, 2003). The Intergovernmental Panel on Climate Change (IPCC) has stated that the world has already warmed approximately 0.72℃ (between 0.49℃ and 0.89℃) over the period of 1951-2012 (Stocker *et al.*, 2013). The majority of this warming is attributed to an increase in anthropogenic greenhouse gases, of which carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and halocarbons (including chlorofluorocarons and hydrochlorofluorocarbons) account for most of the greenhouse effect (Stocker *et al.*, 2013). The concentration of these substances has increased from pre-industrial levels by 40%, 150%, and 20%, respectively (Stocker *et al.*, 2013). Global mean surface temperatures are projected to increase between 1.4℃ and 3.1℃ for 2081-2100 relative to 1986-2005, under RCP6.0, a moderate emission scenario (Stocker *et al.*, 2013). CO_2 is naturally removed from the atmosphere by land and ocean carbon sinks with a half-life of approximately 300 years, although based on current emission scenarios it is unlikely that the Earth will return to pre-industrial era baseline levels within the next 1000 years (Stocker *et al.*, 2013).

Global climate change acts heterogeneously across the Earth's surface, with temperature currently increasing approximately twice as fast over land as over the ocean (Stocker *et al.*, 2013). In general, 20th-century warming was most noticeable in the high arctic and above 45 ° latitude in the northern hemisphere, although land surface warming since the 1970s has affected the entire globe more uniformly (IPCC, 2013). This heterogeneous climate change places species in some regions at greater risk than species in other regions. However, even modest climate change can lead to extinctions of endemic species, particularly in regions of high biodiversity where species tend to occupy narrow ecological (and climatic) niches (Malcolm *et al.*, 2006; Myers *et al.*, 2000).

Increasing temperatures are only one aspect of climate change, and changes in precipitation patterns will be similarly important for plant species (Crimmins *et al.*, 2011). In contrast to temperature increases, which have been relatively uniform across mid-latidude land areas (IPCC, 2013), historical trends in precipitation have been highly heterogeneous. A general increase in global precipitation has been observed since 1901, although this has been accompanied by an increase in the frequency of heat waves and warm spells (IPCC, 2013), driving complex patterns of change in evapotranspirative demand. Several other changes to the water cycle have been projected, including changes in the frequency of extreme precipitation events and decreases in snowpack depth, among others (IPCC, 2013). These changes will have complex and sometimes contrasting effects on plant species.

Projections of future precipitation trends from general circulation models tend to be highly variable. Most models for North America project a decrease in summer precipitation and an increase in winter precipitation, with moderate confidence (IPCC, 2013). Models predict increased aridity and increased drought severity across mid-latitudinal regions, which is in agreement with observations through the 20th century (Stocker *et al.*, 2013). Increased drought frequency is particularly concerning given the narrow hydraulic tolerances of many forest species (Choat *et al.*, 2012), and the agricultural dependence on precipitation. Drought-induced extirpation of native plant species will likely be a major problem in the 21st century (Thomas *et al.*, 2004), although increased drought frequency may also facilitate plant range expansion for species that are restricted by competitive interaction and not by climate (Crimmins *et al.*, 2011). Unfortunately, predicting drought-induced species declines or expansions will be difficult since projections of precipitation trends and drought frequency changes remain uncertain, especially at fine geographic scales (IPCC, 2013).

2.2 Climate change velocity

Climate change velocity is essentially the instantaneous local velocity along Earth's surface needed to maintain constant temperatures (Loarie *et al.*, 2009), measured in km yr^{-1}. This spatial metric presents a biologically-relevant alternative metric to absolute changes in climate values, since it accounts for regional climate differences and the buffering effect of topographic heterogeneity (Ackerly *et al.*, 2010; Dobrowski *et al.*, 2013). For example, substantial migration northward is required to track a temperature increase of 1℃, but a relatively short migration uphill may act as a substitute, where topography permits. Climate change velocity accounts for these topographic variables, making it a more relevant measure of climate change exposure.

The calculation of climate change velocity is straightforward. For any given point, it is the temporal rate of climate change (in ℃ yr^{-1}) divided by two-dimensional spatial rate of climate change (in ℃ km^{-1}). For example, two adjacent 1 km^2 cells with a mean annual temperature difference of 1℃ and warming at a rate of 0.05℃ yr^{-1}, have a climate velocity of 0.05 km yr^{-1}. In practice the spatial temperature gradient is calculated over a 3-by-3 cell area.

Loarie *et al.* (2009) forecast a mean global velocity of 0.42 km yr^{-1} from 2000-2100 under an A1B emission scenario. Similarly, climate change velocity can be calculated for other climate variables; the climate change for mean annual precipitation is 0.22 km yr^{-1} under the same scenario (Loarie *et al.*, 2009). Climate change velocity is anticipated to be highest across plains, while it is lowest in topographically-diverse montane regions. Species occupying flat, high-velocity regions are anticipated to face higher migration requirements corresponding to higher extinction rates, especially for taxa with relatively low migration capacities (IPCC, 2014).

Several studies have followed the Loarie *et al.* (2009) methodology across a variety of spatial scales, time scales, and emission scenarios. Ackerly *et al.* (2010) calculated a mean climate change velocity of 0.27 km yr^{-1} across the southwestern United States from the current period to 2100. Burrows *et al.* (2011) analyzed the climate record from 1960 to 2009, finding a much higher global mean velocity of 2.73 km yr^{-1} at a spatial resolution of 1 °. In contrast to this high velocity, Dobrowski *et al.* (2013) calculated a mean T_{min} velocity of 0.081 km yr^{-1} for the contiguous United States from 1916 to 2005. In a study restricted to Europe, Schueler *et al.* (2014) estimated a mean climate change velocity of 0.47 km yr^{-1} using mean annual temperature. Ecologists have begun to use climate change velocity as a tool for forecasting climate change-induced migration (Burrows *et al.*, 2014; Pinsky *et al.*, 2013). The climate change velocity metric has also been used in the IPCC Fifth Assessment Report to estimate climate change vulnerability across different taxa (IPCC, 2014).

2.3 Multivariate climate change velocity

Climate migrants are anticipated to move predominantly to higher latitudes and altitudes in response to warming temperatures, although observed migration patterns do not always conform to these expectations (Lenoir & Svenning, 2015; Parmesan *et al.*, 1999). Counterintuitive migration patterns are often due to multiple climate variables driving migration in diverging directions (Crimmins *et al.*, 2011; Feeley *et al.*, 2011; Tingley *et al.*, 2012). Recent technical advances in climate change velocity, including the incorporation of multiple climatic factors, have proven more realistic than earlier approaches based on temperature change alone (Burrows *et al.*, 2014).

While useful, the approach documented by Loarie *et al.* (2009) (hereafter the "slope-based approach") is subject to a number of limitations (Hamann *et al.*, 2015). The slope-based approach relies on a 3-by-3 cell grid spatial gradient, which hinders its ability to calculate landscape-scale climate change velocity vectors. For example, true migration requirements may be grossly understated due to nearby migration dead-ends, or "climatic cul-de-sacs" (Hamann *et al.*, 2015). The most common example of such false destinations are mountain tops, where low climate velocities on nearby steep slopes falsely imply low migration requirements that are actually pointing beyond mountaintops. Mountains are considered among the most threatened ecosystems (Colwell *et al.*, 2008; Williams & Jackson, 2007), and mountaintop climatic extinction is already affecting high-elevation plants (Krushelnycky *et al.*, 2013). An inability to account for these false migration destinations means that traditional climate change velocity can underestimate velocities in topographically-diverse regions. Conversely, traditional climate velocity may overestimate velocities in flat regions, where a low spatial climate gradient implies extremely high velocities but in reality a reasonable migration destination is actually just outside of the 3-by-3 cell radius (Hamann *et al.*, 2015).

An alternative approach (hereafter the "distance-based approach") to climate change velocity is presented in Hamann *et al.* (2015). In this study I defined climate change velocity using the distance to nearest analogous climate. Using several climate variables summarized in a principal component analysis (PCA), climate matches are defined using bins of unique climatic variable combinations. Climate change velocities are then calculated as the minimum required geographic distance to a matching cell, divided by the number of years between the baseline climate period and the future projection. In this way, it is not only possible to address the issue of climatic cul-de-sacs (Hamann *et al.*, 2015), but also to measure climate change velocity as a composite of multiple climate variables. Since plant species' climatic constraints are defined by both precipitation and temperature (Crimmins *et al.*, 2011; Lenoir & Svenning, 2015), such a multivariate velocity calculation is necessary. Multivariate climate change velocity conforms broadly to trends identified by Loarie *et al.* (2009). Regions with exceptionally-high or infinite climate change velocity represent disappearing climates, which are primarily located on mountaintops and at high latitudes.

The distance-based approach has several novel applications, including the restriction of suitable destination cells to areas with limited human impact, and calculation of reverse climate change velocity for the identification of valuable climate refugia. Climate change velocities can be calculated as directional vectors using either the slope-based approach or the distance-based approach, the primary application of which is estimating migration directions. An example using the slope-based approach is provided by Burrows *et al.* (2014), who produce maps of climate change velocity for 1960 to 2009 and 2006 to 2100, showing that climate vectors will drive complex migration patterns on a global scale.

2.4 Migration and adaptation capacity of plants

Inevitably, some species will find their distribution in disequilibrium with their historical niche as climate change progresses. Historically, species have accommodated climate change through adaptation or migration. However, accumulating evidence suggests that adaptation to modern climate change will be difficult due to the time required for evolutionary changes (Crisp *et al.*, 2009; Peterson *et al.*, 1999; Wiens & Graham, 2005). The theory of ecological niche conservatism posits that species tend to retain their ancestral ecological niches and environmental distributions (Peterson *et al.*, 1999), with only limited ability to adapt to environmental changes through evolution. In essence, this theory posits that evolutionary adaptation is likely too slow or too infrequent to keep up with the pace of modern climate change, leaving migration as the primary mechanism through which plants will endure modern climate change (Peterson *et al.*, 1999).

Seed dispersal capability is the dominant factor influencing the rate at which plant species migrate (Coutts *et al.*, 2010). Unfortunately, seed dispersal capability is difficult to quantify, even for well-studied species. Modes of seed dispersal, or 'dispersal syndromes', can be roughly grouped into anemochory (dispersal by wind), hydrochory (dispersal by water), zoochory (dispersal by animals), and autochory (self-dispersal), and are sometimes correlated with a species' migration ability (Howe & Smallwood, 1982). For example, ferns propagate via wind-dispersed spores and are exceptionally-well dispersed. As a result their migration is primarily constrained by climate and not geographic dispersal barriers (Qian, 2009). Similarly, animal-aided and water-dispersed plants are more likely to migrate to isolated islands (Sorensen, 1986). Instances of rapid climate change have been correlated with temporary selection for more effective seed dispersal characteristics (Cwynar & MacDonald, 1987).

2.5 Long-distance dispersal

Plants commonly migrate between zero and a few tens of meters per generation, even between a variety of dispersal syndromes (Harper, 1977; Howe & Smallwood, 1982; Willson, 1993). These migration rates are not sufficient for tracking the rapid change that occurred in the late Quaternary (Clark, 1998;

Jackson & Overpeck, 2000), implying that meaningful migration rates may depend on rare seed-dispersal events over exceptionally long distances, known as "long-distance seed dispersal" (Baker, 1955). Even in the absence of wind-dispersal seed adaptations, wind dispersal by storms and updrafts are likely important for trans-continental seed dispersal (Nathan, 2006).

The importance of long-distance seed dispersal in post-glacial recolonization (Clark, 1998) suggests that it will be similarly important for plants tracking modern climate change. Unfortunately, the rarity of such long-distance seed dispersal and the possibility of secondary dispersal can make modelling of these processes difficult (Cain *et al.*, 2000; Nathan & Muller-Landau, 2000). Ecologists typically estimate patterns of long-distance seed dispersal by using genetic markers to identify parent populations or by mark and recapture methods modified for plants (Cain *et al.*, 2000). Alternatives using statistical models are provided by Levey *et al.* (2008), where the authors model the movement of avian seed dispersers in an experimental landscape, and Nathan and Muller-Landau (2000) who use a mechanistic model of wind dispersal to simulate seed dispersal from a stand of trees. These studies have yet to define a practical method of estimating long-distance dispersal capability, partly as a result of difficulties with scale. This is because experimental landscapes are either too small to relate meaningfully to the landscape on which seed dispersal occurs, or too large to be feasible for mark-and-recapture methods (Levey *et al.*, 2008).

As predictive models of long-distance dispersal are lacking (Nathan & Muller-Landau, 2000), a mechanistic framework can provide a reasonable alternative for estimating long-distance dispersal ability. For example, seed terminal velocity, seed abscission characteristics (Nathan *et al.*, 2011) and height of seed release are important for long-distance dispersal of forest tree seeds during prolonged turbulent updrafts (Nathan *et al.*, 2002). Unfortunately, morphological dispersal adaptations are poorly correlated with likelihood of long-distance dispersal (Higgins *et al.*, 2003). Higgins *et al.* (2003) found that long-distance dispersal ability may instead be related to the number of available dispersal vectors. This is supported by Normand *et al.* (2011), who used species distribution modelling and palaeoecological hindcasting' (Svenning *et al.*, 2008) to show that species with fewer dispersal vectors were not able to track climate as effectively following glacial retreat. In this way, long-distance migration capacity may be determined by both flexibility in dispersal syndrome, and specific adaptations for long-distance dispersal (e.g. spores dispersal by ferns).

2.6 Palaeoclimate reconstructions and historical plant migration

Climate change following the Pleistocene Glaciation is of particular importance since the relatively rapid change could provide an indication of how the biosphere will respond to anthropogenic climate change (Overpeck *et al.*, 1992). Glacial retreat from the last glacial maximum (LGM) to the early Holocene is understood to have played an important role in determining modern biodiversity patterns (Ohlemüller *et al.*, 2012). Realistic reconstructions of migration rates from this time may inform modern conservation efforts. Palaeoecologists have traditionally used sedimentary records of fossilized pollen to map historical migration rates, with two well-cited studies estimating range expansion rates between 172 and 214 m yr^{-1} (Davis, 1981; Delcourt & Delcourt, 1987). In reality, migration patterns were complex and driven largely by changes in seasonality (COHMAP Members, 1988).

Sandel *et al.* (2011) made the important discovery that climate change velocity is negatively correlated with species endemism, showing that areas with high climate change velocity in the late Quaternary period are marked by reduced endemism of small-ranged (<250,000 km^2) species. This relationship appears particularly potent for mammals, birds, and amphibians. This is suggestive that small-ranged species were restricted to low-velocity areas due to weak dispersal capabilities, narrow climatic tolerances, or other factors that might restrict range shift capabilities. Similarly, global patterns of endemism in vascular plants are correlated with past records of climate change, to the extent that past records of climate change are a better predictor of plant endemism than annual temperature range (Jansson, 2003). This reinforces the importance of regions with low climate change velocity for the conservation of species under modern climate change, and suggests that we can use climate velocity to pre-emptively identify species and regions at risk.

The palaeobotanist Clement Reid pointed out that oaks could not have migrated to northern Britain from southern Europe using established rates of seed dispersal (Reid, 1899), a problem named Reid's Paradox'. Conventional knowledge has suggested that plants repopulated land left vacant by retreating glaciers through rapid seed dispersion (Clark, 1998). However, most palaeoecologists agree that post-glacial colonization was accomplished through a combination of normal seed dispersal and recolonization from isolated populations in cryptic glacial refugia (Anderson *et al.*, 2006; McLachlan *et al.*, 2005; Svenning & Skov, 2007).

Palaeoecologists question whether the plant community during glacial retreat was in equilibrium with climate (the theory of "dynamic equilibrium") or if plants persisted outside of their climatic niche as they slowly migrated to new, suitable habitats (the theory of "disequilibrium") (Overpeck *et al.*, 1992). Numerous studies support the theory of dynamic equilibrium with vegetation closely tracking climate via rapid migration, (Clark, 1998; Pearson & Dawson, 2003; Webb III, 1986) while others support the theory of disequilibrium and slower migration rates (McLachlan *et al.*, 2005; Normand *et al.*, 2011; Skov & Svenning, 2004). For example, Svenning and Skov (2007) show that tree species in Central and Northern Europe are not yet at equilibrium with climate, and that instead a simple measure of geographical accessibility from glacial refuges explains most of the variation in tree diversity, which more closely aligns with the theory of disequilibrium. However, others suggest that plants track climate closely regardless of geographic accessibility (Normand *et al.*, 2011). The truth lies somewhere between the two theories, and it has implications for the ability of plant species to persist outside of their climatic optimum.

