Search and rescue: detection and mitigation of rare vascular plant species

by

Jacqueline Marie Dennett

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Department of Renewable Resources
University of Alberta

Abstract

Understanding where and when populations occur is the first step to conservation and maintenance of biodiversity. Where human land-use overlaps with populations of conservation concern, population loss may occur, potentially reducing long-term persistence of species, particularly for those that are rare. Understanding the relationship between land-use change and extirpation is therefore essential to guiding conservation, but this can only be achieved through well-designed surveys and monitoring programs. One key aspect of surveys that is often overlooked is the ability to accurately and consistently detect populations, while the success of mitigation practices depends on a clear understanding of what techniques will best ensure the longevity of a given population. In this thesis, I examined factors that affect detection, extirpation of historic populations, and the efficacy of mitigative translocations for rare vascular plants in the oil sands region of Alberta. First, I used two field experiments to better understand and test the effects of scale (1 – 2500 m^2), abundance (plant density), and observer experience on detection rates of rare plants in forested systems. Scale and abundance were the most important determinants of detection for plot-based surveys, whereas previous experience of the observer had limited influence. Plants at low abundance often went unrecorded in large plots (>1000 m^2), even when they were morphologically distinct or flowering. Second, I focused on graminoids and used *Carex* (sedges) as a model group to examine how forest structure and morphology affected detection success for this notoriously challenging group. I found that graminoids were not any less likely to be detected in field surveys than other growth-forms, but greater differences between observers were most related to higher ground cover of forbs and short shrubs. Exploring factors that further affect detection for *Carex*, I found that detection failures were related to local abundance (cover), species morphology, and vegetation cover. In contrast, detection delays of

Carex were less related to morphology, suggesting that cryptic species are likely to go unnoticed where they are present, even with careful searching. Third, I examined the relationship between oil and gas footprint and persistence of rare plant populations in northeastern Alberta by revisiting historical populations across a range of footprint types. I found no correlation between the amount of surrounding oil and gas footprint and extirpation of field-visited populations, suggesting either adjacent development poses little threat or there exists an unpaid extinction debt. Fourth, I conducted experimental translocations for two rare peatland species and monitored their growth and survival over three years. I found high survival across different types of recipient sites, suggesting this technique may be quite suitable for many peatland species, especially *Sarracenia purpurea*. However, poor growth and evidence of stunting in *Carex oligosperma* stresses the importance of conducting translocation with species whose ecology is well understood. When the niche of a species is poorly understood, use of an experimental approach to translocations with detailed monitoring is needed to assess the efficacy of this practice. Key conservation issues within the oil sands area continue to be a lack of consistency in methodology and public reporting for surveys and mitigation. The experiments conducted for this thesis serve to improve our knowledge around rare plant survey practices, rates of population persistence, mitigative efforts, and more broadly contribute to the development of best practices and guidelines for plant conservation within this rapidly changing region of Canada's boreal forest.

Preface

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Table of Contents

List of Tables

List of Figures

Chapter 1: Introduction

The idea that most species are rare is well recognized by ecologists. This reflects observations across systems and geographic areas that relatively few species dominate sites with the majority occurring at low abundance (Whittaker 1972). In addition to local patterns in abundance where most species are rare, the distribution of most species ranges are small (Gaston 2003). Broadly speaking, rarity is product of evolutionary and ecological history, species traits, and our present understanding of species distribution and abundance in time and space (Kunin & Gaston 1997; Murray et al. 2002; Hartley & Kunin 2003). One well-accepted approach to defining rarity is to partition species along axes of abundance and range size (Kunin & Gaston 1997) with Rabinowitz's approach adding a third axis of habitat specificity (Rabinowitz et al. 1986). In addition to ecological definitions of rarity, we often consider the term "rare" to infer a specific conservation status, one of potential concern, which can prompt management actions or convey legal implications (Kunin & Gaston 1997; Master et al. 2012). In Canada, the provincial conservation status for plants is determined using NatureServe methods, which in practice often reflects the abundance-geographic range approach (Master et al. 2012).

Overall, diversity and rarity are low at higher latitudes with the boreal forest representing an extreme in the diversity-latitude relationship (Lessa et al. 2003). This partly reflects this region having been glaciated in recent ice ages, and endemism is uncommon (but see species of the Athabasca Sand Plain (Lamb & Guedo 2012)). Indeed, a large proportion of boreal species that are locally considered rare often have wide geographic ranges and are more abundant in other regions of Canada or the United States (Kershaw et al. 2001). Therefore, species of conservation concern in places like Alberta are often peripheral populations that reach their northern and western range limits in Canada's boreal forest (Kershaw et al. 2001). There are many compelling reasons to conserve species at their range edges, including conservation of novel genes (Hunter & Hutchinson 1994). Although range edge populations may have reduced genetic diversity relative to central ones, they also more often show greater differentiation (Eckert et al. 2008), potentially representing adaptations which could be advantageous in a changing climate. More so, local conservation follows a land ethic (Leopold 1949) and underscores the importance of collective effort in maintaining biodiversity across our landscapes. Therefore, species considered rare within a given jurisdiction are of both ecological and social significance.

An important consideration in the conservation of rare species is their reasons for being rare in the first place, which are often poorly understood (Kunin & Gaston 1997). Attributes such as small geographic ranges, small population sizes, high habitat-specificity, or life-history traits, such as low fecundity and poor dispersal, cause rarity, but can also act to create feedback loops that further promote rarity (Kunin & Gaston 1997; Bevill & Louda 1999). Rare species are more prone to extirpation through stochastic processes and are therefore of greater conservation concern than common ones (Kunin & Gaston 1997; Bevill & Louda 1999; Murray et al. 2002). Loss of individuals or populations can reduce genetic diversity, connectivity between meta-populations, and reduce the overall range and abundance of species, all of which may increase extirpation rates. Because rare species are already vulnerable to population losses, further loss attributed to land use change may be detrimental to their persistence (Stehlik et al. 2007). In the boreal forest of Alberta, rapid land use change resulting from oil and gas extraction has caused loss, alteration, and fragmentation of habitats (Rooney et al. 2012; Dabros et al. 2018). Because this development is ongoing, it is important to accurately gauge the current conservation status, distribution, and abundance of rare species which occupy this region. Currently, the rate of loss of vascular plant populations to oil and gas development is unknown. Obtaining widespread, quality pre-disturbance information or baseline data around rare species is challenging. Where these data are available, it is possible to track population losses and determine relationships between land use change and extirpation (Stehlik et al. 2007; Dolan et al. 2011), among other applications, such as species-distribution modeling and determining landscape patterns in diversity and rarity.

Historically, survey effort within the oil and gas region for vascular plants has been low, due in part to large areas that are inaccessible via road and landcover types which are difficult to traverse. Survey effort has increased in recent decades due to increased access from linear features (e.g., roads, seismic lines), and through mandatory vascular plant surveys conducted as part of pre-disturbance assessments (PDA's) for oil and gas development (Alberta Energy Regulator 2014). These surveys are undertaken prior to development and play an important role in plant conservation and status within the oil sands region. They first provide valuable spatial data on rare plant species occurrence, when they are made public, through institutions such as the Alberta Conservation Information Management System (ACIMS), and second, allow oil and gas companies to practice active or passive mitigation for rare species within lands leased to them for extraction, potentially reducing local losses to development. In order to serve both purposes effectively, PDA surveys must be consistent, accurate, and well-reported. A key component of this consistency is ensuring that species will be detected where they occur through appropriate survey effort (Garrard et al. 2014). However, while survey guidelines in other areas are explicit in advising on survey practices to address detectability of plants (State of New South Wales 2016), guidelines in Alberta are not explicit in suggested expended effort or specifying survey methodology (Alberta Native Plant Council 2012). This introduces issues of imperfect detection in survey data and may diminish the application of mitigation measures where species are overlooked in areas slated for development.

As with other taxa, detection of plants in surveys is imperfect (MacKenzie et al. 2005; Chen et al. 2013; Morrison 2016). Given that plants are static during survey, it has been suggested that imperfect detection is best modeled by approaches distinct from those used in animal surveys (Garrard et al. 2008; Chen et al. 2009). Factors shown to influence plant species detectability include observer experience, plot size, abundance, phenology, habitat attributes, and morphology (Garrard et al. 2008; Chen et al. 2009; Alexander et al. 2012; McCarthy et al. 2013; Morrison 2016). Of these, abundance and scale are likely the greatest determinants of success. This may be especially problematic for rare species which consistently occur in small, scattered populations (Alexander et al. 2012; McCarthy et al. 2013; Garrard et al. 2014). Species morphology is also a source of bias in imperfect detection, where cryptic species (defined in this thesis as species whose morphology blends with the surrounding environment and that of other similar-formed species), even those which are common, are less likely to be detected. Monitoring programs and rare plant surveys may be especially vulnerable to detection-related bias, as in both cases high detection is necessary to ensure favorable conservation outcomes (Moore et al. 2011; Garrard et al. 2014). In order to determine what is rare and to conserve species effectively, we must first detect them reliably. False absences of populations facing threats may lead to local extirpation of populations through failures to avoid or mitigate the damages from future developments.

Of actions currently employed to maintain rare plant populations in the oil sands area, mitigative translocation, the movement of plant material or animals facing destruction from development (Germano et al. 2015), is used infrequently but consistently. Translocation has an extensive record in scientific research (Godefroid et al. 2011; Germano et al. 2015), and movement of plants in particular is a distinctly human practice found throughout history (Grayson 2001; Richardson et al. 2011). Broadly speaking, results from the literature indicate a mixed success in plant re-introduction and augmentation projects (Fahselt 2007; Godefroid et al. 2011; Drayton & Primack 2012). Mitigative translocation may differ from these types of translocations in project scope, planning, monitoring, and thus, efficacy, but these factors have not been meaningfully evaluated. A widely recognized failing of non-mitigative translocation projects is poor recipient site selection, presumably caused by a lack of understanding of species ecology and environmental tolerances, yet mitigative translocations occur on project timelines which reduce the opportunity to consider these factors (Godefroid et al. 2011; Germano et al. 2015). Gaps in knowledge around rare plant surveys and mitigation practices in the oil sands area can lead to overlooked species, local extirpations, and ineffective use of resources.

As oil extraction continues to expand in the province and development continues to overlap with rare species, guidance and information is essential to ensure the maintenance of biodiversity. Future development of guidelines and best practices for surveying and mitigation needs to be informed by work such as what is presented here. Specifically, my thesis addresses the topics of detectability, population loss, and mitigative translocation for rare vascular plants in the oil sands area of Alberta as four ‘data’ chapters that are described briefly below. Each of these chapters is written in manuscript format with an introduction, methods, results, and discussion. Literature cited for all sections of this thesis are provided at the end of the thesis. Scientific nomenclature follows that of Packer, 1983 (Flora of Alberta), or that of the Flora of North America for those species not listed in the Flora of Alberta.

Chapter 1 – "Investigating detection success: lessons from trials using decoy rare plants" focuses on detection of plants given observer experience, plot size (scale), abundance and arrangement, and species appearance. Here I used an experimental approach to estimating plant detection using field trials within controlled arenas. The goal of these trials was to quantify the relationships between detection and these variables using volunteer observers and pre-planted 'decoy' plants.

Chapter 2 – "Detectability of species of *Carex* varies with abundance, morphology, and site complexity" investigates graminoid detection in boreal landcover types using *Carex* as a model group. The goal of this work was to determine what site- and survey-attributes related to overlooking of graminoids, and what influence the gross morphology of *Carex* species had on detection time, failures, and delays.

Chapter 3 – "Proximity to disturbance does not increase extirpation risk of vascular plants in Alberta's oil sands region" discusses the findings of a revisitation study of historical rare vascular plant populations. The goal of this work was to determine the extent of population loss of historic populations (extirpation) in this region, which is presently unknown, and to understand the relationship between historical population status (persistence or extirpation) and surrounding amount of oil and gas footprint.

Chapter 4 – "Early success of mitigative translocation for rare peatland species" examines the results of a three year mitigative translocation experiment for two boreal peatland species where I examined the survival, growth, and flowering of *Sarracenia purpurea* and *Carex oligosperma*. The goal of this experiment was to determine if this practice is suitable for peatland species and if so under what conditions, given that few translocations are reported publicly from the oil sands area.

Chapter 2: Investigating detection success: lessons from trials using decoy rare plants

2.1 Introduction

Surveys conducted by ecologists generate data used in ecological applications. Observations made at the species level provide data used in conservation and management decisions, taxonomic studies, predictive modeling, and other areas of scientific interest; thus, accurate assessment of presence or absence is essential. Biased survey data, i.e., where detection errors are non-random, can severely undermine our ability to conserve, predict, and understand biodiversity on our landscapes (MacKenzie et al. 2002; Chen et al. 2013; Garrard et al. 2014; Lahoz-Monfort et al. 2014). Widely accepted sampling methodologies and statistical approaches reflect the reality of imperfect detection in faunal survey data (MacKenzie et al. 2002; Royle et al. 2005; Lele et al. 2012). Failing to detect a species when it is present (i.e., recording a false absence) is likely equally pervasive in studies of plants (Chen et al. 2013; Morrison 2016), though plant ecologists are among the least likely to consider imperfect detection in statistical analysis (Kellner & Swihart 2014).

Inaccurate or biased plant survey data affects our knowledge of species richness, distribution, demography, rarity, and conservation status (Kéry & Gregg 2003; MacKenzie et al. 2005; Royle et al. 2005; Archaux et al. 2009; Alexander et al. 2012). Bias can arise when observers consistently record false absences for species occurring at low local abundance or with cryptic (defined here as species whose morphology blends with the surrounding environment and that of other similar-formed species) morphology (Alexander et al. 2009; McCarthy et al. 2013). This can result in survey data that only accurately represent abundant, large, or distinct species, causing underestimates of species richness and abundance. Poor monitoring and conservation outcomes may result when detection is not considered in estimating population size and demography from counts of individuals (i.e., life-stage detection bias) (Kéry & Gregg 2003; Alexander et al. 2009). Kéry and Gregg (2003) demonstrated how reduced detection of less obvious individuals in a stable population could result in an erroneous estimated decline of 8%. Costly efforts to eradicate invasive orange hawkweed (*Hieracium aurantiacum* L.) could be undermined when small patches of non-flowering basal rosettes of the species are overlooked (Moore et al. 2011). Large-scale disturbance from resource extraction highlights another potential adverse outcome of false absences, the loss of opportunity for mitigation or conservation and adverse consequences for regional plant species diversity (Garrard et al. 2014). Consideration of imperfect detection in survey planning can be improved by understanding which factors most relate to successful detection across species and environments.

Local abundance is likely the major determinant of the successful detection of plant species (Moore et al. 2011; Alexander et al. 2012; McCarthy et al. 2013). This is logical as the rate of encounter between observers and plants will scale with abundance. Other factors related to imperfect detection include phenology or life-state, morphology, habitat attributes, survey conditions, and the observer (Kéry & Gregg 2003; Chen et al. 2009; Moore et al. 2011; Alexander et al. 2012; Garrard et al. 2013; McCarthy et al. 2013; Ng & Driscoll 2014). Observer effects are well documented in plant surveys (Ahrends et al. 2011; Morrison 2016). Most studies that focused on imperfect detection, or pseudo-turnover, demonstrated an observer effect; results for the effect of previous experience were variable (positive: Ng and Driscoll 2014; Garrard et al. 2014 (negative: Moore et al. 2011; Burg et al. 2015; Morrison and Young 2016)) and others did not consider experience as an explanatory variable (Archaux et al. 2007; Zhang et al. 2014; Bornand et al. 2014). Two studies did not demonstrate an observer effect (Kéry & Gregg 2003;

Chen et al. 2009). Despite these variable results, expert botanists are still believed to be advantageous in reducing the risk of false-absences, and experience is often a condition of employment, particularly in surveys for species of concern (i.e., listed or tracked species at national or sub-national levels). Understanding how species traits, survey attributes, and the observer interact to increase or reduce detectability for plants can help inform design and analysis of survey data and improve plant species conservation.

To address questions of imperfect detection in forested environments, I conducted two decoy field trials in the manner of Moore et al. (2011). Populations of species of interest (decoys) that were not currently growing in the area were planted prior to surveys, permitting manipulation of survey attributes and thus determination of their influence on detection. Controlled trials where the true abundance and location of targets is known have been used in other search-related research, such as spotlight searches for wooden mammal decoys (Sunde & Jessen 2013) and trials using translocated lizards fitted with transmitters (Henke 1998). Detection trials such as these are potentially limited by creating search environments that may not mimic field conditions; however, they permit the manipulation of variables of interest in ways that are unfeasible in uncontrolled surveys and provide excellent learning opportunities.

I employed two trials to examine detection success as a function of plot size, observer experience, abundance and arrangement (clumped or diffuse) of target species, species characteristics, and observer movement paths. Two decoy species were used in each trial, one of which had a more distinct appearance. I hypothesized that observer experience would be positively correlated with detection success; plot area would be negatively correlated with detection when target species abundance was held constant but detection rate in small plots (<100 m^2) would be relatively similar. I expected abundance and arrangement would both affect detection, as clumps are more likely to be readily detected than single individuals, particularly in less showy species, but potential encounters increase when individuals are diffuse.

My research aims and hypothesized outcomes have important implications for survey guidelines and best practices in Alberta, Canada. While ensuring high detection of plants may often require > 1 survey (Moore et al. 2014), logistical constraints in this region often limit effort to a single survey within a growing season. In the case of environmental impact assessments targeting rare species, available resources and short timelines often result in surveys conducted by one or more botanists over a narrow timeframe. In this jurisdiction, guidelines are limited and do not advise on suggested survey effort (Alberta Native Plant Council 2012). Therefore, maximizing detection within a single survey could make the highest impact and greatest contribution to the development and refinement of best practices for surveys, such as setting minimum survey effort requirements (Garrard et al. 2008). I believe that imperfect detection is pervasive in plant surveys and that management of species of concern will be made more effective by incorporating imperfect detection into study design and analysis.

2.2 Methods

2.2.1 Study site and decoy planting methods

Experimental trials took place at the University of Alberta's Woodbend Forest, 20 km west of Edmonton, Alberta, Canada (53.3º, -113.7º). The climate is continental with warm summers (average temperature of ~15º C in the summer months) and average summer precipitation of ~300 mm. Upland forest across this 64 hectare property is predominantly dry to mesic mixed-wood with an overstory of spruce (*Picea* A. Dietr.), aspen or poplar (*Populus* L.) and pine (*Pinus* L.) with moderate shrub cover, mainly *Corylus cornuta* Marsh. While plots differed slightly in tree and shrub density, I considered them to have been effectively similar in structure. Trials ran in the latter half of August to early September in 2015 and 2016.

In both trials I established square survey plots using wooden stakes and rope to deter observers from leaving the plots. Decoy plants were planted at randomly determined locations within the plots, where effort was made to reduce disruption during planting. I watered and checked individuals regularly over both trials and replaced any damaged specimens. I used two target species in each year, *Symphyotrichum lanceolatum* Willdenow and *Viola pedatifida* G. Don (Trial One), and *Allium cernuum* Roth (Alliaceae) and *Petunia* sp. Juss. ‘Red Velour’ (Trial Two) (Appendix 1, Figure A1.1). All species were short-statured (<50 cm in height). In my first trial, I selected two species with different vegetative appearance; no individuals were in flower at the time of the survey. I considered *V. pedatifida* as visually distinct among species at the site given its deeply palmatifid, glaucous leaves. In contrast, individuals of *S. lanceolatum* looked very similar to other *Symphyotrichum* species and *Galium boreale* L., blending well with the surrounding vegetation (cryptic). In Trial Two I selected flowering or fruiting individuals of two distinctly different species. Individuals of *Petunia* sp. were in full bloom with showy, deep red flowers on otherwise short, sprawling plants. This species was selected to represent an extreme in flower showiness. *Allium cernuum* bears a pale, persistent umbel on a long slender scape, although this inflorescence is relatively large in comparison to small-flowered boreal plants, it tends to blend with the environment (cryptic). All individuals of *A. cernuum* had set seed in the characteristic umbel at the time of the trial.

Volunteer observers were recruited through email and word of mouth. In Trial One I targeted individuals with varying seasons of vascular plant survey experience and who had or had not completed field surveys for plants in the summer months preceding the trials. In northern climates, a survey field season is considered approximately 40-60 days. In Trial Two, I recruited individuals who had experience conducting field surveys, but did not require that these observers be experienced with vascular plants (e.g., I accepted individuals with experience surveying amphibians or bryophytes). Immediately prior to beginning their surveys, all observers were shown example specimens of the two decoy species and were able to revisit those specimens throughout the day. I told observers that neither, one, or both species might be present within plots and asked them to record the presence and time of detection, but not abundance, of any target species they encountered. I instructed all observers to survey plots until they felt they had adequately surveyed the area, starting from a fixed corner and using a meandering search pattern. Observers were not asked to make full species inventories, thus upon finding one target species they continued to look for the other, and in the event they encountered both species within a plot, they would terminate the survey. In this sense the searches mimicked field scenarios where observers search for the presence of a short list of target species, such as rare or invasive taxa.

2.2.2 Effects of observer experience and plot size (Trial One)

In Trial One, I focused on manipulating plot size and determining the influence of observer experience. Species abundance was maintained at one individual/species/plot across the following five square plot sizes: 1 m^2, 10 m^2, 100 m^2, 1000 m^2, and 2500 m^2 with three replicate plots per size ($n = 15$). I estimated horizontal cover around each plant using a 2 m range pole, where the number of 10 cm increments > 25% obscured by vegetation was recorded (Griffith & Youtie 1988). Sixteen observers were recruited and categorized as: 1) Expert with >5 seasons of plant survey experience ($n = 4$), 2) Intermediate with 2-3 seasons of general plant survey experience and had completed surveys within the preceding 4 months ($n = 8$), and 3) Intermediate with >2 seasons of experience who had not completed a survey within the last 4 months (i.e., that field season) ($n = 4$). Group 2 (intermediate botanist) aligns with provincial recommendations for taxonomic experience for individuals completing rare plant surveys (Alberta Native Plant Council 2012).

I asked participants to complete surveys in one replicate of each plot size (a requested minimum of 5 plots) and to complete additional plots if they were so inclined. Observers completed 4 to 8 (most often 5) surveys each for a total of 83 surveys; in each of these the observers searched for both target species. The order in which plot sizes were completed and which replicate plot of a given size was surveyed were randomized for each individual, although complete randomization was forgone at the end of the trial to ensure all plots had been surveyed by at least one observer from each experience category. I recorded the order in which surveys were completed by an individual as a continuous variable to account for improved or reduced detection of plants over the day (e.g., improved search image increasing success or observer fatigue reducing success).

2.2.3 Effects of abundance and arrangement (Trial Two)

In the second trial, I maintained a constant plot size of 1000 m^2 (the 4th largest size from the 2015 trial, 33 x 33 m) and recruited 13 observers who had a background in field surveys for target species. I did not require that individuals had previously surveyed for vascular plants specifically, but recorded the number of seasons of vascular plant survey each individual had (i.e., an observer with experience surveying for bryophytes scored a “0”). I manipulated abundance within plots (1, 5, and 10 individuals) and arrangement (clumped or diffuse) of two target species (*A. cernuum* and *Petunia* sp.) across 15 plots using the design illustrated in Figure 2.1. To achieve the desired well-spaced arrangement of individuals for the “diffuse” arrangement, I chose random locations with the restriction that individuals would be planted at least 2 m apart. Individuals were planted together at each randomized location to form the clumped arrangement (of 5 and 10); for *A. cernuum* this resulted in an area of ~ 10 x 10 cm, for *Petunia* sp. the clumps covered an area of ~ 50 x 50 cm. The 13 observers surveyed 3-5 plots each, resulting in 53 surveys where observers searched for both target species. I again recorded the order in which observers completed plot surveys to account for improvement or reduction in detection with increasing surveys completed by an individual. In addition, I asked participants to wear Columbus V990 GPS data loggers (Victory Technology Co., Ltd.) during surveys to generate location data suitable for analyzing observer movements, i.e., proportion of plot searched, speed, and tortuosity.

2.2.4 Statistical approach

Time-to-event (survival) analysis considers the time at which an event (detection) occurs, as well as censored observations, i.e., timed surveys that did not result in detection (right-censored). In traditional survival analysis the influence of covariates upon the likelihood of an event occurring over time can be determined using Cox models (Cox & Oakes 1984). The Cox model framework assumes that a given event will inevitably occur at some time, censored observations therefore represent observation periods that were shorter than the time necessary for the event to occur. This assumption fails in most ecological applications, as the organism of interest may not be present; therefore, Cox models have been weighted by modeled occupancy (Bischof et al. 2014), or more complex Bayesian models have been applied to account for detection, given occupancy (Garrard et al. 2008). In these trials occupancy is known and thus I meet the assumption that all decoy plants would be detected at some survey time.

I first visualized the relationship between detection and plot size (Trial One) and abundance-arrangement levels (Trial Two) using Kaplan-Meier curves, which estimate the cumulative probability of events (Kaplan & Meier 1958). Next, I determined the influence of explanatory variables on the probability of detection over survey time using mixed effect Cox models. I built a single full model for each trial using all explanatory variables and two random effects to account for repeated measures by observers on replicate plots (plot identity) and on observers across plots (observer identity). I first applied this approach using all observations (species identity was included as an explanatory variable), and for each species within a trial separately if species was determined to be an important predictor variable. All analyses were completed in R (Version 3.4.3) (R Core Team, 2016) using the packages 'survival' (version 2.38) (Therneau 2015a) and 'coxme' (version 2.2-5) (Therneau 2015b).

2.3 Results

2.3.1 The influence of observer experience and plot size on detectability (Trial One)

Detection of both species declined rapidly with increasing plot size, falling from 94% in 1 m^2 plots to less than 50% in plots >100 m^2 (10 x 10 meters). Effort, expressed as total survey time divided by plot area (minutes/m^2), declined with increasing plot size, as did detection success (Table 2.1). Kaplan-Meier curves are shown in Figure 2.2, where *V. pedatifida* had higher overall success and faster detection in small plots; less than half the time was required to achieve the same detection in plots of 1 and 10 m^2 for *V. pedatifida* than for *S. lanceolatum* but the accumulation of detection events for both species was similar in larger plots. Censored observations occurred across a range of survey times in plots larger than 100 m^2, differences in survey times were as great as 2 hours (Figure 2.2).

For both trials I considered models of each species separately because species was a significant explanatory variable in a full model built with all observations (Appendix 1, Tables A1.1 and A1.4). Effect sizes for parameters considered for each species in each trial are visualized in Figures 2.3 and 2.4; see Appendix 1, Tables A1.2, A1.3 and A1.5, A1.6 for parameter estimates and p-values. In Trial One, plot size was the most important variable explaining the detection probability of both *V. pedatifida* and *S. lanceolatum*, and was the only significant predictor (*p* <0.001) for detection of *S. lanceolatum* (Figure 2.3a, Table A2). For *S. lanceolatum*, a one unit increase in plot area decreased the detection rate by 0.06 times. There was weak evidence that plant height positively affected the detection of *S. lanceolatum*; the confidence interval for this parameter did not include zero but it was not a significant predictor (Figure 2.3a). For detection of this species, random effect parameters indicate greater variation between individual observers than between replicate plots (SD = 1.25 and 0.02, respectively). Improved model fit over the null model was supported (χ^2 = 88.16, df = 9, *p* <0.001). For *V. pedatifida*, the confidence intervals of plot size, survey order (the order in which plots were surveyed by an observer), and experience category 2 did not include zero, though only survey order and plot size were significant predictors (Figure 2.3b), model fit over null χ^2 = 91.59, df = 9, *p* <0.001. In particular, survey order (exp β = 1.38, SE = 0.11, *p* = 0.002) had a positive relationship with detection probability, where observers were more likely to detect species with an increasing number of surveys, presumably due to improved search image.. There was support for an effect of observer experience for *V. pedatifida*; observers belonging to category 2 (intermediate with recent experience) had a negative influence on detection probability as compared to those in category 1 (experts), but this did not extend to observers in category 3 (intermediate without recent experience). Standard deviation of both random effects was low, SD = 0.02 for both parameters.

2.3.2 The influence of abundance and arrangement on detection success (Trial Two)

Total detection success differed substantially between the showy *Petunia* sp. (94 – 100%) and less distinct *A. cernuum* (0 – 70%) in plots of 1000 m^2 (Table 2.2). *Petunia* sp. demonstrated near perfect detection with little variation among experimental treatments; it was missed on two occasions, both in plots containing only a single individual. Effort (minutes/m^2) expended by observers was relatively consistent between plots, although detection was very rapid in one five-diffuse replicate for *A. cernuum* (Table 2.2). Observers always found *Petunia* prior to finding *A. cernuum*, thus total effort for the plot largely represents time spent searching for *A. cernuum* (Table 2.2). The accumulation of detection events for *Petunia* was notably faster than for *A. cernuum* in all abundance and arrangement combinations, and diffusely arranged *A. cernuum* plants were detected more frequently and rapidly than the same number arranged in clumps (Figure 2.5).

For *Petunia* sp. the variables abundance, arrangement, and survey order were significant predictors (p <0.001, 0.016, and 0.021, respectively) (Figure 2.4a, Table A2.5). Improved model fit over the null was supported ($\chi^2 = 20.86$, df = 6, $p = 0.002$). Abundance and survey order had a positive relationship with detection, where a unit increase in abundance increased the detection rate by 6.5 times. Diffusely arranged individuals resulted in a two-fold increase in the detection rate as compared to clumps. The random effects of observer and plot identity had standard deviations of 0.4 and 0.3, respectively. *Allium cernuum* model parameters indicated that a unit increase in abundance increased the detection rate by 21 times. Although not significant predictors, arrangement and survey order had confidence intervals that did not include zero (Figure 2.4b). Diffusely arranged individuals of *A. cernuum* were twice as likely to be detected as those in clumps. Improved model fit was supported over the null ($\chi^2 = 14.45$, df = 6, $p =$

0.025). The random effect observer id had minimal variation (SD = 0.02), but variation between replicate plots was higher (SD = 1.12). Observers in this trial had quite variable backgrounds (plant surveys within Alberta, Canada, and internationally) and number of seasons of survey experience (range = 0 – 14, median = 3), but again observer experience had no influence (Figure 2.4).

Movement metrics (speed, tortuosity) generated from data loggers were not included in the main analysis due to instances of collection failure; descriptions and analysis using these metrics are given in Appendix 2. I observed uniform speeds across individuals ($\overline{x}$ = 0.14 meters/second, SE = 0.001) and speed and tortuosity ($\overline{x}$ = 0.0006, SE = 0.00006) had no significant influence on detection using mixed-effect Cox models. I observed a trend of *A. cernuum* detections occurring most frequently when <30% of the plot had been searched (Appendix 2).

2.4 Discussion

These detection trials have clearly demonstrated that the probability of detecting cryptic understory species at low density (i.e., 1 individual/1000 m^2) is very low (<35%); this provides further evidence that imperfect detection in plants is pervasive and can be severe (Kéry & Gregg 2003; Chen et al. 2009; Alexander et al. 2012). I observed complete failure at detecting single individuals of *A. cernuum* in 1000 m^2 plots, as compared to 35% success for *V. pedatifida* and 23% for *S. lanceolatum*, this was despite the fact that the latter two species were in a vegetative condition at the time of survey. While phenology is important in detection (Kéry and Gregg 2003; Alexander et al. 2012), it is likely that many species would go undetected when rare within plots and when not bearing showy flowers. Detection of the showier species in both trials was often more rapid (requiring less survey effort) than for the cryptic species, although this trend diminished with increasing plot area in Trial One. In addition, the largest plot size used here (0.25 ha) is smaller than the area typically covered for environmental assessment surveys, suggesting that field surveys may be even less successful than my findings. The importance of survey conditions, observer effects, and plant abundance and plot area (density) varied among species in results of time-to-event analysis.

In both trials I manipulated the density of the target species by maintaining plant abundance while increasing plot area (Trial One), or increasing plant abundance over plots of the same size (Trial Two). The positive relationship between density and detection is a product of increased encounter rate between the observer and a greater number of individuals and is well demonstrated in other work (Moore et al. 2011; Alexander et al. 2012; McCarthy et al. 2013). Manipulating species arrangement in Trial Two indicated that clumps of 5 and 10 individuals were more easily detected than single individuals of *A. cernuum*, presumably due to increased visibility of clustered individuals. However, clumps of 5 and 10 had similar total detection success and rates for both species used in the trial, suggesting that this visual advantage may not scale with clump size. This is supported by the findings of Moore *et al*. (2011), where clumps of 3 and 5 individuals of their target species were detected at an equal rate. These results suggest that surveys targeting species that are known to occur at high local densities or in caespitose growth forms could be successful with less effort than those targeting species that consistently occur at low densities (e.g., some members of the Orchidaceae). Further, including measures of effort along with reported absences of cryptic species will improve the understanding of how species abundance and distribution influence detection, and aid in setting effort requirements for environmental impact assessment surveys (Garrard et al. 2008).