2.7 Novel climates, disappearing climates, and plant assemblages

High rates of climate change following glacial retreat at the LGM were associated with genus-level extinctions of mammals throughout North America and Europe (Stuart *et al.*, 2004). Plant species extinctions in this period have also been identified, but documented extinction events are relatively rare (Jackson & Overpeck, 2000). One example is *Picea critchfieldii*, a previously widespread species of spruce that went extinct following post-glacial climate change (Jackson & Weng, 1999). These extinctions have been variously ascribed to direct exploitation by humans, pathogens, disappearance of suitable climates, or failure to migrate to newly available habitat (Jackson & Weng, 1999).

Climate change will locally introduce climates that are not experienced at present (novel climates) (Williams & Jackson, 2007), which is anticipated to cause the formation of plant assemblages that are compositionally unlike any that currently exist (no-analogue plant communities) (Williams & Jackson, 2007). Conversely, we can expect to see the elimination of certain modern climates (disappearing climates) (Williams & Jackson, 2007), leading to elevated extinction risks as some plant species are forced to adapt or accommodate sub-optimal climates. Palaeoecological research suggests that similar processes took place in the late-Quaternary from 17,000 YBP to 12,000 YBP, where no-analogue plant communities were strongly correlated with novel climates, specifically hypercontinental climates with especially cold winters and warm summers (COHMAP Members, 1988; Williams *et al.*, 2001). Pollen records show that >40% of eastern North America was composed of no-analogue plant communities during this period (Overpeck *et al.*, 1992).

Mapping of disappearing climates and novel climates will be a valuable tool for anticipating the emergence of no-analogue communities. 16% of the terrestrial land surface is already considered novel based on climate change from 1960 through 2009 (Burrows *et al.*, 2014). In a study of western North America, Roberts & Hamann (2012) found that the coast mountains of the Pacific Northwest will experience no-analogue climates with very high precipitation and high summer temperatures. Ackerly *et al.* (2010) mapped modern climate change across California and Nevada, finding that over 98% of protected areas will experience locally-novel climates by 2100. Modern novel and disappearing climates may be correlated with ecosystem instability, presenting both conservation challenges and potential opportunities for establishing non-native species through managed relocation.

2.8 Modern migration and human barriers to migration

Northward migration in correlation with a warming climate has already been observed across a broad range of taxa (Hickling *et al.*, 2005; Parmesan *et al.*, 1999; Parmesan & Yohe, 2003; Thomas & Lennon, 1999; van Herk *et al.*, 2002). Some species have migrated upslope in a similar manner, with cooling from increased elevation replacing cooling from increased latitude (Devi *et al.*, 2008; Feeley *et al.*, 2011; Luckman & Kavanagh, 2000). Traditionally it has been assumed that migration will be northward and upslope, however Crimmins *et al.* (2011) found recent changes in Californian precipitation patterns drove downslope plant migration, even while temperature increases drove upslope migration elsewhere. This knowledge of migration vectors is particularly valuable for the assessment of plant taxa, many of which have lagged behind expected rates of range shift (Chen *et al.*, 2011; Gray & Hamann, 2013; Zhu *et al.*, 2012).

Disappearing climates, declining climates, and high climate velocities all translate to species falling out of climatic suitability. Individual mortality may occur through purely climatic processes, such as increased water stress leading hydraulic failure (Krushelnycky *et al.*, 2013), or through increased competition from non-native species that are newly climatically suitable (Mitchell *et al.*, 2009). Encroaching tree lines threaten high-altitude plants and insects, especially endemics residing in areas that were historically glacial refugia (Dirnböck *et al.*, 2011). There are many modern examples of plant communities lagging behind expected shifts of climate. For example, North American tree populations lagging behind their optimal climate niche by over 100 km (Gray & Hamann, 2013), and French plant communities having responded to less than half of observed temperature trends (Bertrand *et al.*, 2011). Plant taxa lag behind climate velocity because of comparatively-poor migration ability and the influence of geographic barriers, although relatively mobile taxa such as marine communities have tracked climate more closely (Pinsky *et al.*, 2013).

While it remains unclear how plant species tracked post-glacial climate change in the late Quaternary, it is evident that modern climate change will necessitate similar migration (Jackson & Overpeck, 2000). Climate change projected for the coming century is exceptional in both rate and magnitude, comparable to the rate of change during the Younger Dryas stadial approximately 12,000 YBP (Jackson & Overpeck, 2000). Abrupt climate change such as this led to immediate reorganization of terrestrial ecosystems (Tinner & Lotter, 2001). Any modern equivalent of this reorganization will be affected by land use changes such as urban development, agricultural development, infrastructure, and other human activities. Forman and Alexander (1998) estimate that approximately 15-20% of the United States was ecologically impacted by roads as of 1998. Habitat fragmentation can substantially effect velocities of plant migration (Meier *et al.*, 2012), for example, by reducing the ability of large mammals to aid in long-distance dispersal (Markl *et al.*, 2012) or by facilitating movement of wind-dispersed seed down roadways. Additionally, anthropochorhy (human-aided seed dispersal) facilitates essentially unlimited dispersal distances through accidental introduction or managed relocation. The degree to which landscape fragmentation effects seed dispersal will vary between dispersal syndromes, with some dispersal syndromes (i.e. myrmecochory, dispersal by ants) more heavily affected than others (i.e. endozoochory, dispersal through ingestion by birds and mammals) (Honnay *et al.*, 2002).

Human effects on seed dispersal are not limited to effects from habitat fragmentation. The evidence of cryptic refugia occurring during the LGM suggests that isolated northern populations of trees assisted in post-glacial colonization and accelerated range shift (Svenning & Skov, 2007). There is a potential for parallel processes to aid in modern range shift in response to climate change. Outlying populations or even individual plants may exist north of the species' established range, and where planted by humans may even exist outside of their historical northernmost range (Schwartz *et al.*, 2001; Woodall *et al.*, 2009). On the contrary, anthropogenic land use change may have driven the extirpation of many northern refugial plant populations. Dobrowski (2011) identifies local climate refugia as an extension of hindcasting by Atmosphere-Ocean General Circulation Models (AOGCMs), proposing that these locations are more likely to contain refugial populations and that their conservation may aid in latitudinal range shift for plant species. However, there is growing understanding that persistent climate refugia are unlikely under nearly all scenarios of climate change (Hannah *et al.*, 2014) and that conservation planning should not rely on the existence of these refugia.

2.9 Species distribution modelling and the missing migration link

Climate velocity is useful for estimating migration requirements, but it does not accommodate species' adaptive capacity or flexibility in environmental niche (Hamann *et al.*, 2015). Von Humboldt and Bonpland (1807) famously theorized a relationship between climate and plant species distribution, but ecologists have struggled to precisely define this species-environment relationship since. The use of models to quantify the species-environment relationship began in the 1920s (Guisan & Zimmermann, 2000; Pearson & Dawson, 2003), although later development in the computer and statistical sciences was required to advance models from expert-based (non-statistical, non-empirical) to spatially-explicit statistical models (Daly *et al.*, 2008).

Modern correlative ecological models are given a variety of names based on the statistical methods and environmental predictors used, and what I call "species distribution models" (SDMs) has alternately been called: bioclimatic envelope models, ecological niche models, habitat models, or resource selection functions, among various other names (Elith & Leathwick, 2009). In general, SDMs aim to characterize a species' realized niche (Hutchinson, 1957) based on statistical relationships between environmental variables and the species' distribution. SDMs rely on the assumption that "the best indicator of a species' climatic requirements is its current distribution" (Pearson & Dawson, 2003), and it must be stressed that SDMs are only useful where the species being modelled is approximately at equilibrium with the environment used to train the model. Although the use and interpretation of correlative ecological models can be problematic for various reasons (Beale *et al.*, 2008; Schwartz *et al.*, 2006), many ecologists agree that SDMs are powerful for predicting current species distributions when informed by good survey design and model calibration (Elith & Leathwick, 2009; Guisan & Zimmermann, 2000).

SDM applications for hindcasting (extrapolating distributions to historical climates) or forecasting (extrapolating distributions to future climates) involves additional challenges and can be difficult to validate (Elith & Leathwick, 2009). Typically, baseline environmental data is used with species distribution data (presence or presence/abundance records) to develop the model, which is then applied to environmental data derived from projections of clanging climate (Elith & Leathwick, 2009; Guisan & Thuiller, 2005; Midgley *et al.*, 2006; Peterson *et al.*, 2002). There are several major criticisms of SDMs, particularly with regards to this extrapolation to new climate scenarios. First, models based on environmental variables cannot account for biotic interactions, which may restrict a species' range through competition or other processes (Davis *et al.*, 1998a,b). While this criticism is accurate, some ecologists argue that biotic interaction is a minor factor when considered on the landscape scale and especially with regard to plant species; Pearson *et al.* (2002) found good agreement between the distribution of 32 plant species and simulated distributions based on only 5 soil and environmental variables. A second common criticism is that SDMs are poorly-equipped to account for phenotypic plasticity or rapid adaptation in place of range shift (Pearson & Dawson, 2003). Thomas *et al.* (2001) observed rapid evolutionary change in British insects, while Rehfeldt *et al.* (2002) showed that intra-specific genetic variation will lead to increased growth for *Pinus sylvestris* as demographics shift. Existing genetic variation within pacific corals may shelter many species from rising ocean temperatures (Dixon *et al.*, 2015). However, proponents of the theory of niche conservatism argue that rapid adaptation will only be significant for a fraction of species threatened by climate change (Crisp *et al.*, 2009; Peterson *et al.*, 1999; Wiens

& Graham, 2005). European trees responded to Quaternary climate change primarily through migration or extinction, not evolution (Huntley & Webb III, 1989; Svenning, 2003). A third criticism of SDMs is that they do not account for increased levels of atmospheric CO_2, which will cause increased growth rates (Knapp *et al.*, 2001). These increased growth rates are unlikely to heavily affect species distributions, since CO_2 fertilization effects are projected to show diminishing growth-rate returns through 2030 (Cramer *et al.*, 2001).

A critical weakness of SDMs is their limited ability to account for migration processes (Beale *et al.*, 2008). This is a major weakness, since high projected climate change velocity in the 21st century is likely to exceed migration rates (Corlett & Westcott, 2013), even as new habitats become climatically suitable. SDMs have traditionally considered two scenarios regarding species migration ability: unlimited dispersal or no dispersal (Peterson *et al.*, 2002; Thomas *et al.*, 2004; Thuiller *et al.*, 2005). In reality, the migration capability of species will fall somewhere between these two scenarios and will vary greatly even between similar species (Nathan *et al.*, 2011). Various techniques have been proposed to incorporate species migration into distribution projections. For example, Midgley *et al.* (2006) estimate migration rates by dispersal syndrome, comparing these rates against SDM projections. They find that range shift is insufficient for most species to track climate in the African Cape Floristic Region. Alternately, statistical models of dispersal can be incorporated into SDMs to predict migration distances and potential migration shortfalls (Renton *et al.*, 2013; Schurr *et al.*, 2007). However, these options are only sufficient for species with well-defined dispersal capabilities (Renton *et al.*, 2013) and in well-defined environments. Dispersal capabilities are poorly defined for many species and most models, and migration of a significant distance is often dependent on sporadic events such as animal-aided dispersal or long-distance dispersal events (Howe & Smallwood, 1982).

With these limitations in mind, SDMs remain "one of the few practical approaches for forecasting or hindcasting distributions...providing methods and results are rigorously assessed" (Elith & Leathwick, 2009). Accumulating evidence from SDM studies suggest elevated risks of extinction, reduced biodiversity, and latitudinal range shift (Iverson *et al.*, 1999; Thomas *et al.*, 2004). For example, Thuiller *et al.* (2005) evaluated 1,350 European plant species under several climate scenarios, finding that over half could be vulnerable or threatened by 2080. These studies are now supported by observations of species migration (Crimmins *et al.*, 2011; Feeley *et al.*, 2011; Parmesan & Yohe, 2003), usually in accordance with model predictions, although exceptions do occur (Zhu *et al.*, 2012).

Conservation of rare and endemic species can be challenging because their distributions are often poorly-defined, and characterized by few isolated populations. SDMs are particularly poorly suited for forecasting range shift of rare species (e.g., less than 25 observations), despite considerable success in defining the geographic distribution of well-sampled species (Schwartz *et al.*, 2006; Wisz *et al.*, 2008). Machine learning approaches such as Maxent (Phillips *et al.*, 2006) or GARP (Stockwell, 1999) have shown promise for defining the range of some endemic species (Pearson *et al.*, 2007), but ecologists emphasize that even the best statistical methods function poorly with data at course resolutions or where the factor constraining a species' range is not understood (Wisz *et al.*, 2008).

2.10 Conservation, rare species and managed relocation

Conservation spending from international organizations is approximately $1.5 billion USD per year, only 2-32% of which is spent on conservation with global priorities (Halpern *et al.*, 2006). Very few conservation operations explicitly account for climate change, and many of these operations will ultimately be invalidated by changing climate (Iwamura *et al.*, 2013). For example, the biodiversity hotspots identified by Myers *et al.* (2000) have been celebrated as an opportunity to protect 44% of the world's vascular plant species in only 1.4% of the land surface; however, Malcolm *et al.* (2006) use global vegetation models to show that these biodiversity hotspots will become obsolete as climate change progresses, with extinction rates between 1% and 43% through the year 2100.

Rare and endemic species are among the most threatened by climate change (Settele *et al.*, 2014; Thomas *et al.*, 2004), particularly where species occupy narrow environmental niches. Managed relocation represents a powerful conservation tool for these species, with the potential of alleviating extinction from failure to migrate. Although managed relocation remains a controversial tool due to the risk of conservation targets becoming invasive (Ricciardi & Simberloff, 2009), many ecologists agree that the risk of a species becoming invasive is generally low when correctly assessed (Mueller & Hellmann, 2008; Williamson & Fitter, 1996). While migration corridors have been suggested as an option to facilitate natural plant migration, it is unlikely that plant species will be able to keep pace with climate change through normal seed dispersal processes (Hannah *et al.*, 2014; Pearson & Dawson, 2005), necessitating a more direct approach.

Forestry plantations represent an obvious option for testing managed relocation, since seed transfer can be used as a tool for improving productivity without moving species outside of their established ranges (Gray & Hamann, 2013). Hamann *et al.* (2011) used genetic data from provenance trials to model ideal seed zone transfer in western North America. This has promise for increased productivity since forest tree species already lag behind their historical climate niche by approximately 130 km in latitude (Gray & Hamann, 2013). Schreiber *et al.* (2013) performed a large reciprocal transplant experiment in Canada involving *Populus tremuloides*, finding that populations shifted northward produced nearly twice the biomass of local sources.

Government agencies have little policy framework to facilitate the introduction of threatened species outside of their historical range, a serious legal barrier to managed relocation (McLachlan *et al.*, 2007). Overcoming such barriers will require a scientific decision making framework for managed relocation, such as that proposed by Vitt *et al.* (2007). Managed relocation to save charismatic species, such as the Florida torreya, has been performed with some success through the efforts of dedicated conservation organizations (McLachlan *et al.*, 2007). However, effective managed relocation to mitigate climate change requires systematic, science-based prioritization and intervention.

2.11 Vulnerability assessments

Information regarding a species' vulnerability to climate change is important for determining the necessity of human intervention, such as active monitoring of specific populations, establishment of protected areas, or managed relocation. Various climate change vulnerability assessment techniques have been developed, although there is currently no consensus on a preferred approach (Pacifici *et al.*, 2015). It is generally accepted that species' vulnerability is defined by the intersection of climate change exposure, species sensitivity, and species adaptive capacity (Williams *et al.*, 2008). Climate change exposure refers to the magnitude of climate change experienced for a species or population, sensitivity refers to a species' inherent tolerance to climate change, and adaptive capacity refers to a species' ability to adapt to changing climate, usually through evolutionary processes (Williams *et al.*, 2008).