The influence of increased plot area likely affects detection beyond the change in target species density. In Trial One, increasing plot area in a forested site included greater cover of large plants (trees and shrubs), meaning greater physical and visual obstruction for observers, although horizontal cover was not an important predictor variable. In large plots observers must employ more search techniques and maintain a spatial awareness that is different from surveying small plots. Several observers expressed feeling overwhelmed by the physical search area in large plots. I observed a wide range in expended search times in large plots, suggesting that observers differ in their motivations and decision-making for the ‘stop time’ when given the opportunity to survey for unlimited time. Future trials may benefit from more intensive de-briefing interviews with participants post-survey to qualitatively describe such decision making processes, which was not considered in this work. I hypothesize that the effect of search area includes, but extends beyond, the effect of reduced density of the target species to include factors such as observer fatigue, which has been shown to influence aerial detection of mammals (Habib et al. 2012; Ransom 2012). These results suggest that using time-unlimited surveys can only go so far in solving the issue of imperfect detection. Future experiments should consider the search techniques used by observers, perhaps through requesting the use of specific strategies such as dividing the total area into smaller, searchable sections (McCaffrey et al. 2014), or using different plot configurations (e.g., belt transects vs. large, square plots) to search equivalent area. Further, it would be advantageous to explore how effort (minutes/m^2) varies across plots of varying shapes and sizes, where belt transects may, in theory, aid in focusing an observers attention on a smaller search area and thus expended effort may remain higher over square plots of equivalent size.

Considering all four target species and the two trials, the limited relationship between observer experience and detection was surprising, but supported by findings by Moore et al. (2011). While I recorded variation among observers, demonstrated by wide confidence intervals in Kaplan-Meier curves for all species, experience level was not an important explanatory variable; but see *V. pedatifida*. I speculate that the observed variation was instead due to inherent differences, i.e., personality traits or mental attitude. Studies suggest that observer experience or training should be related to detection success (Garrard et al. 2008, 2013; Morrison 2016), and surveys completed by expert botanists are often regarded as more reliable. It is possible that trial conditions negated the ability of experts to outperform less experienced surveyors. For example, many botanists use their knowledge of microsite associations when searching for target species with which they are familiar, but my study did not allow for such associations due to random planting locations. There is further an important consideration that these surveys were highly targeted, to two species in each trial, and it is probable that differences in skill level (experience) of observers may play a greater role in full inventory surveys. Therefore, these results suggest that intermediate and expert observers can achieve similar results in targeted surveys, particularly when they have the opportunity to examine live plants prior to initiating surveys. I found weak, but consistent, evidence of observer improvement over an increasing number of surveys, presumably due to improved search image after early successful detections or observers becoming more familiar with ‘filtering out’ non-target species in the study area. Observers completed surveys over a single day; it is possible that observer improvement over a season could be an important consideration when planning surveys and that observer learning may improve survey results over time. Finally, I observed minimal trampling in survey plots over time and do not suspect trampling improved or reduced plant detection, but note this can be an important consideration in decoy trials.

Although the most rapid and consistent detections occurred in plots of 1 and 10 m^2 in Trial One and in observations of *Petunia* sp. in Trial Two, in both years the majority of plots had at least one successful observation when considering all surveys. It should be noted that *Petunia* sp. detection was exceptional in comparison to the other three species. This could be attributed to the fact that this decoy species is quite distinct in comparison to natural understory boreal species and was also generally familiar to observers; this highlights the importance of careful selection of decoy species traits in trials such as these. In Trial One, only one plot replicate of 2500 m^2 was perfectly undetected for each of *V. pedatifida* and *S. lanceolatum*. In Trial Two, excluding the complete failure in plots with only one individual, only one replicate went without a successful detection of *A. cernuum*. Thus, teams of 3 - 4 observers completing repeat observations in plots could compensate for low individual detection probabilities on a per-site basis and I encourage this survey approach where feasible, as has been suggested in other work (Alexander et al. 2012). I also note that data resulting from such repeat plant surveys, including those collected here, are suitable for estimating detectability using mark-recapture methods when the time of a detection event is either not collected or is not of interest (MacKenzie et al. 2005; Alexander et al. 2012).

In closing, I encourage future decoy trials such as those conducted here and by Moore et al. (2011) to examine relationships between species and survey variables against detection probability in a controlled field setting. As understanding of the pervasiveness and severity of imperfect detection in vascular plants grows, my hope is that future work will more reliably incorporate techniques to address this issue (Kellner & Swihart 2014). I suggest that improvements to field surveys for low abundance species can be achieved through careful consideration of allocation of survey effort, for example, increasing the number of observers within plots and limiting plot size where accurate detection of single individuals is critical (e.g., monitoring applications). Although using small plots may require a trade-off in total area searched, my results suggest that false absences are more likely when species are in low abundance and the survey area is large. I suggest that future research in the field of imperfect detection in plants explore how survey techniques such as using a series of small plots to search a large area in lieu of large plots could improve detection of cryptic species. Considering observer movement using GPS, as was done here, may reveal interesting trends in how observers search plots and when they are most likely to make successful detections (Appendix 2). Recording survey effort through person hours and area covered will not only improve confidence in reported absences, but will add to our understanding of how required effort may fluctuate with species characters and phenological state. Collecting time-to-event data where possible to determine rates of imperfect detection and relevant covariates influencing success in different environments is encouraged.

Table 1.1. Effort, average detection time, and success across plot sizes for both target species, *Symphyotrichum lanceolatum* and *Viola pedatifida*, used in Trial One, $n = 83$.

Plot area	Species	Av. time to detection (min)	No. detections	No. surveys	% success	Median effort (min/m^2)	± SE	Range
1	*S. lanceolatum*	1.1	15	17	88	1.60	± 0.09	1.0 – 3.0
	V. pedatifida	0.7	17	17	100			
10	*S. lanceolatum*	2.4	10	17	59	0.34	± 0.03	0.1 – 0.8
	V. pedatifida	1.1	14	17	82			
100	*S. lanceolatum*	5.8	7	17	41	0.17	± 0.03	0.04 – 0.57
	V. pedatifida	6.9	8	17	47			
1000	*S. lanceolatum*	30.9	4	17	24	0.05	± 0.004	0.02 – 0.1
	V. pedatifida	28.0	6	17	35			
2500	*S. lanceolatum*	60.5	3	15	20	0.04	± 0.003	0.01 – 0.07
	V. pedatifida	56.4	2	15	13			

Table 2.2. Effort, average detection time, and success across plot sizes for both target species, *Petunia* sp. and *Allium cernuum*, used in Trial Two, $n = 53$. "D" denotes diffuse arrangements; "C" denotes clumped arrangements.

Abundance and arrangement	Species	Av. time to detection (min)	No. detections	No. surveys	% success	Median effort (min/m^2)	± SE	Range
1D	*Petunia* sp.	15.3	9	11	82	0.06	± 0.01	0.02 – 0.16
	A. cernuum	-	0	11	0			
5C	*Petunia* sp.	13.7	11	11	100	0.04	± 0.01	0.01 – 0.09
	A. cernuum	34.7	4	11	36			
5D	*Petunia* sp.	8.8	10	10	100	0.02	± 0.01	0.002 – 0.07
	A. cernuum	10.8	7	10	70			
10C	*Petunia* sp.	14.6	10	10	100	0.03	± 0.003	0.02 – 0.05
	A. cernuum	25.7	3	10	30			
10D	*Petunia* sp.	5.7	11	11	100	0.04	± 0.01	0.01 – 0.08
	A. cernuum	23.8	6	11	55			

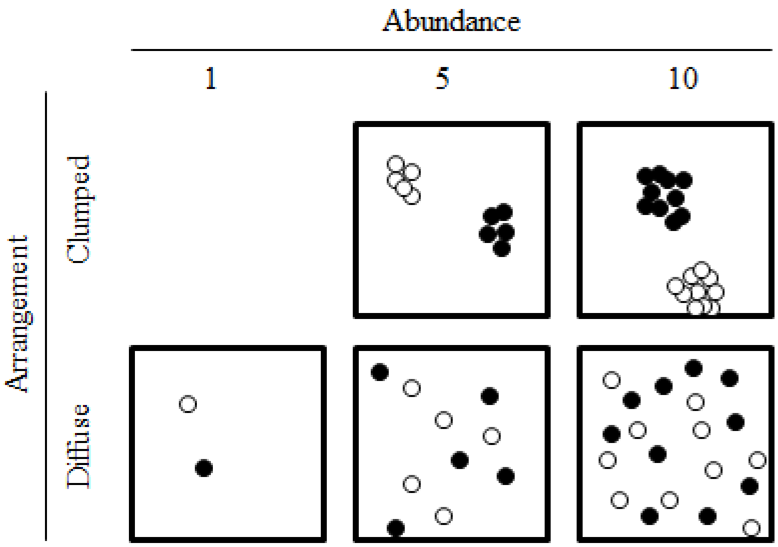

Figure 2.1. Example configuration of decoy plants within square experimental survey plots in Trial Two. Closed circles indicate *Petunia* sp., open circles *Allium cernuum*. This design was replicated three times for a total of 15 experimental plots.

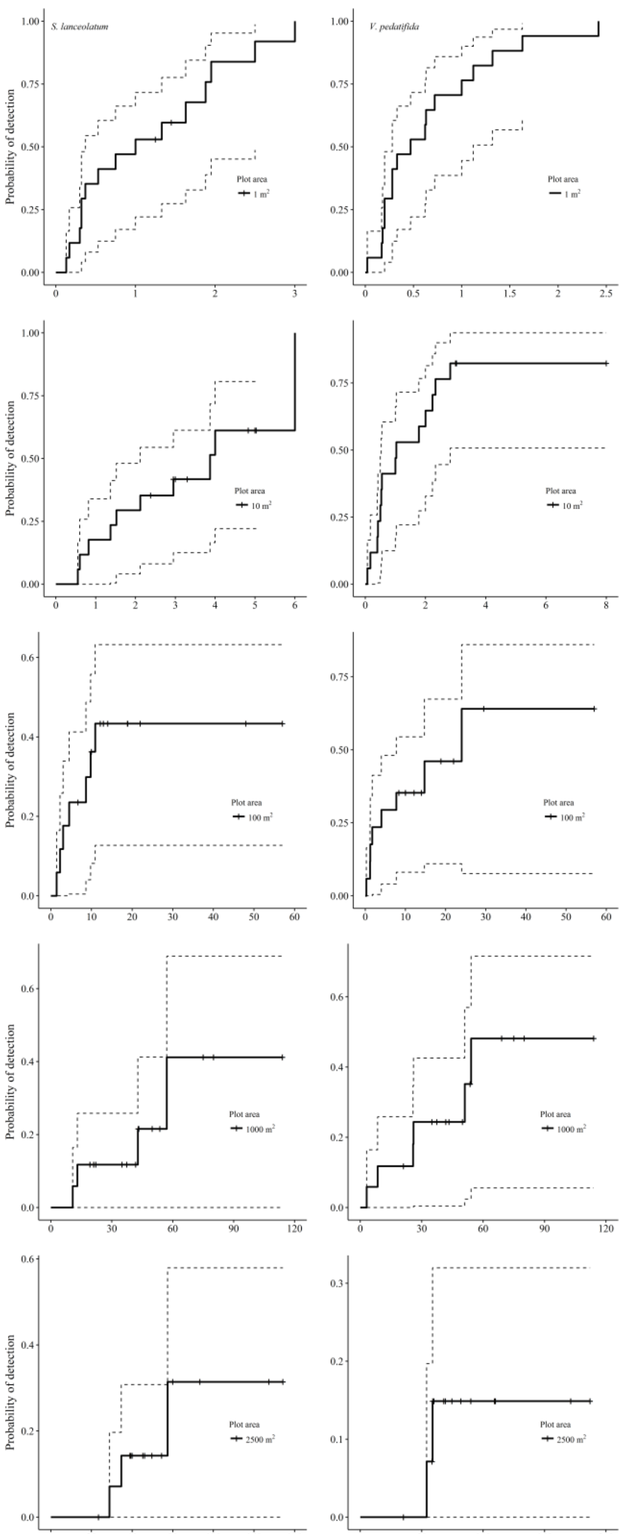

Figure 2.2. Kaplan-Meier curves showing the accumulation of detection events over survey time in Trial One for *Symphyotrichum lanceolatum* and *Viola pedatifida*. Censored observations are shown as vertical ticks along the KM curve, dotted lines show 95% confidence intervals, $n = 83$.

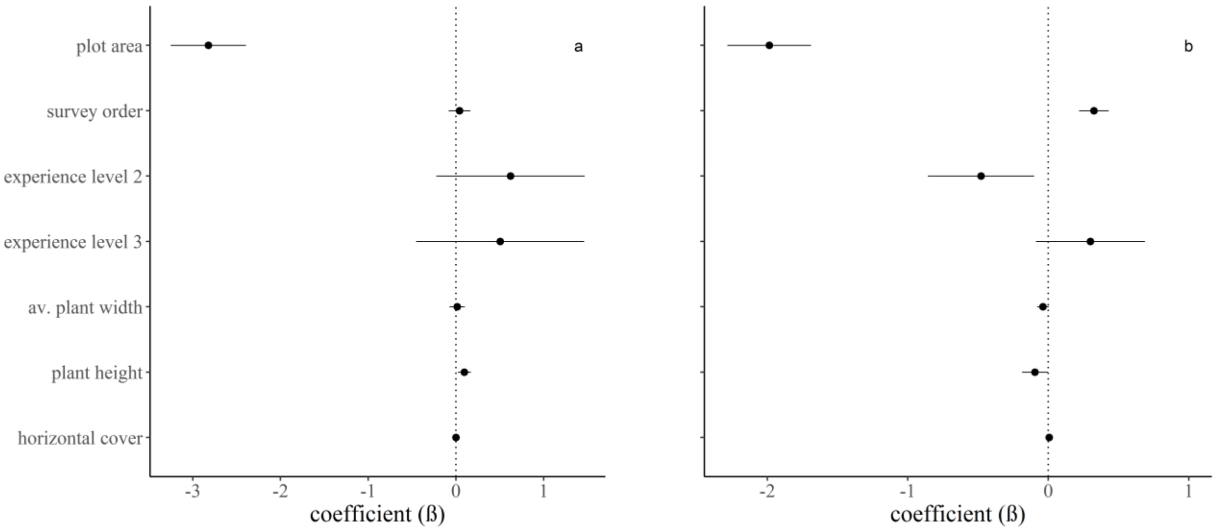

Figure 2.3. Parameter estimates and associated confidence intervals for full models of a) *Symphyotrichum lanceolatum* and b) *Viola pedatifida*. A random effect of observer identity and plot identity were used in both models, see text for standard deviation values. The variable plot area was log transformed in both models

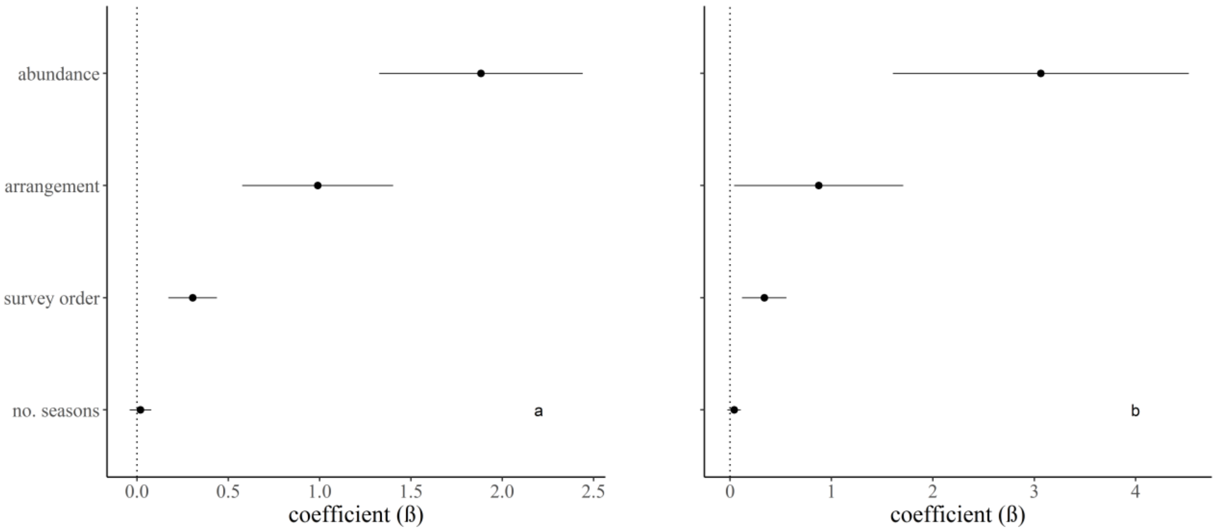

Figure 2.4. Parameter estimates and associated confidence intervals for full models of a) *Petunia* sp. and b) *Allium cernuum*. Random effects of observer identity and plot identity were used in both models, see text for standard deviation values. The variable abundance was log transformed in both models.

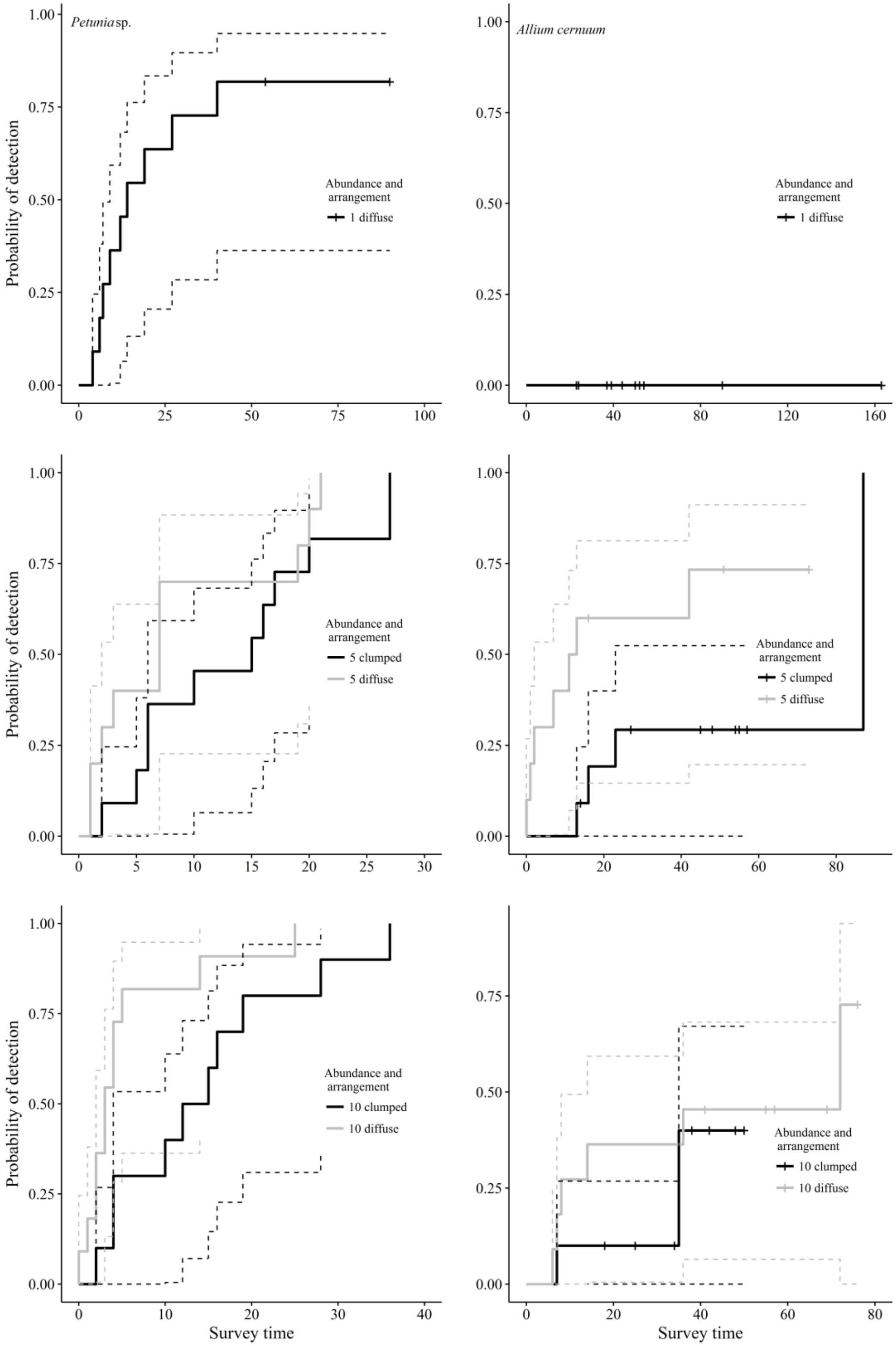

Figure 2.5. Kaplan-Meier curves of detection events over time of *Petunia* sp. and *Allium cernuum* in Trial Two. Censored observations are shown as vertical ticks along the KM curve, dotted lines show 95% confidence intervals, *n* = 53.

Chapter 3: Detectability of species of *Carex* varies with abundance, morphology, and site complexity

3.1 Introduction

Understanding the world around us requires observation and collection of data, yet we know observers to be imperfect in detecting events or patterns where they occur. Issues related to imperfect human observers are common across a number of disciplines including medicine, manufacturing, and ecology (Poulton 1972; Bruno et al. 2015; Lavers et al. 2016). In ecological applications, understanding species' occurrence, abundance, and population dynamics requires that species are consistently detected in surveys (MacKenzie et al. 2005), yet imperfect detection exists and must therefore be accounted for. In practice, those who study static events or species with low mobility (e.g., ground-dwelling arthropods and plants) rarely account for imperfect detection in analyses (Kellner & Swihart 2014). Despite the fact that plants are static once established, previous studies make clear that the assumption of plants being perfectly detectable is often invalid (Chen et al. 2013).

Research on imperfect detection in vascular plants has regularly estimated detection probabilities <0.5 and even as low as 0.09, suggesting that imperfect detection must be considered in survey planning and data analysis (Chen et al. 2009; Moore et al. 2011; Clarke et al. 2012; Ng & Driscoll 2014)**.** Species which are locally abundant or are conspicuous, such as when flowering, have higher detection rates, with abundance generally being the greatest determinant of detection success (Royle & Nichols 2003; Vittoz & Guisan 2007; McCarthy et al. 2013). However, most species are rare (Whittaker 1965; Kunin & Gaston 1997), and species at low abundance in their environment are less likely to be detected in surveys with large plot sizes (Moore et al. 2011; Dennett et al. 2018). The majority of vascular plant surveys are therefore likely to overlook at least some species. Developing strategies to reduce these errors is important to ensure observed data are accurate.

Imperfect detection can generally be categorized as resulting from observer-specific differences (experience, identity, etc.) (Alexander et al. 2009; Bornand et al. 2014; Morrison 2016), species-specific differences such as morphology, abundance, flowering state, and size of individuals (Scott & Hallam 2002; Kéry & Gregg 2003; Clarke et al. 2012; Garrard et al. 2013), and site-specific differences (survey protocol, density of vegetation, management history, etc.) (Garrard et al. 2008; Alexander et al. 2009; Clarke et al. 2012; Burg et al. 2015). When detection probability scales with any one of these characteristics or combinations thereof, survey data become biased, limiting their usefulness. In ecology, the extent to which species- and site-specific attributes act to exacerbate observer errors is of increasing interest (MacKenzie et al. 2005).

Given that imperfect detection differs across species and sites, studies separating these effects are useful for understanding bias and addressing it in survey designs (Kéry & Gregg 2003; Alexander et al. 2009; Garrard et al. 2013). Comparisons of detectability by life stage, flower colour, or other characters help inform field surveys targeting species that share similar traits (Kéry & Gregg 2003; Chen et al. 2013; Garrard et al. 2013). Further, understanding how detection of similar species varies during surveys is important since many rare taxa share traits with their more common congeners, yet their accurate detection is of high conservation value (Kunin & Gaston 1997; Garrard et al. 2014). Evidence suggests that graminoids, here referring to the families Poaceae, Cyperaceae, and Juncaceae (colloquially grasses, sedges, and rushes, respectively) may be more poorly detected than other vascular plants, yet they are an important source of diversity in many systems. Previous work found that graminoids were overlooked more often than other vascular plants, with moderately high mis-identification rates (Archaux et al. 2009). In a Swiss study, grasses had the lowest detection probability among trees, shrubs, and forbs (Chen et al. 2013). Scott and Hallam (2002) reported pseudoturnover (the change in species composition in an area due to differences in observer's species lists) values of 21, 22, and 16% for grasses, sedges, and rushes, respectively, where sedges were the most overlooked group of ground-layer plants. In an analysis of pseudoturnover in alpine environments, Burg et al. (2015) reported 21% of the 62 species most often overlooked by observers were graminoids, although they also demonstrated high detectability of two abundant *Poa* L. (Poaceae) species. Collectively, this suggests that graminoids may be consistently under-detected, likely for reasons related to their morphology and taxonomy.

While these families include remarkable amounts of morphological diversity, differences typically occur at a small scale. Graminoids tend to be thin in profile, with low foliar cover and limited contrast between their vegetative and reproductive parts, thereby representing, as stated by Chen et al. (2013), an "elusive *gestalt*". Further, these families are often intimidating to early-career botanists due to challenges in accumulating the taxonomic knowledge to recognize differences between species, although this may in turn act to dissuade botanists from focusing on them in the field. To address detection of this challenging taxonomic group, I undertook a study of graminoid detection in Canada's boreal forest. First, I estimated the pseudoturnover of graminoids in comparison to other life forms to determine the extent to which these plants are subject to overlooking errors between observers. Next, I used the genus *Carex* L. (Cyperaceae) as a model group to examine variation in detection given species morphology and abundance. My aim in selecting *Carex* was to take advantage of the broad variation in form and niche observed in this genus. Indeed, *Carex* is one of the most diverse angiosperm genera with ~2000 species (Govaerts & Simpson 2002) distributed globally across the earth's terrestrial surface, excluding the Antarctic (Ball & Reznicek 2002). In the study region of northeastern Alberta, Canada, *Carex* occur across the full range of nutrient and moisture conditions and vary significantly in size and form.

Working taxonomic knowledge of *Carex* is slow to accumulate, where some groups remain challenging to identify in the field even for relatively experienced botanists (e.g., section *Ovales*). The process of *Carex* identification for most beginner to intermediate botanists is likely best reflected by a combination of a natural method (grouping similar-appearing *Carex* into coherent search images) followed by the use of keys to confirm the species level identification (Ellis 2011). Therefore, I considered *Carex* detectability using morphological groups, comparing time to detection for these groups and determining factors that influence detection success. Specifically, I sought to answer three questions:

1. How large are detection errors for graminoids in contrast to other life forms in boreal forests, and what survey conditions minimize these errors?
2. To what extent do morphological traits make some groups of *Carex* more readily and accurately detected?
3. Which species- and site-specific traits have the greatest influence on detection failures at sites and delays of detection within sites for *Carex*?

3.2 Methods

3.2.1 Study area and inventory transects

Surveys took place in northeastern Alberta, Canada. Fifty belt transects 100 m in length and 2 m in width (200 m^2) were established in a region spanning from 45 km northeast of Fort McKay

(57°32′23″ N, -111°14′48″ E) to Lac la Biche (54°59′9″ N, 112°0′6″ W), a north-south distance of ~ 300 km. The large geographic span of sites was intentional to reduce local improvement in detection rates due to familiarity with a single area, see Appendix 3, Table A3.1 for site coordinates. I selected sites by generating random points within 2 km of roads in ArcMap 10.2 (ESRI 2015) and then used Google Earth (www.earth.google.com) imagery to stratify by landcover type, or using local knowledge to span the range of possible conditions. Specifically, transects were established in bog, graminoid-, shrub-, and treed-fens, as well as conifer and mixed-wood uplands. Deciduous-dominated upland forests were the only major landcover type avoided due to low representation of *Carex*. Human disturbance was also avoided, but natural disturbances, such as beaver activity and recent wildfires, were included as they represent important habitat for many boreal species. Transect surveys were completed from 2 Jul to 13 Aug 2015. Two observers (JD and JT) generated a complete species inventory of all encountered vascular plants for each transect in 30-minute independent surveys (effort of 0.15 minutes/m^2 per observer), recording the time of detection for each species and the transect segment (1 – 10, 10 × 2 m increments) where it was first detected. Transects were established singly or in pairs <300 m apart, but always set in contrasting habitats when paired. No continuous habitat patch was surveyed twice (e.g., a large fen complex would contain only one transect).

Once each transect was surveyed by both observers, it was then walked a third time for the collection of structural attribute data. At the overall scale of the transect I recorded the weather and assigned a landcover class. Within each 20 m^2 segment (10 segments/transect) I assigned a Domin cover-abundance class (Mueller-Dombois & Ellenberg 1974) for: (1) life forms (short tree (<2 m), tall shrub (>50 cm), short shrub (<50 cm), forb, non-*Carex* graminoid, all *Carex*, moss, and lichen), and (2) each *Carex* species found within that segment. Horizontal cover estimates were obtained using a 2 m cover (Robel) pole (Griffith & Youtie 1988) with banded 10 cm increments at the midpoint of each transect segment. Finally, morphological characters were measured on three representative individuals of each *Carex* species at each transect. Measurements included plant height, leaf width, length and width of the inflorescence or peduncled spike, and the number of pistillate spikes.

While collecting these data, observers had the opportunity to note missed *Carex* observations. *Carex* species missed by one or both observers were noted, as well as detection delays (e.g., both observers found a species in transect segment 8, but it was first present in 2). Thus, this additional time (30 – 60 minutes) acted as a third survey focused on *Carex*, and I made the assumption that all *Carex* species had been captured by the end of this time. At approximately one third of sites cover and site attribute data were collected as a pair, which had the benefit of allowing ongoing calibration of percent cover estimates. When these “third survey” data were collected independently, observers used radios to communicate the presence and location of encountered *Carex* species after both independent surveys had been completed. Both observers had similar field experience at the time of this study (multiple years of vascular plant survey experience and previous *Carex*-specific training). Our familiarity with *Carex* and the flora of the study area was characterized as being upper-level intermediate.

3.2.2 Analysis methods

Pseudoturnover by growth form

I used pseudoturnover, the change in species composition in an area due to differences in observer’s species lists (Nilsson & Nilsson 1985), to estimate the magnitude of overlooking errors for all species encountered (total) and four broad growth forms; tree, shrub, forb, and graminoid. Percent pseudoturnover was calculated using the equation by Nilsson and Nilsson

(1985), $((S_a + S_b) / (S_{aa} + S_{bb})) \times 100$ where S_a and S_b are the number of unique species detected by each observer not detected by the other, and S_{aa} and S_{bb} are the total number of species recorded by each observer respectively. To determine the influence of site-specific variables I related total and graminoid pseudoturnover to attributes of the search environment with generalized linear models (glm's) using the package 'lme4' (Bates et al. 2015) in R version 3.4.3 (R Foundation for Statistical Computing, Vienna, AT). Predictor variables were reported to the transect level by summing species richness and averaging horizontal cover, total vegetative cover, and cover by growth form from the segment level. Statistical models represented related, but distinct hypotheses of site conditions predicted to best explain variation in observer species lists and were evaluated using AIC model selection (Burnham & Anderson 2002). All variables were standardized to their mean to allow for direct comparison of effect sizes between predictors.

Morphological groups and time of detection

All encountered *Carex* species were categorized into six groups based on their gross morphology, grouping by height and general appearance of the inflorescence. I used individual measures collected in the field to visualize the suitability of the grouping scheme using a Principal Components Analysis (PCA) with the package 'vegan' (version 2.4-4) in R. These groups were: aggregated (medium to tall species with spikes aggregated into a head), peduncled (medium height with pistillate spikes on peduncles), sessile (tall, robust aquatics with spikes mainly attached directly to the culm [lower pistillate spikes can be peduncled in some species]), sessile-remote (medium height with well-spaced, sessile pistillate spikes), single-spike (short, unispicate species), and small-aggregated (short to medium height with few, small spikes mostly clustered to a head). Examples of a *Carex* species typical to each group are shown in Appendix 3, Figure A3.1.

To determine if two repeat surveys and additional site time were adequate to detect species within morphological groups, I estimated detection probability for each group using data from all three survey periods, i.e., including those observations made during the ‘third survey’, using Program PRESENCE (version 2.12.12). For each group, if any species assigned to the group was missed by one or both observers, it was coded as an absence; thus, estimates of detection are likely under-estimates for some species within groups.

Finally, considering the time at which observers encounter plants provides meaningful information on the amount of survey effort necessary to achieve high detection success. To explore the influence of morphology on time to detection, I compared detection events across time (cumulative probability) for these groups using Kaplan-Meier (KM) curves (Kaplan & Meier 1958) using the packages ‘survival’ (version 2.38) and ‘survminer’ (version 0.4.1) in R.

Detection success, failure, and delays

As described above, I recorded two types of imperfect detection, detection failures where one or both observers overlooked a species, and detection delays, where a species was detected after its first location along the transect. Given that each case has implications for effective surveying, I considered these two types separately by examining success (detection at first opportunity) vs. detection failures (n = 374), and success vs. detection delays (n = 417). Specifically, the influence of species- and site-specific variables were examined for each question (set of data) using mixed-effect logistic regression (where transect identity was included as a random effect) with exploratory hypotheses ranked for model support using AIC evaluation with the package ‘lme4’ in R. In all cases I included information gained from one observer (e.g., species presence recorded by JT and not by JD), as well as observations collected during additional time spent on each transect. I did not encounter incidents of false presence (misidentification) in these surveys and therefore did not address misidentification in analysis.