Divergent vulnerability assessment methodologies can be subdivided into correlative approaches, mechanistic approaches, trait-based vulnerability assessments (TVAs), and combined approaches (Pacifici *et al.*, 2015). Correlative models assess vulnerability through definition of a species' realized niche based on geographic location (Clark, 1998; Pearson & Dawson, 2003), while mechanistic model use species' functional traits, tolerances, and energy-balance equations to define a species niche (Kearney & Porter, 2009). While these two methods are data-intensive and their use require a high level of expertise, TVAs are seen as relatively easy to implement by contrast, and thus have been adopted by some conservation and government agencies to rapidly assess the vulnerability of multiple species (Pacifici *et al.*, 2015).

The NatureServe Climate Change Vulnerability Index (Young *et al.*, 2010) is one such example of a TVA, with elements of a combined approach. This vulnerability index involves assessment of 21 separate parameters contributing to a species' climate exposure, sensitivity, and adaptive capacity (Young *et al.*, 2010). However, even the relatively sparse data requirements for TVAs may be prohibitive for understudied and rare species, since data gaps are common even for relatively well-studied species (Foden *et al.*, 2013).

Most authors agree that life history traits and in particular dispersal ability are the most important factors governing a species vulnerability (Schloss *et al.*, 2012; Foden *et al.*, 2013). If this is true, it may be possible to substitute a simple assessment of dispersal ability combined with a measure of climate change exposure as a simplified vulnerability assessment. Many TVAs use absolute value of temperature or precipitation change as a measure of climate exposure Beardmore & Winder (2011), however climate change velocity representing migration requirements would be more appropriate when compared directly against dispersal ability. This streamlined approach would have particular utility for species with significant data gaps.

3. Methods

3.1 Climate data

Climate datasets were generated using the ClimateWNA software package (Hamann *et al.*, 2013; Wang *et al.*, 2012), available for anonymous download at http://tinyurl.com/ClimateWNA. The software provides lapse rate-adjusted climate surfaces at any resolution and in any projections, based on Parameter-elevation Regressions on Independent Slopes Model (PRISM) climate grids (Daly *et al.*, 2008). ClimateWNA further overlays anomaly surfaces for historical data and future projections from atmosphere-ocean global circulation models (AOGCMs). I used seven AOGCMs projections for the A2 emission scenario from the CMIP3 dataset referenced in the IPCC's Fourth Assessment Report (IPCC, 2007): CCMA CGCM3.1, CSIRO MK3.0, IPSL CM4, MIROC3.2 HIRES, MPI ECHAM5, NCAR CCSM3.0, and UKMO HADGEM. These seven models were chosen based on resolution, validation statistics, and representation of predictions for the study area according to Stralberg *et al.* (2015). The results were averaged into an aggregate estimate for concise reporting.

All climate datasets were generated in Lambert Conformal Conic projection at a 1 km resolution. Climate variables selected for the multivariate velocity analysis included: mean annual temperature, mean temperature of the warmest month, mean temperature of the coldest month, mean annual precipitation, mean growing season (May to September) precipitation, annual compound moisture index, summer (June, July, August) compound moisture index, degree days above 5℃, and number of frost-free days. A baseline period of 1961-1990 (hereafter "normal period") was used as the climate normal and 2041-2070 (hereafter "2050s") was used as my climate projection.

3.2 Multivariate climate change velocity and velocity trajectories

Multivariate climate change velocity surfaces were calculated using the protocol described in Hamann *et al.* (2015). Whereas univariate velocities are typically calculated as the temporal gradient of climate change divided by the spatial gradient of climate, Hamann *et al.* (2015) measured climate change velocity using the distance to nearest analogous climate. This is accomplished by performing a principal component analysis (PCA) on multiple climate variables, and defining analogous climates as unique combinations of the PCA bins. This is done for both present and future climates, with matching PCA bins scores used to indicate a suitable migration destination. Climate change velocities are then calculated as the minimum required geographic distance to a matching cell divided by the number of years between the baseline climate period and the future projection. Velocity trajectories are simply calculated as the compass direction between the origin cell and the future climate match. Principal component scores have the advantage that correlated climate variables do not erroneously influence climate matching and the resulting climate change velocity. Velocity calculations were completed using climate datasets for all of North America west of 100°W to avoid arbitrary edge-effects associated with the boundary of Alberta, however results are presented only to the extent of Alberta. Velocity data is available for anonymous download at http://tinyurl.com/VelocityWNA.

3.3 Rare species data

I used rare vascular plants in Alberta, Canada as a case study to demonstrate my methods of ranking climate change vulnerability. Of the different major terrestrial taxa, plants are considered the most vulnerable to migration stresses associated with climate change due to their poor dispersal capability (IPCC, 2014). The Alberta Conservation Information Management System ("ACIMS") is a biodiversity information database managed by Alberta Environment and Sustainable Resource Development (2014) that contributes to North America's NatureServe system. ACIMS tracks species occurrence locations across Alberta using NatureServe protocols (Young *et al.*, 2010). Rare plant species in this database are any species with a NatureServe subnational conservation rank of vulnerable (S3), imperiled (S2) or critically imperiled (S1). We used the Plant List v1.1 (http://www.theplantlist.org) as the primary data source to define species authority, from which we used 'accepted' or 'unresolved' names. Where this source was inconclusive, we used ITIS (the Integrated Taxonomic Information System, http://www.itis.gov) to find species authority.

Dispersal capability of each species was categorized using the NatureServe Climate Change Vulnerability Index dispersal criteria (Young *et al.*, 2010). Young *et al.* (2010) classify species into seven categories according to how dispersal ability putatively affects climate change vulnerability, ranging from "greatly increases vulnerability", such as gravity-dispersed seed, to "greatly decreases vulnerability", such as avian-assisted seed dispersal. Dispersal mechanisms for the plant species covered in this study were obtained from the United States Department of Agriculture PLANTS database (USDA and NRCS, National

Plant Data Team, 2015), the Encyclopedia of Life database (Encyclopedia of Life, 2014), and botanical references. Following Young *et al.* (2010), classification is based on the primary dispersal mechanism of the plant and excludes dispersal mechanisms that rely on rare events that may result in extreme long-distance-dispersal. In adopting the approach by Young *et al.* (2010) to my species portfolio, I consolidated the two most extreme classes (classes 1 and 2, and classes 6 and 7) for a total of 5 vulnerability classes based on dispersal.

3.4 Habitat Fragmentation

I used the Human Footprint Map of Alberta (Alberta Biodiversity Monitoring Institute, 2012) to derive habitat fragmentation for the year 2010. The human footprint refers to the temporary or permanent transformation of native ecosystems to human land uses (e.g. built-up area, roads, agricultural land, and surface mines). In addition, I considered water and snow/ice land cover as barriers to migration, obtained from the 2010 ABMI Land Cover Map (Castilla *et al.*, 2014). Both datasets were derived from SPOT 5 satellite imagery at a resolution of 50 m. I consider areas classified as 'barren' and 'exposed' as available' in my fragmentation index, since the dispersal of seeds are not normally inhibited by these features (Bacles *et al.*, 2006). Habitat fragmentation (resistance to natural seed dispersal) was estimated as the proportion of habitat not available' to plants within a 10 km moving window.

3.5 Vulnerability ranking

I defined an index of 'migration stress' as the product of habitat fragmentation and climate change velocity at any given point in the landscape. Lacking empirical data to weigh the effects of habitat fragmentation versus climate change velocity in a species-specific way, I assigned them equal weight by means of normalizing each index. Migration stress was reclassified into five equal ordinal groups, as was seed dispersal capacity. Class 1 populations were anticipated to experience the highest migration stress, while class 5 populations were anticipated to experience the lowest.

Subsequently, the climate change vulnerability index was calculated for each species as the multiplicative interaction of migration stress class (threat classes) and seed dispersal class (capacity for response). Class 1 represents the most vulnerable species, characterized by high climate velocities, high habitat fragmentation, and weak dispersal capability, whereas class 25 represents the least vulnerable group. I used a multiplicative index since the two factors are unlikely to have additive effects without interactions (Renton *et al.*, 2013). High migration stress is likely to be disproportionately exacerbated by low dispersal capacity.

4. Results

4.1 Climate change velocity

Climate change velocities varied markedly between Alberta's Natural Regions, with the highest velocities in the Grassland Natural Region, a mean of 6.9 km yr^{-1} (Fig. 4). The Parkland Natural Region also exhibited high climate velocities, with a mean of 5.3 km yr^{-1} (Fig. 4). Overall climate change velocity for Alberta averaged 3.6 km yr^{-1}, although climate change velocity was spatially variable within Natural Regions (Fig. 4). The Rocky Mountain Natural Region had the highest variance, with climate velocity ranging from 0.05 km yr^{-1} to 20.8 km yr^{-1}. Mountaintop species were forecasted to experience the highest climate velocities in the province while montane climate shifts occurred at a significantly reduced rate (Fig. 2). Vulnerability of these mountaintop species is highly dependent on the population's elevation and dispersal mechanism. For example, some high-elevation species experience high climate change velocity (Fig. 2), but have adaptations for wind dispersal and are likely to benefit from their high elevation. Conversely, high-elevation gravity-dispersed species are among the most threatened (Table 1).

Climate change velocity magnitude and vectors were both heavily influenced by regional topographic effects. Climate vectors were most divergent in topographically-diverse regions such as the Rocky Mountain Natural Region. Climate vectors generally conformed to expectation of latitudinal climate shifts, with an average bearing of 346°(NNW; Fig. 2). However, the Rocky Mountain and Foothills Natural Regions exhibited an upslope climate velocity towards the continental divide, generally at an overall lower climate velocity than the other Natural Regions. This divided climate vectors in Alberta into two distinct groups: those directed northward to boreal climates, and those directed towards the Rocky Mountains. Velocity vectors in the Foothills Natural Region nearly always pointed southwest towards mountainous destinations.

4.2 Habitat fragmentation and migration stress

Several regions of Alberta had disproportionately high habitat fragmentation (Fig. 1, Fig. 4). The Grassland and Parkland Natural Regions (south-central Alberta) had a mean fragmentation of 56% and 78% respectively. Many parts of these regions exhibited near 100% habitat loss due to agricultural land use, forming a significant barrier to migration for species incapable of long-distance dispersal. Mean fragmentation of other Natural Regions was much lower: 27% in the Foothills; 22% in the Boreal; 16% in the Canadian Shield; and 9% in the Rocky Mountains. Habitat fragmentation showed significant variation within Natural Regions, with the most variation within the Grassland and Boreal Forest Natural Regions (Fig. 1).

Migration stress was highest in unprotected regions of the Parkland Natural Region, although all six Natural Regions exhibited high migration stress across part of their extent (Fig. 5). Protected areas had lower migration stress than unprotected regions, but this was not consistent across all Natural Regions (Fig. 5). Mountaintop climate extinction was represented by several high migration stress outliers in the Rocky Mountain Natural Region, however these high outliers were rarely higher than the migration stresses exhibited throughout much of the Parkland Natural Region.

4.3 Vulnerability ranking

The five most vulnerable rare species were spiked lobelia (*Lobelia spicata*), lance-leaved loosestrife (*Lysimachia hydriba*), low cinquefoil (*Potentilla plattensis*), goosefoot (*Chenopodium atrovirens*), and clammy hedge-hyssop (*Gratiola neglecta*) (Table 2). None of these species, however, are imperiled on a global scale. Only 36 plant species in Alberta were listed with a NatureServe conservation status of globally vulnerable (G3), imperiled (G2) or critically imperiled (G1). Macoun's cinquefoil (*Potentilla macounii*) and McCalla's braya (*Braya humilis ssp. maccallae*) represented the only G1-ranked species assessed in Alberta. Macoun's cinquefoil had the fourth-highest vulnerability of the 36 globally threatened species and is anticipated to spread seed passively, making it a priority target for considering conservation interventions. Of these globally-threatened species, three species had the highest vulnerability ranking (highest migration stress class, lowest dispersal ability, "extremely vulnerability"), and an additional five had very high vulnerability rankings (middle migration stress class, lowest dispersal ability, "highly vulnerable"). Table 1 summarizes rare Alberta plant species that occupy the highest category of climate change vulnerability, regardless of global conservation status (full records in Appendix B).

5. Discussion

5.1 Regional climate change velocity trends

My climate velocity estimates conform to the expectation that areas of high topographic and climatic heterogeneity reduce species' migration requirements for modern climate change (Loarie *et al.*, 2009). In some cases, this topographic heterogeneity, coupled with migration corridor planning, may be sufficient for species conservation through normal seed dispersal processes. Conventional ecological knowledge is that plants migrate a few tens of meters per year under normal seed dispersal circumstances (Howe & Smallwood, 1982; Willson, 1993), and some ecologists suggest that strong dispersers may be able to migrate up to 1,500 m yr^{-1} (Vittoz & Engler, 2008) (Vittoz & Engler, 2008). However, nearly a quarter of the plant populations studied here are forecasted to experience mean climate change velocities over 5.0 km yr^{-1}. These populations represent natural targets for ex situ conservation.

Lower climate velocities in topographically-diverse regions (e.g. Rocky Mountain and Foothills Natural Region) are reflective of the ready availability of upslope migration pathways and a greater variety of locally-available microclimates (Ackerly *et al.*, 2010). However, my results show that the high magnitude of forecasted climate change will invalidate upslope migration in many cases, essentially extending mountaintop extinction to encompass larger areas. Mountaintop extinction will be especially potent where treelines expand to encompass previously-exposed areas (Dirnböck *et al.*, 2011). There is still a potential for topographic variation to act as migratory "stepping stones" (Hannah *et al.*, 2014), and isolated microrefugia may allow for plants to persist as holdout populations. However, it will also be important to continue monitoring of existing plant populations, since holdout populations are unlikely to persist permanently without aggressive global climate change mitigation (Hannah *et al.*, 2014).

A key observation is the high variability of climate change velocities and resultant migration stress across the study area. Alberta features very diverse landscapes including the Rocky Mountains, the boreal plains, and the grasslands of western Canada, resulting in projected climate velocities that differ in several orders of magnitude. I forecast climate change velocities of up to 10 km yr^{-1} in some parts of the study region, far exceeding observed rates of plant migration (Chen *et al.*, 2011; Feeley *et al.*, 2011). While some studies have emphasized the importance of mean dispersal distance (Coutts *et al.*, 2010), my results suggest that climate change velocity may become dominant as velocities exceed dispersal rates even for strong dispersers.

5.2 Regional fragmentation

As expected, my model shows that habitat fragmentation increases climate change vulnerability. Adjusting climate change velocity by habitat fragmentation is especially useful where a species' migratory mechanism and pathway is unknown. Models of species vulnerability, species distribution models included, often fail to account for migration retardation by habitat fragmentation (Midgley *et al.*, 2006). This shortcoming may lend itself to systematic underestimation of climate change vulnerability. Populations in the Alberta's

Grassland and Parkland Natural Regions, where mean climate velocities exceed 5.0 km yr^{-1}, are most at risk of climate-related extirpation. These regions feature little topographic heterogeneity, which limits the availability of climatic refugia or migratory stepping stones. Furthermore, Grassland and Parkland habitat is highly fragmented compared to the rest of Alberta, with both of these Natural Regions featuring mean fragmentation exceeding 50% (Fig. 1, Fig. 4). Even independently, high habitat fragmentation or high climate change velocities could threaten many plant species (Hannah *et al.*, 2014; Pearson & Dawson, 2005). Southeastern Alberta features both of these risk factors, and rare Grassland or Parkland species are unlikely to migrate with sufficient velocity to stay in sync with climate change, except perhaps when accelerated by extreme long-distance dispersal. Increased climatic stress may result in a decline of fitness and eventually local extirpation. My results suggest that migration corridor planning would be futile for these Natural Regions. Species conservation may be best served through ex situ conservation, including considering managed relocation.

Fragmentation levels of 80% or higher can completely inhibit species' migration capacity, even for strong dispersers under a mild climate change scenario (Renton *et al.*, 2013). Normal seed dispersal processes can thus be interrupted across isolated regions of heavy land use. Meanwhile, linear fragmentation features such as roads may actually accelerate the seed dispersal of wind-dispersed seeds and facilitate long-distance dispersal through updrafts (Damschen *et al.*, 2014). Thus, my habitat fragmentation measure would be improved by an assessment of patch connectivity using fine-scale fragmentation data, similar to the methodology used by Nuñez *et al.* (2013). Multivariate climate change velocity is useful for this application, since it provides a migration vector that can be compared against fragmentation features. This will be an essential step in the planning of migration corridors.

5.3 Target species

In the case study for Alberta, I identified spiked lobelia (*Lobelia spicata*) as the highest-risk species of those studied. Nonetheless, Lobelia spicata is listed as "apparently secure" on a global (i.e. full range) scale (Encyclopedia of Life, 2014; Natureserve, 2012). Conservation priorities should, of course, not be determined by threats for individual population alone. While vulnerability rankings, such as those proposed in this study, may be used to identify threatened populations, the global conservation status of a species is an important consideration as well.

For a globally-oriented conservation strategy, Macoun's cinquefoil (*Potentilla macounii*) and McCalla's braya (*Braya humilis ssp. maccallae*) are the most vulnerable. These species have a vulnerability rank of 3 and 4 respectively, and are critically-imperiled (G1-ranked) globally, ideal targets for ex situ conservation. Secondary targets may include western false gromwell (*Onosmodium molle*), Lyall's scorpionweed (*Phacelia lyallii*), and alpine poppy (*Papaver pygmaeum*), which are among the highest vulnerability class and are considered globally-vulnerable (G3-ranked). Weighing local and global conservation goals of species will remain an important responsibility of conservation managers who operate within their local jurisdictions.

6. Thesis Conclusions

6.1 General summary

The climate change vulnerability assessment described herein provides a simple method for rapidly assessing the climate change vulnerability of plant species and populations. Complex vulnerability assessments are often impractical to apply extensively because of missing species-specific data, particularly for rare, understudied species (Pacifici *et al.*, 2015). Migration stress is spatially-defined, which reduces an assessment of population vulnerability down to two input parameters: location and dispersal ability. Admittedly, this is less precise, however it represents a practical way to make a first vulnerability assessment for a large number of species. Such an assessment may be used as a filter with which to guide monitoring or a more thorough assessment for species that have been identified as particularly vulnerable.

Migration stress has several novel extensions for improving correlative vulnerability assessments and trait-based vulnerability assessments. For example, the Natureserve Climate Change Vulnerability Index Correlative (Young *et al.*, 2010) measures climate exposure as the absolute value of temperature change. Migration stress or even climate change velocity would be an improved measure of climate exposure, since they account for species maintaining their niche through migration. Similarly, correlative vulnerability assessments typically represent species migration as nonexistent or unlimited (Peterson *et al.*, 2002; Thomas *et al.*, 2004). Spatial variations in migration stress may assist in preemptively identifying regions where newly-suitable habitat is not accessible, whether due to habitat fragmentation or excessive distance.

My research indicates that ex situ conservation (including managed relocation) of some globally-endangered species may be the only plausible conservation option for some populations if climate change continues as predicted. Identifying suitable future habitat for managed relocation will be a difficult task. First, it will be necessary to incorporate factors such as climate tolerance, specificity of edaphic requirements, and mycorrhizal and pollinator associations, among a host of other potentially important biotic and abiotic factors. This remains a challenge especially for rare species where it is difficult to even define their basic environmental tolerances (Schwartz *et al.*, 2006). In setting conservation objectives, migration prescriptions must overcome policy and administrative barriers, including international boundaries, private land ownership, conflicts with industrial resource extraction, and ethical considerations (Schwartz *et al.*, 2012; Vitt *et al.*, 2010). Success in this respect will require an open dialogue between stakeholders on how best to effectively managed species movement.

6.2 Study limitations and future research

An important limitation is that I treat adaptive capacity as uniform across all populations (i.e. not factoring into the vulnerability ranking). Unfortunately, species' capacity for adaptive or genetic response are normally unknown and are understood to vary between populations (Savolainen *et al.*, 2007). Plants can acclimate to some degree to changed environments through plastic physiological or morphological responses, or they may adapt genetically over multiple generations to new climates (Chen *et al.*, 2011). This remains a modelling challenge, although many ecologists suggest that adaptive capacity has been historically overestimated and that adaptation will only rarely match the magnitude of modern climate change (Crisp *et al.*, 2009; Peterson *et al.*, 1999; Wiens & Graham, 2005).

Categorizing dispersal ability is a highly subjective process, and represents a limitation of this methodology. Mean dispersal distance is perhaps the strongest indicator of a species' ability to track climate change (Coutts *et al.*, 2010), but it is also one of the most difficult factors to assess. The ambiguous relationship between dispersal syndrome and stochastic long-distance dispersal is an important knowledge gap, especially since many authors suggest that long-distance dispersal is the main mechanism behind plant species range shift (Cain *et al.*, 2000; Clark, 1998). Long-distance dispersal is difficult to predict and problematic to incorporate into vegetation models or species distribution models (Cain *et al.*, 2000; Nathan & Muller-Landau, 2000). For example, whitebark pine (*Pinus albicaulis*) lacks specific morphological adaptations for long-distance dispersal, but is regularly dispersed to sites of forest disturbance by Clark's Nutcracker (Keane *et al.*, 1990). While whitebark pine is forecasted to experience high climate change velocities (mean of 3.2 km yr^{-1} across Alberta), it is sheltered by "very strong" dispersal ability. Further research into species-specific mean migration distance is a viable solution, although it will rely heavily on expert knowledge.

Several other minor limitations represent potential improvements to this methodology. First, this study uses habitat fragmentation as a proxy for habitat connectivity, where a more involved assessment may assess habitat connectivity directly. Such an assessment could take advantage of multivariate climate change velocity vectors to map migration trajectories (Burrows *et al.*, 2014), adjusting these trajectories using a cost-distance algorithm to identify realistic migration corridors (Nuñez *et al.*, 2013). A second limitation is that ACIMS (2014) rare species location is of mixed quality, with some data being logged before the ubiquity of modern GPS tools. Under ideal circumstances it would be possible to verify the exact location of each population in question, since climate velocity and habitat fragmentation are spatially-determined.

Further research is needed to better define the relationship between seed dispersal strategy and migration ability. While this relationship has been ambiguous, observational studies of species migrating in response to 21st-century climate change may prove useful. Secondly, the integration of species distribution models with migration stress is a promising research direction. Since species distribution models often approximate migration as either universal or nonexistent (Midgley *et al.*, 2006), such an integration promises to identify vulnerable species that were previously thought safe.

6.3 Applications and recommendations for Alberta

Based on the intersection of climate change velocity, habitat fragmentation, and dispersal ability, I identify at least two species that should be considered for ex situ conservation: Macoun's cinquefoil (*Potentilla macounii*) and McCalla's braya (*Braya humilis ssp. Maccallae*). These two species are critically-endangered on a global scale, and have a very high vulnerability rank. Recommended targets for further investigation include western false gromwell (*Onosmodium molle*), Lyall's scorpionweed (*Phacelia lyallii*), and alpine poppy (*Papaver pygmaeum*), which are vulnerable on a global scale and are among the highest vulnerability class within Alberta.

An alternative to this would be to prioritize conservation of species threatened on a provincial level, rather than a global level. Determining conservation priorities on such a scale is generally not recommended, since the high magnitude of projected climate change velocities will force native Albertan species to migrate outside of their home range, and potentially out of the province. Should such conservation priorities be desirable, we identified spiked lobelia (*Lobelia spicata*), lance-leaved loosestrife (*Lysimachia hybrida*) and low cinquefoil (*Potentilla plattensis*) as the three most vulnerable species. Further investigation is recommended for these species and the other species listed in Table 2.

An important conclusion for Alberta conservation planning is that the high magnitude of forecasted climate change will invalidate upslope migration, often within the 21st century. While the topographic heterogeneity of the Foothills and Rocky Mountain Natural Regions may provide migratory stepping stones for migrating species, most upslope migration pathways are finite due to mountaintop "climatic cul-de-sacs" (Hamann *et al.*, 2015). It will be especially critical to monitor Rocky Mountain plant populations where encroaching tree lines threaten mountaintop climatic extinction in previously-exposed areas (Dirnböck *et al.*, 2011).

While mountaintop extinction is a well-known threat, my results show that migration stress is often even higher in flat, agricultural regions. Plant populations in the Grassland and Parkland Natural Regions are anticipated to experience migration stresses similar to high-elevation populations, and should be considered for ex situ conservation. This is especially true for populations isolated by land-use change. While land use change and habitat fragmentation are important for defining seed dispersal permeability, future research might consider using climate change velocity and land cover data to model source and sink populations, explicitly defining potential migration routes.

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Table 1: Plant species of highest vulnerability in Alberta, Canada with a global conservation status of vulnerable (G3), imperiled (G2), or critically imperiled (G1).

Species name	**Authority**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Onosmodium molle	Michx.	4.9	69	Very Weak	1*1 = 1	S2	G3
Phacelia lyallii	(A.Gray) Rydb.	1.4	6	Very Weak	1*1 = 1	S2	G3
Papaver pygmaeum	Rydb.	1.9	3	Very Weak	1*1 = 1	S2	G3
Potentilla macounii	Rydb.	5.2	33	Very Weak	3*1 = 3	S1	G1
Sisyrinchium septentrionale	E.P.Bicknell	5	32	Very Weak	3*1 = 3	S3	G3
Conimitella williamsii	(D.C.Eaton) Rydb.	2.4	13	Very Weak	3*1 = 3	S2	G3
Lupinus minimus	Hook.	1.3	19	Very Weak	3*1 = 3	S1	G3
Pedicularis flammea	L.	3.4	4	Very Weak	3*1 = 3	S2	G3
Chrysosplenium iowense	Rydb.	4.2	23	Very Weak	4*1 = 4	S3	G3
Braya humilis ssp. maccallae	J.G. Harris	3.7	8	Very Weak	4*1 = 4	S1	G1
Arnica louiseana	Farr	2.4	4	Weak	2*2 = 4	S1	G3
Stellaria americana	(Porter ex B.L. Rob.) Standl.	1.8	1	Strong	1*4 = 4	S1	G3
Braya humilis ssp. porsildii	J.G. Harris	6.8	15	Very Weak	5*1 = 5	S1	G3
Draba porsildii	Mulligan	4.2	5	Very Weak	5*1 = 5	S1	G3
Draba macounii	O.E.Schulz	5.4	4	Very Weak	5*1 = 5	S2	G3
Draba ventosa	A.Gray	4.9	3	Very Weak	5*1 = 5	S2	G3
Botrychium lineare	W.H.Wagner	1.3	4	Very Strong	1*5 = 5	S1	G2
Erigeron lackschewitzii	G.L.Nesom & W.A.Weber	2.4	12	Weak	3*2 = 6	SU	G3
Prenanthes sagittata	Tjitrosoedirdjo	0.9	11	Neutral	2*3 = 6	S2	G3
Carex cordillerana	Saarela & B.A.Ford	4.1	14	Weak	4*2 = 8	S1	G3
Packera contermina	(Greenm.) J.F.Bain	3.6	6	Weak	4*2 = 8	S3	G3
Poa gracillima	Vasey*	1	10	Strong	2*4 = 8	S2	G2
Salix raupii	Argus	2	4	Strong	2*4 = 8	S1	G2
Puccinellia distans ssp. hauptiana	(Jacq.) Parl.	2.8	92	Weak	5*2 = 10	S1	G3
Erigeron radicatus	Hook	4.7	9	Weak	5*2 = 10	S2	G3
Botrychium paradoxum	W.H. Wagner	2.1	4	Very Strong	2*5 = 10	S1	G3
Stellaria arenicola	Raup*	3.4	6	Strong	3*4 = 12	S1	G3
Pellaea gastonyi	Windham	2.7	16	Very Strong	3*5 = 15	S1	G2
Pinus albicaulis	Engelm.	3.2	7	Very Strong	3*5 = 15	S2	G3
Botrychium ascendens	W.H. Wagner	2.8	8	Very Strong	3*5 = 15	S2	G3
Botrychium pallidum	W.H. Wagner	4.4	23	Very Strong	4*5 = 20	S1	G3
Botrychium campestre	W.H. Wagner & Farrar	3.5	28	Very Strong	4*5 = 20	S1	G3
Arenaria longipedunculata	Hultén	5.7	14	Strong	5*4 = 20	S1	G3
Botrychium crenulatum	W.H. Wagner	4.2	16	Very Strong	4*5 = 20	S1	G3
Botrychium spathulatum	W.H. Wagner	4.1	16	Very Strong	4*5 = 20	S2	G3
Botrychium pedunculosum	W.H. Wagner	4.7	11	Very Strong	5*5 = 25	S1	G2

* - Species as listed in ACIMS is a synonym, and now goes by a different scientific name

Table 2: Locally-threatened plant species by highest climate change vulnerability in Alberta, Canada. The full table is provided in Appendix B.

Species name	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**
Lobelia spicata	Lam.	1	7.9	86	Very Weak	1*1 = 1
Lysimachia hybrida	Michx.	2	6.6	78	Very Weak	1*1 = 1
Potentilla plattensis	Nutt.	3	8.2	55	Very Weak	1*1 = 1
Chenopodium atrovirens	Rydb.	4	5.6	80	Very Weak	1*1 = 1
Gratiola neglecta	Torr.	5	5.4	80	Very Weak	1*1 = 1
Cyperus squarrosus	L.	6	7.2	59	Very Weak	1*1 = 1
Rorippa curvipes var. truncata	(Jeps.) Rollins*	7	6.1	67	Very Weak	1*1 = 1
Viola pedatifida	G. Don	8	6.3	64	Very Weak	1*1 = 1
Thelesperma subnudum var. marginatum	(Rydb.) T.E. Melchert ex Cronquist*	9	4.7	80	Very Weak	1*1 = 1
Hedyotis longifolia	(Gaertn.) Hook.*	10	5	71	Very Weak	1*1 = 1

* - Species as listed in ACIMS is a synonym, and now goes by a different scientific name

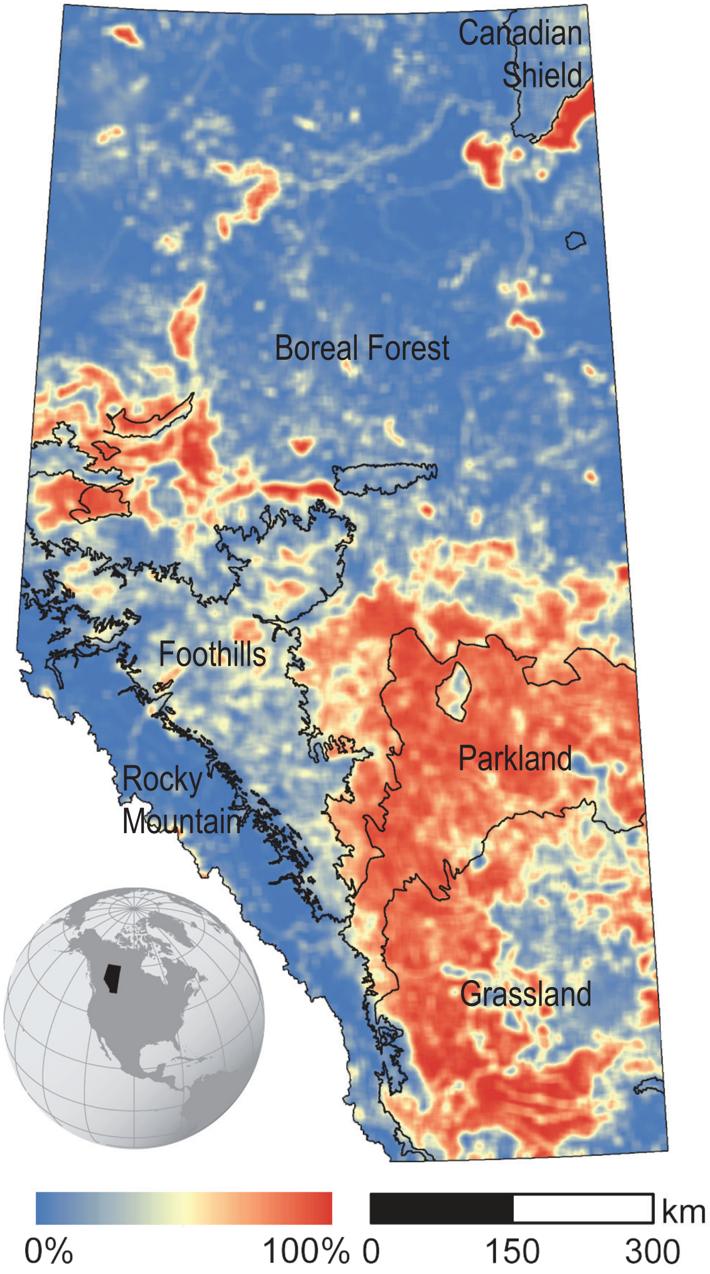

Figure 1: Study area with divisions into major ecological regions and habitat fragmentation (%). The fragmentation index relevant for barriers to migration includes anthropogenic land conversion, water bodies and permanent snow or ice cover.