3.3 Results

3.3.1 Experimental detection transects and pseudoturnover

I recorded 260 vascular plant species across all 50 transects, 36 (13.8%) of which were species of *Carex.* Species richness of *Carex* ranged from 1 – 10 per transect with a mean of 4.5 (SE = 0.4), while total species richness ranged from 15 – 81 species per transect with a mean of 34.8 (SE = 2.2). Detection failures of *Carex* were low with detection success for both observers at 92%. Out of 227 unique instances of *Carex* species presence, JD missed 22 (9.7 % detection failure), while JT only 15 (6.6 %). In eight instances both individuals missed a *Carex* sp. during their individual survey, but detected it later when spending additional time estimating cover and assessing site attributes. Although detection failures were low, delayed detections within a site (transect segments) were more common with JD recording 39 instances (17.2%) of detection delay, and JT recording 41 instances (18.1%).

When considering all species of vascular plants, average pseudoturnover rate was 11.6% (range: 2.2 – 23.3, SE = 0.7) with little variation in pseudoturnover across life forms. Graminoids were marginally lower than others at 10.0% (range: 0 – 33.3, SE = 1.3), while forbs were marginally higher than others at 12.7% (range: 0 – 38.5, SE = 1.4) (Figure 3.1). Model ranking suggested weak support for site complexity (horizontal and total vegetation cover) in explaining total pseudoturnover. However, horizontal cover (visual obstruction measured using a Robel pole) was positively related to pseudoturnover, while total vegetation cover (the sum of estimated cover values for growth forms, not including horizontal cover, per transect) was negatively associated to pseudoturnover. When considering only graminoids, pseudoturnover was best explained with a positive relationship with forb and short shrub cover, representing low, leafy plants which would obscure graminoids with their foliage, and forb cover having almost twice the effect of short shrub cover (Tables 1 and 2).

3.3.2 Morphological groups and time of detection

Morphological groups had meaningful differences across field measured traits when visualized with a PCA (Figure 3.2) (Appendix 3, Table A3.2 for summary of field measurements), with obvious separation across the axes representing plant height and leaf width. Detection probability, estimated using data from both repeat surveys and additional site time, varied across morphological groups. Sessile *Carex* were the most reliably detected, while small-aggregated *Carex* were most poorly detected, where even three survey periods were not sufficient to achieve >95% detectability for half of morphological groups (Table 3.3). Kaplan-Meier curves of detection rates indicated that aggregated and sessile *Carex* accumulated detections rapidly, while small-aggregated and single-spike *Carex* required nearly the full survey time to reach the same detection probability achieved within 10 minutes for sessile *Carex* (Figure 3.3). The two observers in this study did not significantly differ in their total detection success, nor in their accumulation of detection events across survey time, again compared using Kaplan-Meier curves (Figure 3.4).

3.3.3 Detection success, failure, and delay

Ten candidate models explaining variation in success vs. detection failure ($n = 374$) and vs. delays ($n = 417$) were compared based on species- and site-specific predictors (Table 3.4). All models included *Carex* species abundance (here, mid-point of Domin cover class for each species in each transect segment) as abundance is a well-known determinant of detection success. Detection success over failures was best explained by total vegetation cover in the segment where the species was first present, its morphology, and the segment cover (abundance) of that species (Table 3.4). Specifically, *Carex* species cover had the greatest, positive influence on detection success (β = 14.5, SE = 3.56). Total vegetation cover was inversely related to detection success (β = - 0.40, SE = 0.20), and the morphological groups aggregated and sessile-remote were most similar to sessile *Carex,* while single-spike, peduncled, and small-aggregated had reduced detection in comparison to this group (Table 3.5). Models containing morphological traits were not supported in explaining detection success over delays. The best supported model instead contained variables representing physical obstructions at sites (i.e., short tree and tall shrub cover). Short tree cover was positively associated with detection success over delays (but note a large SEs, Table 3.5), and tall shrub cover was inversely related to success. Increasing cover of each *Carex* species in the block they were first present was again the most important parameter in explaining immediate detection (β = 1.28, SE = 0.43), although this parameter was not as influential as in models explaining detection success over failure (Table 3.5).

3.4 Discussion

High *Carex* detection (> 90%) can be achieved for some species in field surveys using narrow (2 m wide) belt transects with abundance, morphology, and site-specific variables affecting detection success. Overlooking errors between two similar, well-trained observers were low relative to published estimates with graminoids consistently having the lowest pseudoturnover among four major life forms (Leps & Hadincova 1992; Scott & Hallam 2002; Morrison 2016), although this is potentially a product of the focus of this experiment on graminoids themselves, and thus may represent a best-case scenario among field surveys. Total pseudoturnover was weakly supported by site complexity, while graminoid pseudoturnover was best explained by low cover. Time to detection can be related to species morphology, and the use of morphological groupings may be relevant to future survey planning or techniques for adjusting detectability in analyses (Lele et al. 2012; Sólymos et al. 2012). Here, two repeat surveys and additional site time focused on *Carex* was still not sufficient to achieve detection probabilities of >95% for half of the morphological groups considered, suggesting that surveys which employ less effort than what was used here (i.e., 0.15 minutes/m^2 per observer per independent survey + extra site time amounting to an additional 0.15 – 0.3 minutes/m^2) are likely well below this threshold. Species- and site- specific factors differ in their effect on an observer's overall detection in a plot versus delaying detection within a plot along segmented belt transects. Abundance, morphology, and surrounding vegetation cover best explain detection failures in general, while abundance and site structure best explain delays in detection within a plot. These results speak to the importance of considering species- and site-specific attributes in survey planning and data collection.

Low rates of pseudoturnover in boreal forest surveys are encouraging. While forbs were more often overlooked, there was little variation among groups, suggesting no bias in detectability among life forms. From a site-specific perspective, pseudoturnover was higher in sites with more complex vegetation, being negatively related to total vascular plant cover. This may reflect observers being more cautious in detecting species when vegetation cover was high. In contrast, pseudoturnover was positively related to horizontal cover perhaps reflecting increased distractions at sites with a more complex understory. However, these results were only marginally more supported than a null model of constant detectability. Increased graminoid pseudoturnover at sites with higher forb and short-shrub cover is logical given that the larger foliage of these plants can obscure thin-profile graminoids. It is interesting to note that these results differ from previous studies in different environments. For example, pseudoturnover within an 11 person team was best explained by observer-specific variables, with virtually no influence of site-specific factors in open alpine areas (Burg et al. 2015). Detection is also strongly related to abundance (Royle & Nichols 2003), and estimates of pseudoturnover may in part reflect random processes of encounters between observers and very scarce species in survey plots (Dennett et al. 2018). It is possible, considering published examples, that achieving rates of pseudoturnover of less than 10%, or especially 5%, is unlikely given some level of inherent randomness in surveys (Leps & Hadincova 1992; Scott & Hallam 2002; Vittoz & Guisan 2007; Burg et al. 2015; Morrison 2016).

Observer success in recognizing graminoids, and *Carex* specifically, is more closely tied to taxonomic knowledge than for more easily identified vascular plants. In cases where taxonomic certainty is problematic or observers lack experience, there may be significant impediments to estimating graminoid detectability. For example, Garrard et al. (2013) did not include grasses in their detection analysis due to high uncertainty in species identification, which points to where the most serious gaps are. Therefore, I suggest using morphological groups as an approach to understanding detectability for difficult taxa. Gross morphological groupings may more closely mimic the recognition process for *Carex* in this study area, whereby some species are instantaneously recognized (e.g., *C. aquatilis* Wahlenb.), but others are recognized as groups of similar-formed species (e.g., *C. concinna* R. Br., *C. deflexa* Hornem. and *C. peckii* Howe). Here, robust, large species of *Carex* were 1.3-times more likely to be detected than those with a short, small inflorescence and we showed detection bias against short, small inflorescence species, even after additional time spent at sites. Surveys with poor detectability of smaller graminoid species could therefore bias conservation assessments of rare species or local land use decisions, and care must be taken through increased effort or number or observers to ensure adequate detection.

Morphological groups are further useful in characterizing the effects of morphology on detection failures and delays. Abundance was the most important determinant of success over failure, with a small negative influence of total surrounding vegetation cover and species morphology that differs from large, robust *Carex*. In contrast, abundance and site structure (short trees and tall shrubs), but not morphology, affected the probability of delayed observations within a site with abundance having a more moderate effect on detections. Transects with greater cover of short trees and tall shrubs represent physical impediments to human movements within plots that should reduce detections. One key characteristic of boreal forests is that tall shrub cover is common, posing an impediment to observer movement within the forest, especially in shrub-fens and shrub-swamps. It may be the case that tall shrubs increase delays because of a required lapse in visual and mental focus on plants while navigating through dense sites. However, I observed a weak positive effect of higher short tree cover on detection success over that of delays in detection. It is possible that this is a product of environments with greater amounts of short tree cover also having a sparser understory (e.g., regenerating *Pinus banksiana* Lamb. forests), and thus *Carex* contrasted more with their environment and were more readily sighted. Together, these results support the well-known relationship between abundance and detection success (Royle & Nichols 2003; McCarthy et al. 2013), as well as the importance of considering morphology and site structure in the allocation of survey effort (Moore et al. 2011; Garrard et al. 2013; Ng & Driscoll 2014).

It is important to consider survey area and effort per unit area in interpreting these results. These survey plots (belt transects) were small (200 m^2) and effort was high (0.15 minutes/m^2 per observer) relative to other studies estimating observer differences and plant detection (Moore et al. 2011; Zhang et al. 2014), yet I still did not achieve detection probabilities of 95% for half of the morphological groups considered, and detection of small-aggregated *Carex* was much poorer than the other morphological groups at 0.82. The effort used in this study exceeds that of surveys typically used for environmental impact assessments, but helps establish benchmarks of when high detection rates are achievable. Further, my results highlight a benefit of well-trained, similar observers. Previous studies that have used pairs of experienced observers also found no evidence of observer effects, as was documented here (Kéry & Gregg 2003; Chen et al. 2009), but see Bornand et al. (2014) who found meaningful differences between a team of two observers, attributed to different levels of experience. Although I attribute observer success here to similarity in mental attitude and previous training, observer experience has not always related to improved detection in other examples, and it may be that observer effects are less prevalent than anticipated in some field studies. Dennett et al. (2018) found limited to no influence of observer experience in targeted surveys, despite a wide range of number of years of vascular plant experience among 29 individuals. Moore et al. (2011) found variation among a team of 12 observers in targeted surveys but could not attribute this to their previous survey experience, and a consistency experiment using 41 observers failed to detect an effect of observer background with reasonable consistency within the group (Ringvall et al. 2005). However, this latter experiment used very small plot sizes of 0.33 m^2 and 0.01 m^2. Thus, questions of the influence of observer experience are ongoing with my study representing an example of a consistent pair of observers.

Inferring species occupancy and trend from survey data requires accurate detection, yet variability in detection is the norm. I encourage future work focused on detecting graminoids or other challenging taxa (e.g., Asteraceae or Ophioglossaceae). A clear benefit in using *Carex* as a model genus is that many species can occupy a single site, allowing for direct comparison of the effect of morphology of similar plants under the same site conditions. One consideration not made in this study was the inclusion of growth form (i.e., cespitose and rhizomatous). This may further influence *Carex* detection, but varies within some species given environmental conditions and therefore should be measured on a per-site basis, and I encourage future work to consider this trait for *Carex* and other graminoids. This study of graminoid detection in boreal environments found no bias in detection among life-forms, the potential suitability of using morphological groupings for difficult genera, and the importance of biases associated with abundance of the target species, as well as site-specific factors in influencing detection success and delays. I hope that this example encourages future work in forested systems using this and other focal taxa.

Table 3.1. Generalized linear models relating pseudoturnover of all species (top) and graminoids (bottom) to site-specific variables representing different conditions of the search environment (*n* = 50).

Total pseudoturnover			
Hypothesis	**Model**	***K***	**ΔAIC**
Site complexity	horizontal cover + mean transect cover	2	0.00
Null	null	0	2.02
Richness	transect species richness	1	2.12
Low cover	mean forb cover + mean short shrub cover	2	4.37
High cover	mean tall shrub cover + mean short tree cover	2	5.17
Habitat	habitat class	1	11.20
Graminoid pseudoturnover			
Hypothesis	**Model**	***K***	**ΔAIC**
Low cover	mean forb cover + mean short shrub cover	2	0.00
Graminoid prevalence	mean graminoid cover	1	2.58
Null	null	0	3.69
Graminoid richness	graminoid richness	1	5.62
Habitat	habitat class	1	4.10
High cover	mean tall shrub cover + mean short tree cover	2	6.86
Site complexity	horizontal cover + mean transect cover	2	7.01

Table 3.2. Parameters of the best supported generalized linear models relating pseudoturnover of all species (top) and graminoids (bottom) (*n* = 50). All variables were standardized to their mean prior to their inclusion in candidate models so that their values can be compared among each other for their effect size.

Total pseudoturnover	**β**	**SE**
horizontal cover	1.25	0.81
total mean transect cover	-1.89	0.79
Graminoid pseudoturnover	**β**	**SE**
forb cover	3.10	1.24
short shrub cover	1.52	1.24

Table 3.3. Detection probabilities (± SE) estimated using both repeat surveys (30 minutes each) and additional 'third survey' time (30 to 60 minutes) for 50 experimental transects in Program PRESENCE across six morphological groupings of *Carex* ($n = 417$).

Morphological group	**Detection probability ± SE**
Sessile	0.99 ± 0.01
Aggregated	0.99 ± 0.01
Sessile-remote	0.98 ± 0.02
Peduncled	0.91 ± 0.03
Single-spike	0.87 ± 0.05
Small-aggregated	0.82 ± 0.04

Table 3.4. Mixed-effect logistic regression models relating detection success vs. failure (n = 374) (top) or delay (n = 417) (bottom) to species- and site-specific variables. All variables were standardized to their mean prior to inclusion in candidate models. A random effect of transect identity was used in all candidate models. The morphological group 'sessile' was withheld as the reference for the variable "morphological group".

Detection failure

Hypothesis	Models	*K*	ΔAIC
Vegetation cover and *Carex* morphology	total segment cover + morphological group + species segment cover	4	0.00
Morphological group	morphological group + species segment cover	3	2.10
Low cover	short shrub segment cover + forb segment cover + species segment cover	4	9.44
Richness	transect species richness + species segment cover	3	15.44
High cover	short tree segment cover + tall shrub segment cover + species segment cover	4	16.42
Species cover	species segment cover	2	17.30
Site complexity	total segment cover + horizontal cover + species segment cover	4	17.58
Observer	observer identity + species segment cover	3	17.70
Carex prevalence	total *Carex segment* cover + species segment cover	3	18.24
Null	1 + transect random effect	1	57.94

Detection delay

Hypothesis	Models	*K*	ΔAIC
High cover	short tree segment cover + tall shrub segment cover + species segment cover	4	0.00
Site complexity	total segment cover + horizontal cover + species segment cover	4	0.20
Species cover	species segment cover	2	7.54
Carex prevalence	total *Carex segment* cover + species segment cover	3	8.00
Richness	transect species richness + species segment cover	3	8.73
Observer	observer identity + species segment cover	3	9.53
Low cover	short shrub segment cover + forb segment cover + species segment cover	4	11.38
Morphological group	morphological group + species segment cover	3	14.25
Vegetation cover and *Carex* morphology	total segment cover + morphological group + species segment cover	4	15.33
Null	1 + transect random effect	1	22.22

Table 3.5. Parameters of the most supported mixed-effect logistic regression models of detection success vs. failure (*n* = 374) (top) or delay (*n* = 417) (bottom) of *Carex*. A random effect of transect was used in all candidate models. The morphological group ‘sessile’ (large, robust aquatics) was withheld as the reference for the variable “morphological group”.

Detection failure	**β**	**SE**
species segment cover	14.50	3.56
total segment cover	-0.40	0.20
morphological group		
Aggregated	-0.04	1.47
Peduncled	-1.34	1.13
Sessile-remote	0.31	1.28
Single-spike	-2.34	1.18
Small-aggregated	-2.40	1.09
Detection delay	**β**	**SE**
species segment cover	1.27	0.43
short tree segment cover	0.26	0.23
tall shrub segment cover	-0.55	0.21

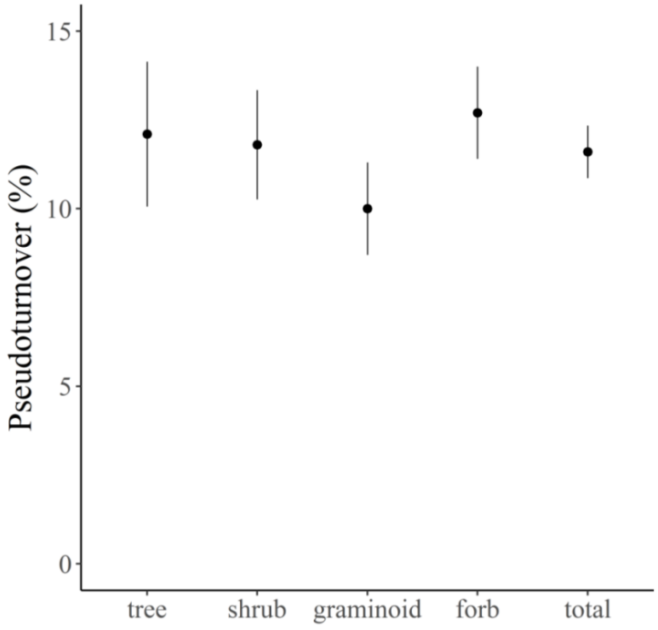

Figure 3.1. Average (and standard error) percent pseudoturnover (the difference in observer's species lists due to overlooking) by life forms for all encountered vascular plant species for 50 transects 100 m in length and 2 m in width.

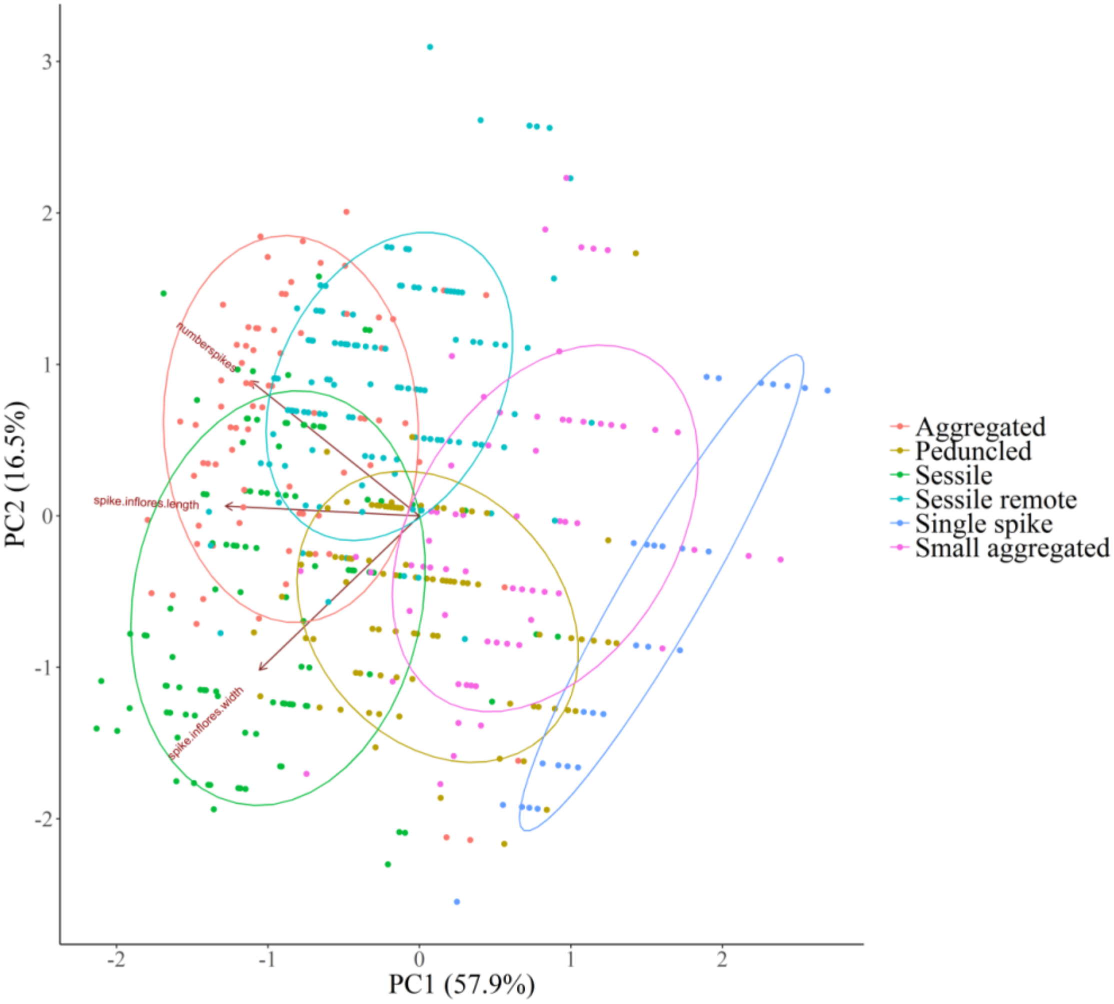

Figure 3.2. Principal Components Analysis (PCA) of six morphological groupings of *Carex* species. Each point represents an individual plant measured in the field (n = 593). Measurements used in the PCA include plant height, leaf width, spike/inflorescence length, width, and number of spikes. Means for each species are summarized in Appendix 3, Table A3.2. Axis 1 is related most to plant height, while Axis 2 is related most to leaf width.

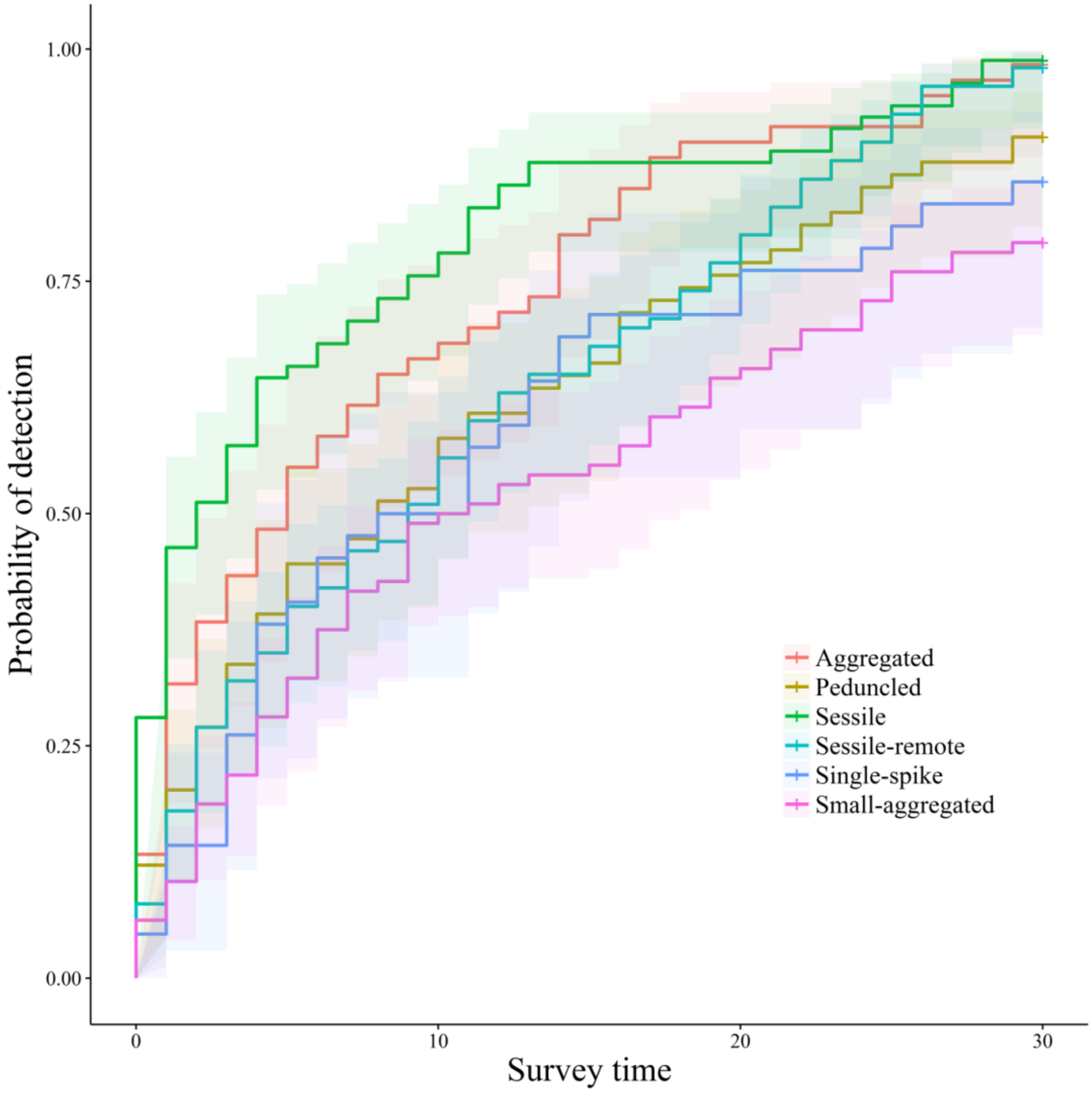

Figure 3.3. Kaplan-Meier curves (where the y-axis represents the estimated probability of an event occurring at a given survey time, up to censoring) representing the accumulation of detection events for six morphological groupings of *Carex* over 30 minutes of survey time over a 200 m^2 belt transect ($n = 454$). Confidence intervals are shown in shaded bands.

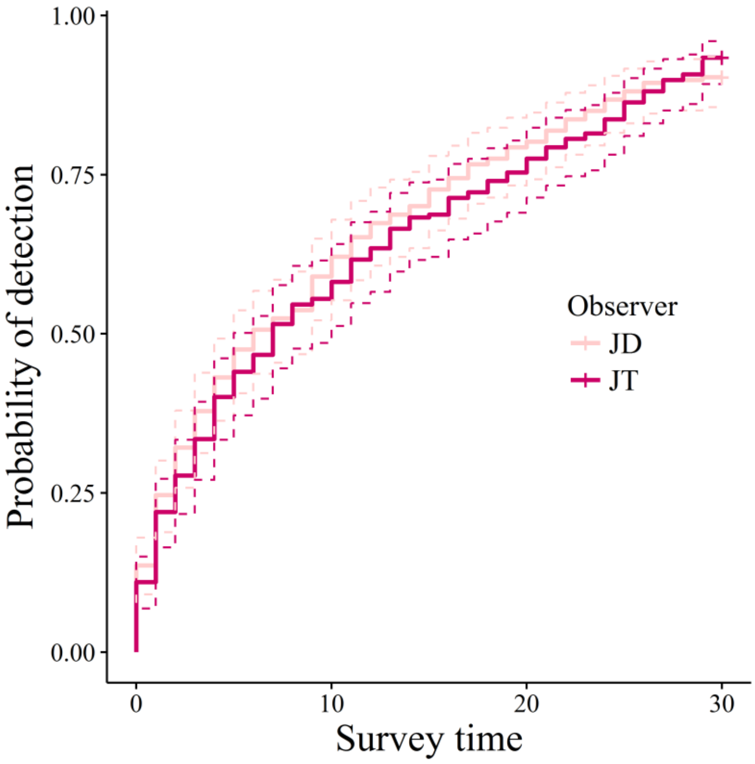

Figure 3.4. Kaplan-Meier curves plotting the accumulation of detection events of all *Carex* across 30 minutes of survey time over a 200 m^2 belt transect (n = 227). Confidence intervals are shown as dashed lines with no differences found between observers.

Chapter 4: Persistence of rare vascular plants in Alberta's Oil Sands Area

4.1 Introduction

Maintaining rare species at provincial and national scales is of significant cultural and ecological value. Creating and meeting conservation and management goals for rare species requires accurate categorizations of species' rarity and conservation status. A number of definitions of rarity exist (Rabinowitz et al. 1986; Kunin & Gaston 1997), as well as methods for ranking conservation status (Master et al. 2012). However, all of these approaches rely on data representing the location and status of populations for each species of interest. Historical records of occurrences, such as those from herbaria or other organizations, are often used to rank the rarity status of species, as well as estimate extinctions, population losses, and range contractions (Pyke & Ehrlich 2010; Master et al. 2012; Nualart et al. 2017). Human activity and changes in land use, such as those resulting in habitat loss and fragmentation, are the primary cause of local extirpation of populations (Cincotta et al. 2000; Stehlik et al. 2007; Godefroid et al. 2014). For instance, a revisitation study for 63 historic populations of a single vascular plant species in Switzerland observed 24% of populations had been lost which was related to increased levels of agriculture and fragmentation (Lienert et al. 2002). The extirpation of these historic populations can bias the conservation ranking of species if those same records are included in ranking schemes. This would make a species appear more prevalent than it really is and thus potentially result in a lower (less at risk) conservation status.

In Alberta, Canada, historical records of rare plant, fungi, and lichen species locations are maintained by the Alberta Conservation Information Management System (ACIMS). Species-level observations are submitted by the public, most often by amateur botanists or those employed by consulting, government, and research agencies. ACIMS assesses these data prior to inclusion within their database and then uses these occurrences to determine the rarity status of species at the sub-national level. ACIMS uses NatureServe methods to assign sub-national ranks (S-ranks) to all native species for which data are available (Master et al. 2012). The rank calculator used in this method includes entry fields for, among others, range extent, area of occupancy, number of occurrences (populations), population size, habitat specificity, and population trends. Although the calculator is comprehensive, many, if not most, species have major data gaps for these attributes. As such, ranking is often based primarily on 2 factors: the range extent as determined by a minimum convex polygon of known populations, and the number of occurrences within this geographic area (Master et al. 2012, L. Allen, pers. comm.). This process determines the conservation status of species from S1-S5, with S1 being exceptionally vulnerable to extirpation (critically imperiled) and S5 being common and widespread (secure). Uncertainty is expressed through combined ranks, e.g., S1S2. Additional ranks indicate when species are unable to be assessed due to extinction, provincial extirpation, lack of taxonomic resolution, or insufficient data.

Population records for rare vascular plants are often collected and submitted to ACIMS by consultants as part of a pre-disturbance assessment (PDA) rare plant survey conducted on oil and gas leases (Alberta Native Plant Council 2012; Alberta Energy Regulator 2014). Oil and gas extraction is a substantial disturbance in Alberta resulting in habitat loss and fragmentation (Rooney et al. 2012; Dabros et al. 2018). Given the nature of PDA surveys, population records available for rare species often fall within proposed development footprints resulting in direct or indirect disturbances to populations of conservation concern. Therefore, these records may be more likely to experience changes in land use, and hence population threats, than those not occurring within proposed developments and identified by the general public, government, or academics. For example, seismic lines, linear features which are characteristic of oil and gas exploration, have been shown to influence growth, diversity, and competition among vascular plants (Dabros et al. 2017), while surface mining operations represent large footprints which remove all surficial soil layers (Rooney et al. 2012). This raises concerns regarding the use of records associated with pre-disturbance assessments in provincial ranking of rarity and conservation status. Inclusion of populations that are more likely to be extirpated by disturbances could first inflate the total number of records for a species, as well as potentially exaggerate the provincial geographic range if disjunct populations are lost to disturbance, potentially resulting in misapplied rarity status for species.

To address these concerns, I measured the amount of human footprint within and surrounding historical rare vascular plant populations for the oil sands region and field visited 62 of these populations during peak flowering periods for each species in northeast Alberta. At each field site, I identified whether historic rare plant populations persisted or not to test whether oil sands developments pose threats to known rare plant populations. More specifically, I asked whether land use activities, record age, or initial reported size of population relate to patterns in rare plant persistence. I predicted that persistence of rare vascular plants will be negatively related to age of record and the amount of surrounding area covered by human footprint (i.e., land which has been modified for either oil and gas production or other anthropogenic activities), while being positively related to historic population size. I further anticipated that historical populations falling within conventional mining leases would be subject to higher levels of footprint then those on in-situ leases.

4.2 Methods

4.2.1 Study area and included datasets

To examine the potential relationship between oil and gas extraction and rare plant population persistence, I focused on the Oil Sands Area (OSA), located in northeastern Alberta (Figure 4.1). The OSA is comprised mostly of boreal forest and covers roughly one fifth of the province (21% or 140,000 km^2) and includes all major oil sands deposits in Alberta and eight natural sub-regions. Oil (bitumen) is extracted via two main methods, conventional surface mining which occurs only where bitumen deposits are close to the surface and in-situ extraction, which typically uses steam assisted gravity drainage (SAGD) or other solvents to extract deeper bitumen deposits via wells. Areas where surfacing mining is possible are limited to only 4,800 km^2 (3.4% of the OSA) surrounding the areas of Fort McMurray and Fort McKay (Figure 4.1). These two extraction methods differ greatly in their footprint, thus I considered areas leased for mining (surface) and in-situ development separately. I obtained publicly available provincial oil sands lease and protected area boundaries from the Government of Alberta (Alberta Environment and Parks, 2018) and classified rare plant populations by lease type (mining vs. commercial in-situ) and protected area status. Lease boundaries were current to 2015, protected areas to 2018.

Locations of rare vascular plant populations were obtained from the publicly available ACIMS database (Alberta Environment and Parks, 2017). This public dataset includes spatially represented populations (polygons) of single species ranked from S1 to S3 (where S1 is critically imperiled and S3 is vulnerable) which are currently ‘tracked’ by ACIMS, and includes 4956 observations of 407 species. Here, I considered only those populations which fell within the Oil Sands Area boundary, resulting in 209 populations of 49 unique species, the most recent records having been submitted in 2015. Populations are represented as polygons in a Geographic

Information System (GIS) whose shape and area relate to either the originally reported population extent, reported uncertainty in the location, or mapped shapefiles submitted directly to ACIMS. Further information includes the date of observation (record age) and polygon area. Populations may be made up of more than one polygon, e.g., clusters of an orchid species in a small local area. For the purposes of this analysis, all sub-polygons of a single record were considered a single population. Thus, the term population as it is used here reflects observations submitted to ACIMS as a single entry. This does not reflect the spatial distribution of individuals within these mapped polygons, or the differences in species abundance between sub-polygons of a single record.

The publicly available Human Footprint Mapping Layer (2014) from the Alberta Biodiversity Monitoring Institute (ABMI) (ABMI 2016) was used to determine the amount and type of footprint present across ACIMS populations. Footprint includes all forms of anthropogenic disturbance, including soft features, such as man-made clearings and trails, as well as industrial features, such as permanent facilities. Within the Oil Sands Area, 132 classifications are used to describe footprint types (e.g., landfill, oil sand mines, transmission line, etc.) and the classification was current to 2014. For the purposes of concisely summarizing disturbance types within the region, I simplified this classification to 15 broader categories (e.g., residential, in-situ structure), and separated those related to oil and gas vs. those related to other activities (Appendix 4, Table A4.1).

Next, I determined the amount and type of disturbance within originally reported population polygons by comparing ACIMS population polygons with the ABMI Human Footprint Mapping Layer in ArcMap version 10.3.1 (ESRI 2015). Concentric ring buffers were created around all populations at radii of 10 m (0-10 m), 100 m (10-100 m), and 1 kilometer (100 m - 1 km), representing adjacent, neighbouring, and distant disturbances. These were again compared with the ABMI Human Footprint Mapping Layer (overlay analysis) to quantify the amount of footprint (later simplified to 15 categories [Appendix 4, Table A4.1] and oil and gas or non-oil and gas) in each buffer class. Given that population polygons were of different sizes (area), I determined the proportion of disturbed area by dividing the total disturbed area by population or buffer area. I compared the proportion of both oil and gas and non-oil and gas footprint between original polygons and all three buffer classes for records on and off lease areas, and between mining and in-situ leases using unpaired Wilcoxon Rank Sum tests.

4.2.2 Assessing status of historic rare plant populations in the field

Field crews visited 62 historic ACIMS populations (sites) representing 26 vascular plant species within the Oil Sands Area between June and August of 2016 (n = 42) and 2017 (n = 20) (Figure 4.2). Sites were stratified based on logistical constraints and selected to encompass a range of landcover and disturbance types, including terrestrial and aquatic areas. In each year a team of two observers with previous vegetation experience were trained to recognize target species prior to survey. Teams visited each population during the period when species were expected to be flowering to increase detectability (Kéry et al. 2006; Garrard et al. 2014). At terrestrial sites (n = 49), surveyors performed time unlimited searches of a circular plot with a radius of 50 m around the center of each originally reported polygon (maximum search area of 7,850 m^2). Field crews only visited sites which had a high degree of spatial accuracy in the ACIMS database, as it is unfeasible to search polygons whose mapped extent is large (e.g., up to kilometers in radius) and whose spatial accuracy is poor. Where a single population, as the term is used here, was represented by sub-polygons, a centroid was created for each sub-polygon and the search protocol was applied to each centroid. If any individuals of the target species were re-located in any of the sub-polygons, that population was considered extant. For those records represented by a single polygon, only a single centroid was created and searched from. When a site had been cleared with major soil disturbance (e.g., a wellpad surface), the cleared area was given a precursory scan and the search radius was established around the edge of the feature (e.g., beginning from the vegetated edge of wellpad). Transect tapes and a handheld GPS were used to ensure that the search radius was adhered to and the total search area was covered. For aquatic open water sites ($n = 14$), a small inflatable boat was used for all surveys with one observer paddling in concentric rings inward from the wetland margin while the other observer searched for the species. Observers recorded the time of detection when target species were encountered, and the total time spent surveying where species were not encountered. Although many historical populations were in close proximity to one another, only 3 were directly overlapping at a single site within a protected area (La Saline Natural Area), thus, in the vast majority of cases observers searched for a single target species.

At each population location I assigned a broad landcover type based on 5 pre-determined categories. These included (1) aquatic (i.e., open water), (2) lowland (fen or bog), (3) upland (a wide category consisting largely of mixedwood stands), (4) riparian (lake or river edges), and (5) highly-altered. A site was classified as highly-altered where the soils had been modified by human activity such that they were no longer in a natural state (e.g., gravel wellpad surfaces, pavement). Disturbance, both natural and anthropogenic, was present across all habitat types with the highly-altered classification only assigned to those sites modified to the extent described. To improve confidence in reported extirpations, four populations reported extirpated in 2016 were re-surveyed in 2017 to confirm absence of the target species. This was additional to the 20 historical populations surveyed in 2017.

4.2.3 Statistical analysis of field data

To assess the effect of human footprint on persistence of rare plant populations surveyed in the field, I considered the proportion of oil and gas or non-oil and gas human footprint within and surrounding originally reported polygons and three concentric ring buffers (0 -10 m, 10 - 100m, and 100 m -1 km) to determine at which scale footprint correlated with plant population persistence. Additional predictor variables expected to influence persistence included record age, most prevalent disturbance type within the original polygon, and landcover type (Table 4.1). Persistence of rare plant populations across all sites was assessed using logistic regression where the detection of rare plants in field visits was represented as a "1" and undetected populations as a "0". Candidate explanatory variables are those described above (Table 4.1) and all continuous variables were log transformed. All analyses were conducted in R version 3.4.3 (R Core Team, 2018) using the package 'lme4' (Bates et al. 2015) with a series of candidate models compared using Akaike Information Criteria (AIC) (Burnham & Anderson 2002). I employed single-variable models to explore which of these candidate variables was best supported in explaining observed persistence of rare plant populations.

Given that detectability of rare plants cannot be assumed to be perfect (MacKenzie et al. 2002; Kéry et al. 2006; Chen et al. 2013), reported rates of persistence are likely under-estimated, although I do not expect bias in detectability among sites based on site covariates tested here. To better understand the difference in effort expended to detect small vs. large populations in the field, I compared survey effort (time) with population size of detected target species using generalized linear models (regression). Both variables were log transformed prior to analysis to normalize variables.

4.3 Results

4.3.1 Relationship between survey effort and population abundance

Surveyors searched the specified area exhaustively at all 62 field sites. At maximum, 21 person hours were spent at one site (a total effort of 0.16 minutes/m^2), although in many cases (n = 31) the target species was detected within five minutes of survey time. Survey times of '0' were applied when the species was located prior to beginning the formal search, i.e., upon arrival. Supporting my expectation, expended survey time was negatively related to population size (r^2 = 0.48, p <0.001). Median total search time per observer when species were encountered was <1 minute (range: 0 – 110, $\bar{x}$ = 13.8, SE = 3.69), but where species were absent median search time was 180 minutes (range: 5 – 630, $\bar{x}$ = 215, SE = 60.07). All populations >50 individuals (n = 22) were detected within three minutes of survey time (median <1, range: 0 – 3, $\bar{x}$ = 0.18, SE = 0.14).

4.3.2 Assessing oil sands footprints in proximity to historic rare plant populations

Approximately half (109, 52%) of regional ACIMS populations fall within areas currently leased to oil and gas companies. Of these on lease populations, 42% (46) occur on leases designated for surface mining extraction of bitumen (mines) and 58% (63) occur on in-situ leases. At present, only 9% (18 populations of 14 species) of rare plant populations currently fall within provincially protected areas. Notably, of 102 in-situ leases within the Oil Sands Area, roughly one-quarter (22%, 23) have a reported rare plant population within their boundary (average lease area of 157 km^2), as compared to nearly three quarters (12) of the 15 mining leases (average area of 172 km^2). Recorded age of occurrences (by the date at which they were first submitted to ACIMS) is relatively recent, with a mean age of 12 years (range = 2 – 86, SE = 0.82).

Anthropogenic footprint is relatively common among the 209 historical plant populations in the oil sands area with just under half of populations (45%, 94) having some amount of oil and gas footprint within the original population's polygon. Of these, 23% (22) have greater than half their originally reported extent overlapped. At adjacent (0 - 10 m), neighbouring (10 - 100 m), and distant (100 m - 1 km) distances, 48% (101), 70% (147), and 95% (199) of populations have some amount of oil and gas footprint respectively. Non-oil and gas footprint (e.g., residential, agriculture) within population polygons was less frequent, with 22% (46) of populations having directly overlapping non-oil and gas footprint. At adjacent, neighbouring, and distant distances, 28%, 42%, and 69% of populations have some amount of non-oil and gas footprint respectively.

The two most common footprint types overlapping populations were soft linear features (e.g., seismic and power lines) and in-situ structures (e.g., wellpads), both mainly associated with in-situ extraction (Table 4.1). Despite the large size of surface mining operations, only four populations were shown to have an oil sands mine as their most prevalent disturbance type within the original polygon (Table 4.1). Results of unpaired Wilcoxon rank sum tests for oil and gas footprint between on and off lease areas suggest significant differences at all scales, where those populations on lease have higher proportions (Figure 4.2a, Table 4.2). Comparing in-situ to mine leases again demonstrated that the proportion of oil and gas footprint was significantly different at all scales, where populations on in-situ leases have a greater proportion of their immediate to distant area disturbed (Figure 4.2b, Table 4.2). Using the same comparison to determine differences in the prevalence of non-oil and gas footprint demonstrates that on and off lease areas have no significant difference at any scale (Figure 4.2c, Table 4.2). Populations on in-situ leases have significantly higher proportions of non-oil and gas footprint at all scales excluding within the original population as compared to those on mine leases (Figure 4.2d, Table 4.2).

4.3.3 Field surveys of population persistence

Rare plant populations were successfully detected at 48 of 62 sites (77%) (Table 4.3). Of the 14 presumed extirpations, eight were located within oil sands leases resulting in estimated on-lease persistence of 79%, versus six located off lease for an estimated persistence of 75%. Estimated persistence for the 26 rare species examined varied among landcover types ranging from open water wetlands to conifer-dominated uplands (Table 4.4). Surprisingly, all populations persisted in aquatic-related habitats (open water wetlands, riparian, and lowland, $n = 22$), while 63% persisted in upland sites ($n = 29$), and 64% persisted in highly-altered sites ($n = 11$) (Table 4.4).

Results of logistic regression with AIC model comparison of nine candidate logistic regression models suggested that disturbance metrics (proportion and most prevalent disturbance type) did not relate to local patterns in persistence (Table 4.5, Figure 4.3). Footprint proportions within buffers were too highly correlated to be included within the same model (Table 4.5). The most supported model contained landcover type, reflecting perfect persistence in wetland and lowland habitats (Table 4.6). Initial population size was reported to ACIMS for only 45 populations and thus was examined separately. Model comparison using initial population size, record age, and their interaction, is reported for this reduced dataset (Table 4.7). Larger initial reported populations had a positive effect on persistence with no evidence of an interaction with record age (Table 4.8, Figure 4.4), where a 10-fold increase in population size resulted in a 2-fold increase in the probability of persistence. Finally, additional revisits in 2017 of four sites recorded as extirpations in 2016 found only one false-absence (failure to detect the target species when it was present), in the case of a single *Potentilla bimundorum* Soják growing approximately 100 m from the original population centroid on a sparsely-vegetated powerline.

4.4 Discussion

4.4.1 Extinction debt in in the Oil Sands Area of Alberta, Canada

Over half of the rare plant population records (209) within the Oil Sands Area occur within lands leased to oil and gas activity, with high frequency of overlapping and nearby oil and gas footprint. This indicates that the records maintained by ACIMS for this region are, in large part, a product of environmental impact assessments completed by industry and consultants prior to developing an area, and that these populations occur in a landscape with widespread human footprint. Field surveys of 62 of these populations indicated an overall estimated persistence rate of 77%, which I acknowledge may remain an under-estimate given the potential that some populations were overlooked during field surveys. I found that records in aquatic-related habitats, such as riparian areas and small open water wetlands, always persisted, while I observed an estimated persistence rate of only 64% for populations in uplands areas. This is interesting, as wetland-associated species would be expected to be particularly susceptible to hydrologic changes caused by road construction (Miller et al. 2015), or further from eutrophication caused from human activity (Kneitel & Lessin 2010). It is possible that wetland areas may be avoided during construction in some cases due to practical constraints, or that hydrological changes have yet to cause extirpation of these populations, as it has been noted that adverse effects of road construction may take decades to become obvious, i.e., there is a time lag between construction and reduction of populations (Findlay & Bourdages 2000). Notably, two wetland plant populations, although persisting, appeared in poor health at the time of survey.

Previous work has correlated increased human activity, particularly intensive agricultural practices or urbanization, with population loss and/or species extinction (Lienert et al. 2002; Stehlik et al. 2007; Van Calster et al. 2008; Dolan et al. 2011; Pergl et al. 2012), yet disturbance metrics which represent land use change surrounding or overlapping populations were not supported in explaining field observed persistence. There are therefore two important arguments to consider before concluding that persistence of rare vascular plants in this region does not relate to human footprint, specifically oil and gas related footprints. First, the means by which oil and gas activity may negatively affect plants, for example hydrological changes due to road construction (Miller et al. 2015), fragmentation effects from extensive seismic line networks (Dabros et al. 2017), or aerial deposition of contaminated dust from construction or processing of oil resources (Mullan-Boudreau et al. 2017), were not directly measured here. These factors would in theory be equivalently likely to affect populations throughout the region, on both on and off lease areas. Instead the metric of proportion of surrounding footprint best represents direct removal of vegetation due to construction or local impacts such as soil alteration or canopy removal. Although sites on oil and gas lease areas were anticipated to be at greater risk of extirpation due to direct removal of vegetation or other immediate effects due to increased construction within lease boundaries, I did not observe any trend of lower persistence between on and off lease populations (79 and 75%, respectively), Therefore, it appears that currently, rare vascular plant populations are rarely immediately lost to construction activities within lease boundaries, and no more so than those occurring within off-lease areas.

Extinction debt, the concept that populations are doomed to extirpation or indeed, entire species to extinction yet they persist on the landscape while ‘paying off’ this debt, is a second important aspect to consider and has been noted in other revisitation work to historical plant populations (Stehlik et al. 2007; Godefroid et al. 2014). In some cases, extinction debts for forest plants have been shown to persist for up to 100 years or more (Vellend et al. 2006). In the Oil Sands Area, development and expansion of the oil and gas industry has been occurring since the

1970's, a time span of less than 50 years (Bott 2004). Further, road effects on wetland plants in Canada have been shown to exhibit this lag (Findlay & Bourdages 2000). The background rate of extirpation for boreal plants in this region is unknown, thus comparison of my observed persistence rates to other areas with low disturbance is not feasible. However, it should be noted that all of the extirpations observed here occurred within upland areas, and an estimated extirpation rate of 35% for upland forest records which are relatively recent in age is not negligible, and may indicate that extinction debt is present in this region.

Based upon the observed 14 extirpations, all occurring in uplands, the causes of seven are evident and the remainder unknown. Three extirpations on oil and gas lease areas were in highly-altered habitats where vegetation removal and habitat destruction were the presumed cause of extirpation (*Malaxis paludosa* Sw., *Carex vulpinoidea* Michx., and *Lactuca biennis* (Moench) Fern.) accounting for only 5% of field surveyed populations. The remaining four instances where the cause of extirpation was evident included one observation in a crop field where a small wetland previously containing the annual species *Gratiola neglecta* had been drained and planted, and three populations of two perennial orchid species which had been severely burned in 2011 (*Cypripedium acaule* Ait., *n* = 2) and 2016 (*Spiranthes lacera* (Raf.) Raf., *n* = 1). It is important to note the remaining seven unknown-cause extirpations were of species capable of vegetative dormancy (*Spiranthes lacera*, *Cypripedium acaule*, *Sceptridium oneidense* (Gilbert) Holub., and *Botrychium crenulatum* W.H. Wagner), although the extent to which they exhibit this trait in the study region is unknown. Thus, confidence in assumed extirpations of these species is lower than others. Even if we consider these seven records as potentially erroneous, it remains that upland forests in this region have incurred an estimated extirpation rate of 20% over a relatively short time span.

4.4.2 Species traits in revisitation studies of historical rare plant populations

Species traits may further play a role in the relationship between human footprint and rare plant populations in the Oil Sands Area. Previous work has found correlation between species traits and extirpation (Saar et al. 2012), although Godefroid et al. (2014) cautions that extrinsic factors, such as habitat loss, rather than reproductive traits, are the major drivers of species decline. Of the 26 species surveyed in the field here, three are noted as ruderal species (*Cardamine parviflora* L.*,* *Carex vulpinoidea*, and *Potentilla bimundorum*) and one is associated with habitats expected to experience high levels of natural disturbance (*Lathyrus palustris* L.*,* a species associated with riparian margins of lakes and rivers) (Grime 1979). Broadly, boreal forest plants tend to be stress-tolerant (Grime 1979), and this biome is characterized by frequent disturbance events, such as wildfire and insect outbreaks with short, cool growing seasons. Conditions which typically follow natural disturbance may, in the case of ruderal species, be mimicked by soft disturbed features, such as vegetated wellpad edges or road ditches, which provide exposed mineral soil and removal of trees resulting in higher light conditions. Ruderal species, for example *Carex vulpinoidea* and *Potentilla bimundorum*, are associated with ditches and waste areas, and therefore areas of high light and exposed mineral soil. Field observations of these three species were of healthy populations growing in moderately to highly disturbed habitats (e.g., pavement cracks at an airport and gravel substrate surrounding wellpads). Thus, when anticipating the relationship between rare vascular plants and human footprint, such as oil sands development, species traits are an important consideration, and disturbance features may, in fact, provide habitat for a number of ruderal species which are of conservation concern. Relating to extinction debt, as noted in other work, some species may be less susceptible to decline under human development, or may be slow to realize their extinction debt given their traits (Vellend et al. 2006).

4.4.3 Survey effort in the Oil Sands Area and non-mandatory reporting

It is likely that the ACIMS database underrepresents regional vascular plant populations due to low survey effort and inconsistent reporting. Only 209 records of rare vascular plants are available for the Oil Sands Area, a density of 0.001 occurrences/km^2, likely in part a product of accessibility, as large areas have little to no road access and thus survey effort is low. It is likely that the ACIMS database is therefore biased to higher survey effort in areas of greater development, i.e., close to roads, an issue common to many conservation datasets (Stolar & Nielsen 2015). While PDA surveys are mandatory for some project developments within the Oil Sands Area, submission of records to ACIMS is only encouraged, rather than mandated, and thus this dataset is likely representative of only a portion of the rare vascular plant populations which exist in the region, even within lease areas. Further, there are important considerations in how survey effort is applied within lease areas. Regionally, only 22% of in-situ leases report a rare plant population, and these occur at a density of 0.004 occurrences/km^2 over the total area leased to this extraction type. This low frequency and density may reflect first that lease areas are almost never fully developed, and survey effort would only be applied to areas where construction is planned, therefore the ACIMS database may under-reflect populations within lease areas. Second, between 2011 and 2015 the number of oil sands leases in the region increased from 61 to 115 with only three of these for mining operations and 56 for in-situ. It is possible that newly approved oil leases are still undergoing development and records of rare plant populations may be forthcoming as development of these areas begins.

Considering mining leases, perhaps due to their small number within the region, mines in fact report a higher density of rare plant population occurrences, 0.02 occurrences/km^2, and report a population much more frequently than do in-situ leases. However, because oil sands mines represent major, continuous footprints (up to 200 km^2) within the Oil Sands Area, removing vegetation and surficial soil layers, it is logical to anticipate that their construction would lead to local population loss. This low density of records is likely to relate to effort in surveys, as it is improbable that pre-disturbance surveys could cover the entirety of a mining operation prior to construction within a short time span (i.e., a group of individuals working within a limited growing season), and therefore we would expect lower effort per unit of developed land than would be expected for smaller, more compact in-situ developments (where a typical wellpad is roughly a quarter hectare in size and thus feasible to search within a short time span). Although I had anticipated greater losses to surface mines, given their larger, more intensive footprint, this was not observed in field surveys, where instead persistence was equivalent across these two lease types. However, it is important to note that although in-situ mining disturbs less area than surface mines, there are arguments that the effects of fragmentation caused by in-situ practices (Dabros et al. 2018) and increased demand for natural gas to supply in-situ development, results in approximately equivalent area of land disruption (Jordaan et al. 2009), but fragmentation was not considered as a metric in this study.

While the overall regional survey effort and application of effort within lease areas may be inconsistent within the ACIMS database, and the degree to which observed populations are not reported to ACIMS is unknown, I argue that their application here remains valuable, as these data are the primary and most comprehensive source for historical vascular plant records available within this region. However, I acknowledge that these inconsistences may make establishing relationships between footprint and vascular plants challenging. In future, mandatory reporting is highly encouraged for PDA survey results, and ongoing education of the importance of submission of records to ACIMS may further promote a greater number of records submitted by those surveying for personal or academic purposes in the region. These data can be applied to gain understanding in not only persistence given surrounding footprint as examined here, but regional rare plant distribution and relationships to landscape features, thus their collection and submission has important value.

4.4.4 Survey effort and detection of target species

There was a strong relationship between search time and abundance that may help guide future resurvey efforts. All populations' ≥50 individuals were detected almost immediately, with a maximum effort of 0.0004 minutes/m^2. In contrast, maximum effort expended where the target species was located was nearly two hours to locate six individuals within the search area, an effort of 0.01 minutes/m^2, nearly 25 times greater effort than was expended for a large population. This illustrates that survey results for species known to occur in high abundance are likely more reliable than those for species which occur at low abundance when search effort is not recorded (McCarthy et al. 2013); a common practice in rare plant surveys conducted in this region. When populations are expected to be small, such as with *Lactuca biennis*, higher survey effort will be required to ensure adequate detection and achieve confidence in reported absences. If future surveys for rare plants consistently report effort expended based on population size and the presence or absence of target species, we can begin to understand the differences in required effort for small populations and ensure confidence in reported absences, and therefore use this information to set minimum survey effort as seen in Australian guidelines (State of New South Wales 2016). The development of these relationships is a key reason to support improved consistency in reporting of survey effort. In these field visits, high survey effort at sites with reported absences, particularly those with species capable of vegetative dormancy, and the team's demonstrated ability to detect small populations where present provided further confidence in these results. However, I acknowledge that as with most plant studies, these findings represent underestimates of persistence.

4.4.5 The role of long term monitoring in assessing population extirpations within the Oil Sands Area and future revisitation efforts

It is important to consider that population recruitment, health, and species longevity are beyond the scope of this project. These results represent a snapshot in time of a rapidly developing region (Rooney et al. 2012), and long-term monitoring would be required to fully understand the dynamics of extirpation of rare plant populations in the Oil Sands Area. It is therefore possible that over time trends may appear in the relationship between footprint and extirpation that were not observed in this study, i.e., the realization of extinction debt or loss due to direct effects of ongoing development. Finally, encountered populations were small (≤30 individuals) for 22 of the 48 persisting populations surveyed in the field. Thirteen of these small populations occurred on lease areas. Sites with small populations are likely the most vulnerable of those assessed with extirpation possible through stochastic processes (Shaffer 1987; Kunin & Gaston 1997), meaning that my estimate of population persistence may be over-estimated if these populations are subject to greater losses over time.

Although I did not correlate oil and gas footprint with population extirpation, an estimated persistence rate of 77% demonstrates a concern for the ongoing use of these data in rarity ranking or other conservation applications. If my initial findings from this survey are representative of the condition of ACIMS recorded rare plant populations across the region, it is feasible that some species could, or may in the future, have misapplied ranks if historical records are not verified prior to use in ranking exercises. It is evident that extirpated populations are currently included in the provincial dataset used here (23% of records), and I would encourage continual monitoring of populations or revisitation prior to future conservation status assessments that use these data (Master et al. 2012). Thus, I suggest the development of revisitation standards for assessing the status of rare plant populations in the Oil Sands Area.

If future monitoring efforts of these populations are made in this region, I would suggest the following framework for development of a prioritization scheme for revisitation. First, those species ranked S1 or S2, and additionally any species ranked at G3 or higher (regardless of their provincial ranking), should be the highest priority for revisitation. Species with ruderal traits, such as those noted here, should be downgraded in priority, provided they do not meet these criteria. Next, initially reported population size can act as a second priority, as these populations may be more prone to extirpation due to stochastic events, but further that they will require greater survey effort than large populations. As such, multiple observers or repeated visits over years may be required to ensure adequate effort has been expended to declare them extirpated. In closing, the relationships between oil and gas footprint, species traits, and population longevity are complex in this region and are almost certainly not equivalent among species, nor fully understood given the short time frame of oil and gas expansion and the young age of regional records. Further work concerning boreal plant traits is of interest, in particular where species with ruderal traits living under frequent disturbance regimes from fire and insects may show less response to human footprint than those in other areas.

Table 4.1. The most prevalent simplified footprint types directly overlapping reported Alberta Conservation Information Management System population polygons in the Oil Sands Area. , where 108 of 209 regional records have directly overlapping footprint (52%). See Appendix 4, Table A4.1 for description of full classification. Note that polygons could contain more than one footprint type and only the most prevalent is summarized below.

Footprint type	No. of polygons with overlapping footprint (*n* = 108)
linear feature	48
in-situ structure	15
agriculture	8
road	8
forestry	6
oil and gas structure	6
miscellaneous industrial	4
oil sands mine	4
residential	3
airport	2
reclaimed land	2
cleared/disturbed ground	1
residential	1

Table 4.2. Results of unpaired Wilcoxon Rank Sum tests for the proportion of oil and gas and non-oil and gas footprint within originally reported Alberta Conservation Information Management System population polygons on and off lease areas (left), and between in-situ and mining leases (right) in the Oil Sands Area.

	On vs. off lease areas		In-situ vs. mine leases	
Oil and gas footprint	*W*	*p*	*W*	*p*
original polygon	4224.0	0.002	915.5	<0.001
10 m	3507.0	<0.001	940.5	0.001
100 m	3068.0	<0.001	901.0	<0.001
1 km	2734.0	<0.001	797.0	<0.001
Non-oil and gas footprint	*W*	*p*	*W*	*p*
original polygon	5955.0	0.111	1271	0.107
10 m	6090.0	0.063	1173.5	0.021
100 m	5469.5	0.961	1133.5	0.034
1 km	5217.5	0.590	1030	0.009

Table 4.3. Observations of 26 species surveyed at 62 field sites in the Oil Sands Area. Average population and population range denote field estimates of encountered populations of each species. * denotes those observations where individuals lacked the reproductive structures necessary to confirm species-level identification and thus it is assumed that the initial identification of the population was correct. All other records were confirmed to the species level.

Species	Common name	S-rank	No. records	No. persisting (%)	average pop^n where persisting	Pop^n range
Botrychium crenulatum	Scalloped grape-fern	S3	1	0 (0)	-	-
*Cardamine parviflora**	Small bittercress	S2	1	1 (100)	5	-
Carex oligosperma	Few-fruited sedge	S3	2	2 (100)	1000	1000
Carex vulpinoidea	Fox sedge	S3	3	2 (67)	8	6 - 10
Cypripedium acaule	Stemless lady's-slipper	S3	9	6 (67)	60	1-180
Dryopteris cristata	Crested shield fern	S3	5	3 (60)	36	15 - 85
Elodea canadensis	Canada waterweed	S2	2	2 (100)	515	30 - 1000
Eutrema salsuginum	Mouse-ear cress	S1	1	1 (100)	200	-
Gentianopsis detonsa spp. raupii	Northern fringed gentian	S1	1	1 (100)	150	-
Gratiola neglecta	Clammy hedge-hyssop	S3	2	1 (50)	3	-
Houstonia longifolia	Long-leaved bluets	S3	1	1 (100)	1	-
Isoetes echinospora	Northern quillwort	S2	2	2 (100)	17	5 - 30
Lactuca biennis	Tall blue lettuce	S3	4	2 (50)	1.5	1-2
Lathyrus palustris	Marsh vetchling	S1	1	1 (100)	100	-
Liparis loeselii	Loesel's twayblade	S2	1	1 (100)	30	-
Malaxis paludosa	Bog adder's-mouth	S2S3	2	1 (50)	35	-
Najas flexilis	Slender naiad	S3	5	5 (100)	100	100
Nymphaea leibergii	Pygmy water-lily	S2	4	4 (100)	62	50-100
Nymphaea tetragona	White water-lily	S2	1	1 (100)	75	-
Phegopteris connectilis	Northern beech fern	S3	1	1 (100)	75	-
Plantago maritima	Sea-side plantain	S1	1	1 (100)	50	-
Polygaloides paucifolia	Fringed milkwort	S2	5	5 (100)	464	20 - 1000
*Potentilla bimundorum**	Branched cinquefoil	S2	2	2 (100)	150	1 - 300
Sceptridium oneidense	Blunt-lobe grape-fern	S1	2	1 (50)	20	-
Spartina pectinata	Prairie cord grass	S2	1	1 (100)	1	-
Spiranthes lacera	Northern slender ladies'-tresses	S2	3	0 (0)	-	-

Table 4..4. Persistence among 62 historical rare vascular plant populations of 26 species in the Oil Sands Area by landcover type and current land use.

	Upland		*Altered*		*Riparian*		*Lowland*		*Aquatic*	
	no. sites	persisting (%)	no. sites	persisting (%)	no. sites	persisting (%)	no. sites	persisting (%)	no. sites	persisting (%)
In-situ lease	10	6 (60%)	7	5 (71%)	1	1 (100%)	3	3 (100%)	5	5 (100%)
Mine lease	3	2 (67%)	2	1 (50%)	2	2 (100%)	1	1 (100%)	4	4 (100%)
Protected area	4	4 (100%)	-	-	-	-	-	-	1	1 (100%)
Public or private land	12	7 (58%)	2	1 (50%)	-	-	1	1 (100%)	4	4 (100%)
Total no. sites	30	19 (63%)	11	7 (64%)	3	3 (100%)	5	5 (100%)	14	14 (100%)

Table 4.5. Results of AIC model selection of ten single predictor candidate logistic regression models of rare plant population persistence (*n* = 62). All continuous variables were log-transformed prior to inclusion.

Model	*K*	AIC	ΔAIC
landcover type	1	61.78	0.00
oil related disturbance within 10 m of population	1	68.24	6.46
null	0	68.24	6.46
oil related disturbance within ACIMS population	1	69.20	7.42
oil related disturbance within 100 m of population	1	69.65	7.87
oil related disturbance within 1000 m of population	1	69.93	8.15
record age	1	69.95	8.17
land use	1	70.60	8.82
simplified disturbance type	1	82.33	20.55

Table 4.6. Model parameters of the best supported logistic regression model of rare plant population persistence in the Oil Sands area (*n* = 62). The category 'upland' was withheld as reference for the variable "landcover type". Observations in aquatic, lowland, and riparian margin sites had perfect persistence.

Model parameter	β	SE	*p*
Intercept	0.64	0.39	0.100
Landcover type			
Aquatic	Perfect persistence		
Lowland	Perfect persistence		
riparian margin	Perfect persistence		
highly-altered	-0.08	0.74	0.911

Table 4.7. Results of AIC model selection of three candidate logistic regression models of rare plant population persistence using a reduced dataset, (*n* = 45). All continuous variables were log-transformed prior to inclusion.

Model	*K*	AIC	ΔAIC
initially reported population size	1	42.92	0.00
initially reported population size + record age	2	44.82	1.90
initially reported population size * record age	3	45.53	2.61
Null	0	49.67	6.75

Table 4.8. Model parameters of the best supported logistic regression model shown in Table 4.7 (*n* = 45). Initially reported population size was log-transformed prior to inclusion in models.