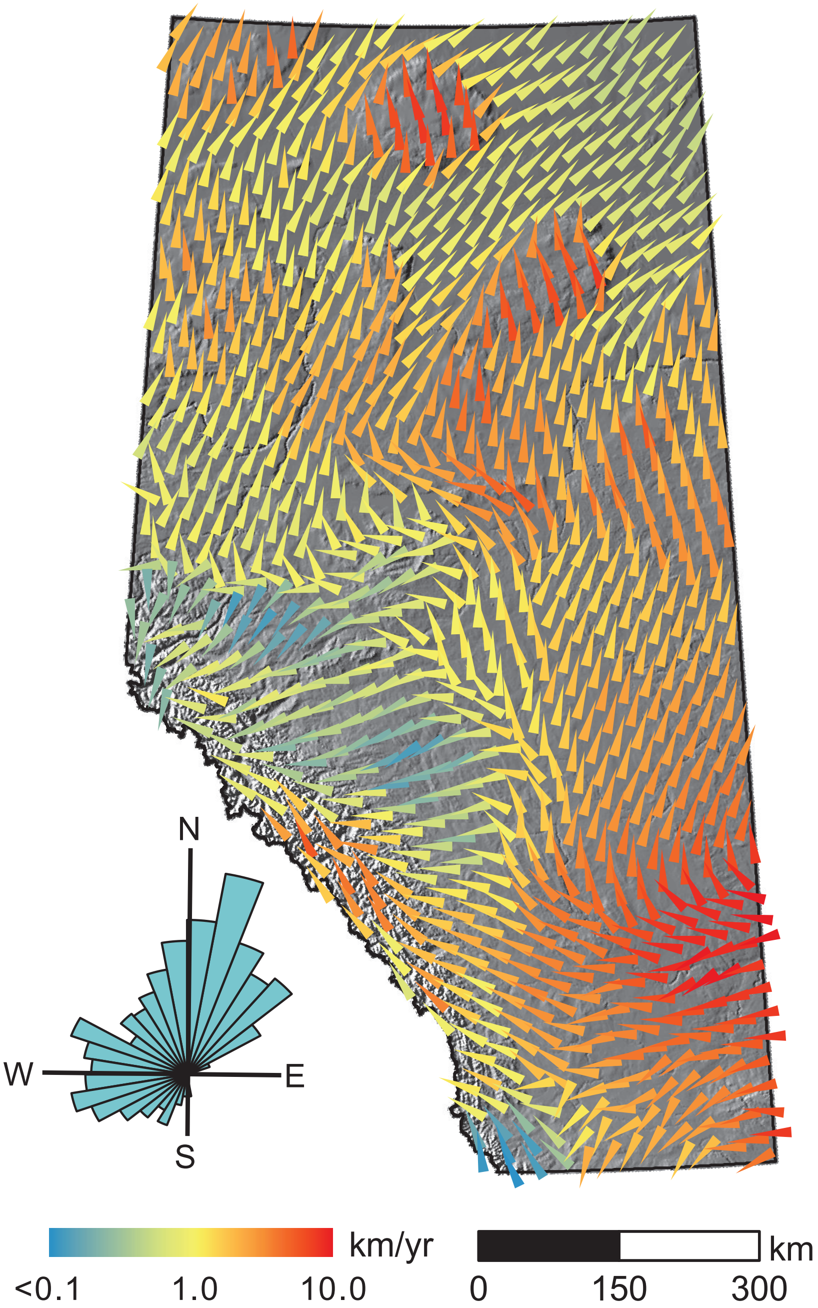

Figure 2: Multivariate climate velocity vectors in Alberta summarized at 25 km resolution for a 2050s ensemble projection. Vectors are based on a continental-scale analysis to avoid Alberta boundary artifacts. The rose plot is the equivalent of a histogram, indicating the prevailing directions of climate shifts in Alberta.

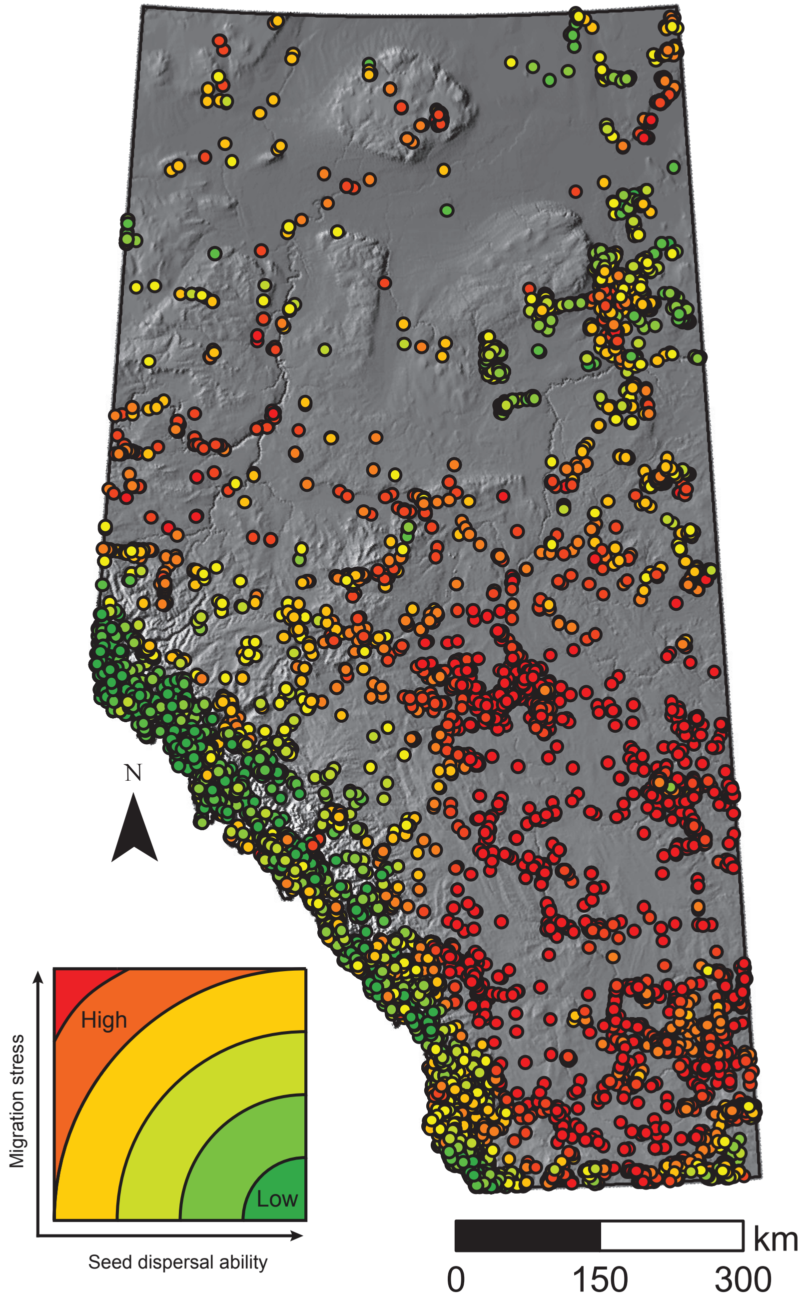

Figure 3: Migration stress of rare plant populations across Alberta, Canada between the normal period and 2050s. (Inset) Climate vulnerability is estimated by the product of dispersal capability and migration stress.

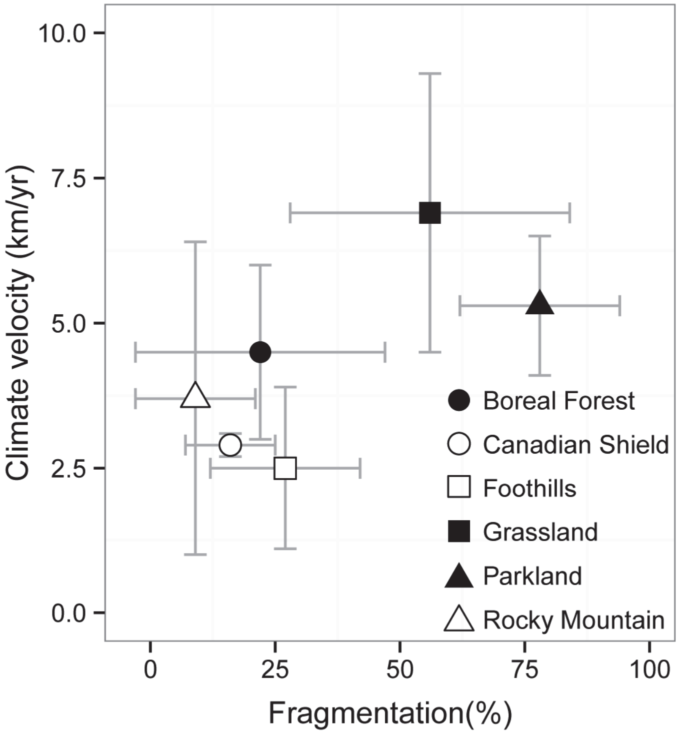

Figure 4: Mean multivariate climate velocity and fragmentation by Alberta Natural Region, summarized by 1 km^2 cells. Dots represent the data mean, whiskers represent the standard deviation.

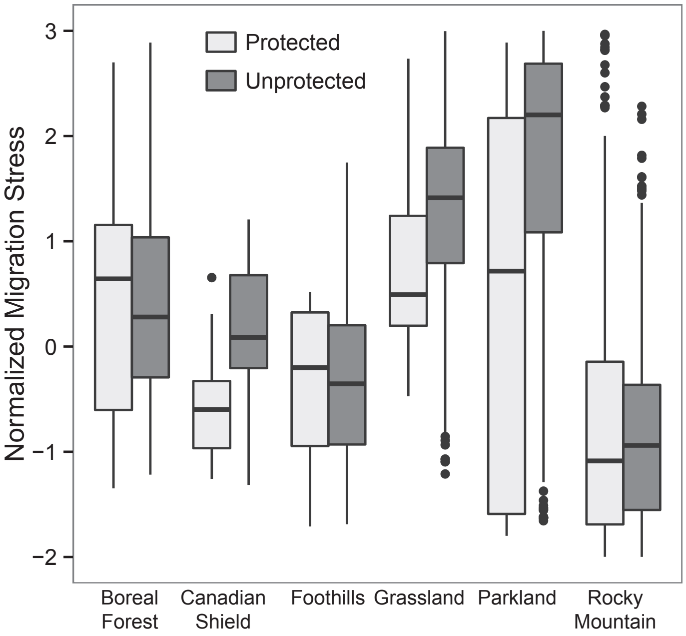

Figure 5: Migration stress of rare plant species by Alberta Natural Region within and outside of protected areas. Note that the box-plots may not be fully representative of the entire ecoregion, but summarize known occurrences of rare species as shown in Figure 3.

Appendix A - Sample Code

R code for calculating multivariate climate velocity, adapted from Hamann *et al.* (2015) for Alberta case study. present1, present2, future1, and future2 represent principal components 1 and 2 for present and future climate datasets. Refer to Hamann *et al.* (2015) for further documentation. Note that this example code does not include calculation of climate vectors.

```
library(SDMTools)     \# install package to read and write ESRI ASCII grids
library(yaImpute)     \# install package for k-nearest neighbour (kNN) search

present1 <- asc2dataframe("C:\Your Path\PC1_6190.asc") # principal component grids
present2 <- asc2dataframe("C:\Your Path\PC2_6190.asc")
future1  <- asc2dataframe("C:\Your Path\PC1_2020s.asc")
future2  <- asc2dataframe("C:\Your Path\PC2_2020s.asc")

idxy <- cbind(id=1:nrow(present1),present1[,1:2])   \# data frame of IDs and XY coords
b <- (max(present1$var.1)-min(present1$var.1))/120  \# bin size for 120 PC1 bins

p1 <- round(present1$var.1/b)               # convert PC1 to 120 bins via rounding
p2 <- round(present2$var.1/b)               # convert PC2 to <120 bins via rounding
f1 <- round(future1$var.1/b)                # same for future PC1
f2 <- round(future2$var.1/b)                # same for future PC2
p  <- paste(p1,p2)                          # PC1/PC2 combinations in present climate
f  <- paste(f1,f2)                          # PC1/PC2 combinations in future climate
u  <- unique(p)[order(unique(p))]           # list of unique PC1/PC2 combinations

sid <- c()                                  # empty vector for source IDs
tid <- c()                                  # empty vector for target IDs
d   <- c()                                  # empty vector for distances

for(i in u){                          # loop for each unique PC1/PC2 combination
  pxy <- idxy[which(p==i),]           # coordinates of i-th combination in present
  fxy <- idxy[which(f==i),]           # coordinates of i-th combination in future
  sid <- c(sid, pxy$id)               # append i-th PC1/PC2 combination to previous

  if(nrow(fxy)>0){                    # kNN search unless no-analogue climate
     knn <- data.frame(ann(as.matrix(fxy[,-1]), as.matrix(pxy[,-1]), k=1)$knnIndexDist)
     tid <- c(tid, fxy[knn[,1],"id"]) # the IDs of the closest matches
     d <- c(d, sqrt(knn[,2]))         # their corresponding geographic distances
     }
  else {                              # else statement for no-analogue climates
     tid <- c(tid, rep(NA,nrow(pxy))) # flag destinations as missing for no analogues
     d <- c(d, rep(Inf,nrow(pxy)))    # flag distances as infinity for no analogues
     } }
```

```
sxy <- merge(sid, idxy, by.y="id", all.x=T, all.y=F, sort=F)[2:3]  # source coordinates
txy <- merge(tid, idxy, by.y="id", all.x=T, all.y=F, sort=F)[2:3]  # target coordinates
names(txy)=c("target_y","target_x")


# write output table in CSV format with source and target coordinates and distances
outtab <- cbind(id=sid, sxy, txy, distance=d)
write.csv(outtab, "output.csv", row.names=F)


# writes out log10 velocities and distances multiplied by 100 in ESRI ASCII format
# conversion: -200=0.01km, -100=0.1km, 0=1km, 100=10km, 200=100km etc.
out=merge(present1[,1:2], outtab[,c(2,3,6)], by=c("y","x"), sort=F)
out$distance[out$distance==Inf] <- 10000  # sets no analogue to 10,000km
out$distance[out$distance==0] <- 0.5  # sets zero distance to 0.5km (1/2 cell size)
out$logDist=round(log10(out$distance)*100)
out$logSpeed=round(log10(out$distance/50)*100)
dataframe2asc(out)
```

Appendix B - Complete species vulnerability records

Table 3: Plant vulnerability data for rare Alberta species is reported here by: ACIMS (Alberta Environment and Sustainable Resource Development, 2014) species name (*Species*); species authority (*Authority*); vulnerability rank relative to all other assessed species (*Rank*); mean climate velocity across all populations (*Vel. (km/yr)*); mean habitat fragmentation across all populations (*Frag. (%)*); categorical seed dispersal capability, ranging from very weak to very strong (*Seed Dispersal*); categorical vulnerability rank, with lower numbers indicating higher vulnerability (*Vulnerability*); NatureServe subnational conservation rank (*Provincial Status*); and NatureServe global conservation rank (*Global Status*).