Model parameter	β	SE	*p*
Intercept	0.43	0.49	0.390
initially reported population size	0.69	0.30	0.019

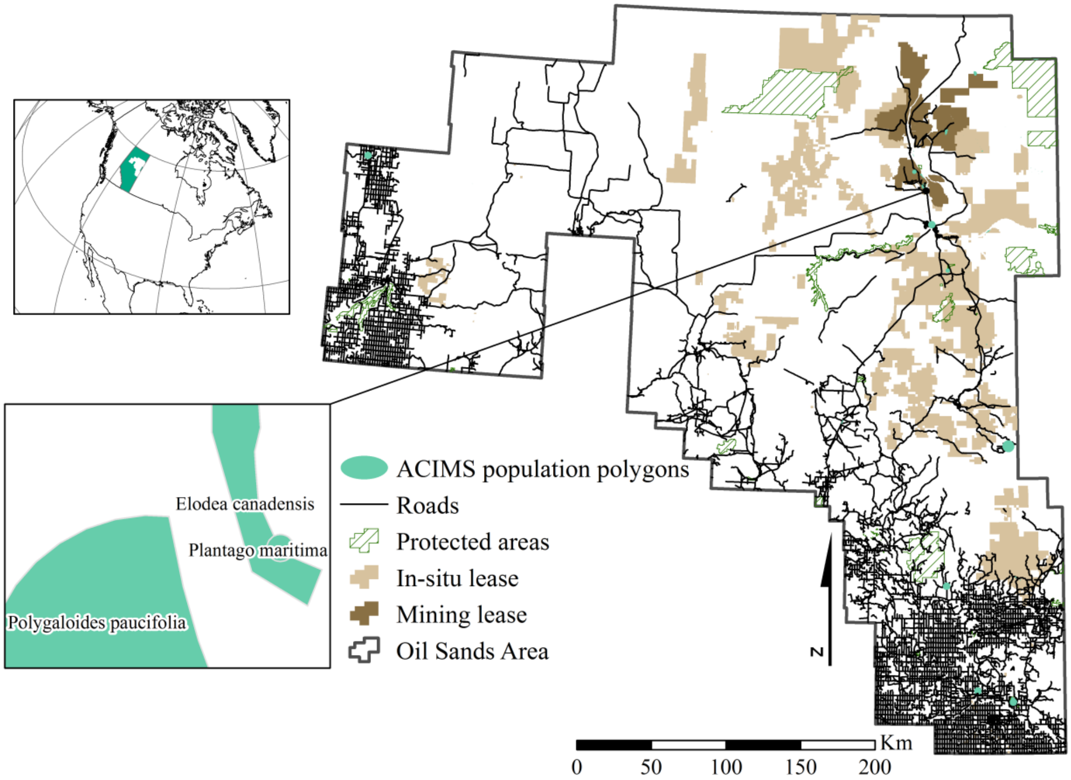

Figure 4.1. The Oil Sands Area of Alberta, Canada, showing lands leased for the extraction of oil via in-situ or conventional mining and protected areas. Upper inset shows Alberta relative to North America. Lower inset shows typical rare plant population polygons, which can be irregular in shape and of varying size.

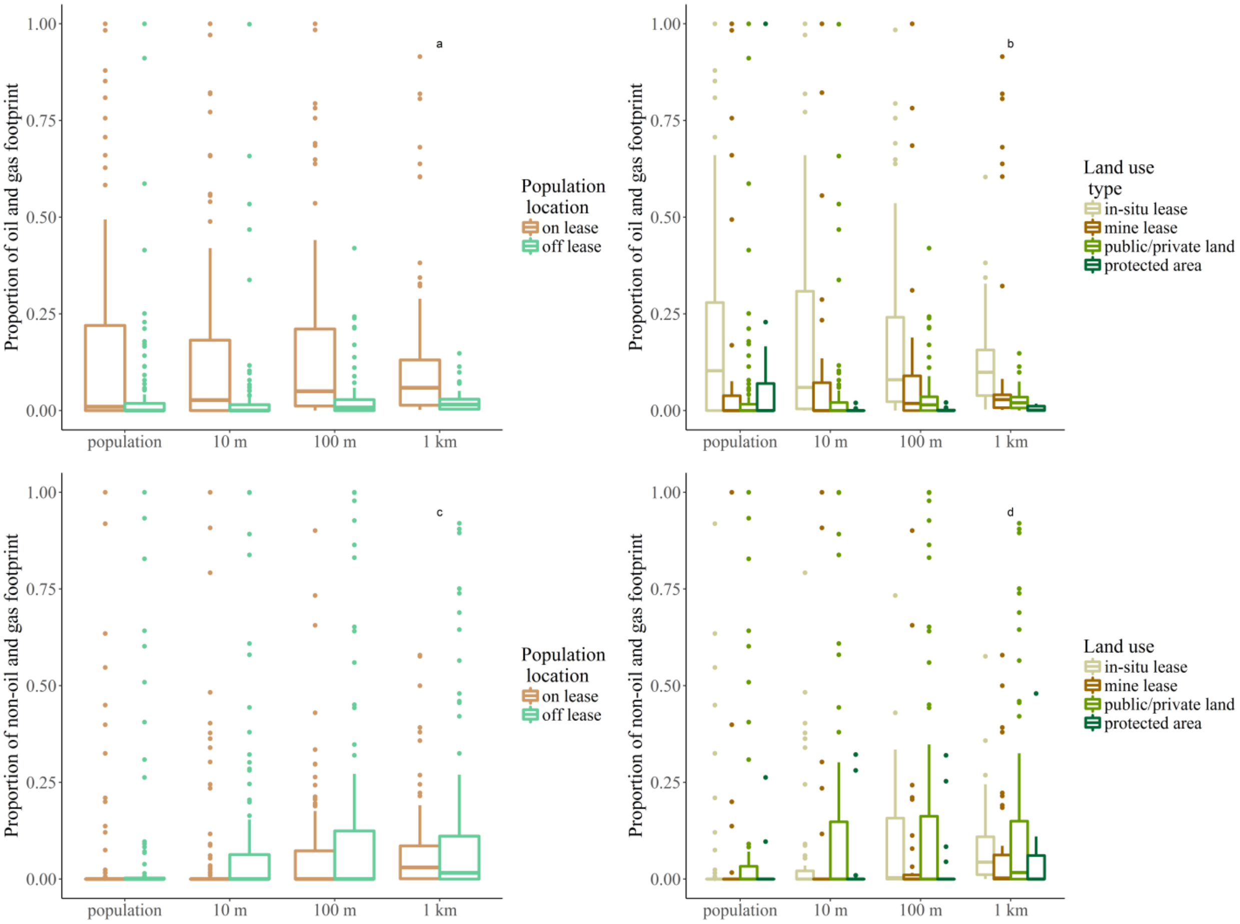

Figure 4.2. Boxplots of proportion of human footprint by oil and gas vs. non-oil and gas footprints overlapping and surrounding for 209 rare vascular plant populations in the Oil Sands Area for both on and off lease areas and by land use type.

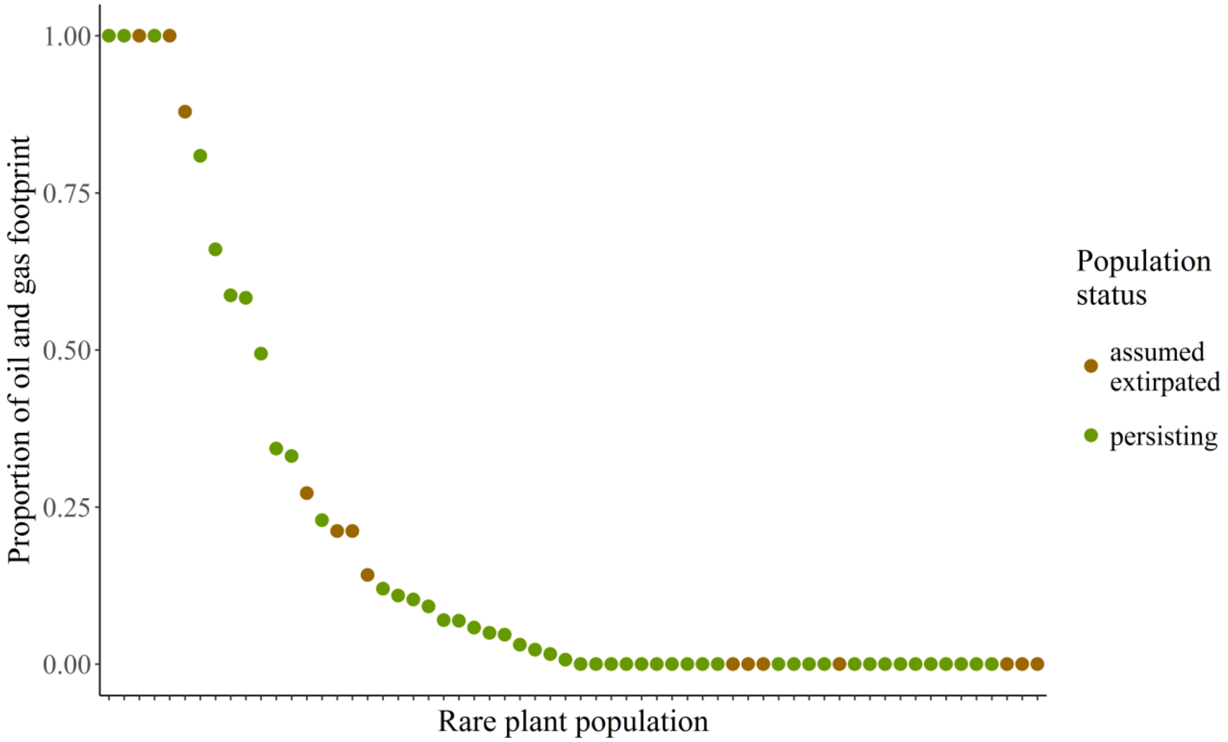

Figure 4.3. Persistence and assumed extirpations at 62 field sites of historical rare vascular plant populations across a range of oil and gas footprint proportions, ordered by the proportion of oil and gas related footprint directly overlapping the originally reported population polygon..

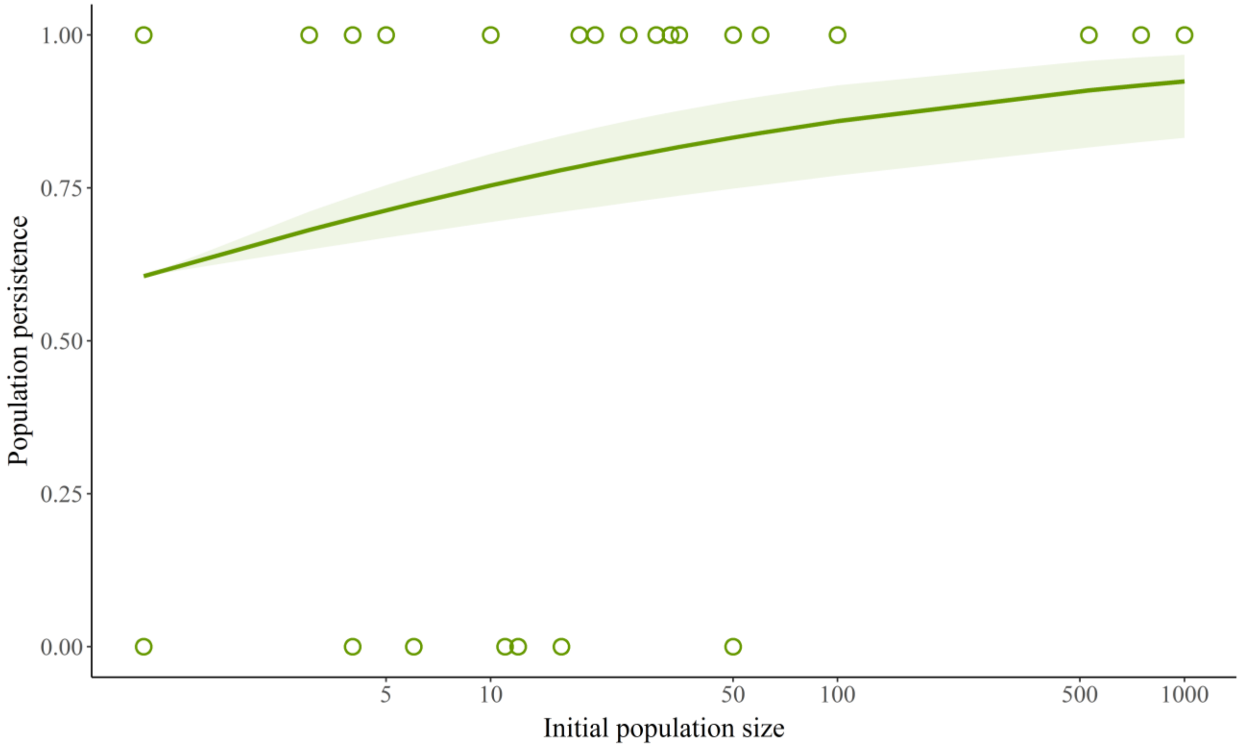

Figure 4.4. Scatterplot of persisting vs. assumed extirpated field-observed rare vascular plant populations given their initially reported population size ($n = 45$). Fitted model relationship (Table 4.8) shown in green with shaded confidence interval. X-axis shown in log-10 scale.

Chapter 5: Early success of mitigative translocation for rare peatland species

5.1 Introduction

Translocations, the movement of plant or animal material, have long been conducted by humans for aesthetic, cultural, personal, or conservation purposes (Grayson 2001; Richardson et al. 2011). Where translocation is used as a conservation measure, objectives include re-introduction or augmentation of populations of endangered or at-risk species (Godefroid et al. 2011), assisted-migration (McLachlan et al. 2007), and mitigative translocation (Germano et al. 2015). Mitigative translocations move individuals or populations facing imminent destruction or habitat alteration from development (Germano et al. 2015). These have only recently been distinguished from other transplanting activities with few published examples, while literature on general species reintroduction and augmentation is relatively abundant (Godefroid et al. 2011). The success of those projects is often poor, perhaps due to the fact that in all cases excluding augmentation, recipient sites must be selected based on best-guesses or historical records of where a species may be able to grow and reproduce successfully (Godefroid et al. 2011; Drayton & Primack 2012).

Currently, there is debate and no clear consensus on the use of mitigative translocation to rescue populations that are at risk of development. While this practice has received support as a tool to conserve endangered species, it is most often recommended against as a mitigation technique due to concerns regarding larger issues of habitat loss and low reported success rates (Vallee et al. 2004; Fahselt 2007; Maslovat 2009). However, many believe that mitigative translocation can be a valuable tool to prevent local extirpation despite these obstacles (Germano et al. 2015). Factors suggested to increase the success rate of re-introduction and augmentation translocations include use of adult plants from healthy donor populations, establishing large founder populations, and applying site preparations (Godefroid et al. 2011). Poor recipient site selection is a suspected cause of failure in many unsuccessful trials, highlighting the importance of evaluating species' niche and biotic and abiotic site factors in selecting appropriate recipient sites (Bottin et al. 2007; Godefroid et al. 2011). However, mitigative translocations do not always allow for careful consideration of these factors.

First, follow up monitoring is generally lacking, presumably due to a lack of funding for monitoring or high turnover within industries, such as oil and gas, leading to a loss of information around transplant records. Second, public reporting of projects, even those that do receive monitoring, is rare. This is also true among more carefully planned projects (Godefroid & Vanderborght 2011). Lack of reporting is a major detriment to mitigative translocation and may increase the failure rate of future efforts by reducing learning opportunities, and further makes evaluating the overall efficacy of these actions challenging (Germano et al. 2015). Finally, these projects are often more time limited than reintroduction or augmentation projects. This limits the pre-translocation planning process resulting in recipient sites being selected quickly, in some cases without knowledge of the ecology of the species being translocated or consideration of methodology. While these statements are true across taxa and industries, in Alberta, major footprint resulting from oil and gas development has prompted the infrequent but consistent use of mitigative translocation for rare vascular plants. These projects are costly, as are many mitigative efforts, and it is therefore imperative to understand their efficacy and allocate resources appropriately (Germano et al. 2015).

Despite these potential obstacles, well-planned mitigative translocations in the oil sands region have the capacity to inform definitions of environmental tolerances of boreal species. Boreal environments are unique in that they are often dominated by peatlands and wetlands, landcover types which have not been the focus of translocation research in Canada (Clements 2013), but are important sources of biodiversity, ecosystem services, and house rare species (Whitehouse & Bayley 2005; Kuglerova et al. 2014). Further, peatlands are likely to be disrupted during oil and gas development with minimal likelihood of successful reclamation due to the complexity of replicating hydrological flow regimes (Rooney & Bayley 2011; Rooney et al. 2012; Raab & Bayley 2013). Given the conservation focus and knowledge gaps associated with this landcover type, I conducted experimental translocations for two rare peatland obligate species with notably different life-history traits, *Sarracenia purpurea* L. and *Carex oligosperma* Michx. These species were selected because the amenability of *S. purpurea* to transplanting has been documented in other regions of Canada (Hardwick & Giberson 1996), but very little is known regarding the ecology of *C. oligosperma*, thus it was possible to compare the success of mitigative translocation for two species whose ecology was reasonably well, and poorly, understood, respectively. An important objective in this experiment was to mimic conditions characteristic of mitigative translocation, specifically a relatively narrow timeframe to select recipient sites and limited knowledge of the ecology of *C. oligosperma*.

My specific objectives were to first determine transplant survival and overall efficacy of this practice for these two rare species, which could be indicative of other peatland species with similar traits. Given literature suggesting that similarity between donor and recipient sites may positively influence translocation outcomes (Lawrence & Kaye 2011), sites were selected to vary in their similarity to donor sites and thereby test whether differences in vegetation assembly and water chemistry (pH, nutrient, and mineral status) affects transplant survival and health. Sites most similar to the donor were predicted to have higher survival rates under the assumption that these rare species are habitat specialists. Relating easily measured variables such as these to transplant success can inform recipient site selection in future translocation projects in the region. Further, success in my transplanting methodology can inform future guidelines and best practices for boreal plant translocations.

5.2 Methods

5.2.1 Donor and recipient study sites

Donor populations were selected from known large (>1000 individuals), healthy populations that had been encountered during previous research (see Zhang *et al.* 2014). Three independent donor and recipient sites were selected for each species. Each focal species therefore had six experimental sites. No donor sites contained both focal species and no recipient sites had existing populations. Recipient sites were purposefully chosen to differ from one another and from donor sites in vegetation assemblage and inferred nutrient status (i.e., rich vs. poor fen). This was done to in part test the influence of recipient site characteristics, where poor recipient site selection is a suspected cause of failure in translocation projects, and to further mimic the realities of mitigation translocation in areas of rapid development, where a short project timeline is likely to reduce the amount of time available to search for a recipient site which matches closely with donor conditions. All six *S. purpurea* sites are located in the vicinity of Conklin, Alberta. Three *C. oligosperma* sites were located near Fort Mackay, Alberta, while the remaining three were located near Conklin (Figure 5.1), site locations are reported in Appendix 5, Table A5.1.

5.2.2 Study design, removal, and planting methodology

Translocations were conducted between late August and mid-September of 2014, after hot summer conditions had passed for the season. I selected 70 transplants at each donor site for both species. Twenty of these transplants were removed and immediately replanted within each donor site as a control for the effect of transplanting (hereafter referred to as control transplants). The remaining 50 transplants from each donor site were distributed among the three recipient sites in groups of 17, 17, and 16. Therefore, each recipient site for each species had a founder population of 50 individuals (Franklin 1980), from three different donor locations (Figure 5.2). In total, 210 transplants of each species were transplanted. No transplants were moved between donor locations.

To limit damage to the donor population, selected individuals were taken from as small an area as possible with no individuals closer than 2 m to prevent overlap of vegetation plots. As both species were abundant (>1000 individuals) at all six donor locations, this resulted in removal from an area of roughly 400 m^2. Replanting was conducted over a similarly sized area at all three recipient sites per species. Prior to removal, each transplant was given an identification code with a metal washer attached to a loop of string and flagging tape for relocation. A 0.25 m^2 quadrat (0.5 x 0.5 m PVC frame) was then placed around the transplant and percent cover was estimated for all species within the quadrat. Transplants were cut from the peat as small monoliths averaging 30 x 30 cm with substrate attached. Healthy adult plants were removed with a focus on obtaining significant amounts of root material rather than the precise removal of a single individual. *Carex oligosperma* is strongly rhizomatous and thus each transplant contained multiple vegetative and flowering stems, most likely ramets of a single genet. Transplants of *S. purpurea* often contained more than one individual. When transplants were first removed at their donor site, vegetative and flowering stems of *C. oligosperma* and pitchers of *S. purpurea* were counted and recorded. Plants were transported in coolers or tubs with icepacks between donor and recipient sites, with a typical travel time of 4-6 hours, but in one case transplants (from OD2 and OD3 to OR1 and OR2) were stored in coolers for 12-14 hours.

Planting of both species in peat substrate was straightforward. I cut slits in the peat (through the roots of other plants), widened them by hand if necessary, and packed the transplant in with a moderate amount of force to avoid air space around the roots. Vegetation plots with cover estimates (0.25 m^2) were repeated when plants were transplanted, giving two complete vegetation surveys for each transplant. This was also completed for donor site controls. In 2016 transplant tags were replaced with improved permanent markers using metal pigtails 0.5 to 0.75 m in length with attached engraved metal tags bearing the transplants identification code.

5.2.3 Spring site visits and water chemistry sampling

Transplants were monitored for three years post-planting (2015 – 2017), with two visits to each of the 12 donor and recipient sites per year. In early June of each year spring relocation checks (re-locating and flagging transplants) and water chemistry sampling were conducted at all sites. Water chemistry analysis was used to determine the nutrient and mineral status of donor and recipient sites. At each site, a transect was placed in the orientation that water could be expected to flow (e.g., perpendicular to open water or upland slopes) through the peatland. This transect was set out to bisect the founder population at recipient sites and the control individuals at donor sites and was generally 15 – 20 m in length. Then, three water samples were collected from piezometers inserted approximately 30 centimeters into the peat at the beginning, mid-point, and end of each transect. Piezometers were siphoned out upon insertion, left to refill, and then siphoned into plastic sampling containers approximately 500 mL in size. I determined pH and temperature in the field at each piezometer station from a separate collection container using an EcoSense ® pH 100A meter manufactured by YSI (Yellow Springs, Ohio, USA). Samples were not filtered or preserved in the field and were shipped within two days of collection. All analyses were conducted by the Biogeochemical Analytical Service Laboratory (BASL) at the University of Alberta for the following: the nutrients nitrite and nitrate (NO_2- and NO_3-, respectively), total nitrogen (TN), total Kjeldahl nitrogen (TKN), total phosphorus (TP), and the minerals sodium (Na+), potassium (K+), calcium (Ca_2+), and magnesium (Mg_2+). Total Kjeldahl nitrogen was used in analysis as it represents organic nitrogen plus ammonium and ammonia, and may best represent the organic nitrogen available to plants in their environment (Vitt & Chee 1990).

5.2.4 Annual monitoring of survival, growth, and flowering at donor and recipient sites

The second annual visit to determine survival, growth, and flowering of transplants occurred between late July and early August of each year at all 12 experimental sites. By this time of year *C. oligosperma* is fully mature but has not begun to shed perigynia and *S. purpurea* flowers are mature or beginning to senesce. I defined transplants to be deceased when no green stems were produced in *C. oligosperma* and all pitchers of *S. purpurea* were completely brown (non-living tissue). Flowering and survival were recorded as binary variables. Growth in *S. purpurea* was determined by counting the number of living pitchers. Pitchers can persist for one to two years in this species (Gotelli & Ellison 2009). For *C. oligosperma* I determined growth by counting all vegetative and flowering stems and measuring the tallest stem (flowering or vegetative). Leaves in this species senesce annually and new above ground material is produced each spring (Ryser & Kamminga 2009). When *C. oligosperma* produced flowers, I recorded the gender and number of spikes, and the average length and width of female spikes using calipers.

The identification of control transplants at *C. oligosperma* donor sites was complicated by the species' rhizomatous growth form. I was unable to determine if shoots in the region of the original transplant tag originated from the transplant or from neighbouring individuals. Further, rapid burial at one donor site (OD3) resulted in the loss of original transplant markers after a single season. Due to these factors these data were not considered reliable and are not included in analysis. Instead, I measured 30 random wild individuals at each donor site in each monitoring year (2015 – 2017), to estimate height and inflorescence characters (gender and number of spikes, and average length and width of female spikes) of non-transplanted individuals under normal donor site conditions.

5.2.5 Statistical analysis of growth and flowering

Differences in transplant survival at the end of the monitoring period among recipient sites and donor groups (i.e., the identity of each transplant's natal site) were assessed using chi-squared tests in R, version 3.4.3 (R Core Team 2017) . To examine the possible negative effects caused by transplanting, I compared annual growth (the change in the number of pitchers between monitoring years) of control transplants of *S. purpurea* (those re-planted at their natal site) to those at recipient sites using a generalized linear mixed model with a random effect of site and transplant identity, where predictor variables were transplant status, i.e., "control" or "transplant", and year. For *C. oligosperma*, given that no true controls for the transplanting process were able to be used in this experiment (failure to re-located *C. oligosperma* re-planted within the donor sites), I instead visualized differences between wild, un-transplanted individuals growing at donor sites and transplants in terms of their height and the length and width of their pistillate spikes.

I visualized the relationships between the existing vegetation assembly and water chemistry at donor and recipient sites with a Canonical Correspondence Analysis using the package 'vegan', version 2.4-6 in R, where vascular plant species were related to nutrients, minerals, pH and temperature. Using vegetation data (0.25 m^2 plots) collected at donor and recipient sites for each transplant, I calculated Bray-Curtis dissimilarity for each transplant using the package ‘vegan’ in R as a measure of vegetation change between donor and recipient sites, i.e., how different the local conditions were for each individual transplant between sites.

I then used a two-step model building and evaluation process to assess the relationships between annual growth and flowering of transplants at recipient sites and recipient site characters for both species. Annual growth represented the change in the number of stems (vegetative or flowering) produced between years for *C. oligosperma,* and pitchers in *S. purpurea*, and flowering was a binary variable for both species. First, I constructed a set of eight generalized linear mixed-effect and mixed-effect logistic regression models representing water chemistry variables and growth and flowering, respectively. Given that water chemistry variables were highly correlated, each model contained a single predictor variable. I ranked support for these models using Akaike Information Criterion (AIC) (Burnham & Anderson 2002) to determine which variable most influenced growth and flowering, respectively. I included a random effect of year and transplant identity in all candidate models to account for variation between years and repeated measures on individual transplants. Due to major differences in the scale of some water chemistry variables and the random effect variables, I standardized all water chemistry variables to their mean prior to inclusion in candidate models. The linear relationships between water chemistry variables at recipient sites and annual growth were visualized by plotting observations (Appendix 5, Figures A5.1 and A5.2).

In the second step of model building and evaluation, I evaluated transplant growth and flowering against vegetation change, initial transplant size, year, and the best supported water chemistry variable from the first step of model evaluation in generalized linear mixed-effect and mixed-effect logistic regression models, respectively. Candidate models represented 1) beta diversity (Bray-Curtis dissimilarity) and total vegetation cover at recipient sites, 2) beta diversity and the change in total vegetation cover between donor and recipient sites, 3) the most supported water chemistry variable and donor and recipient site identity (site effects), 4) monitoring year alone, and 5) initial transplant size, measured by number of pitchers in *S. purpurea* and total number of stems in *C. oligosperma* when they were first transplanted. In this step, year was included as a covariate in all models as a proxy for variation between monitoring years in variables not measured here, such as summer precipitation and temperature. I again used a random effect of transplant identity in all models to account for repeated measures of individual transplants. A pseudo R^2 was estimated for all top ranked candidate models in both steps using the package 'MuMIn', version 1.40.0, and function 'rsquared.GLMM' for generalized linear mixed-effect models and the package 'piecewiseSEM', version 1.2.1, and function 'sem.model.fits' for logistic regression models (Nakagawa & Schielzeth 2013).

5.3 Results

5.3.1 Survival, growth, and flowering of transplants

Transplant survival was high for both species, with total survival of 97% for *S. purpurea* and 85% for *C. oligosperma* after three years of monitoring (Table 5.1). Total survival was highly similar between recipient sites or donor groups for *S. purpurea*, such that chi-squared tests were not reliable (expected frequencies <5) (Table 5.1, see Figure 5.6). No significant differences in survival between recipient site and donor group for *C. oligosperma* were observed ($\chi^2 = 2.70$, $p = 0.259$ and $\chi^2 = 4.15$, $p = 0.125$, respectively). *Carex oligosperma* survival declined annually, by a maximum of 8% (Table 5.1). Flowering rates for *S. purpurea* declined slightly over monitoring years. Flowering was reduced in 2016, the wettest summer relative to 2015 and 2017 for *C. oligosperma*. Not all transplants were successfully relocated at recipient sites in a given monitoring year (Table 5.1). For *C. oligosperma*, four transplants were never relocated at recipient sites, and four were only relocated in the first monitoring year. Relocation of *S. purpurea* was similar, with one transplant never relocated and four only relocated in the first monitoring year. These transplants were removed from analysis as their survival could not be reliably determined in the field.

As previously discussed, 60 control transplants from *S. purpurea* donor sites (20 transplants/site) were used to investigate the effect of transplanting with site conditions held constant (Table 5.2). Total survival for control individuals was 100% throughout the monitoring period. Flowering rate among controls was similar to transplants moved to recipient sites in all monitoring years (Tables 5.1 and 5.2), as was annual growth across all years (Figure 5.3). A single mixed-model fit to describe annual growth in control transplants and transplants at recipient sites as a function of year and status (control vs. transplant) indicated that status had a limited relationship to explaining differences in growth ($\beta = 0.15$, SE = 0.59, $p = 0.80$) whereas year was a more influential predictor variable ($\beta = 1.10$ and -1.25, and SE = 0.60 and 0.59, and $p = 0.07$ and 0.04 for 2016 and 2017 respectively), where 2015 was withheld as the reference category. Model fit over the null was supported (ΔAIC = 12.24). In 2017, the driest year of the monitoring period, I observed divergence in growth for *S. purpurea,* with two recipient sites showing a slight reduction in the average number of pitchers and one (SR3) showing an average increase (Figure 5.6), otherwise growth was relatively consistent and largely positive among sites. Anecdotally, *S. purpurea* transplants appeared vigorous at all sites.

In contrast, *Carex oligosperma* transplants appeared noticeably stunted in 2015 and 2016. Visualizing height and inflorescence characters between wild (30 individuals/donor site/year) and transplanted *C. oligosperma* showed that transplants increased, on average, in height and female spike length and width over monitoring years, while wild populations appeared to be declining in these same characters (Figure 5.4). As such, initial large differences in spike length and width in 2015 and 2016were less pronounced in 2017, but transplants remained noticeably shorter than wild individuals in all years (Figure 5.4).

Finally, growth over years among recipient sites varied, with an overall trend of major reduction in aboveground biomass for *C. oligosperma* the year after transplanting, and continued reduction or minimal growth in later years (Figure 5.5). As of 2017, nearly a third of transplants had three or fewer stems, and anecdotally, roughly 10% of transplants appear to have ‘taken’; that is begun to produce ramets outside the margin of the originally transplanted material. Based on donor group (i.e., the donor site from which transplants originated), growth across monitoring years was similar for all three donor groups for both species (Figures 5.5 and 5.6).

5.3.2 Water chemistry, vegetation, and growth and flowering at recipient sites

Major differences in water chemistry (pH and major nutrients) were observed between recipient and donor sites, and among recipient sites for both species, suggesting I was successful in choosing recipient sites which covered a nutrient gradient (Appendix 5, Figures A2.3 and A2.4). It should be noted that reported nitrogen levels were very low in 2017 relative to previous estimates at all sites for both *C. oligosperma* and *S. purpurea*, likely due to sampling or processing error or possibly, large seasonal fluctuation. Vegetation assemblages, as related to water chemistry variables, were similar between donor sites and one recipient site (SR2) for *S. purpurea*, while SR1 and SR3 were less similar to donor sites (Figure 5.7). For *C. oligosperma*, one donor and recipient were more similar (OD1 and OR1), while the remainder were not (Figure 5.8).

For *S. purpurea*, results of mixed-effect linear and logistic regression models representing relationships between annual growth and flowering (respectively) and individual water chemistry variables at recipient sites suggested that magnesium was an important mineral which related positively to annual growth (ΔAIC = 46.27 over null, Pseudo $R^2_m = 0.12$ (fixed effects), and $R^2_c = 0.13$ (random effects), $\beta = 2.18$, SE = 0.29, $p < 0.001$), the random effects of year and transplant identity had standard deviations of 0.64 and 0, respectively. Results of water chemistry model selection are shown in Appendix 5, Table A5.1. Scatter plots between water chemistry measures and annual growth (Appendix 5, Figure A5.1) indicate that variables correlated with magnesium, such as calcium and pH, also related to annual growth in *S. purpurea*. Total phosphorus was best related to flowering for this species (ΔAIC of 11.99 over null model, $\beta = 0.44$, SE = 0.12, $p < 0.001$) where the random effects of year and transplant identity had standard deviations of 0 and 0.58, respectively (model selection shown in Appendix 5, Table A5.2). However, this model explained relatively little variation in flowering among transplants (R^2_m of 0.05 and $R^2_c = 0.14$), suggesting that flowering may better relate to individual plant responses or other factors at recipient sites.

Comparison of candidate linear mixed-effect models for annual growth indicated that site effects, specifically magnesium concentration, monitoring year, and donor and recipient site identity, were most supported in explaining variation in annual growth (Table 5.3, model parameters shown in Table 5.4) (ΔAIC = 33.65 over null, $R^2_m = 0.12$, and $R^2_c = 0.12$), where magnesium was the only significant predictor. The random effect of transplant identity had a standard deviation of zero, indicating that individual variation was negligible in explaining annual growth. Flowering was similarly supported, where total phosphorous, year, and donor and recipient site identity best explained variation in flowering (Table 5.5, model parameters shown in Table 5.6) (ΔAIC = 20.14 over null, $R^2_m = 0.18$, $R^2_c = 0.46$). Flowering was reduced for transplants from donor site SD3 relative to SD2, and at both recipient sites in reference to SR2

(Table 5.1). The random effect of transplant identity had a standard deviation of 0.53, which, combined with a higher pseudo R^2 value for random effects, suggests that transplant level variation drives flowering for *S. purpurea* transplants.