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Lobelia spicata	Lam.	1	7.9	86	V. Weak	1*1 = 1	S1	G5
Lysimachia hybrida	Michx.	2	6.6	78	V. Weak	1*1 = 1	S2	G5
Potentilla plattensis	Nutt.	3	8.2	55	V. Weak	1*1 = 1	S1	G4
Chenopodium atrovirens	Rydb.	4	5.6	80	V. Weak	1*1 = 1	S1	G5
Gratiola neglecta	Torr.	5	5.4	80	V. Weak	1*1 = 1	S2	G5
Cyperus squarrosus	L.	6	7.2	59	V. Weak	1*1 = 1	S1	G5
Rorippa curvipes var. truncata	(Jeps.) Rollins*	7	6.1	67	V. Weak	1*1 = 1	S1	G5
Viola pedatifida	G. Don	8	6.3	64	V. Weak	1*1 = 1	S2	G5
Thelesperma subnudum var. marginatum	(Rydb.) T.E. Melchert ex Cronquist*	9	4.7	80	V. Weak	1*1 = 1	S1	G5
Hedyotis longifolia	(Gaertn.) Hook.*	10	5.0	71	V. Weak	1*1 = 1	S2	G4
Onosmodium molle	Michx.	11	4.9	69	V. Weak	1*1 = 1	S2	G3
Rorippa curvipes	Greene	12	5.9	55	V. Weak	1*1 = 1	SU	G5
Downingia laeta	(Greene) Greene	13	4.8	68	V. Weak	1*1 = 1	S2	G5
Nuttallanthus canadensis	(L.) D.A.Sutton	14	8.4	37	V. Weak	1*1 = 1	S1	G5
Heliotropium curassavicum	L.	15	6.3	44	V. Weak	1*1 = 1	S1	G5
Polanisia dodecandra	(L.) DC.	16	6.8	40	V. Weak	1*1 = 1	S2	G5
Suckleya suckleyana	(Torr.) Rydb.	17	6.7	39	V. Weak	1*1 = 1	S1	G5
Potentilla finitima	Kohli & Packer*	18	8.6	29	V. Weak	1*1 = 1	S1	G5
Lilaea scilloides	(Poir.) Hauman	19	6.7	36	V. Weak	1*1 = 1	S1	G5
Chenopodium desiccatum	A.Nelson	20	6.4	37	V. Weak	1*1 = 1	S1	G5
Boykinia heucheriformis	(Rydb.) Rosend.	21	1.5	8	V. Weak	1*1 = 1	S2	G4
Viola praemorsa ssp. linguifolia	(Nutt.) M.S. Baker & J.C. Clausen ex M. Peck*	22	1.2	9	V. Weak	1*1 = 1	S2	G5
Aquilegia jonesii	Parry	23	1.2	8	V. Weak	1*1 = 1	S2	G4
Montia parvifolia	(Moc. ex DC.) Greene	24	0.9	10	V. Weak	1*1 = 1	S1	G4
Phacelia lyallii	(A.Gray) Rydb.	25	1.4	6	V. Weak	1*1 = 1	S2	G3

Appendix B - Continued

Species	Authority	Rank	Vel. (km/yr)	Frag. (%)	Seed Dispersal	Vulnerability	Provincial Status	Global Status
Townsendia condensata	Parry ex Parry	26	1.4	5	V. Weak	1*1 = 1	S2	G4
Mertensia longiflora	Greene	27	1.2	6	V. Weak	1*1 = 1	S2	G4
Papaver pygmaeum	Rydb.	28	1.9	3	V. Weak	1*1 = 1	S2	G3
Juncus regelii	Buchenau	29	0.9	4	V. Weak	1*1 = 1	S1	G4
Suksdorfia ranunculifolia	(Hook.) Engl.	30	1.0	3	V. Weak	1*1 = 1	S2	G5
Tellima grandiflora	(Pursh) Douglas ex Lindl.	31	0.6	1	V. Weak	1*1 = 1	S1	G5
Castilleja parviflora	Bong.	32	1.7	0	V. Weak	1*1 = 1	S1	G5
Elodea bifoliata	H.St.John	33	4.7	79	Weak	1*2 = 2	S2	G4
Atriplex powellii	S.Watson	34	6.1	60	Weak	1*2 = 2	S1	G4
Almutaster pauciflorus	(Nutt.) Á.L ove & D.L ove	35	7.8	46	Weak	1*2 = 2	S2	G4
Bromus latiglumis	(Shear) Hitchc.	36	4.9	60	Weak	1*2 = 2	S1	G5
Panicum wilcoxianum	Vasey	37	6.2	47	Weak	1*2 = 2	S1	G5
Gentiana fremontii	Torr.	38	5.1	53	Weak	1*2 = 2	S2	G4
Sphenopholis obtusata	(Michx.) Scribn.	39	5.8	45	Weak	1*2 = 2	S2	G5
Veronica catenata	Pennell	40	4.6	56	V. Weak	2*1 = 2	S2	G5
Bacopa rotundifolia	(Michx.) Wettst.	41	6.7	38	Weak	1*2 = 2	S1	G5
Cryptantha celosioides	(Eastw.) Payson	42	5.0	50	Weak	1*2 = 2	S2	G5
Rorippa sinuata	(Nutt.) Hitchc.	43	4.4	55	V. Weak	2*1 = 2	S1	G5
Mirabilis linearis	(Pursh) Heimerl	44	7.7	30	Weak	1*2 = 2	S2	G5
Ellisia nyctelea	(L.) L.	45	5.1	44	V. Weak	2*1 = 2	S2	G5
Calylophus serrulatus	Nutt	46	7.1	31	V. Weak	2*1 = 2	S2	G5
Townsendia exscapa	(Richardson) Porter	47	6.7	33	V. Weak	2*1 = 2	S2	G5
Boisduvalia glabella	(Nutt.) Walp.	48	6.8	32	V. Weak	2*1 = 2	S2	G5
Cyperus schweinitzii	Torr.	49	6.5	33	V. Weak	2*1 = 2	S2	G5
Cuscuta gronovii	Willd. ex Roem. & Schult.	50	7.6	27	V. Weak	2*1 = 2	S1	G5
Draba reptans	(Lam.) Fernald	51	7.2	27	V. Weak	2*1 = 2	S1	G5
Oenothera flava	(A. Nelson) Garrett	52	6.7	29	V. Weak	2*1 = 2	S2	G5
Oxytropis lagopus var. conjugans	Barneby	53	4.9	38	V. Weak	2*1 = 2	S1	G4
Rorippa tenerrima	Greene	54	5.2	34	V. Weak	2*1 = 2	S1	G5
Pedicularis racemosa	Douglas ex Benth.	55	2.2	10	V. Weak	2*1 = 2	S1	G5
Douglasia montana	A.Gray	56	1.9	11	V. Weak	2*1 = 2	S1	G4
Oxytropis campestris var. davisii	(L.) DC.	57	2.2	9	V. Weak	2*1 = 2	S2	G5
Lithophragma parviflorum	(Hook.) Nutt.	58	1.5	11	V. Weak	2*1 = 2	S2	G5

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Phlox gracilis ssp. gracilis	(Hook.) Greene*	59	1.1	13	V. Weak	2*1 = 2	S1	G5
Potentilla multisecta	(S.Watson) Rydb.	60	2.3	6	V. Weak	2*1 = 2	S2	GU
Heuchera glabra	Willd. ex Roem. & Schult.	61	1.8	7	V. Weak	2*1 = 2	S1	G5
Triantha occidentalis ssp. montana	(C.L.Hitchc.) Packer	62	1.9	6	V. Weak	2*1 = 2	S1	G4
Triantha occidentalis ssp. brevistyla	(C.L.Hitchc.) Packer	63	1.8	7	V. Weak	2*1 = 2	S1	G4
Romanzoffia sitchensis	Bong.	64	2.2	5	V. Weak	2*1 = 2	S2	G4
Carex paysonis	Clokey	65	1.6	7	Weak	1*2 = 2	S1	G4
Braya purpurascens	(R.Br.) Bunge ex Ledeb.	66	2.4	4	V. Weak	2*1 = 2	S1	G4
Suksdorfia violacea	A. Gray	67	0.9	11	V. Weak	2*1 = 2	S1	G4
Agrostis mertensii	Trin	68	2.7	4	V. Weak	2*1 = 2	S2	G5
Draba densifolia	Nutt.	69	2.0	5	V. Weak	2*1 = 2	S1	G5
Packera subnuda	(DC.) Trock & T.M.Barkley	70	1.4	6	Weak	1*2 = 2	S2	G5
Epilobium glaberrimum ssp. fastigiatum	Barbey	71	1.2	6	Weak	1*2 = 2	S1	G4
Physocarpus malvaceus	(Greene) Kuntze	72	0.6	12	V. Weak	2*1 = 2	S1	G4
Torreyochloa pallida var. pauciflora	(J. Presl) J.I. Davis	73	0.8	10	Weak	1*2 = 2	S1	G5
Hypericum scouleri ssp. scouleri	Hook.	74	0.9	7	Weak	1*2 = 2	S1	G4
Spiraea splendens	Baumann ex K.Koch	75	0.9	7	Weak	1*2 = 2	S1	G5
Ranunculus nivalis	L.	76	2.9	2	V. Weak	2*1 = 2	S1	G5
Trillium ovatum	Pursh	77	0.3	15	V. Weak	2*1 = 2	S1	G5
Carex infirminervia	Naczi	78	1.1	4	Weak	1*2 = 2	S1	G5
Gayophytum racemosum	Torr. & A.Gray	79	1.2	3	Weak	1*2 = 2	S1	G5
Antennaria luzuloides	Torr. & A.Gray	80	1.3	3	Weak	1*2 = 2	S1	G5
Carex heteroneura var. epapillosa	(Mack.) F.J.Herm.	81	1.5	2	Weak	1*2 = 2	S1	G5
Saussurea americana	D.C.Eaton	82	0.6	1	Weak	1*2 = 2	S1	G5
Oryzopsis canadensis	(Poir.) Torr. ex A.Gray	83	4.5	76	Neutral	1*3 = 3	S1	G5
Geranium carolinianum	L.	84	3.8	59	V. Weak	3*1 = 3	S1	G5
Cardamine parviflora	L.	85	4.2	51	V. Weak	3*1 = 3	S1	G5
Camissonia breviflora	(Torr. & A.Gray) P.H.Raven	86	8.0	25	V. Weak	3*1 = 3	S1	G5

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Anagallis minima	(L.) E.H.L.Krause	87	7.6	26	V. Weak	3*1 = 3	S1	G5
Potentilla macounii	Rydb.	88	5.2	33	V. Weak	3*1 = 3	S1	G1
Elatine triandra	Schkuhr	89	6.0	27	V. Weak	3*1 = 3	S1	G5
Sisyrinchium septentrionale	E.P.Bicknell	90	5.0	32	V. Weak	3*1 = 3	S3	G3
Arabidopsis salsuginea	(Pall.) N.Busch*	91	6.7	24	V. Weak	3*1 = 3	S1	G4
Juncus nevadensis	S.Watson	92	4.7	34	V. Weak	3*1 = 3	S1	G5
Linanthus septentrionalis	H.Mason.	93	6.5	24	V. Weak	3*1 = 3	S2	G5
Penstemon fruticosus var. scouleri	(Lindl.) Cronquist	94	3.1	13	V. Weak	3*1 = 3	S2	G5
Conimitella williamsii	(D.C.Eaton) Rydb.	95	2.4	13	V. Weak	3*1 = 3	S2	G3
Juncus parryi	Engelm.	96	2.8	11	V. Weak	3*1 = 3	S2	G4
Saxifraga odontoloma	Piper	97	2.3	12	V. Weak	3*1 = 3	S1	G5
Hippuris montana	Ledeb. ex Rchb.	98	2.8	10	V. Weak	3*1 = 3	S1	G4
Primula egaliksensis	Wormsk.	99	2.3	12	V. Weak	3*1 = 3	S2	G4
Lupinus minimus	Hook.	100	1.3	19	V. Weak	3*1 = 3	S1	G3
Potentilla villosa	Pall. ex Pursh	101	3.3	6	V. Weak	3*1 = 3	S2	G5
Lewisia pygmaea var. pygmaea	(A. Gray) B.L. Rob.	102	3.2	5	V. Weak	3*1 = 3	S2	G5
Montia linearis	(Douglas ex Hook.) Greene	103	0.9	17	V. Weak	3*1 = 3	S1	G5
Pedicularis flammea	L.	104	3.4	4	V. Weak	3*1 = 3	S2	G3
Philadelphus lewisii	Pursh	105	0.7	17	V. Weak	3*1 = 3	S1	G5
Lupinus polyphyllus	Lindl.	106	0.7	17	V. Weak	3*1 = 3	S1	G5
Utricularia cornuta	Michx.	107	3.4	3	V. Weak	3*1 = 3	S1	G5
Lechea intermedia var. depauperata	Legg.	108	3.0	3	V. Weak	3*1 = 3	S1	G5
Saxifraga flagellaris ssp. setigera	(Pursh) Tolm.	109	3.4	2	V. Weak	3*1 = 3	S2	G4
Oryzopsis exigua	Thurb.	110	1.4	5	Neutral	1*3 = 3	S1	G5
Pedicularis oederi	Vahl	111	3.1	0	V. Weak	3*1 = 3	S1	G5
Bidens vulgata	Greene	112	7.8	58	Strong	1*4 = 4	SU	G5
Osmorhiza longistylis	(Torr.) DC.	113	5.3	71	Strong	1*4 = 4	S2	G5
Muhlenbergia racemosa	(Michx.) Britton, Stern & Poggenb.	114	5.0	66	Strong	1*4 = 4	S2	G5
Bidens frondosa	L.	115	7.4	42	Strong	1*4 = 4	S2	G5
Doellingeria umbellata var. pubens	(Mill.) Nees	116	4.1	74	Weak	2*2 = 4	S2	G5
Atriplex canescens	(Pursh) Nutt.	117	6.3	45	Strong	1*4 = 4	SU	G5

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Bouteloua curtipendula	(Michx.) Torr.	118	4.4	59	Weak	2*2 = 4	S1	G5
Picradeniopsis oppositifolia	(Nutt.) Rydb. ex Rydb.	119	4.7	54	Weak	2*2 = 4	S1	G5
Spergularia salina	J. Presl & C. Presl	120	6.6	38	Strong	1*4 = 4	S2	G5
Aristida purpurea var. longiseta	(Steud.) Vasey*	121	5.3	47	Strong	1*4 = 4	S2	G5
Munroa squarrosa	(Nutt.) Torr.	122	7.8	30	Weak	2*2 = 4	S2	G5
Amaranthus californicus	(Moq.) S.Watson	123	5.0	46	Weak	2*2 = 4	S1	G4
Marsilea vestita	Hook. & Grev.	124	7.4	30	Weak	2*2 = 4	S2	G5
Wolffia columbiana	H.Karst.	125	4.6	48	Weak	2*2 = 4	S2	G5
Carex crawei	Dewey ex Torr.	126	4.7	46	Weak	2*2 = 4	S2	G5
Ambrosia acanthicarpa	Hook.	127	7.1	30	Weak	2*2 = 4	S2	G5
Puccinellia cusickii	Weath.	128	8.0	26	Weak	2*2 = 4	SU	G5
Cryptantha kelseyana	Greene	129	8.0	26	Weak	2*2 = 4	S1	G4
Shinnersoseris rostrata	(A.Gray) Tomb	130	7.7	26	Weak	2*2 = 4	S2	G5
Crepis occidentalis	Nutt.	131	5.1	39	Weak	2*2 = 4	S2	G5
Astragalus kentrophyta var. kentrophyta	A.Gray	132	6.9	23	V. Weak	4*1 = 4	S2	G5
Pinguicula villosa	L.	133	6.9	20	V. Weak	4*1 = 4	S2	G4
Castilleja sessiliflora	Pursh	134	5.5	23	V. Weak	4*1 = 4	S1	G5
Lupinus wyethii	S.Watson	135	5.9	21	V. Weak	4*1 = 4	S1	G5
Juncus stygius var. americanus	(Buchenau) Hultén	136	3.7	33	V. Weak	4*1 = 4	S2	G5
Juncus brevicaudatus	(Engelm.) Fernald	137	3.8	32	V. Weak	4*1 = 4	S2	G5
Suaeda moquinii	(Torr.) Greene	138	5.2	20	V. Weak	4*1 = 4	S2	G5
Chrysosplenium iowense	Rydb.	139	4.2	23	V. Weak	4*1 = 4	S3	G3
Anemone quinquefolia	L.	140	2.5	22	V. Weak	4*1 = 4	S1	G5
Phacelia linearis	(Pursh) Holz.	141	2.7	19	V. Weak	4*1 = 4	S2	G5
Agrostis exarata	Trin.	142	3.0	16	V. Weak	4*1 = 4	S2	G5
Campanula aparinoides	Pursh	143	4.0	12	V. Weak	4*1 = 4	S1	G5
Lathyrus palustris	L.	144	3.5	10	V. Weak	4*1 = 4	S1	G5
Braya humilis ssp. maccallae	J.G. Harris	145	3.7	8	V. Weak	4*1 = 4	S1	G1
Agrostis humilis	Vasey	146	3.9	7	V. Weak	4*1 = 4	S2	G4
Nemophila breviflora	A. Gray	147	1.3	21	V. Weak	4*1 = 4	S1	G5
Artemisia tridentata	Nutt.	148	2.2	11	Weak	2*2 = 4	S2	G5
Allium geyeri	S.Watson	149	1.0	24	V. Weak	4*1 = 4	S2	G4
Saxifraga nivalis	L.	150	4.0	5	V. Weak	4*1 = 4	S2	GU