For *C. oligosperma*, results indicated that water chemistry variables relate poorly to annual growth in this species, where the best ranked model, containing pH, was equivalent to a null model predicting equal average growth everywhere ($\Delta AIC = 1.53$, $\beta = -0.43$, $SE = 0.19$, $p = 0.03$). Further, this model showed low fit with regard to fixed effects ($R^2_m = 0.009$, $R^2_c = 0.38$), where the standard deviations of year and transplant identity were 2.74 and 0, respectively. The same was true for variation in flowering, where the best ranked model (again including pH) was equivalently supported to a null model of average flowering everywhere (ΔAIC of 0.08, $\beta = -0.22$, $SE = 0.15$, $p = 0.15$), with low fit ($R^2_m = 0.009$, $R^2_c = 0.35$), and the random effects of year and transplant identity had standard deviations of 0.95 and 0.92, respectively. Thus, water chemistry variables were not included in the next step of model building and it is likely that annual and individual variation explain more variation in these parameters for this species. See Appendix 5, Tables A5.3 and A5.4, for model selection of water chemistry variables and Figure A5.2, for scatter plots of water chemistry variables and annual growth, which indicate limited correlation.

Comparison of models for annual growth showed the greatest support for initial transplant size (number of stems) and monitoring year (Table 5.7, model parameters shown in Table 5.8), where these variables were negatively and positively related to growth, respectively. This is a product of largely negative growth (annual decrease in above ground material, Figure 5.5), where transplants which had many stems in 2014 were greatly reduced in 2015 and showed limited growth or continuing decline in following years ($\Delta AIC = 167.99$, $R^2_m = 0.42$, $R^2_c = 0.42$).

Transplant identity had a standard deviation of zero, suggesting similar trends across transplants. Beta diversity, total cover at recipient sites, and monitoring year best explained variation in flowering among transplants, where all but the monitoring year 2016 had a negative relationship to flowering, and sites which are more similar to a transplants donor may be favorable to flowering (ΔAIC = 50.60, $R^2_m = 0.26$, $R^2_c = 0.39$, model comparison shown in Table 5.9, model parameters shown in Table 5.10). The random effect of transplant identity had a standard deviation of 0.85. Combined with higher fit of random effects, this suggests that individual variation also affects flowering patterns in *C. oligosperma* transplants.

5.4 Discussion

This transplanting experiment is considered successful after three years of monitoring, with high survival of *S. purpurea* and *C. oligosperma* (97% and 85%). Considering that there are obvious differences in pH, nutrients, and minerals among recipient sites, it does not appear that these factors influence survival, suggesting that both of these rare species are more dispersal-limited than environmentally limited within the study region. Previous work with orchids similar to the approach taken here, where recipient sites were selected based on vegetation assemblage and pH, showed relatively low success but did link vegetation similarity between donor and recipient site to success for one species (Brzosko et al. 2018). Further, the similarity of functional plant groups between source and recipient sites was linked to success in translocation of *Castilleja levisecta* Greenm. after two years of monitoring, although success rates were lower than what is reported here (Lawrence & Kaye 2011). Despite notable differences in vegetation assemblage among recipient sites, species turnover (beta diversity), the change in vegetation cover, and the total cover of plants at recipient sites was only supported in influencing flowering in *C. oligosperma*, and did not influence growth in either of these rare peatland species.

Of the water chemistry variables considered here, it is probable that minerals influence the growth *S. purpurea*, with transplants showing the highest annual growth in 2017 at SR3, a rich fen with higher concentration of magnesium and calcium than the two other recipient sites. Interestingly, this site also showed significant vegetation and water chemistry differences from all donor sites, suggesting that differences in vegetation assembly are not a reliable field measure to indicate recipient site suitability for this species. Further, in previous reciprocal transplant work with this species, it was found that *S. purpurea* was nitrogen-limited (Bott et al. 2004), but I found no support of nitrogen levels being more influential than other nutrients and minerals within the peatlands considered here. Flowering was best related to variation between years, recipient sites, and in particular, a large effect of transplant identity. It is likely that transplants may vary in flowering not only at the individual level but potentially in response to variables not measured here, such as water table level or light.

Sarracenia purpurea is a well-studied species shown to be amenable to transplanting within Canada (Hardwick & Giberson 1996) and capable of colonizing new habitats, to the extent of being considered an invasive plant in parts of Europe (Parisod et al. 2005). Most significantly, it is known to occur across a gradient of peatland types (Karberg & Gale 2013). My results align with these observations of *S. purpurea* and highlight the potential value of mitigative translocation for species known to be tolerant of a range of conditions and amendable to the transplanting process. Using such species is an important consideration in existing translocation guidelines in other jurisdictions (Vallee et al. 2004; Maslovat 2009)

For *C. oligosperma*, water chemistry variables expected to influence growth and flowering showed limited correlation, suggesting this species may not be sensitive to changes in nutrient and mineral status. Given observations of *C. oligosperma* growing in acidic habitats (all donor sites had a pH <5.0), I had anticipated that pH and minerals, such as calcium, would influence the survival, growth, and flowering of this species. However, no water chemistry variable was supported in explaining variation in growth and flowering, and growth is relatively consistent across recipient sites. The strong negative relationship between growth, monitoring year, and the initial number of stems in a given transplant is a product of large initial decline in large transplants, i.e., those with a large number of stems at the time of transplanting could have greater annual losses than those with a small number. Large initial declines stabilized in 2016 and 2017, with minimal net gain or loss in ensuing years across recipient sites, and remarkable similarity in 2017 among donor groups.

Carex oligosperma tends to dominate and form rhizomatous mats in the peatlands where it has been observed in the study region, with low cover of other vascular plant species. I anticipated that *C. oligosperma* may therefore be influenced by change in vegetation assemblage and total vascular cover between donor and recipient sites, reflecting higher levels of inter-specific competition to which it may not be well adapted. Model results suggest that flowering is reduced at sites which are most dissimilar to the donor site from which transplants originated, indicating a potential negative influence of competition from novel species and negative effect of vegetation dissimilarity for this species. Thus, there is some indication that in order to maximize flowering for this species, which could improve the probability of the production of a successful second generation, recipient sites should be similar to the donor. While transplant identity appeared negligible in models of growth, identity explained more variation in flowering that did fixed effects, suggesting that individual responses to transplanting or potentially, unmeasured variation, may better relate to flowering.

Monitoring year was an important variable influencing both growth and flowering, and I suggest this may be related to weather patterns over the monitoring period. The observed decline in the size of wild plants of *C. oligosperma* suggests this species may be sensitive to factors such as temperature or precipitation, potentially reflected by significant contributions of year in growth and flowering models. Cumulative rainfall between May and August was 130 – 170 mm less in 2015 and 2017 than 2016 for the Fort McMurray area (Environment Canada), representing drought conditions in both of these years where peatland water tables were potentially reduced. Donor sites for *C. oligosperma* and other northern wetlands where this species has been observed often had standing water, especially in 2014, the year of transplanting. In subsequent monitoring years however, these wetlands were much drier during the late summer visit. To my knowledge, there is no published information on the habitat preferences of *C. oligosperma*, other than being an obligate wetland species, and none on response to climatic factors. It is possible that the observed reduction in flowering in 2016 was related to precipitation, and that decline in the size of wild individuals over these three years reflects adverse responses to drought. Finally, in one of few published observations of *C. oligosperma*, the species was shown to germinate at very low rates (<1%) despite reasonable seed viability (33%) in peatland reclamation experiments in Quebec (Laberge et al. 2015). This is in contrast to greater germination and establishment of two ecologically similar species, *Carex limosa* L. and *Carex magellanica* Lam. (Laberge et al. 2015). Although the use of seed in translocation often yields poor results (Godefroid et al. 2011), these findings, in addition to my own, could potentially indicate a lack of amenability of *C. oligosperma* to movement from its natal site. It is probable that the lack of rhizomatous connections at recipient sites was a factor in the observed stunting of *C. oligosperma* transplants, although as noted no true control for the effects of transplanting was considered here.

The high survival of transplants after three years of monitoring and across variable recipient sites demonstrated here is unusual among examples from the literature, which more frequently report high success in the first year after translocation with an ensuing rapid decline (Drayton & Primack 2000; Godefroid et al. 2011; Cypher et al. 2014). In a 2011 review, Godefroid et al. reported an average success rate of translocations (including re-introduction and augmentation projects) of roughly 80% after three to four years from published literature, but when combined with results from a survey, this figure decreased to 50%. However, it is important to note that survival, even after three years, may not always correlate with long term success (Drayton & Primack 2012), and an important measure of translocation success is reproduction and germination. Flowering rates for *C. oligosperma* and *S. purpurea* are higher than what has been observed in other translocations, a hopeful indicator for successful future reproduction (Godefroid et al. 2011). Production of a second generation at recipient sites would confirm the ability of both of these species to regenerate under recipient site conditions, another dimension of site suitability. No new, separately rooted individuals of *C. oligosperma* growing outside the transplant margin were ever noted at recipient sites, however, locating vegetative stems of this sedge can be challenging and thus plants would likely need to flower to be detected. There were observations of *S. purpurea* seedlings; however these always occurred within the margin of the original transplant, on surviving peat substrate translocated from the original donor site. Until new individuals are observed growing outside transplant margins, this project can only consider success as survival of original transplants.

Therefore, these transplants are successful with the caveat that further decline in the number of stems produced annually by *C. oligosperma* may lead to eventual transplant failure. Findings for *C. oligosperma* highlight the importance of monitoring growth or another health metric in transplants, especially when species tolerances are unknown (Maslovat 2009). It appears reasonable to assume that over a longer timeframe then what is considered here *S. purpurea* may form self-sustaining populations at recipient sites. Given this, I suggest restricting mitigative translocations to species whose ecology is relatively well understood, although this is uncommon for many species of conservation concern (Maslovat 2009). When mitigative translocations can focus on species likely to survive and establish in recipient sites, their efficacy is increased, and monitoring schemes may be able to invest revisitation efforts on a longer time rotation than for those species whose amenability to transplanting is unknown. While this work had the benefit of being able to monitor individuals at the original donor site, in the case of real-world mitigative translocations the donor site will be destroyed, emphasizing that monitoring transplants and reporting findings is critical in order to gain understanding of what factors may increase success (Germano et al. 2015). Translocation of species whose tolerances are unknown may be best approached on an experimental basis, perhaps through site preparations or varying recipient site characters as was done here, rather than considered an active conservation strategy. This may be the most effective use of resources available for mitigative translocations and provide the greatest learning opportunities.

Finally, future work with mitigative translocations may benefit by incorporating strategies learned from these experimental transplants. First, issues with relocation are a noted problem in translocations within the oil sands region (L. Halsey, pers. comm, K. Mackenzie pers.comm). I was relatively successful in locating transplants, however planting on a grid pattern and using tall, permanent markers is strongly encouraged to prevent failure to relocate and extensive time spent searching for transplants, both of which could decrease the efficacy of a monitoring program. Species traits and relocation should be considered, as in the case of strongly rhizomatous growth limiting the ability to determine control transplants of *C. oligosperma* at donor sites. I further suspect that translocations of peatland species may be more successful than those of upland species when monitoring and follow up resources are limited. While this is speculative, post-transplant watering is known to be important for survival of transplanted upland species (Godefroid et al. 2011), and short-timeframe mitigation projects may not be able to provide this level of support. Peatlands are an extensive landcover type in the oil sands area of Alberta, and mitigative translocation of peatland species is known to occur, if rarely (K. Mackenzie, pers. comm). Assuming peatlands are selected with some knowledge of average water table fluctuation over time and that watering upland species is a limiting factor, it may be that eliminating the need to water could increase success of transplants, although published work on rare peatland species translocation is scarce. This work has hopefully begun to address questions related to peatland mitigative translocations and I look forward to future experiments with other peatland species.

Table 5.1. Survival, growth, and flowering counts at recipient sites of transplanted *Sarracenia purpurea* and *Carex oligosperma* over three monitoring years.

Sarracenia purpurea						***Carex oligosperma***					
Site	Relocated	Living	%	Flowering	%	Site	Relocated	Living	%	Flowering	%
2015						2015					
SR1	50	50	100	21	42	OR1	46	44	96	17	37
SR2	49	49	100	25	51	OR2	50	45	90	18	36
SR3	45	45	100	18	40	OR3	47	46	98	15	32
Total	**144**	**143**	**99**	**64**	**44**	**Total**	**143**	**135**	**94**	**50**	**35**
2016						2016					
SR1	48	47	98	8	17	OR1	47	42	89	4	9
SR2	49	48	98	22	45	OR2	50	41	82	4	8
SR3	48	48	100	28	58	OR3	45	42	93	7	16
Total	**145**	**143**	**99**	**58**	**40**	**Total**	**142**	**125**	**88**	**15**	**11**
2017						2017					
SR1	47	44	94	8	17	OR1	47	43	91	28	60
SR2	48	47	98	30	63	OR2	49	39	80	13	27
SR3	48	48	100	5	10	OR3	45	38	84	17	38
Total	**143**	**139**	**97**	**43**	**30**	**Total**	**141**	**120**	**85**	**58**	**41**

Table 5.2. Flowering, growth, survival, and relocation rates for 60 control transplants of *Sarracenia purpurea* across three donor sites.

Site	Relocated	Living	%	Flowering	%
2015					
SD1	20	20	100	9	45
SD2	18	18	100	7	39
SD3	20	20	100	5	25
Total	**58**	**58**	**100**	**21**	**36**
2016					
SD1	16	16	100	8	50
SD2	17	17	100	12	71
SD3	20	20	100	2	10
Total	**53**	**53**	**100**	**22**	**42**
2017					
SD1	17	17	100	6	35
SD2	18	18	100	11	61
SD3	20	20	100	4	20
Total	**55**	**55**	**100**	**21**	**38**

Table 5.3. Results of model comparison for candidate generalized linear mixed-effect models expected to influence annual growth over three years of transplanted *Sarracenia purpurea* across three monitoring years (*n* = 304). All models included a random effect transplant identity.

Name	Model	*K*	AIC	ΔAIC
site effects	magnesium concentration + donor site + recipient site + year	5	1919.49	0.00
vegetation 1	beta diversity + total cover at recipient site + year	4	1949.45	29.96
Year	Year	2	1950.36	30.87
vegetation 2	beta diversity + change in cover between donor and recipient site + year	4	1951.20	31.71
Null	Null	1	1953.14	33.65
initial size	initial number of pitchers + year	3	1955.69	36.20

Table 5.4. Parameters of the best supported generalized linear mixed-effect model explaining variation in growth for transplanted *Sarracenia purpurea* across three monitoring years (*n* = 304). The random effect of transplant identity had a SD = 0.

Parameter	β	SE	*p*
Intercept	3.45	0.97	<0.001
Magnesium concentration (mg/L)	4.38	2.58	0.090
Year			
2016	0.19	0.87	0.830
2017	-0.90	0.79	0.267
Donor site			
SD1	0.70	0.88	0.429
SD3	-0.59	0.79	0.454
Recipient site			
SR1	1.82	2.58	0.480
SR3	-3.84	3.96	0.321

Table 5.5. Results of model comparison for candidate mixed-effect logistic regression models expected to influence flowering of transplanted *Sarracenia purpurea* across three monitoring years (n = 307). All models included a random effect of transplant identity.

Name	model	K	AIC	ΔAIC
site effects	total phosphorus + donor site + recipient site + year	5	383.88	0.00
initial size	initial number of pitchers + year	3	394.59	10.71
Null	null	1	404.02	20.14
Year	year	2	405.76	21.88
vegetation 1	beta diversity + total cover at recipient site + year	4	does not converge	
vegetation 2	beta diversity + change in cover between donor and recipient site + year	4	does not converge	

Table 5.6. Parameters of best supported candidate mixed-effect logistic regression model explaining variation in flowering for transplanted *Sarracenia purpurea* across three monitoring years (n = 307). The random effect of transplant identity had a SD = 0.53.

Parameter	β	SE	p
Intercept	1.21	0.61	0.046
Total phosphorus (μg/L)	-0.30	0.36	0.395
Year			
2016	-0.39	0.32	0.221
2017	-0.85	0.60	0.155
Donor site			
SD1	0.07	0.36	0.842
SD3	-1.16	0.34	<0.001
Recipient site			
SR1	-1.71	0.66	0.009
SR3	-1.14	0.58	0.049

Table 5.7. Results of model comparison for candidate generalized linear mixed-effect models expected to influence annual growth of transplanted *Carex oligosperma* across three monitoring years (*n* = 327). All models included a random effect of transplant identity.

Name	model	*K*	AIC	ΔAIC
initial size	initial number of stems + year	3	1707.87	0.00
site effects	donor site + recipient site + year	4	1764.87	57.00
Year	year	2	1773.25	65.38
vegetation 1	beta diversity + total cover at recipient site + year	4	1776.95	69.08
vegetation 2	beta diversity + change in cover between donor and recipient site + year	4	1778.51	70.64
Null	null	1	1875.86	167.99

Table 5.8. Parameters of the best supported generalized linear mixed-effect model explaining variation in growth of transplants *Carex oligosperma* across three monitoring years (*n* = 327). The random effect of transplant identity had an SD = 0.

Parameter	β	SE	*p*
Intercept	-2.15	1.60	0.597
Year			
2016	4.61	0.43	<0.001
2017	4.79	0.43	<0.001
initial number of stems	-0.27	0.03	<0.001

Table 5.9. Results of model comparison for candidate mixed-effect logistic regression models expected to influence flowering of transplanted *Carex oligosperma* across three monitoring years (*n* = 332). All models included a random effect of transplant identity.

Name	model	*K*	AIC	ΔAIC
vegetation 1	beta diversity + total cover at recipient site + year	4	371.09	0.00
vegetation 2	beta diversity + change in cover between donor and recipient site + year	4	372.74	1.64
Year	year	2	377.02	5.92
initial size	initial number of stems + year	3	378.73	7.64
Null	null	1	421.69	50.60
site effects	donor site + recipient site + year	4	model fails to converge	

Table 5.10. Parameters of best supported candidate mixed-effect logistic regression model explaining variation in flowering for transplanted *Carex oligosperma* across three monitoring years (n = 332). The random effect of transplant identity had a SD = 0.85.

Parameter	β	SE	p
Intercept	0.96	0.56	0.084
Year			
2016	-1.99	0.42	<0.001
2017	0.51	0.30	0.093
beta diversity	-1.39	0.61	0.023
total cover at recipient site	-0.02	0.01	0.169

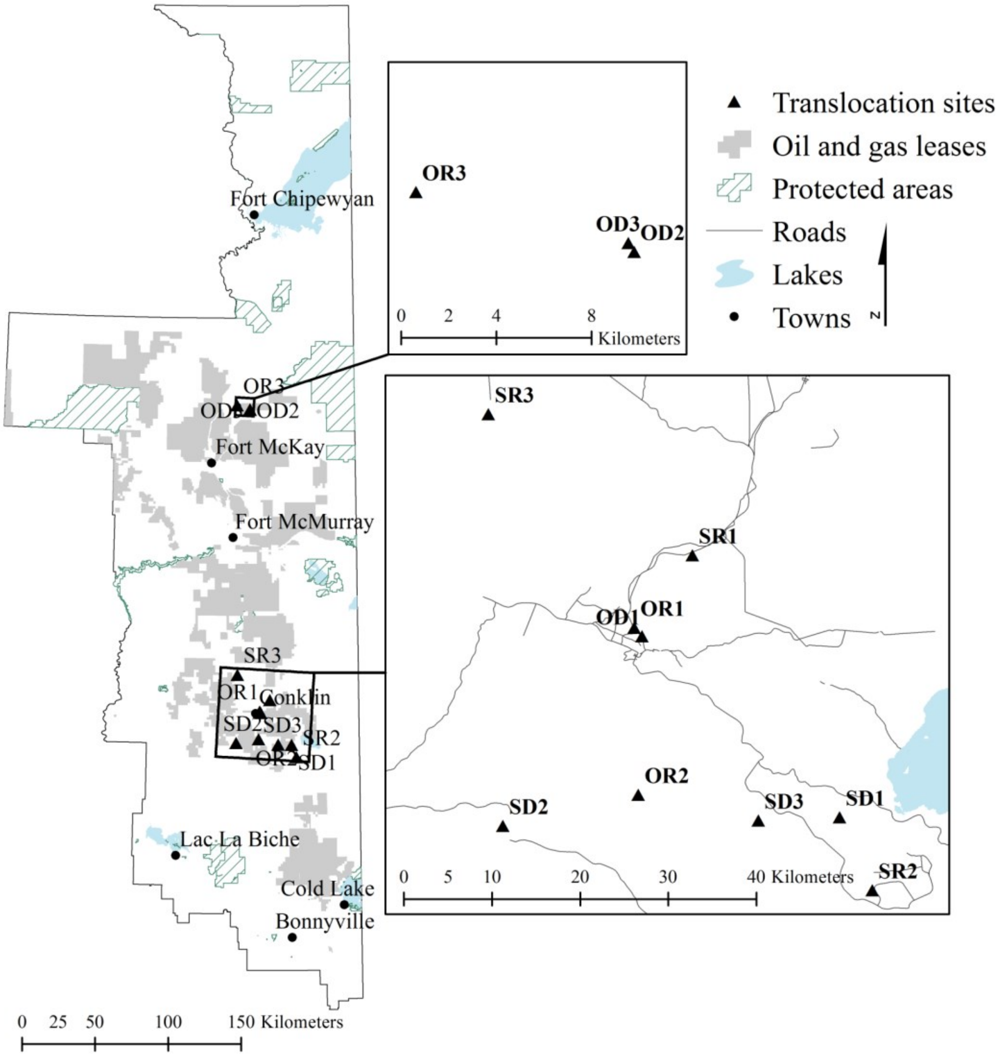

Figure 5.1. Location of 12 experimental translocation sites in northeastern Alberta. *Sarracenia purpurea* sites are denoted by S, *Carex oligosperma* sites by O. The letters R and D refer to recipient and donor sites, respectively. Numbers 1, 2, and 3 indicate replicates.

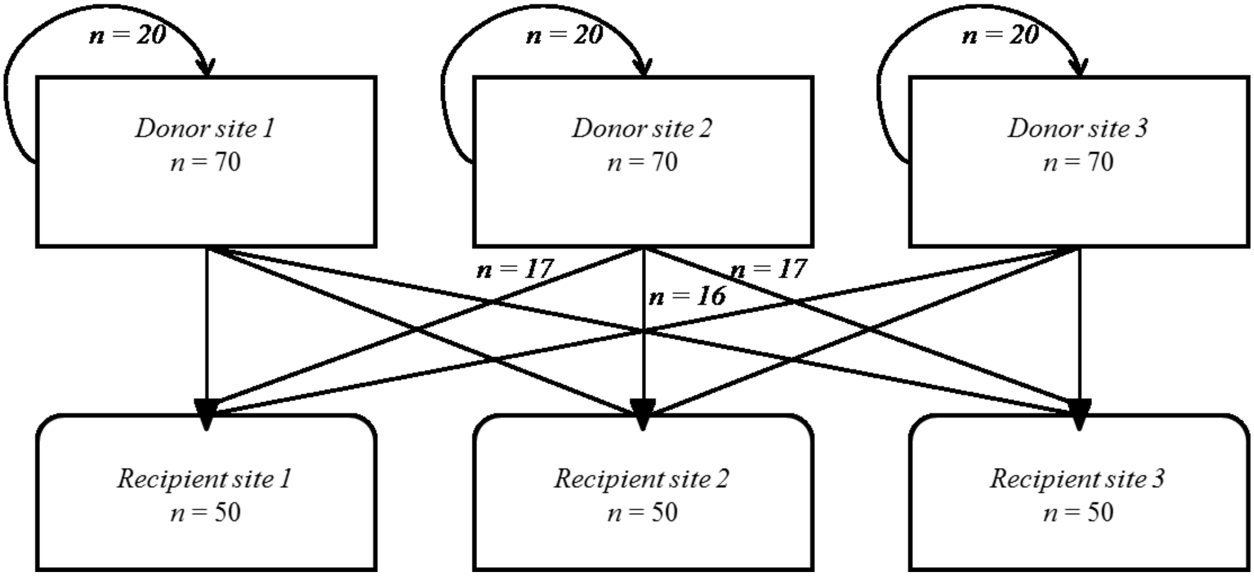

Figure 5.2. Study design schematic used in mitigative transplantations, where 210 transplants for each *Sarracenia purpurea* and *Carex oligosperma* were moved among three donor and recipient sites, respectively. Note that the 20 individuals of *C. oligosperma* transplanted at each donor site were not included in later analysis due to difficulty in relocation.

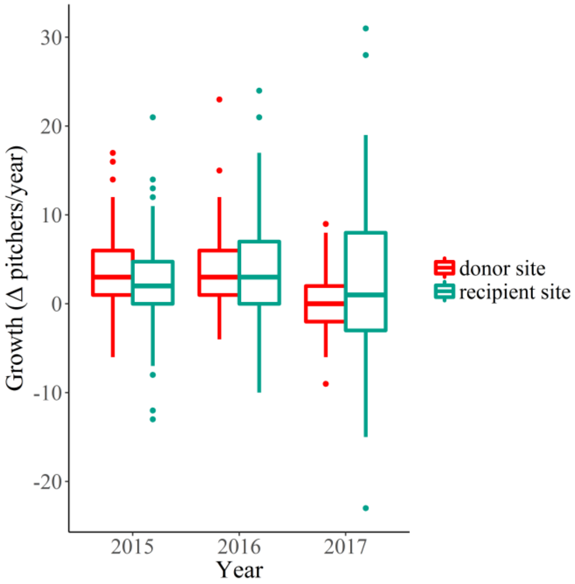

Figure 5.3. Boxplots of annual growth in transplanted *Sarracenia purpurea* over three years of monitoring. Donor site individuals are those transplants immediately replanted at their natal site, recipient site are those moved to one of three recipient sites.

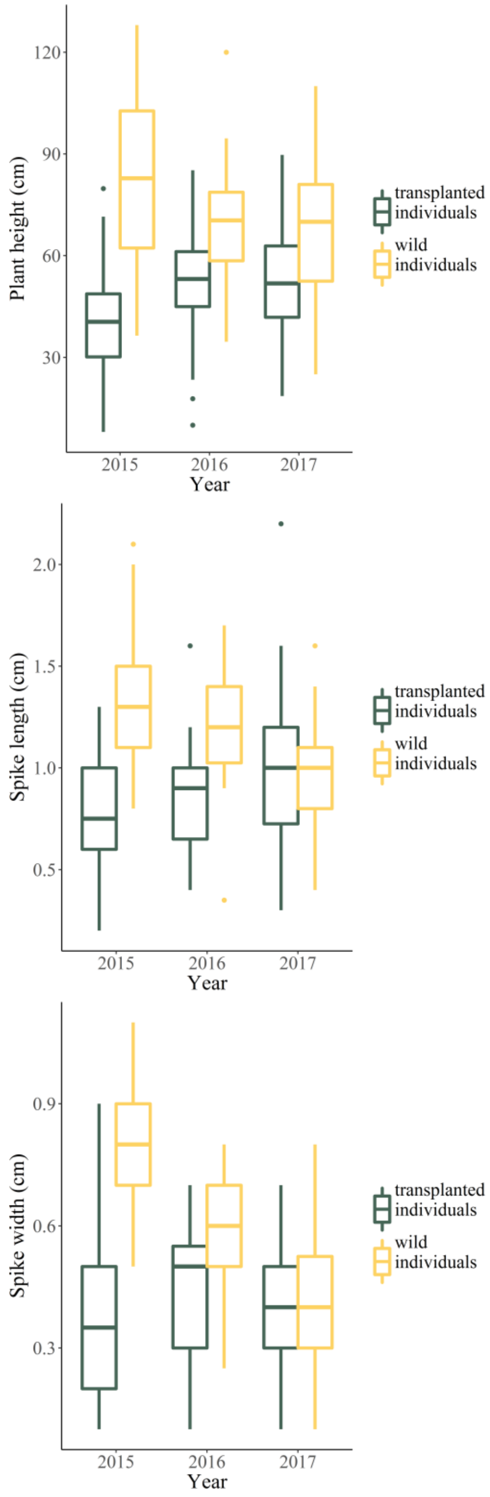

Figure 5.4. Boxplots of plant height and female spike length and width between transplanted *Carex oligosperma* and wild individuals occurring naturally at donor sites (30 individuals/donor site/year).

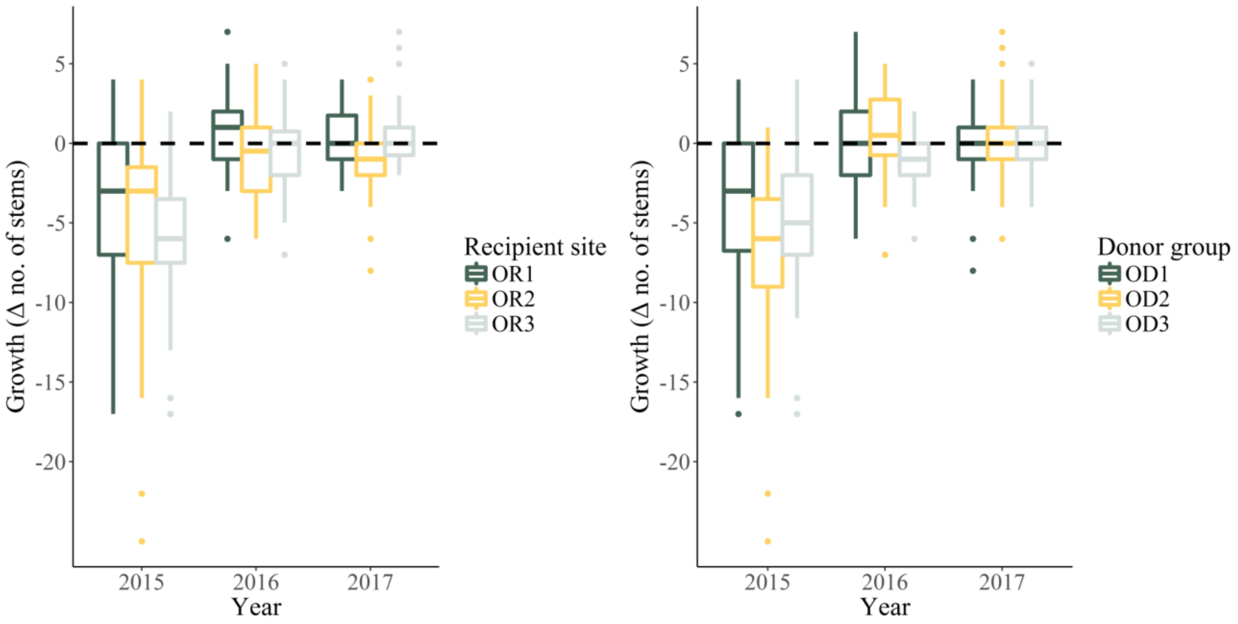

Figure 5.5. Boxplots of growth over three monitoring years, as measured by the change in the number of stems produced by each transplant, across recipient sites for *Carex oligosperma.* Note that donor group refers to the identity of the natal site of transplants at recipient sites.

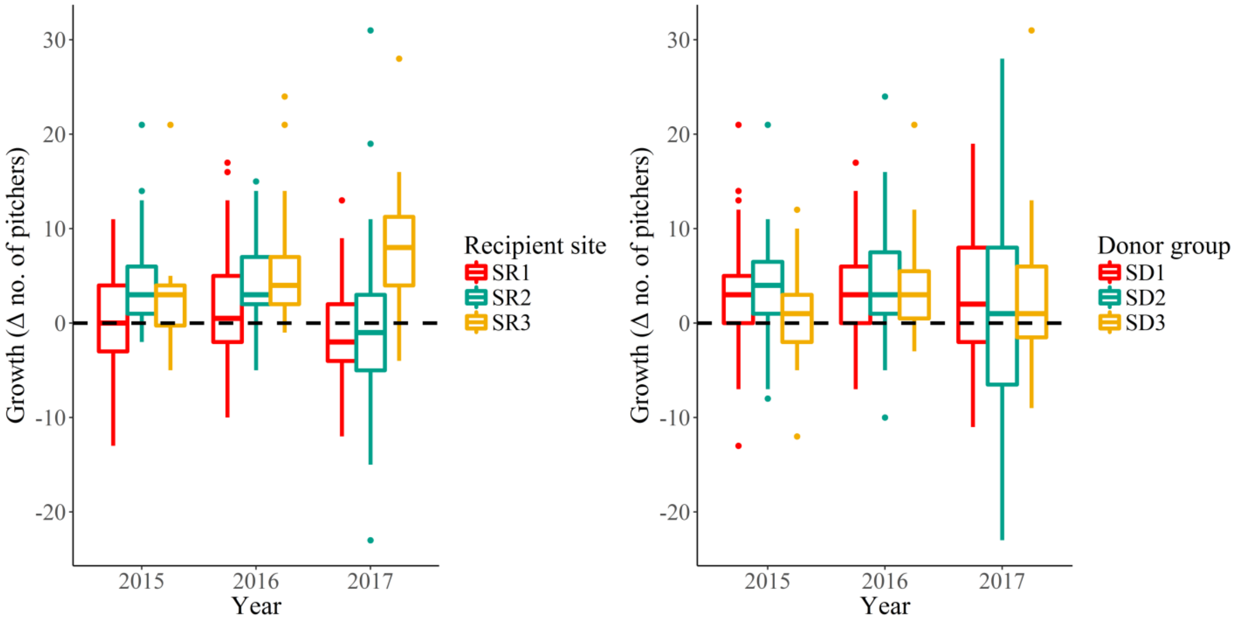

Figure 5.6. Boxplots of growth over three monitoring years, as measured by the change in the number of pitchers on a transplant, across recipient sites for *Sarracenia purpurea.* Note that donor group refers to the identity of the natal site of transplants at recipient sites.