Appendix B - Continued

Species	Authority	Rank	Vel. (km/yr)	Frag. (%)	Seed Dispersal	Vulnerability	Provincial Status	Global Status
Epilobium leptocarpum	Hausskn.	151	1.7	13	Weak	2*2 = 4	S1	G5
Arabis lemmonii	S.Watson*	152	3.7	5	V. Weak	4*1 = 4	S2	G5
Carex podocarpa	R.Br.	153	2.4	8	Weak	2*2 = 4	S2	G4
Lewisia rediviva	Pursh	154	0.9	22	V. Weak	4*1 = 4	S1	G5
Pedicularis langsdorfii ssp. arctica	(R. Br.) Pennell*	155	3.9	5	V. Weak	4*1 = 4	S2	G5
Iliamna rivularis	(Douglas) Greene	156	0.8	21	V. Weak	4*1 = 4	S2	G5
Erigeron flagellaris	A.Gray.	157	2.7	6	Weak	2*2 = 4	S1	G5
Gentiana calycosa	Grisbe.	158	1.9	8	Weak	2*2 = 4	S1	G4
Draba juvenilis	Kom.	159	3.8	4	V. Weak	4*1 = 4	S2	G4
Erigeron divergens	Torr. & A.Gray	160	1.4	9	Weak	2*2 = 4	S1	G5
Arnica longifolia	D.C.Eaton	161	1.8	7	Weak	2*2 = 4	S2	G5
Carex preslii	Steud.	162	1.5	8	Weak	2*2 = 4	S2	G4
Polypodium hesperium	Maxon	163	0.8	13	Weak	2*2 = 4	S1	G5
Microseris nutans	(Hook.) Sch.Bip.	164	0.9	11	Weak	2*2 = 4	S2	G5
Pseudognaphalium microcephalum	(Nutt.) Anderb.	165	0.7	14	Weak	2*2 = 4	SU	GU
Sedum divergens	S. Watson	166	2.6	4	Weak	2*2 = 4	S2	G5
Arnica louiseana	Farr	167	2.4	4	Weak	2*2 = 4	S1	G3
Carex misandra	R.Br.*	168	2.2	4	Weak	2*2 = 4	S1	G5
Cirsium scariosum	(Poir.) Nutt.	169	1.2	7	Strong	1*4 = 4	S2	G5
Stellaria obtusa	Engelm.	170	1.0	7	Strong	1*4 = 4	S1	G5
Salix stolonifera	Coville	171	1.2	6	Strong	1*4 = 4	S1	G4
Brickellia grandiflora	(Hook.) Nutt.	172	0.8	9	Strong	1*4 = 4	S1	G5
Artemisia furcata var. furcata	M.Bieb.	173	2.4	3	Weak	2*2 = 4	S1	GU
Sagina nivalis	(Lindblad) Fr.	174	1.9	3	Strong	1*4 = 4	S1	G5
Festuca subulata	Trin.	175	0.7	7	Strong	1*4 = 4	S1	G5
Festuca viviparoidea ssp. krajinae	Krajina ex Pavlick	176	1.4	4	Strong	1*4 = 4	S1	G4
Stellaria americana	(Porter ex B.L. Rob.) Standl.	177	1.8	1	Strong	1*4 = 4	S1	G3
Quercus macrocarpa	Michx.	178	6.5	83	V. Strong	1*5 = 5	SU	G5
Botrychium matricariifolium	(Döll) A. Braun ex W.D.J. Koch	179	4.9	86	V. Strong	1*5 = 5	S1	G5
Eleocharis engelmannii	Steud.	180	4.5	81	V. Strong	1*5 = 5	S1	G4
Rubus x paracaulis	L.H. Bailey (pro sp.)	181	4.6	77	V. Strong	1*5 = 5	S1	GU
Fraxinus pennsylvanica	Marshall	182	7.1	47	V. Strong	1*5 = 5	S1	G5
Ruppia cirrhosa	(Petagna) Grande	183	5.0	57	V. Strong	1*5 = 5	S1	G5

Appendix B - Continued

Species	Authority	Rank	Vel. (km/yr)	Frag. (%)	Seed Dispersal	Vulnerability	Provincial Status	Global Status
Luzula groenlandica	Böcher	184	3.0	66	V. Weak	5*1 = 5	S1	G4
Chenopodium watsonii	A.Nelson	185	8.9	19	V. Weak	5*1 = 5	S1	G5
Potentilla multifida	L.	186	3.2	40	V. Weak	5*1 = 5	S1	G5
Tanacetum bipinnatum ssp. huronense	(Nutt.) Breitung*	187	3.1	40	V. Weak	5*1 = 5	S2	G4
Luzula rufescens	Fisch. ex E.Mey.	188	2.9	43	V. Weak	5*1 = 5	S1	G5
Luzula acuminata	Raf.	189	2.3	49	V. Weak	5*1 = 5	S1	G5
Braya humilis ssp. porsildii	J.G. Harris	190	6.8	15	V. Weak	5*1 = 5	S1	G3
Viola pallens	(Banks ex DC.) Brainerd*	191	3.0	33	V. Weak	5*1 = 5	S2	G5
Astragalus bodinii	E.Sheld.	192	5.1	17	V. Weak	5*1 = 5	S1	G4
Ranunculus glaberrimus	Hook.	193	5.8	14	V. Weak	5*1 = 5	S2	G5
Lomatium cous	(S. Watson) J.M. Coult. & Rose	194	8.2	10	V. Weak	5*1 = 5	S1	G5
Pedicularis sudetica	Willd.	195	8.8	8	V. Weak	5*1 = 5	S1	G5
Chenopodium incanum	(S.Watson) A.Heller	196	8.2	8	V. Weak	5*1 = 5	S1	G5
Potentilla hookeriana	Lehm.	197	4.4	13	V. Weak	5*1 = 5	S2	G4
Potentilla subjuga	Rydb.	198	5.6	10	V. Weak	5*1 = 5	S1	G4
Geranium erianthum	DC.	199	4.7	11	V. Weak	5*1 = 5	SU	G5
Saxifraga oregana var. montanensis	(Small) C.L. Hitchc.	200	4.4	12	V. Weak	5*1 = 5	SU	G4
Camassia quamash var. quamash	(Pursh) Greene	201	1.7	27	V. Weak	5*1 = 5	S2	G5
Castilleja pallida	(L.) Kunth	202	5.9	8	V. Weak	5*1 = 5	SU	G5
Lechea minor var. depauperata	(Hodgdon) B. Boivin	203	4.4	9	V. Weak	5*1 = 5	S2	G4
Lithophragma glabrum	Nutt.	204	5.0	8	V. Weak	5*1 = 5	S2	G4
Potentilla drummondii	Lehm.	205	4.2	9	V. Weak	5*1 = 5	S2	G5
Camissonia andina	(Nutt.) P.H.Raven	206	5.3	7	V. Weak	5*1 = 5	S1	GU
Draba novolympica	Payson & H.St.John	207	4.8	6	V. Weak	5*1 = 5	S2	G5
Papaver radicatum ssp. kluanense	(D. Löve) D.F. Murray	208	5.6	4	V. Weak	5*1 = 5	S2	G4
Juncus biglumis	L.	209	4.3	5	V. Weak	5*1 = 5	S2	G5
Draba porsildii	Mulligan	210	4.2	5	V. Weak	5*1 = 5	S1	G3
Draba macounii	O.E.Schulz	211	5.4	4	V. Weak	5*1 = 5	S2	G3
Draba ventosa	A.Gray	212	4.9	3	V. Weak	5*1 = 5	S2	G3
Campanula uniflora	L.	213	4.8	3	V. Weak	5*1 = 5	S2	G4

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Athyrium alpestre var. americanum	(Butters) Lellinger*	214	1.2	8	V. Strong	1*5 = 5	S1	G4
Vaccinium ovalifolium	Sm.	215	1.6	5	V. Strong	1*5 = 5	S2	G5
Boschniakia rossica	(Cham. & Schltdl.) B.Fedtsch.	216	5.8	1	V. Weak	5*1 = 5	S1	G5
Botrychium lineare	W.H.Wagner	217	1.3	4	V. Strong	1*5 = 5	S1	G2
Mimulus tilingii	Regel	218	1.3	4	V. Strong	1*5 = 5	S1	G5
Polygonum douglasii ssp. austiniae	(Greene) M.E.Jones*	219	1.0	2	V. Strong	1*5 = 5	S1	G4
Botrychium x watertonense	W.H.Wagner	220	1.1	1	V. Strong	1*5 = 5	S1	GU
Mimulus breweri	(Greene) Coville	221	0.1	2	V. Strong	1*5 = 5	S1	G5
Carex vulpinoidea	Michx.	222	3.9	58	Weak	3*2 = 6	S2	G5
Potamogeton strictifolius	A.Benn.	223	4.0	52	Weak	3*2 = 6	S2	G5
Carex nebrascensis	Dewey	224	4.4	47	Weak	3*2 = 6	S2	G5
Potamogeton obtusifolius	Mert. & W.D.J.Koch	225	4.2	44	Weak	3*2 = 6	S2	G5
Carex hystericina	Muhl. ex Willd.	226	4.0	46	Weak	3*2 = 6	S1	G5
Polygala paucifolia	Willd.	227	4.0	45	Weak	3*2 = 6	S1	G5
Elodea canadensis	Michx.	228	4.8	36	Weak	3*2 = 6	SU	G5
Lomatogonium rotatum	(L.) Fr. ex Fernald	229	4.6	38	Weak	3*2 = 6	S2	G5
Carex garberi	Fernald	230	7.2	24	Weak	3*2 = 6	S2	G5
Isoetes echinospora	Durieu	231	4.7	32	Weak	3*2 = 6	S2	G5
Erigeron hyssopifolius	Michx.	232	2.6	15	Weak	3*2 = 6	S1	G5
Artemisia borealis	Pall.*	233	2.8	11	Weak	3*2 = 6	S2	G5
Epilobium lactiflorum	Hausskn.	234	2.8	11	Weak	3*2 = 6	S2	G5
Arnica parryi	A.Gray	235	3.2	9	Weak	3*2 = 6	S2	G5
Carex scoparia	Willd.	236	3.0	10	Weak	3*2 = 6	S1	G5
Erigeron lackschewitzii	G.L.Nesom & W.A.Weber	237	2.4	12	Weak	3*2 = 6	SU	G3
Antennaria aromatica	Evert	238	3.0	8	Weak	3*2 = 6	S2	G4
Arnica amplexicaulis	Nutt.*	239	2.8	8	Weak	3*2 = 6	S2	G5
Melica spectabilis	Scribn.	240	1.5	13	Neutral	2*3 = 6	S2	G5
Carex vesicaria	L.	241	0.9	16	Weak	3*2 = 6	S1	G5
Epilobium luteum	Pursh	242	2.8	5	Weak	3*2 = 6	S1	G5
Carex lachenalii	Schkuhr	243	3.3	4	Weak	3*2 = 6	S2	G5
Adenocaulon bicolor	Hook.	244	0.7	19	Weak	3*2 = 6	S2	G5
Erigeron trifidus	Hook.	245	2.8	4	Weak	3*2 = 6	S3	G5
Melica smithii	(Porter) Vasey	246	0.9	12	Neutral	2*3 = 6	S1	G4

Appendix B - Continued

Species	Authority	Rank	Vel. (km/yr)	Frag. (%)	Seed Dispersal	Vulnerability	Provincial Status	Global Status
Prenanthes sagittata	Tjitrosoedirdjo	247	0.9	11	Neutral	2*3 = 6	S2	G3
Rumex paucifolius	Nutt.	248	3.3	0	Weak	3*2 = 6	S1	G5
Panicum leibergii	(Vasey) Scribn.	249	3.7	74	Weak	4*2 = 8	S1	G5
Hypericum majus	(A.Gray) Britton	250	3.7	46	Weak	4*2 = 8	S2	G5
Potamogeton foliosus	Raf.	251	3.7	41	Weak	4*2 = 8	S2	G5
Carex lacustris	Willd.	252	3.6	38	Weak	4*2 = 8	S2	G5
Hymenopappus filifolius	Hook.	253	4.9	21	Weak	4*2 = 8	S2	G5
Gentianopsis detonsa ssp. raupii	(Rottb.) Ma	254	3.5	28	Weak	4*2 = 8	S1	G4
Carex heleonastes	Ehrh. ex L.f.	255	3.7	25	Weak	4*2 = 8	S2	G4
Eupatorium maculatum	L.	256	4.3	20	Weak	4*2 = 8	S1	G5
Carex arcta	Boott	257	3.5	20	Weak	4*2 = 8	S1	G5
Monotropa hypopithys	L.	258	3.9	18	Weak	4*2 = 8	S2	G5
Carex adusta	Boott	259	3.4	21	Weak	4*2 = 8	S1	G5
Crepis intermedia	A.Gray	260	2.9	24	Weak	4*2 = 8	S2	G5
Carex cordillerana	Saarela & B.A.Ford	261	4.1	14	Weak	4*2 = 8	S1	G3
Gnaphalium viscosum	Kunth	262	4.0	13	Weak	4*2 = 8	SU	GU
Potamogeton nodosus	Poir.	263	3.5	14	Weak	4*2 = 8	S1	G5
Carex lenticularis var. dolia	(M.E.Jones) L.A.Standl.	264	4.2	11	Weak	4*2 = 8	S1	G5
Carex aperta	Boott	265	3.3	14	Weak	4*2 = 8	S1	G4
Epilobium saximontanum	Hausskn.	266	3.7	11	Weak	4*2 = 8	S1	G5
Panicum acuminatum	Sw.	267	3.9	10	Weak	4*2 = 8	SU	G5
Alopecurus alpinus	Vill.	268	1.6	22	Weak	4*2 = 8	S2	G5
Carex petasata	Dewey	269	3.5	9	Weak	4*2 = 8	S1	G5
Packera contermina	(Greenm.) J.F.Bain	270	3.6	6	Weak	4*2 = 8	S3	G3
Agoseris lackschewitzii	Douglass M.Hend. & R.K.Moseley*	271	4.1	5	Weak	4*2 = 8	S2	G4
Trisetum wolfii	Vasey*	272	2.5	9	Strong	2*4 = 8	S1	G4
Trisetum montanum	Vasey	273	2.1	10	Strong	2*4 = 8	S1	G4
Poa stenantha	Trin.	274	2.5	7	Strong	2*4 = 8	S1	G5
Galium bifolium	S.Watson	275	1.4	10	Strong	2*4 = 8	S1	G5
Osmorhiza purpurea	(J.M. Coult. & Rose) Suksd.	276	1.7	8	Strong	2*4 = 8	S2	G4
Trisetum canescens	Buckley*	277	0.8	15	Strong	2*4 = 8	S2	G5
Poa gracillima	Vasey*	278	1.0	10	Strong	2*4 = 8	S2	G2
Pyrola picta	Sm.	279	1.0	11	Strong	2*4 = 8	S1	G4