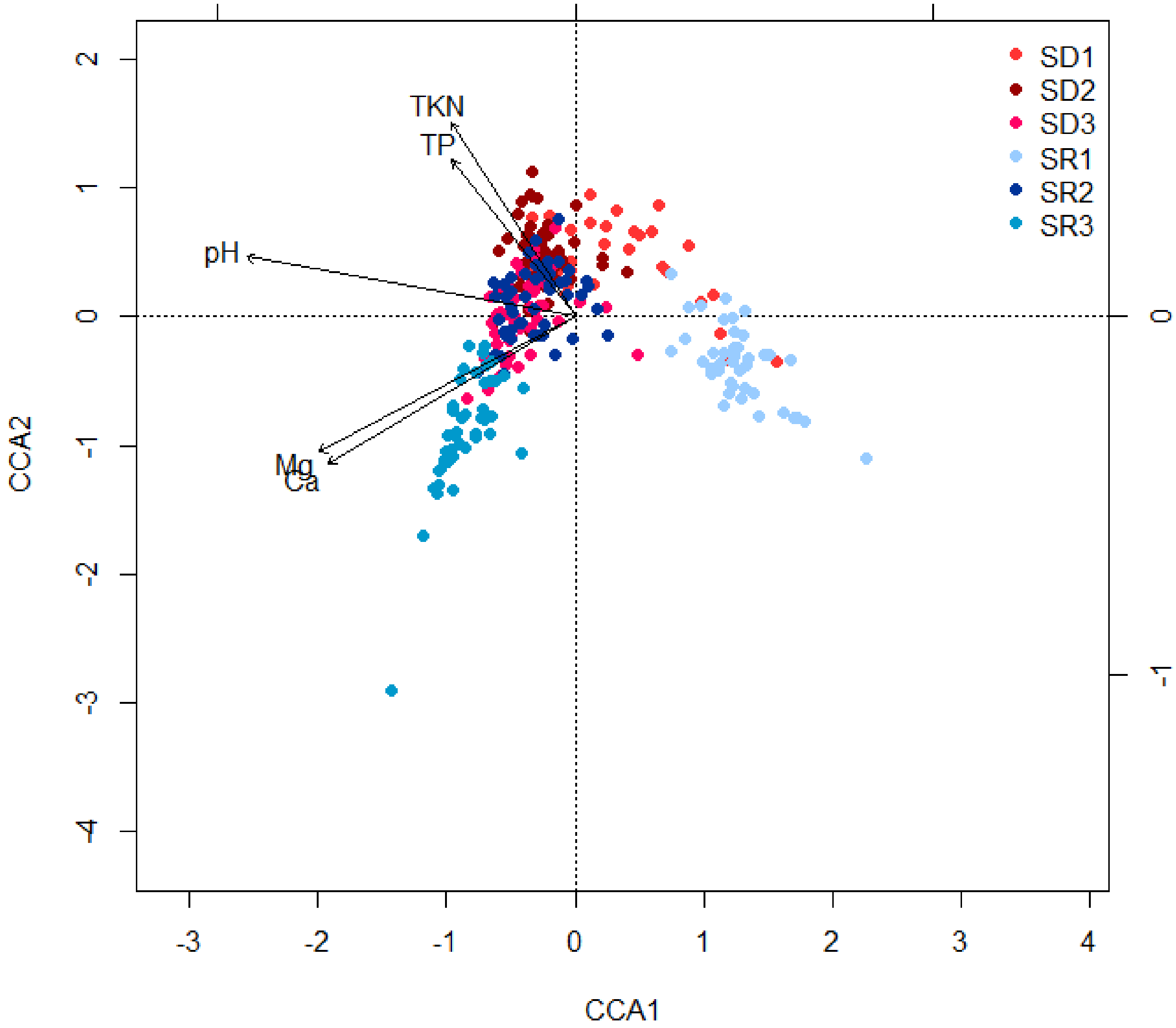

Figure 5.7. Canonical correspondence analysis of nutrient and mineral concentrations and vegetation assemblage of donor and recipient sites for *Sarracenia purpurea.* The first axis relates to pH (46%) and the second to total Kjeldahl nitrogen (21%). Variation explained (mean squared contingency coefficient) was 0.23.

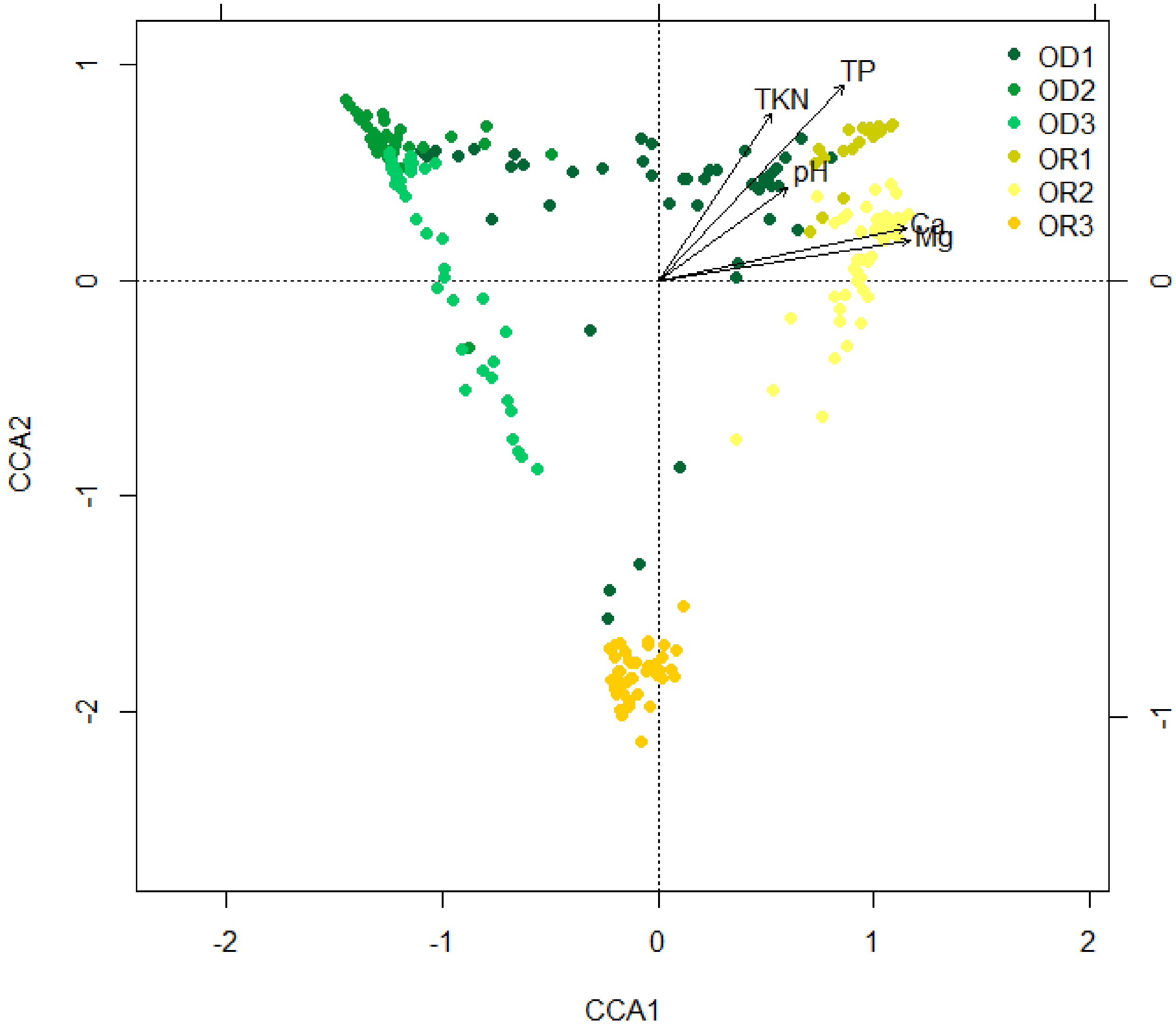

Figure 5.8. Canonical correspondence analysis of nutrient and mineral concentrations and vegetation assemblage of donor and recipient sites for *Carex oligosperma.* The first axis relates to pH (80%), the second to total Kjeldahl nitrogen (66%). Variation explained (mean squared contingency coefficient) was 0.53.

Chapter 6: Conclusion

In this thesis, I considered the detection, persistence, and mitigation of rare vascular plant populations. These three topics relate to actions that enable conservation, specifically, surveying, monitoring, and mitigation. Developing survey methodology and recommendations to effectively and efficiently survey and mitigate for rare plants is challenging, and in some cases understanding of these practices is still developing. My findings, combined with those of previous study on these important topics, provide ideas of how to best improve these actions and help inform the development of future guidelines.

In Chapter 2 I discuss controlled detection trials designed to understand how scale, abundance, and observer and species-related factors influence detection. Field trials using controlled surveys where variables of interest such as scale are investigated are uncommon (Moore et al. 2011). My results suggest that observers are unlikely to detect species at low abundance over large search areas, that repeat surveys may compensate for low detection probability, and that flowering or showy plant characters cannot guarantee success when plants are scarce. Specifically, the outcomes of these trials speak to potential tradeoffs in total area searched, which we most often wish to maximize in hopes of encountering species, and the likelihood of detection over such areas. My detection trials, which build on those of Moore et al. (2011), and others investigating survey success given observer traits and species abundance (Alexander et al. 2011, Garrard et al. 2014, McCarthy et al. 2012), clearly demonstrate that, excluding exceptionally showy species such as *Petunia*, the amount of effort that individuals naturally expend over large plots is generally insufficient to achieve reasonable detection success. Specifically, there was a general trend of effort expenditure of 0.03 – 0.06 minutes/m^2 over plots of 1000 m^2, i.e., 30 to 60 minutes, which most commonly yielded 30 – 50% success in these targeted surveys. In smaller plots with higher effort, I observed reasonable detection success with efforts ranging from 0.1 - 3.0 minutes/m^2 for plots of 1 and 10 m^2.

I believe a key message in these results is that survey effort is highly variable among observers, regardless of their background, and should be standardized in surveys. Effort and success declined rapidly with increasing plot size, and therefore I would advocate for the use of small plots or belt transects to search areas in lieu of covering greater amounts of area to ensure that effort remains high. Although this was not directly tested here, it seems a reasonable hypothesis that these smaller search areas would act to focus the observer and force their attention to a small area, rather than becoming overwhelmed by expansive search areas.

Further, these findings directly relate to future survey guidelines for the province of Alberta. Survey guidelines that explicitly address required survey effort (time and area) were published by the Government of New South Wales, Australia in 2016 (State of New South Wales 2016), those for Alberta were last updated in 2012 (ANPC 2012). I believe the province and botanical community in Alberta should aim to create explicit guidelines such as these and incorporate findings from recent work concerning imperfect detection (Garrard et al. 2008; Chen et al. 2013; Zhang et al. 2014). Gathering data to inform such guidelines could be done through collections of detailed survey data from individuals (or consulting firms) conducting rare plant surveys. This should include a mandatory submission of survey effort (survey time and area), but could also extend to collecting other information at survey sites such as GPS data logger points or mapped polygons of search area with associated effort, especially when plants are not encountered. Currently, one of ANPC's most explicit guidelines focuses on the training and background of the observer (Alberta Native Plant Council 2012), but my results demonstrate that, at least in targeted surveys for single species, experience may not convey a significant advantage.

In Chapter 3 I again focused on detection, here in an uncontrolled field setting using full inventory surveys, in contrast to my second Chapter which employed targeted surveys. Here I had an emphasis on graminoid morphology and abundance and forest structure. My examination of graminoids and use of *Carex* as a model group was prompted by previous findings of reduced detection or high taxonomic uncertainty among these thin-profile plants when compared to other growth-forms (Chen et al. 2013; Garrard et al. 2013). My results indicated that abundance, site structure (vegetation density), and species morphology influence detection failures. Morphology did not influence delays, which suggests that very short-term (being spatial or temporal in this sense) overlooking (for instance, becoming momentarily distracted by site attributes) by an individual will occur regardless of species form, but complete failure to detect is far more likely for cryptic species. Minimizing distractions for the observer, or recognizing that delays are more likely where observers are physically impeded by tall shrubs and allocating additional effort in complex sites will improve detection.

When we recognize the magnitude by which broad traits can reduce detection, as I have demonstrated here between large aquatic and short, cryptic species of sedge, we can begin to quantify potential bias across a broader range of species, or correct for imperfect detection post-survey using morphology as predictor of detection probability (Lele et al. 2012). I believe my work provides insight for how to approach challenging groups of species who share morphology during surveys. Specifically, species can be assigned into broad groupings based on morphology or functional traits which reflect different levels of mandatory effort, as is suggested for different life-forms in the recently published New South Wales plant survey guidelines (State of New

South Wales 2016). Based on my findings with short *Carex*, even an attribute as simple as average height for a species could be used to delineate groups in lieu of any more detailed information. This could further extend to performing inventory surveys using a strata-based approach, where observers search each strata (height category, e.g., 0 – 20 cm) independently, generating a species list for each strata, and thus focus their attention on short species during survey, as is done for a broad scale monitoring program in New Zealand (Green & McNutt 2012).

Here, I applied a constant survey effort of 0.15 minutes/m^2 over 200 m^2 belt transects for each observer, and achieved both low pseudoturnover (a promising finding indicating minimal differences in observer species lists) and reasonably high detection of challenging species (but note a detection bias against short *Carex* with a small inflorescence). Combined with the results from my targeted surveys in Chapter 2, I would suggest that effort of 0.15 minutes/m^2 could be used as a working lower limit for recommended survey effort until further data become available. This level of effort is high given what is currently exerted in the boreal region of Alberta. For example, the Alberta Biodiversity Monitoring Institute currently uses a survey effort of 0.008 minutes/m^2 to create species inventories over plots of 2500 m^2. This is far lower than what observers naturally exhibited in my targeted trials, and lower than the effort which I deemed reasonable and reported encouraging findings in species inventory belt transects. It is interesting to note that the maximum effort in terms of hours/hectare suggested by the New South Wales survey guidelines amounts to 0.03 minutes/m^2 within dense vegetation (presumably forested areas in some cases, or those with higher shrub cover) while the New Zealand monitoring scheme using height strata is time unlimited and does not suggest explicit effort.

My collective findings, moreover, demonstrate that consistent detection of large, showy species and those with large local populations can be achieved at lower effort, in some cases less than half that required to detect cryptic ones. This has implications for future targeted survey planning, such as those for noxious weeds or rare species. Surveys for large, showy species can be made more efficient by allocating less survey effort where they are targeted, in Chapter 4 I report that nearly all populations >50 individuals were detected upon arrival, although in this case observers were returning to sites where the species was known to have previously occurred. Survey findings should always be reported with associated survey effort, as this will allow for evaluation of reported absences. Collectively, I believe my results provide insights into detection in forested boreal environments, and I hope they will promote further investigation of these interesting questions.

Once populations of rare species are identified, we hope that they will persist through time either without intervention, or through mitigation employed to preserve them where they overlap with development. Historical population records have been used previously to estimate species loss in relation to land-use changes, where findings suggest that such changes are linked to reduced geographic ranges or extirpation (Lienert et al. 2002; Lienert & Fischer 2003; Stehlik et al. 2007). Interestingly, in Chapter 4 I found no correlation between rare plant extirpation and a direct measure of oil and gas footprint, a major land-use change in the boreal forest. It is likely that an extinction debt exists in the oil and gas region for some species, particularly given the young age of many developments, relatively recent construction, and observed reduction in upland species populations of up to 40%. But I also noted that some species, specifically those with ruderal traits (Grime 1979), could be promoted through the types of features that oil and gas footprints create.

My investigation of the amount and type of footprint surrounding all historic rare plant populations in the oil sands region provides a baseline that future work can build upon. Given recent increases in the number of in-situ leases approved by the province, I would anticipate the proportion of disturbance around rare plant populations to increase, and my reporting of current levels can provide a starting point to track change. Further, if an extinction debt exists, populations are likely in the process of paying off this debt, and it is clearly demonstrated that these lags can persist well into the future from features such as roads changing local hydrology, a common occurrence in this region (Findlay & Bourdages 2000; Miller et al. 2015). Given ongoing development and my findings of an estimated 77% persistence rate of historical vascular plant populations in the Oil Sands Area, future monitoring efforts are important to develop an understanding of rate of loss over time and ensure the veracity of these historical records before they are used to rank species conservation status. I would suggest a monitoring program be developed, where resources allow, and focus on a range of species with dissimilar traits (i.e., ruderal vs. stress-tolerant) at varying levels of footprint, while prioritization those which are provincially and globally rare (e.g., S1/S2 or G3 or higher), and that such efforts would provide valuable information regarding population loss over time.

Mitigation practices in the oil sands area cover a range of passive and active techniques. In Chapter 5 I examined mitigative translocation (Germano et al. 2015) as one active technique. While translocation has a well-documented, lengthy history (Grayson 2001; Godefroid et al. 2011), this study addressed two factors rarely considered in the literature. First, short-timeline (i.e., less than a single growing season for planning and completion) mitigative translocation for vascular plants is poorly documented, and second, to my knowledge there are very few examples of translocation of peatland species, and none that specifically address transplanting in peatlands as a mitigation strategy or use sample sizes as large as what was used here. In the case of my study, translocation appeared to be successful in terms of survival of two species, *Sarracenia purpurea* and *Carex oligosperma*, after three years of monitoring. Both species appear to be more dispersal-limited than environmentally-limited, not uncommon in a region as young as the boreal forest. Future studies using peatland species could reveal whether these plants are more amenable to translocation than upland species, potentially due to their not requiring post-transplant monitoring. My findings of large initial reduction in stems of *Carex oligosperma* with limited growth in ensuing monitoring years indicates that monitoring health or growth may be as important as survival in understanding success, especially when there is little ecological information available about the species being transplanted. Dissimilarity in donor and recipient sites in terms of vegetation assemblage, the measure most easily and readily assessed in the field, does not appear to be an important determinant of success and therefore cannot be used to guide recipient site selection for these two species, as has been the case in other examples (Lawrence & Kaye 2011).

Currently, no guidelines exist for translocation in the province, but see Maslovat (2009) for well-researched best practices for British Columbia. Critical pieces to be incorporated into best practices at the provincial level and for the Oil Sands Area include minimum founder population guidelines, mandatory monitoring periods of at minimum three years where this practice is undertaken, and most critically, standardized reporting of measures such as survival, growth, and establishment of a second generation. This will allow determination of factors influencing success across projects and species and ensure that the extensive resources invested in translocation efforts achieve conservation goals (Germano et al. 2015).

Overall, my thesis provides important information related to how we search for rare plants and when these searches may be most effective, an example for revisitation surveys to ascertain longevity of rare vascular plants in the oil sands area, and a framework for evaluating mitigation translocation when species traits are poorly understood.

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Appendices

Appendix 1

Figure A1.1. The four decoy species used in Trials One and Two, clockwise from upper left: *Symphyotrichum lanceolatum*, *Viola pedatifida*, *Petunia* sp., *Allium cernuum*.

Table A1.1. Parameters of a full mixed effect cox model of the detectability of both target species from Trial One. *Symphyotrichum lanceolatum* was used as the reference category for the variable "species". Plant height is the maximum height of the plant; plant width is the average width of the plant. Plot area was $\log_{10}$ transformed, (*n* = 83).

Parameter	coef (β)	SE (β)	exp (β)	*p*
plot size	-1.98	0.22	0.14	<0.001
species	0.98	0.48	2.67	0.039
survey order	0.15	0.08	1.16	0.057
experience level 2	-0.11	0.37	0.90	0.770
experience level 3	0.15	0.41	1.16	0.720
plant height	0.04	0.04	1.04	0.350
average width	-0.03	0.04	0.97	0.370
horizontal cover	0.00	0.01	1.00	0.770
Random effect	SD			
plot	0.34			
observer	0.42			

Table A1.2. Parameters of a full mixed effect cox model of the detectability of *Symphyotrichum lanceolatum* in Trial One. Plant height is the maximum height of the plant; plant width is the average width of the plant. Plot area was $\log_{10}$ transformed, (*n* = 83).

Parameter	coef (β)	SE (β)	exp (β)	*p*
plot size	-2.82	0.43	0.06	<0.001
survey order	0.04	0.12	1.04	0.750
experience level 2	0.62	0.85	1.86	0.460
experience level 3	0.50	0.96	1.65	0.600
plant height	0.10	0.07	1.10	0.200
average width	0.01	0.09	1.01	0.870
horizontal cover	0.00	0.01	1.00	0.910
Random effect	SD			
plot	0.02			
observer	1.25			

Table A1.3. Parameters of a full mixed effect cox model of the detectability of *Viola pedatifida* in Trial One. Plant height is the maximum height of the plant; plant width is the average width of the plant. Plot area was $\log_{10}$ transformed, (n = 83).

Parameter	coef (β)	SE (β)	exp (β)	*p*
plot size	-1.99	0.30	0.14	<0.001
survey order	0.33	0.11	1.38	0.002
experience level 2	-0.48	0.38	0.62	0.210
experience level 3	0.30	0.39	1.35	0.440
plant height	-0.09	0.09	0.91	0.310
average width	-0.04	0.04	0.96	0.360
horizontal cover	0.01	0.01	1.01	0.640
Random effect	SD			
plot	0.02			
observer	0.02			

Table A1.4. Parameters of a full mixed effect cox model of the detectability of both species used in Trial Two. *Petunia* was used as the reference in the variable “species”. Plant height is the maximum height of the plant; plant width is the average width of the plant. Clumped was the reference category for the variable “arrangement”. Abundance was $\log_{10}$ transformed, (n = 53).

Parameter	coef (β)	SE (β)	exp (β)	*p*
species	-2.29	0.32	0.10	<0.001
abundance	1.88	0.51	6.52	<0.001
arrangement	0.75	0.36	2.12	0.037
survey order	0.32	0.11	1.37	0.005
no. seasons	0.03	0.04	1.03	0.420
Random effect	SD			
plot	0.40			
observer	0.17			

Table A1.5. Parameters of a full mixed effect cox model of the detectability of *Petunia* sp. in Trial Two. Plant height is the maximum height of the plant; plant width is the average width of the plant. Clumped was the reference category for the variable "arrangement". Abundance was $\log_{10}$ transformed, (n = 53).

Parameter	coef (β)	SE (β)	exp (β)	*p*
abundance	1.88	0.56	6.57	0.001
arrangement	0.99	0.41	2.69	0.016
survey order	0.31	0.13	1.36	0.021
no. seasons	0.02	0.06	1.02	0.740
Random effect	SD			
plot	0.33			
observer	0.40			

Table A1.6. Parameters of a full mixed effect cox model of the detectability of *Allium cernuum* in Trial Two. Plant height is the maximum height of the plant; plant width is the average width of the plant. Clumped was the reference category for the variable “arrangement”. Abundance was $\log_{10}$ transformed, (n = 53).

Parameter	coef (β)	SE (β)	exp (β)	*p*
abundance	3.06	1.46	21.41	0.036
arrangement	0.87	0.83	2.40	0.290
survey order	0.34	0.22	1.40	0.120
no. seasons	0.04	0.07	1.04	0.540
Random effect	SD			
plot	1.12			
observer	0.02			

Appendix 2

As stated, I asked participants to wear Columbus V990 GPS data loggers (Victory Technology Co., Ltd.) during surveys in Trial Two to collect location data suitable for creation and analysis of movement paths. To relate detection to movement patterns, I measured observer movements as effective search paths in a GIS (ESRI 2015). Specifically, I created steps from data logger GPS waypoints (sample intensity of 1 location per second) using Geospatial Modeling Environment (Beyer 2015) and calculated tortuosity from these steps. Next, lines were buffered by a 1 m radius (2 m wide path) in ArcMap (ESRI 2015). I consider this path size conservative, but representative, of effective visual search distance in forested plots with tall shrub cover. Search area by each individual in each plot was then calculated as the proportion of each plot searched (total area of buffered path divided by plot size). Speed and tortuosity of search paths for each individual in each plot were calculated in R (R Core Team, 2016) using the package ‘adehabitatLT’ (Calenge 2006).

I was unable to include observer movement metrics in the main analysis as I did not collect seven observations from three individuals (their data logger was turned off during the survey), leaving only 12 individuals with 46 movement paths for analysis. Therefore, I fit a single full model containing an observers’ average speed within survey plots and tortuosity to determine the relative influence of these two parameters on detection. As in other models in the main analysis, I included observer and replicate plot as random effects in all candidate models. Models were built using the package ‘coxme’ (version 2.2-5) (Therneau 2015b) in R (R Core Team, 2016). Tortuosity and average speed did not differ among observers and were not related to detection success, the model built with the reduced dataset using these predictor variables was not supported over a null model ($\chi^2 = 7.1$, df = 4, $p = 0.13$) (Table A2.1). I observed uniform speeds across individuals ($\bar{x}$ = 0.14 meters/second, SE = 0.001). I visualized proportional search area as it relates to success (Figure A2.1), as search area is inherently confounded with success. Searches were ended when both target species were located, thus individuals who quickly located both species searched a small proportion of the plot.

Interestingly, there was a trend for the majority of *Allium cernuum* detections to occur when approximately 30% of the plot had been surveyed; surveys which covered a greater proportion of the plot most often resulted in a false absence (Figure A2.1). Chen *et al*. (2009) estimated detection probabilities of 95% could be achieved when the survey path approached 20% of the total search area (20 x 20m quadrat) in their study of nine target shrub species. Although I did not observe high detection at this threshold, I suggest that this search proportion may reflect the point at which the observer has seen, though not completely searched, the entirety of the plot. I hypothesize that observer fatigue may relate to the observed trend; searches may be more effective in their early stages before an observer becomes fatigued (Habib et al. 2012; Ransom 2012). It would be interesting to explore the influence of proportional search area and observer saturation in future study via the use of GPS data loggers (as was done here) to determine if this threshold applies across habitat types and plot sizes. Further, the use of GPS units on observers can allow for reporting of survey effort, for example in pre-disturbance surveys conducted as part of an environmental impact assessment.

Table A2.1. Parameters of a mixed-effect cox model fitted with movement metrics from Trial Two, *n* = 53. Av. speed is the average speed of an observer within a survey plot.

Parameter	coef (β)	SE (β)	exp (β)	*p*
av. speed	0.82	2.67	21.41	0.760
tortuosity	284.68	308.28	$4.33e^{123}$	0.360
Random effect	SD			
plot	0.66			
observer	0.02			

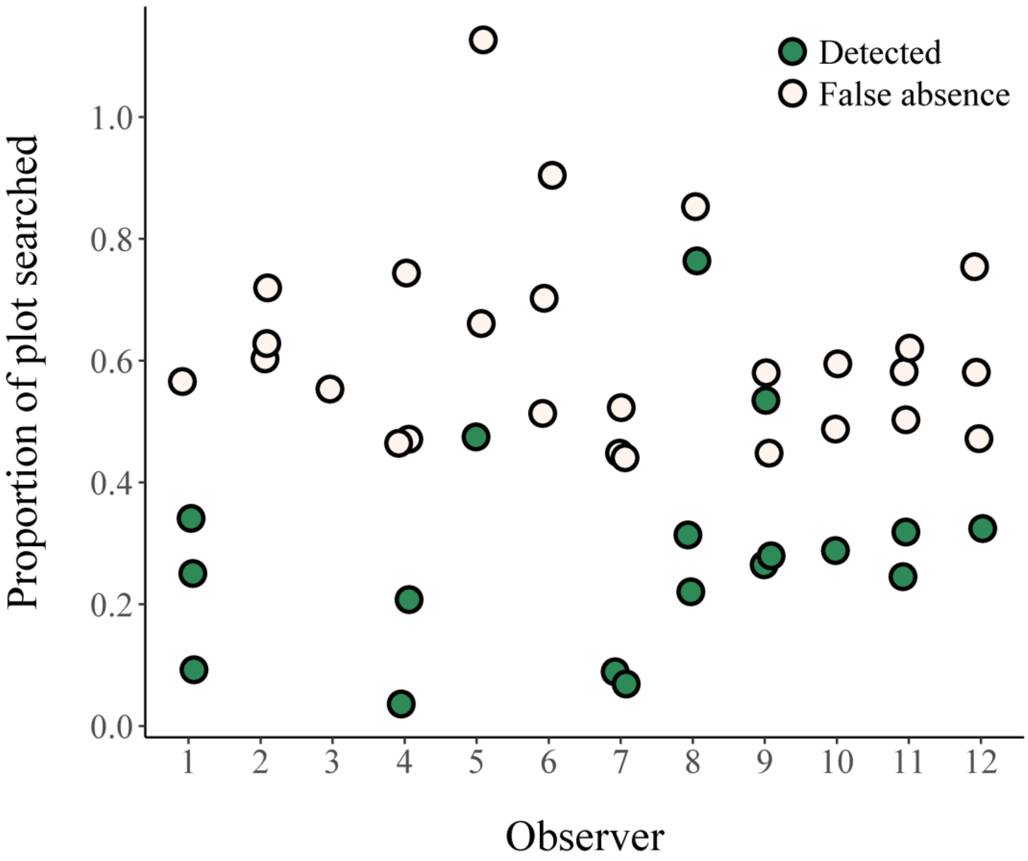


Figure A2.1. Proportion of the plot searched by each observer in Trial Two. Proportion of the plot represents effort (m^2) expended searching for *Allium cernuum*, and points are coloured by success in detection (teal), or false absences (pink).

Appendix 3

Table A3.1. Site locations for 50 experimental transects visited in 2015. Coordinates are given in UTM (Zone 12U), NAD83. Location is start of transect, bearing in the direction in which the 100 m transect was laid out.