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Festuca altaica	Trin. ex Ledeb.	280	2.9	3	Strong	2*4 = 8	S2	G5
Salix raupii	Argus	281	2.0	4	Strong	2*4 = 8	S1	G2
Najas flexilis	(Willd.) Rostk. & W.L.E.Schmidt	282	4.3	48	Neutral	3*3 = 9	S2	G5
Streptopus roseus	Michx.*	283	1.7	16	Neutral	3*3 = 9	S1	G4
Bolboschoenus fluviatilis	(Torr.) Soják	284	4.2	81	V. Strong	2*5 = 10	S1	G5
Puccinellia distans ssp. hauptiana	(Jacq.) Parl.	285	2.8	92	Weak	5*2 = 10	S1	G3
Spartina pectinata	Bosc ex Link	286	5.4	37	V. Strong	2*5 = 10	S1	G5
Carex echinata ssp. echinata	Murray	287	2.9	65	Weak	5*2 = 10	S1	G5
Carex lenticularis var. lenticularis	Michx.	288	2.8	63	Weak	5*2 = 10	S1	G5
Lactuca biennis	(Moench) Fernald	289	3.4	45	Weak	5*2 = 10	S2	G5
Carex pedunculata	Muhl. ex Willd.	290	2.5	54	Weak	5*2 = 10	S1	G5
Lycopodiella inundata	(L.) Holub	291	3.1	41	Weak	5*2 = 10	S2	G5
Atriplex truncata	(Torr.) A.Gray	292	8.9	13	Weak	5*2 = 10	S1	G5
Potamogeton robbinsii	Oakes	293	2.7	43	Weak	5*2 = 10	S1	G5
Carex umbellata	Willd.	294	3.2	33	Weak	5*2 = 10	S2	G5
Epilobium halleanum	Hausskn.	295	3.2	32	Weak	5*2 = 10	S1	G5
Isoetes occidentalis	L.F. Hend.	296	6.0	15	Weak	5*2 = 10	S1	G4
Isoetes x truncata	(A.A. Eaton) Clute	297	6.0	15	Weak	5*2 = 10	S1	GU
Isoetes maritima	Underw.	298	6.2	14	Weak	5*2 = 10	S1	G4
Carex supina	Willd. ex Wahlenb.	299	3.0	29	Weak	5*2 = 10	S1	G5
Asclepias viridiflora	Raf.	300	5.6	15	Weak	5*2 = 10	S1	G5
Carex glacialis	Mack	301	5.9	13	Weak	5*2 = 10	S2	G5
Crepis atribarba	A.Heller	302	5.1	13	Weak	5*2 = 10	S2	G5
Diphasiastrum sitchense	(Rupr.) Holub	303	5.2	12	Weak	5*2 = 10	S2	G5
Carex oligosperma	Michx.	304	5.5	11	Weak	5*2 = 10	S3	G5
Carex incurviformis var. incurviformis	Mack.	305	4.6	12	Weak	5*2 = 10	S2	G4
Antennaria corymbosa	E.E.Nelson	306	4.7	11	Weak	5*2 = 10	S1	G5
Carex platylepis	Mack	307	5.3	9	Weak	5*2 = 10	S1	G4
Carex mertensii	Prescott ex Bong.	308	1.9	26	Weak	5*2 = 10	S2	G5
Carex saximontana	Mack.	309	7.4	6	Weak	5*2 = 10	S1	G5
Erigeron ochroleucus var. scribneri	Nutt.	310	4.6	10	Weak	5*2 = 10	S1	G5
Erigeron radicatus	Hook	311	4.7	9	Weak	5*2 = 10	S2	G3
Carex illota	L.H.Bailey	312	4.3	8	Weak	5*2 = 10	S1	G4

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Nothocalais cuspidata	(Pursh) Greene	313	6.3	5	Weak	5*2 = 10	S2	G5
Erigeron pallens	Cronquist	314	5.6	6	Weak	5*2 = 10	S2	G4
Epilobium clavatum	Trev.	315	4.5	6	Weak	5*2 = 10	S2	G5
Cheilanthes gracillima	D.C.Eaton	316	1.7	13	V. Strong	2*5 = 10	S1	G4
Mimulus floribundus	Douglas ex Lindl.	317	2.0	11	V. Strong	2*5 = 10	S1	G5
Dryopteris filix-mas	(L.) Schott	318	1.3	15	V. Strong	2*5 = 10	S1	G5
Platanthera stricta	Lindl.	319	1.5	11	V. Strong	2*5 = 10	S2	G5
Adiantum aleuticum	(Rupr.) C.A. Paris	320	1.4	12	V. Strong	2*5 = 10	S2	G5
Tsuga heterophylla	(Raf.) Sarg.	321	1.9	9	V. Strong	2*5 = 10	S1	G5
Piperia unalascensis	(Spreng.) Rydb.*	322	1.7	8	V. Strong	2*5 = 10	S2	G5
Polygonum minimum	S. Watson	323	1.4	9	V. Strong	2*5 = 10	S2	G5
Selaginella wallacei	Hieron.	324	0.8	15	V. Strong	2*5 = 10	S1	G5
Taxus brevifolia	Nutt.	325	0.7	14	V. Strong	2*5 = 10	S1	G4
Botrychium paradoxum	W.H. Wagner	326	2.1	4	V. Strong	2*5 = 10	S1	G3
Oryzopsis micrantha	(Trin. & Rupr.) Thurb.	327	8.4	21	Neutral	4*3 = 12	S2	G5
Cerastium brachypodum	(Engelm. ex A.Gray) B.L.Rob. ex Britton	328	7.2	24	Strong	3*4 = 12	S1	G5
Symphyotrichum eatonii	(A.Gray) G.L.Nesom	329	4.6	28	Strong	3*4 = 12	S2	G5
Nymphaea leibergii	(Morong) Morong	330	3.6	28	Neutral	4*3 = 12	S1	G5
Nymphaea tetragona	Georgi	331	3.5	18	Neutral	4*3 = 12	S1	G5
Plantago maritima	L.	332	3.2	14	Strong	3*4 = 12	S1	G5
Thuja plicata	Donn ex D.Don	333	2.4	17	Strong	3*4 = 12	S2	G5
Deschampsia elongata	(Hook.) Munro	334	1.4	20	Strong	3*4 = 12	S1	G5
Salix alaxensis var. alaxensis	(Andersson) Coville	335	3.0	8	Strong	3*4 = 12	S2	G5
Festuca occidentalis	Hook.	336	1.1	20	Strong	3*4 = 12	S1	G5
Salix commutata	Bebb	337	2.8	8	Strong	3*4 = 12	S2	G5
Stellaria arenicola	Raup*	338	3.4	6	Strong	3*4 = 12	S1	G3
Elymus scribneri	(Vasey) M.E.Jones	339	3.2	6	Strong	3*4 = 12	S2	G5
Salix tyrrellii	Raup	340	3.4	5	Strong	3*4 = 12	S1	G5
Trisetum cernuum	Trin.	341	1.0	16	Strong	3*4 = 12	S2	GU
Mimulus glabratus	Kunth	342	6.7	24	V. Strong	3*5 = 15	S1	G5
Malaxis paludosa	(L.) Sw.	343	4.2	34	V. Strong	3*5 = 15	S1	G4
Dryopteris cristata	(L.) A. Gray	344	4.7	27	V. Strong	3*5 = 15	S1	G5
Streptopus streptopoides	(Ledeb.) Frye & Rigg	345	2.6	38	Neutral	5*3 = 15	S1	G5
Eriogonum cernuum	Nutt.	346	6.8	14	Neutral	5*3 = 15	S2	G5
Prenanthes alata	(Hook.) D. Dietr.	347	2.6	31	Neutral	5*3 = 15	S1	G5

Appendix B - Continued

Species	Authority	Rank	Vel. (km/yr)	Frag. (%)	Seed Dispersal	Vulnerability	Provincial Status	Global Status
Eriogonum pauciflorum	Pursh	348	5.4	11	Neutral	5*3 = 15	SU	G5
Silene antirrhina	L.	349	2.9	17	V. Strong	3*5 = 15	S1	G5
Gymnocarpium disjunctum	(Rupr.) Ching	350	2.7	16	V. Strong	3*5 = 15	S1	G5
Pinus flexilis	E.James	351	3.2	14	V. Strong	3*5 = 15	S2	G4
Pellaea gastonyi	Windham	352	2.7	16	V. Strong	3*5 = 15	S1	G2
Cypripedium montanum	Douglas ex Lindl.	353	2.1	17	V. Strong	3*5 = 15	S2	G4
Mimulus ringens	L.	354	2.9	12	V. Strong	3*5 = 15	S1	G5
Listera convallarioides	(Sw.) Nutt. ex Elliott*	355	2.8	11	V. Strong	3*5 = 15	S2	G5
Botrychium hesperium	(Maxon & R.T. Clausen) W.H. Wagner & Lellinger	356	2.5	12	V. Strong	3*5 = 15	SU	G4
Ribes laxiflorum	Pursh	357	2.8	10	V. Strong	3*5 = 15	S2	G5
Pinus albicaulis	Engelm.	358	3.2	7	V. Strong	3*5 = 15	S2	G3
Schoenoplectus heterochaetus	(Chase) Soják	359	3.1	8	V. Strong	3*5 = 15	S1	G5
Larix occidentalis	Nutt.	360	2.4	9	V. Strong	3*5 = 15	S2	G5
Listera caurina	Piper*	361	1.2	19	V. Strong	3*5 = 15	S1	G4
Botrychium ascendens	W.H. Wagner	362	2.8	8	V. Strong	3*5 = 15	S2	G3
Eriophorum callitrix	Cham. ex C.A.Mey.	363	3.2	2	V. Strong	3*5 = 15	S2	G5
Salix sitchensis	Sanson ex Bong.	364	3.7	53	Strong	4*4 = 16	S1	G5
Danthonia spicata	(L.) Roem. & Schult.	365	3.9	45	Strong	4*4 = 16	S2	G5
Symphyotrichum campestre	(Nutt.) G.L.Nesom	366	3.5	18	Strong	4*4 = 16	S2	G5
Calamagrostis lapponica	(Wahlenb.) Hartm.	367	4.0	15	Strong	4*4 = 16	S1	G5
Glyceria elata	(Nash) M.E.Jones	368	1.9	25	Strong	4*4 = 16	S2	G4
Stellaria umbellata	Turcz.	369	3.8	6	Strong	4*4 = 16	S1	G5
Poa lettermanii	Vasey	370	3.9	5	Strong	4*4 = 16	S1	G4
Minuartia elegans	(Cham. & Schltdl.) Schischk.	371	3.7	2	Strong	4*4 = 16	S1	G4
Liparis loeselii	(L.) Rich.	372	3.9	44	V. Strong	4*5 = 20	S1	G5
Leymus mollis	(Trin.) Pilg.	373	2.9	58	Strong	5*4 = 20	S2	G5
Sagina nodosa	(L.) Fenzl	374	2.9	58	Strong	5*4 = 20	S1	G5
Eleocharis elliptica	Kunth	375	3.9	41	V. Strong	4*5 = 20	S2	G5
Festuca lenensis	Drobow	376	8.6	17	Strong	5*4 = 20	SU	G4
Arctagrostis arundinacea	(Trin.) Beal	377	7.8	16	Strong	5*4 = 20	S2	G5
Botrychium michiganense	**	378	4.2	30	V. Strong	4*5 = 20	SU	GU
Gymnocarpium jessoense	(Koidz.) Koidz.	379	3.4	36	V. Strong	4*5 = 20	S1	G5
Sagittaria latifolia	Willd.	380	3.6	32	V. Strong	4*5 = 20	S2	G5

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Botrychium pallidum	W.H. Wagner	381	4.4	23	V. Strong	4*5 = 20	S1	G3
Botrychium campestre	W.H. Wagner & Farrar	382	3.5	28	V. Strong	4*5 = 20	S1	G3
Pellaea glabella ssp. simplex	(Butters) Á. Löve & D. Löve	383	3.6	26	V. Strong	4*5 = 20	S2	G4
Arenaria longipedunculata	Hultén	384	5.7	14	Strong	5*4 = 20	S1	G3
Phegopteris connectilis	(Michx.) Watt	385	3.7	20	V. Strong	4*5 = 20	S2	G5
Botrychium crenulatum	W.H. Wagner	386	4.2	16	V. Strong	4*5 = 20	S1	G3
Cystopteris montana	(Lam.) Bernh. ex Desv.	387	3.4	20	V. Strong	4*5 = 20	S2	G5
Botrychium pinnatum	H. St. John	388	3.3	20	V. Strong	4*5 = 20	S3	G4
Botrychium spathulatum	W.H. Wagner	389	4.1	16	V. Strong	4*5 = 20	S2	G3
Spiranthes lacera	(Raf.) Raf.	390	4.3	15	V. Strong	4*5 = 20	S1	G5
Stellaria crispa	Cham. & Schltdl.	391	2.2	27	Strong	5*4 = 20	S2	G5
Cypripedium acaule	Aiton	392	3.9	15	V. Strong	4*5 = 20	S3	G5
Brasenia schreberi	J.F.Gmel	393	3.4	17	V. Strong	4*5 = 20	S1	G5
Cynoglossum virginianum var. boreale	(Fernald) Cooperr.*	394	1.6	36	Strong	5*4 = 20	S1	G5
Mimulus guttatus	DC.	395	3.4	16	V. Strong	4*5 = 20	S2	G5
Sparganium hyperboreum	Laest. ex Beurl.	396	3.0	17	V. Strong	4*5 = 20	S1	G5
Woodsia glabella	R. Br. ex Richardson	397	3.1	16	V. Strong	4*5 = 20	S1	G5
Botrychium oneidense	(Gilbert) House	398	3.8	13	V. Strong	4*5 = 20	S1	G4
Poa laxa ssp. banffiana	Soreng	399	5.3	9	Strong	5*4 = 20	S1	G5
Botrychium lanceolatum	(S.G. Gmel.) Ângström	400	3.4	15	V. Strong	4*5 = 20	S2	G5
Botrychium simplex	E. Hitchc.	401	3.4	14	V. Strong	4*5 = 20	S2	G5
Blysmus rufus	(Huds.) Link	402	3.3	15	V. Strong	4*5 = 20	S1	G5
Pinus monticola	Douglas ex D.Don	403	2.2	20	V. Strong	4*5 = 20	S1	G4
Pellaea glabella ssp. occidentalis	(E.E. Nelson) Windham	404	3.5	13	V. Strong	4*5 = 20	S1	G4
Salix lanata ssp. calcicola	(Fernald & Wiegand) Hultén*	405	4.2	9	Strong	5*4 = 20	S1	G4
Cryptogramma stelleri	(S.G. Gmel.) Prantl	406	3.4	10	V. Strong	4*5 = 20	S2	G5
Festuca minutiflora	Rydb.	407	4.9	7	Strong	5*4 = 20	S2	G5
Hordeum pusillum	Nutt.	408	6.8	5	Strong	5*4 = 20	SU	G5
Rhynchospora capillacea	Torr.	409	3.2	78	V. Strong	5*5 = 25	S1	G4
Trichophorum clintonii	(A.Gray) S.G.Sm.	410	3.2	62	V. Strong	5*5 = 25	S1	G4
Sparganium fluctuans	(Engelm.) B.L.Rob.	411	3.0	64	V. Strong	5*5 = 25	S1	G5

Appendix B - Continued

Species	**Authority**	**Rank**	**Vel. (km/yr)**	**Frag. (%)**	**Seed Dispersal**	**Vulnerability**	**Provincial Status**	**Global Status**
Sparganium glomeratum	(Laest. ex Beurl.) Beurl.	412	2.5	46	V. Strong	5*5 = 25	S1	G4
Scirpus pallidus	(Britton) Fernald	413	5.8	9	V. Strong	5*5 = 25	S1	G5
Botrychium pedunculosum	W.H. Wagner	414	4.7	11	V. Strong	5*5 = 25	S1	G2
Pellaea glabella	Mett. ex Kuhn	415	4.4	11	V. Strong	5*5 = 25	S2	G5
Silene involucrata	(Cham. & Schltdl.) Bocquet	416	4.4	2	V. Strong	5*5 = 25	S1	G5
Rhododendron lapponicum	(L.) Wahlenb.	417	5.6	7	Unknown	Unknown	S2	G5
Koenigia islandica	L.	418	2.7	6	Unknown	Unknown	S1	G4
Loiseleuria procumbens	(L.) Loisel.	419	1.8	4	Unknown	Unknown	S2	G5

* - Species as listed in ACIMS is a synonym, and now goes by a different scientific name

** - Species listed in ACIMS is not a widely recognized species name