Transect	Easting	Northing	Bearing	Date	Encountered species
1	458501	6095122	180	4/7/2015	*C. aquatilis, C. canescens, C. disperma, C. gynocrates, C. magellanica, C. tenuiflora*
2	437839	6094145	70	5/7/2015	*C. aquatilis, C. atherodes, C. canescens, C. diandra, C. lacustris, C. utriculata*
3	435961	6093639	180	5/7/2015	*C. siccata, C. tonsa*
4	463899	6087265	0	12/7/2015	*C. aquatilis, C. gynocrates, C. magellanica, C. prairea, C. tenuiflora*
6	458864	6091768	180	12/7/2015	*C. aquatilis, C. aurea, C. capillaris, C. concinna, C. disperma, C. gynocrates*
7	474312	6083432	0	13/7/2015	*C. deflexa, C. disperma, C. gynocrates, C. vaginata*
8	461177	6079470	350	13/7/2015	*C. aquatilis, C. atherodes, C. canescens, C. diandra, C. disperma, C. magellanica, C. sartwellii, C. utriculata*
9	461312	6079317	180	13/7/2015	*C. brunnescens, C. canescens, C. concinna, C. deweyana, C. disperma, C. media, C. vaginata*
10	427021	6090212	180	14/7/2015	*C. aquatilis, C. chordorrhiza, C. diandra, C. interior, C. limosa, C. magellanica, C. prairea, C. tenuiflora, C. utriculata*
11	436046	6095782	180	14/7/2015	*C. richardsonii, C. siccata, C. tonsa*
12	436032	6095445	270	14/7/2015	*C. chordorrhiza, C. lasiocarpa, C. limosa, C. rostrata, C. tenuiflora*
13	437617	6069144	0	15/7/2015	*C. siccata*
14	447743	6094716	270	15/7/2015	*C. chordorrhiza, C. limosa, C. rostrata*
15	494579	6168064	90	16/7/2015	*C. siccata*
16	494495	6168019	310	16/7/2015	*C. aquatilis, C. canescens, C. deflexa, C. siccata*
17	481174	6376395	180	17/7/2015	*C. aquatilis, C. canescens, C. chordorrhiza, C. limosa, C. rostrata*
18	481175	6376553	270	17/7/2015	*C. deflexa, C. foena, C. siccata, C. tonsa*
19	480631	6277681	0	17/7/2015	*C. richardsonii, C. siccata, C. tonsa*
20	454327	6292704	230	18/7/2015	*C. deflexa, C. disperma*
21	454408	6292632	250	18/7/2015	*C. aquatilis, C. canescens, C. diandra, C. disperma, C. leptalea, C. utriculata*
22	452943	6294751	0	19/7/2015	*C. brunnescens, C. deflexa*
23	452818	6294650	270	19/7/2015	*C. aquatilis, C. canescens, C. chordorrhiza, C. diandra, C. limosa, C. magellanica, C. tenuiflora*

24	455420	6292091	130	19/7/2015	*C. aquatilis, C. canescens, C. disperma, C. gynocrates, C. leptalea, C. media, C. vaginata*
25	432002	6195359	0	20/7/2015	*C. aquatilis, C. chordorrhiza, C. limosa, C. magellanica, C. pauciflora, C. rostrata, C. trisperma*
26	449026	6294611	270	25/7/2015	*C. aquatilis, C. disperma, C. gynocrates, C. tenuiflora*
27	462018	6290944	180	25/7/2015	*C. aquatilis, C. canescens, C. capillaris, C. disperma, C. gynocrates, C. leptalea, C. magellanica, C. tenuiflora*
28	488162	6254598	110	26/7/2015	*C. aquatilis, C. diandra, C. disperma, C. leptalea, C. magellanica*
29	487907	6254599	270	26/7/2015	*C. deflexa*
30	483239	6246321	180	27/7/2015	*C. limosa, C. magellanica, C. pauciflora*
31	483082	6246332	260	27/7/2015	*C. disperma, C. trisperma*
32	482827	6248104	270	27/7/2015	*C. deflexa*
33	467074	6236014	220	28/7/2015	*C. siccata*
34	467145	6236063	135	28/7/2015	*C. aquatilis, C. brunnescens, C. canescens, C. disperma, C. magellanica, C. tenuiflora, C. utriculata, C. vaginata*
35	497267	6249182	180	29/7/2015	*C. aquatilis, C. chordorrhiza, C. diandra, C. disperma, C. gynocrates, C. leptalea, C. magellanica, C. tenuiflora*
36	508527	6187481	180	29/7/2015	*C. deweyana, C. sprengelii*
37	545671	6193784	0	30/7/2015	*C. siccata*
38	545619	6193784	275	30/7/2015	*C. canescens, C. disperma, C. gynocrates, C. leptalea, C. magellanica, C. tenuiflora*
39	544129	6190798	270	30/7/2015	*C. aquatilis, C. chordorrhiza, C. diandra, C. disperma, C. gynocrates, C. interior, C. leptalea, C. limosa, C. magellanica, C. tenuiflora*
40	529726	6185992	200	31/7/2015	*C. aquatilis, C. canescens, C. diandra, C. utriculata*
41	529927	6186106	200	31/7/2015	*C. deweyana, C. peckii, C. vaginata*
42	490134	6377706	205	1/8/2015	*C. aquatilis, C. bebbii, C. canescens, C. diandra, C. prairea, C. utriculata*
43	489830	6377909	0	1/8/2015	*C. foena, C. siccata, C. tonsa*
44	470478	6236308	135	3/8/2015	*C. canescens, C. limosa, C. magellanica, C. rostrata*
45	470687	6236350	90	3/8/2015	*C. brunnescens, C. deflexa, C. foena*
46	464080	6246339	40	4/8/2015	*C. brunnescens, C. canescens, C. disperma, C. leptalea, C. loliacea, C. magellanica, C. vaginata*
47	464116	6246051	90	4/8/2015	*C. brunnescens, C. concinna, C. disperma, C. gynocrates, C. media, C. vaginata*
48	519251	6147336	240	12/8/2015	*C. chordorrhiza, C. diandra, C. lasiocarpa, C. limosa, C. rostrata*
49	515466	6141931	335	12/8/2015	*C. aquatilis, C. chordorrhiza, C. diandra, C. lasiocarpa, C. limosa, C. rostrata*

50	477272	6173089	315	13/8/2015	*C. deweyana*
51	482573	6176870	38	13/8/2015	*C. deweyana, C. peckii*

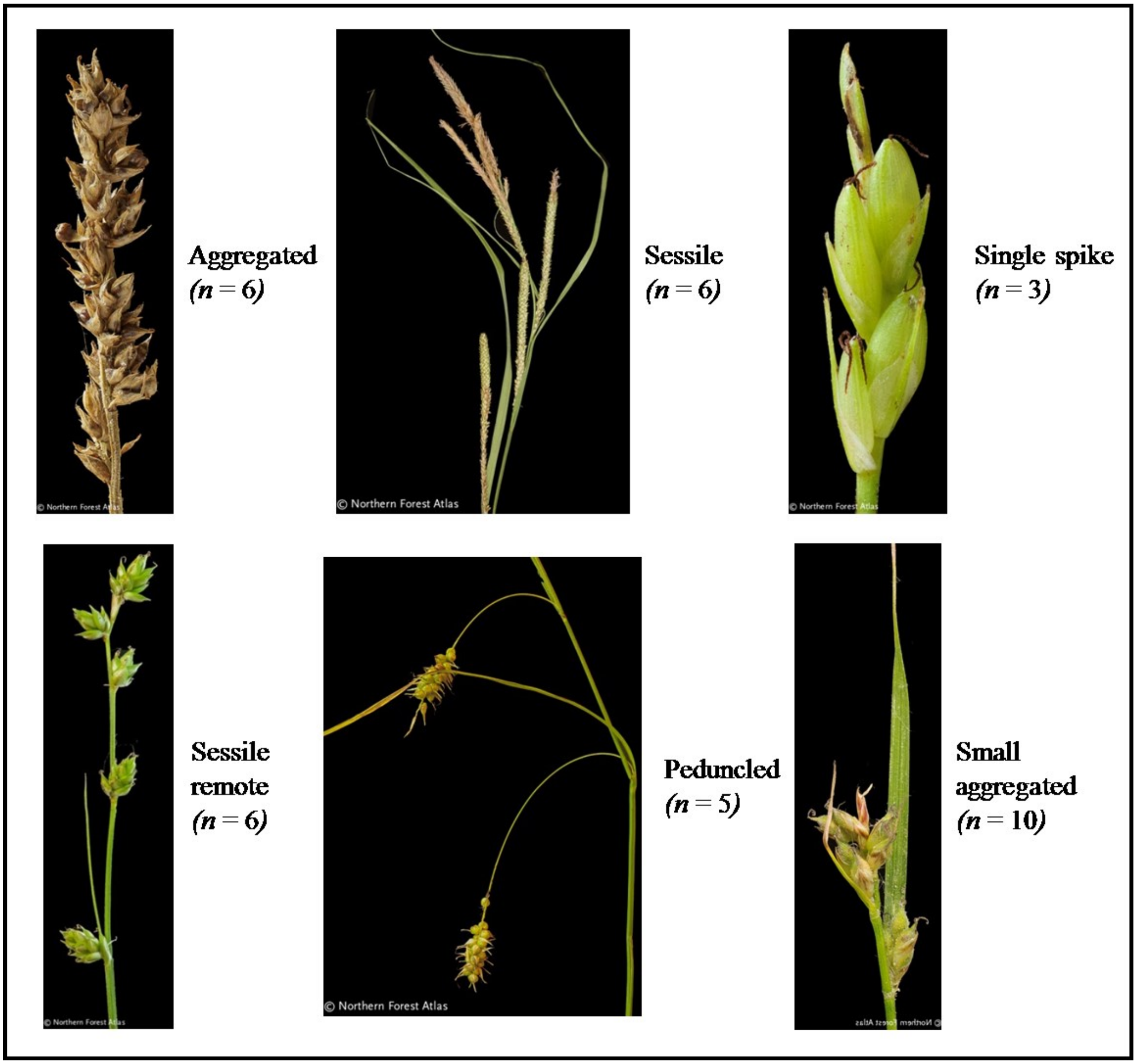
Table A3.2. Summarized number of occurrences, number of measured individuals, median time to detection, and plant characters for 36 *Carex* species found across 50 experimental transects in Northeastern Alberta, Canada. Table S1. Measurements of 660 individuals were used to summarize plant height and leaf width. Only those *Carex* bearing pistillate spikes (593) were used in summarizing reproductive characters including spike or inflorescence length, width, and number of spikes.

Carex sp.	No. sites recorded	No. individuals measured	No. measured w/ ♀ spikes	Mean time to detection	Median time to detection	Mean height	± SE	Range	Mean leaf width	± SE	Range	Mean summed spike or inflorescence length[1]	± SE	Range	Mean spike or inflorescence width[2]	± SE	Range	Mean no. spikes	± SE	Range	Morphological group
C. aquatilis	18	63	51	3.9	1	61.2	2.54	12.5 - 124.5	0.4	0.02	0.1 - 0.7	11.6	0.7	1.6 - 28	0.4	0.01	0.2 - 0.6	2.9	0.12	3-4	Sessile
C. atherodes	2	6	6	12.7	12	86.9	7.51	60.5 - 110	0.6	0.03	0.5 - 0.7	12.8	0.88	10.5 - 16	1	0.09	0.8 - 1.1	3.7	2.12	3-4	Sessile
C. aurea	1	3	3	17.5	17.5	10.2	0.78	9.1 - 11.7	0.2	0.03	0.2 - 0.3	1.8	0.44	1.3 - 2.7	0.8	0.22	0.45 - 1.2	3.3	0.33	5-9	Small aggregated
C. bebbii	1	3	3	2	2	54	6.57	42.2 - 64.9	0.3	0.03	0.3 -0.4	2.1	0.15	1.8 - 2.3	1.1	0.15	0.8 - 1.3	7	1.16	3-5	Aggregated
C. brunnescens	5	17	14	19.2	20	27.6	1.86	16 - 43.5	0.2	0.01	0.1 - 0.3	3	0.23	1.8 - 4.1	0.3	0.03	0.2 - 0.5	4.3	0.16	4-7	Sessile remote
C. canescens	16	47	47	9.2	6.5	36.2	1.05	22 - 58.3	0.2	0.01	0.1 - 0.3	3.9	0.13	2.2 - 5.9	0.4	0.02	0.1 - 0.8	5.2	0.13	1-2	Sessile remote
*C. capillaris**	3	10	6	9	9	30.7	1.58	26 - 36.5	0.3	0.04	0.1 - 0.4	3	0.8	1.4 - 6.4	0.3	0.03	0.1 - 0.3	1.7	0.21	1-5	Peduncled
C. chordorrhiza	9	29	25	7.3	4	19.3	1.13	6.5 - 37	0.1	0.01	0.1 - 0.2	0.8	0.04	0.5 - 1.2	0.3	0.02	0.1 - 0.5	2.1	0.32	1-2	Small aggregated
C. concinna	3	8	7	14.8	15.5	6	0.71	3.5 - 9.3	0.2	0.01	0.2 - 0.3	0.7	0.18	0.2 - 1.4	0.3	0.03	0.2 - 0.4	1.4	0.2	1-3	Small aggregated
C. deflexa	7	21	19	9.7	9	9.1	0.73	3.5 - 15.1	0.2	0.01	0.1 - 0.2	0.8	0.05	0.4 - 1.2	0.3	0.04	0.1 - 0.6	2.1	0.18	3-5	Small aggregated
C. deweyana	3	15	7	11.9	13.5	24.3	3.09	12 - 53	0.2	0.02	0.1 - 0.4	3.8	0.43	2.4 - 5.6	0.4	0.12	0.1 - 1	4.4	0.3	5-10	Sessile remote
C. diandra	12	37	36	6.1	3	59.9	2.9	37.9 - 110	0.2	0.01	0.1 - 0.5	2.6	0.14	0.5 - 4.4	0.5	0.03	0.2 - 0.9	7.9	0.21	2-6	Aggregated
C. disperma	19	56	56	8.4	7	22	0.87	10.5 - 35.4	0.1	0.01	0.1 - 0.3	2	0.07	0.8 - 3.2	0.3	0.01	0.1 - 0.5	3.6	0.12	3-8	Sessile remote
C. foena	3	9	9	7.8	7.5	49.1	7.38	18 - 76.8	0.2	0.03	0.1 - 0.3	3.1	0.32	2.2 - 4.7	0.7	0.12	0.2 - 1.2	5.2	0.49	1-5	Aggregated
C. gynocrates	10	33	23	9.5	6	11.5	0.47	6.4 - 16.8	0.1	0	0.1 - 0.1	0.9	0.06	0.4 - 1.4	0.4	0.04	0.1 - 0.6	1	0	1-1	Single spike
C. interior	2	6	6	4.3	4.5	25.2	1.28	22 - 29	0.1	0	0.1 - 0.1	1.5	0.02	1.4 - 1.5	0.5	0.02	0.4 - 0.5	3	0	3-3	Small aggregated
*C. lacustris**	1	3	3	0	0	86.7	6.01	75 - 95	0.7	0.12	0.5 - 0.9	14.3	1.7	11.6 - 17.4			not collected				Sessile
*C. lasiocarpa**	3	8	8	2.5	1	53.1	4.16	33 - 70	0.1	0.01	0.1 - 0.2	2.5	0.2	1.5 - 3.3	0.4	0.04	0.3 - 0.6	1.5	0.19	1-2	Sessile
C. leptalea	8	24	24	10.4	8	15.5	1.53	4.1 - 34.5	0.1	0	0.1 - 0.1	0.5	0.03	0.3 - 1	0.2	0.02	0.1 - 0.3	1	0	1-1	Single spike
*C. limosa**	11	31	31	7.1	4	28.1	1.09	17.5 - 41	0.1	0.01	0.1 - 0.2	1.7	0.1	0.9 - 4	0.4	0.01	0.3 - 0.5	1.3	0.08	1-2	Peduncled
C. loliacea	1	2	2	13	13	24	3	21 - 27	0.2	0	0.2 - 0.2	1.5	0.25	1.2 - 1.7	0.5	0.05	0.4 - 0.5	2.5	0.5	2-3	Sessile remote
*C. magellanica**	15	42	42	9.3	5	32.9	1.25	20 - 56	0.3	0.01	0.1 - 0.5	2.5	0.1	0.8 - 4.4	0.5	0.02	0.3 - 0.9	2.4	0.12	1-4	Peduncled
C. media	3	6	6	18.7	22	38.4	4.98	27 - 62	0.3	0.03	0.1 - 0.3	1.1	0.11	0.7 - 1.5	0.6	0.05	0.5 - 0.8	3	0.26	2-4	Small aggregated
C. pauciflora	2	6	6	3.75	3.5	18	1.46	14.5 - 24	0.1	0.02	0.1 - 0.2	1	0.1	0.7 - 1.4	0.6	0.08	0.4 - 0.9	1	0	1-1	Single spike
C. peckii	2	6	3	7	7	9.7	0.59	6.6 - 11.3	0.2	0	0.2 - 0.2	1.1	0.19	0.7 - 1.3	0.2	0	0.2 - 0.2	2	0	2-2	Small aggregated
C. prairea	3	9	9	7.4	5	50.9	4.69	26.5 - 70	0.2	0.02	0.1 - 0.3	4.5	0.19	3.5 - 5.1	0.4	0.04	0.2 - 0.6	7.7	0.33	6-9	Aggregated
C. richardsonii	2	6	6	18.7	20	8.7	0.93	4.9 - 11.6	0.3	0.02	0.2 - 0.3	2.3	0.19	1.8 - 2.9	0.1	0.03	0.1 - 0.2	2.3	0.33	2-3	Small aggregated
*C. rostrata**	7	22	22	9.4	5	45.2	1.73	26.5 - 55.9	0.3	0.02	0.2 -0.4	7.8	0.6	3.9 - 17.2	0.8	0.03	0.6 - 1.1	2	0.09	1-3	Sessile
C. sartwellii	1	3	3	0.5	0.5	70.7	8.87	56.8 - 87.2	0.4	0.03	0.4 - 0.5	4.5	0.76	3.1 - 5.7	0.6	0.15	0.4 - 0.9		not collected		Aggregated
C. siccata	8	30	20	10.3	8.5	27.1	1.66	10.5 - 50.1	0.2	0.01	0.1 - 0.3	1.8	0.09	1 - 2.6	0.5	0.04	0.2 - 0.7	4.6	0.53	1-8	Aggregated
*C. sprengelii**	1	3	3	11	11	88	4.16	80 - 94	0.3	0	0.3 - 0.3	5.3	0.2	5.1 - 5.6	0.5	0	0.5 - 0.5	3	0	3-3	Peduncled
C. tenuiflora	11	33	33	10.9	7	23	1.36	12.5 - 39.1	0.1	0.01	0.1 - 0.2	0.9	0.15	0.5 - 5.5	0.5	0.03	0.2 - 0.9	2.2	0.08	1-3	Small aggregated
C. tonsa	4	14	10	7.2	2.5	6.9	0.63	4.8 - 13	0.2	0.02	0.1 - 0.3	0.6	0.05	0.3 - 0.8	0.3	0.05	0.2 - 0.5	2	0	2-2	Small aggregated
C. trisperma	3	9	9	10.3	4	24.7	2.79	14.8 - 39.5	0.1	0.01	0.1 - 0.2	3	0.38	1.8 - 5.6	0.4	0.03	0.2 - 0.5	2.3	0.17	2-3	Sessile remote
*C. utriculata**	7	22	17	8.14	5.5	63.9	4.35	40 - 97.5	0.5	0.04	0.3 - 0.9	11.2	1.4	4.2 - 24.7	0.9	0.03	0.6 - 1.1	2.8	0.14	1-3	Sessile
*C. vaginata**	7	18	18	11.6	14	28.9	1.38	19 -37.5	0.3	0.02	0.2 - 0.5	4.1	0.6	1.9 - 13.4	0.5	0.04	0.3 - 0.3	2.1	0.14	1-3	Peduncled

[1] For those species marked with an asterisk, I measured the length of the largest peduncled pistillate spike rather than the total inflorescence, as this is the unit most recognizable on the plant and peduncled spikes tend to be well spaced from one another. For all remaining species I measured the length of the total inflorescence (i.e,. all spikes), where spikes were singular, aggregated, or overlapping.

[2] For those species marked with an asterisk, I measured the width of the same peduncled pistillate spike measured for length. For the remaining species I measured the inflorescence at its average width point.

Figure A3.1. Example of a *Carex* species typical to each of six morphological groups, with the number of species per group shown in brackets (see table A3.2 for species identities). Morphological groups were assigned using field measured characters of height, leaf width, and inflorescence measurements (Table A3.2), as well as the general appearance of the species. All photos shown here are taken from the Northern Forest Atlas (www.northernforestatlas.org).



Appendix 4

Table A4.1. Reclassification of the Alberta Biodiversity Monitoring Institute's Human Footprint Mapping layer disturbance classes, (*n* = 132) to 15 simplified classes based on their broader affiliation and if the disturbance type is related to oil and gas.

ABMI Human Footprint Mapping Feature type	Simplified classification	Oil and gas related
AIRP-RUNWAY	airport	non-oil and gas
BORROWPIT-DRY	reservoir	oil and gas
BORROWPIT-DRY	reservoir	oil and gas
BORROWPITS	reservoir	oil and gas
BORROWPITS	reservoir	oil and gas
BORROWPIT-WET	reservoir	oil and gas
BORROWPIT-WET	reservoir	oil and gas
CAMPGROUND	recreation	non-oil and gas
CAMPGROUND	recreation	non-oil and gas
CAMP-INDUSTRIAL	oil and gas structure	oil and gas
CANAL	agriculture	non-oil and gas
CFO	agriculture	non-oil and gas
CLEARING-UNKNOWN	cleared/disturbed ground	non-oil and gas
CLEARING-WELLPAD-UNCONFIRMED	cleared/disturbed ground	non-oil and gas
COUNTRY-RESIDENCE	residential	non-oil and gas
CROP	agriculture	non-oil and gas
CULTIVATION_ABANDONED	agriculture	non-oil and gas
CUTBLOCK	forestry	non-oil and gas
DISTURB_VEG	cleared/disturbed ground	non-oil and gas
DUGOUT	reservoir	non-oil and gas
DUGOUT	reservoir	non-oil and gas
FACILITY-OTHER	miscellaneous industrial	non-oil and gas
FACILITY-UNKNOWN	miscellaneous industrial	non-oil and gas
GOLFCOURSE	recreation	non-oil and gas
GOLFCOURSE	recreation	non-oil and gas
GREENSPACE	recreation	non-oil and gas
GREENSPACE	recreation	non-oil and gas
GRVL-SAND-PIT	miscellaneous industrial	oil and gas
INTERCHANGE-RAMP	road	non-oil and gas

LAGOON	reservoir	non-oil and gas
LAGOON	reservoir	non-oil and gas
LANDFILL	residential	non-oil and gas
LOW-IMPACT-SEISMIC	linear feature	oil and gas
MILL	miscellaneous industrial	non-oil and gas
MINES-OILSANDS	oil sands mine	oil and gas
MINES-PITLAKE	oil sands mine	oil and gas
MISC-OIL-GAS-FACILITY	oil and gas structure	oil and gas
OIL-GAS-PLANT	oil and gas structure	oil and gas
OPEN-PIT-MINE	oil sands mine	oil and gas
PEAT	forestry	non-oil and gas
PIPELINE	linear feature	oil and gas
PRE-LOW-IMPACT-SEISMIC	linear feature	oil and gas
RECREATION	recreation	non-oil and gas
RECREATION	recreation	non-oil and gas
RESERVOIR	reservoir	non-oil and gas
RESERVOIR	reservoir	non-oil and gas
RESIDENCE_CLEARING	residential	non-oil and gas
RIS-AIRP-RUNWAY	airport	non-oil and gas
RIS-BORROWPITS	reservoir	oil and gas
RIS-BORROWPITS	reservoir	oil and gas
RIS-CAMP-INDUSTRIAL	oil and gas structure	oil and gas
RIS-CLEARING-UNKNOWN	cleared/disturbed ground	non-oil and gas
RIS-DRAINAGE	oil sands mine	oil and gas
RIS-FACILITY-OPERATIONS	miscellaneous industrial	oil and gas
RIS-FACILITY-UNKNOWN	miscellaneous industrial	oil and gas
RIS-MINES-OILSANDS	oil sands mine	oil and gas
RIS-OILSANDS-RMS	oil sands mine	oil and gas
RIS-OVERBURDEN-DUMP	oil sands mine	oil and gas
RIS-PIPELINE	linear feature	oil and gas
RIS-PLANT	oil and gas structure	oil and gas
RIS-RECLAIMED-CERTIFIED	reclaimed land	oil and gas
RIS-RECLAIMED-CERTIFIED	reclaimed land	oil and gas
RIS-RECLAIMED-PERMANENT	reclaimed land	oil and gas

RIS-RECLAIMED-PERMANENT	reclaimed land	oil and gas
RIS-RECLAIMED-TEMP	reclaimed land	oil and gas
RIS-RECLAIMED-TEMP	reclaimed land	oil and gas
RIS-RECLAIM-READY	reclaimed land	oil and gas
RIS-RECLAIM-READY	reclaimed land	oil and gas
RIS-ROAD	road	non-oil and gas
RIS-SOIL-REPLACED	oil sands mine	oil and gas
RIS-SOIL-SALVAGED	oil sands mine	oil and gas
RIS-TAILING-POND	oil sands mine	oil and gas
RIS-TAILING-POND	oil sands mine	oil and gas
RIS-TANK-FARM	oil and gas structure	oil and gas
RIS-TRANSMISSION-LINE	linear feature	non-oil and gas
RIS-UTILITIES	oil and gas structure	non-oil and gas
RIS-WASTE	oil sands mine	oil and gas
RIS-WASTE	oil sands mine	oil and gas
RIS-WELL	in-situ structure	oil and gas
RIS-WINDROW	oil sands mine	oil and gas
RIS-WINDROW	oil sands mine	oil and gas
RLWY-ABANDONED	railway	non-oil and gas
RLWY-ABANDONED	railway	non-oil and gas
RLWY-DBL-TRACK	railway	non-oil and gas
RLWY-DBL-TRACK	railway	non-oil and gas
RLWY-MLT-TRACK	railway	non-oil and gas
RLWY-MLT-TRACK	railway	non-oil and gas
RLWY-SGL-TRACK	railway	non-oil and gas
RLWY-SGL-TRACK	railway	non-oil and gas
RLWY-SPUR	railway	non-oil and gas
RLWY-SPUR	railway	non-oil and gas
ROAD-GRAVEL-1L	road	non-oil and gas
ROAD-GRAVEL-2L	road	non-oil and gas
ROAD-PAVED-1L	road	non-oil and gas
ROAD-PAVED-2L	road	non-oil and gas
ROAD-PAVED-3L	road	non-oil and gas
ROAD-PAVED-4L	road	non-oil and gas
ROAD-PAVED-5L	road	non-oil and gas
ROAD-PAVED-DIV	road	non-oil and gas
ROAD-PAVED-UNDIV-1L	road	non-oil and gas
ROAD-PAVED-UNDIV-2L	road	non-oil and gas
ROAD-UNCLASSIFIED	road	non-oil and gas
ROAD-UNIMPROVED	road	non-oil and gas
ROAD-UNPAVED-2L	road	non-oil and gas

ROAD-WINTER-ACCESS	road	non-oil and gas
ROUGH_PASTURE	agriculture	non-oil and gas
RUNWAY	airport	non-oil and gas
RURAL-RESIDENCE	residential	non-oil and gas
SUMP	reservoir	oil and gas
SURROUNDING-VEG	airport	non-oil and gas
TAILING-POND	oil sands mine	oil and gas
TAILING-POND	oil sands mine	oil and gas
TAME_PASTURE	agriculture	non-oil and gas
TRAIL	linear feature	oil and gas
TRAIL-ATV	linear feature	non-oil and gas
TRANSFER_STATION	miscellaneous industrial	non-oil and gas
TRANSMISSION-LINE	linear feature	non-oil and gas
TRUCK-TRAIL	linear feature	non-oil and gas
URBAN-INDUSTRIAL	miscellaneous industrial	non-oil and gas
URBAN-RESIDENCE	residential	non-oil and gas
VEGETATED-EDGE-RAILWAYS	railway	non-oil and gas
VEGETATED-EDGE-RAILWAYS	railway	non-oil and gas
VEGETATED-EDGE-ROADS	road	non-oil and gas
WELL-ABAND	in-situ structure	oil and gas
WELL-BIT	in-situ structure	oil and gas
WELL-CASED	in-situ structure	oil and gas
WELL-CLEARED-DRILLED	in-situ structure	oil and gas
WELL-CLEARED-NOT-DRILLED	in-situ structure	oil and gas
WELL-DRILLED-OTHER	in-situ structure	oil and gas
WELL-GAS	in-situ structure	oil and gas
WELL-OIL	in-situ structure	oil and gas
WELL-OTHER	in-situ structure	oil and gas

Appendix 5

Table A5.1. Site locations of 12 experimental sites used in mitigative translocation of two species, *Sarracenia purpurea* (sites denoted by 'S') and *Carex oligosperma* (sites denoted by 'O'). Donor locations containing naturally occurring populations are denoted by 'D', recipients, those to which 50 transplants of each species were moved, respectively, by 'R'. Coordinates are given in UTM (Zone 12U), NAD83.

Site	Easting	Northing
OD1	495310	6167003
OD3	487962	6375058
OD2	488184	6374672
OR1	494492	6168017
OR2	493868	6149058
OR3	479182	6377713
SD1	516512	6145190
SD2	478329	6146433
SD3	507302	6145366
SR1	501572	6175829
SR2	519746	6136716
SR3	479380	6193157

Table A5.1. Candidate model selection of annual growth as related to water chemistry variables for transplanted *Sarracenia purpurea* across three monitoring years (n = 407). All models included a random effect of year and transplant identity.

model	K	AIC	ΔAIC
magnesium	3	2615.00	0.00
calcium	3	2619.08	4.07
temperature	3	2621.52	6.52
pH	3	2625.65	10.65
sodium	3	2645.57	30.57
potassium	3	2656.07	41.07
total Kjeldahl nitrogen	3	2658.05	43.04
null	2	2661.27	46.27
total phosphorus	3	2662.23	47.23

Table A5.2. Candidate model selection of flowering as related to water chemistry variables for transplanted *Sarracenia purpurea* across three monitoring years (*n* = 430). All models included a random effect of year and transplant identity.

model	*K*	AIC	ΔAIC
total phosphorus	3	560.24	0.00
potassium	3	561.66	1.42
pH	3	564.37	4.13
sodium	3	565.00	4.76
magnesium	3	571.82	11.57
total Kjeldahl nitrogen	3	571.99	11.75
null	2	572.24	12.00
temperature	3	572.49	12.24
calcium	3	572.60	12.35

Table A5.3. Candidate model selection of annual growth as related to water chemistry variables for transplanted *Carex oligosperma* across three monitoring years (*n* = 359). All models included a random effect of year and transplant identity.

model	*K*	AIC	ΔAIC
pH	3	1950.44	0.00
calcium	3	1950.67	0.24
magnesium	3	1950.75	0.32
total Kjeldahl nitrogen	3	1950.97	0.53
temperature	3	1951.18	0.74
total phosphorus	3	1951.89	1.46
null	2	1951.97	1.53
sodium	3	1954.36	3.92
potassium	3	1954.63	4.20

Table A5.4. Candidate model selection of flowering as related to water chemistry variables for transplanted *Carex oligosperma* across three monitoring years (*n* =430). All models included a random effect of year and transplant identity.

model	*K*	AIC	ΔAIC
pH	3	449.31	0.00
potassium	3	449.40	0.08
null	2	449.40	0.08
calcium	3	449.55	0.24
total Kjeldahl nitrogen	3	449.64	0.32
magnesium	3	449.94	0.62
temperature	3	450.98	1.66
total phosphorus	3	451.14	1.83
sodium	3	451.29	1.98

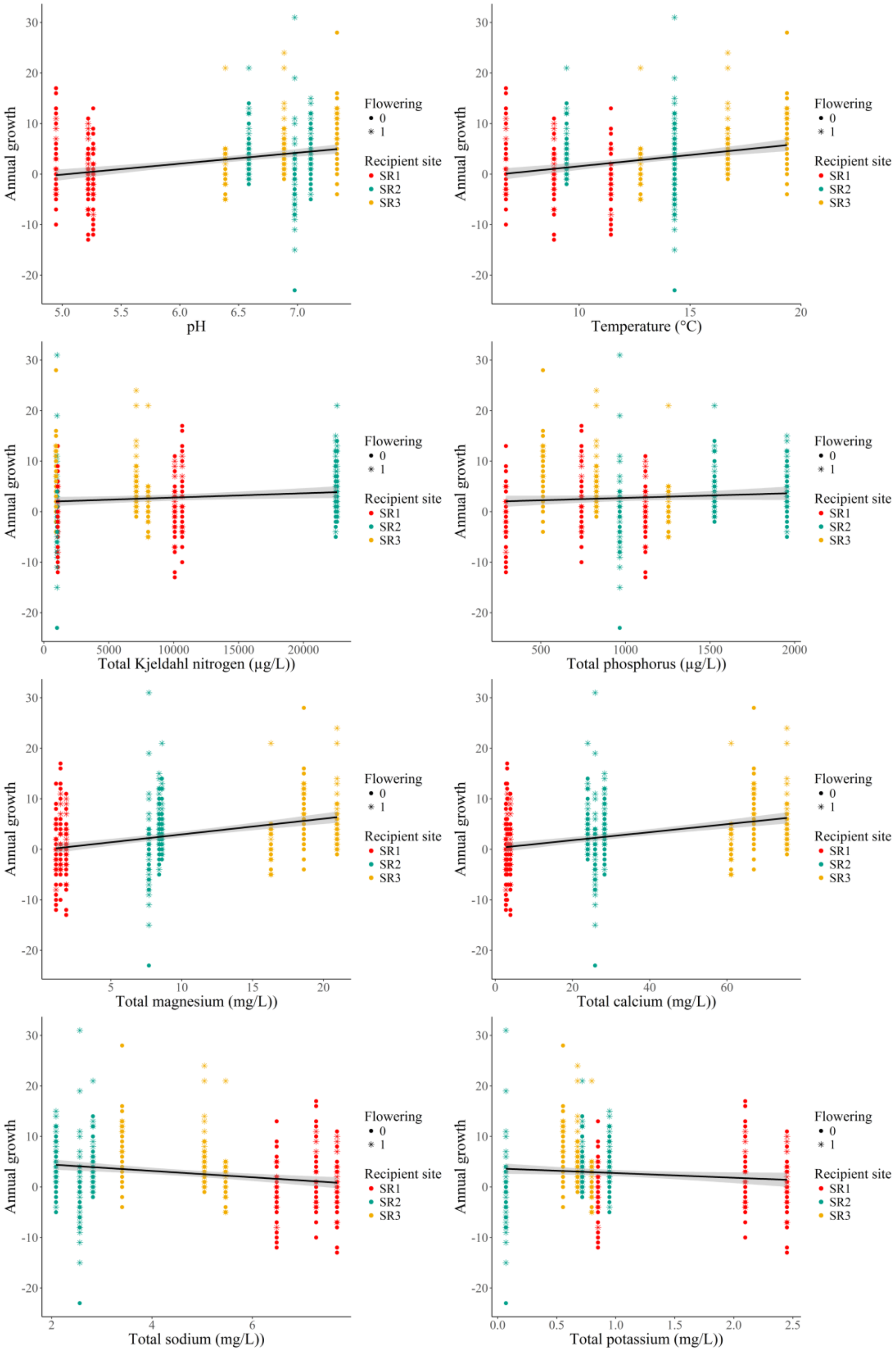


Figure A5.1. Scatter plots of annual growth and water chemistry (pH, temperature, and major nutrients) at recipient sites for *Sarracenia purpurea*. Linear regression lines are shown in black, with confidence intervals in shaded grey.

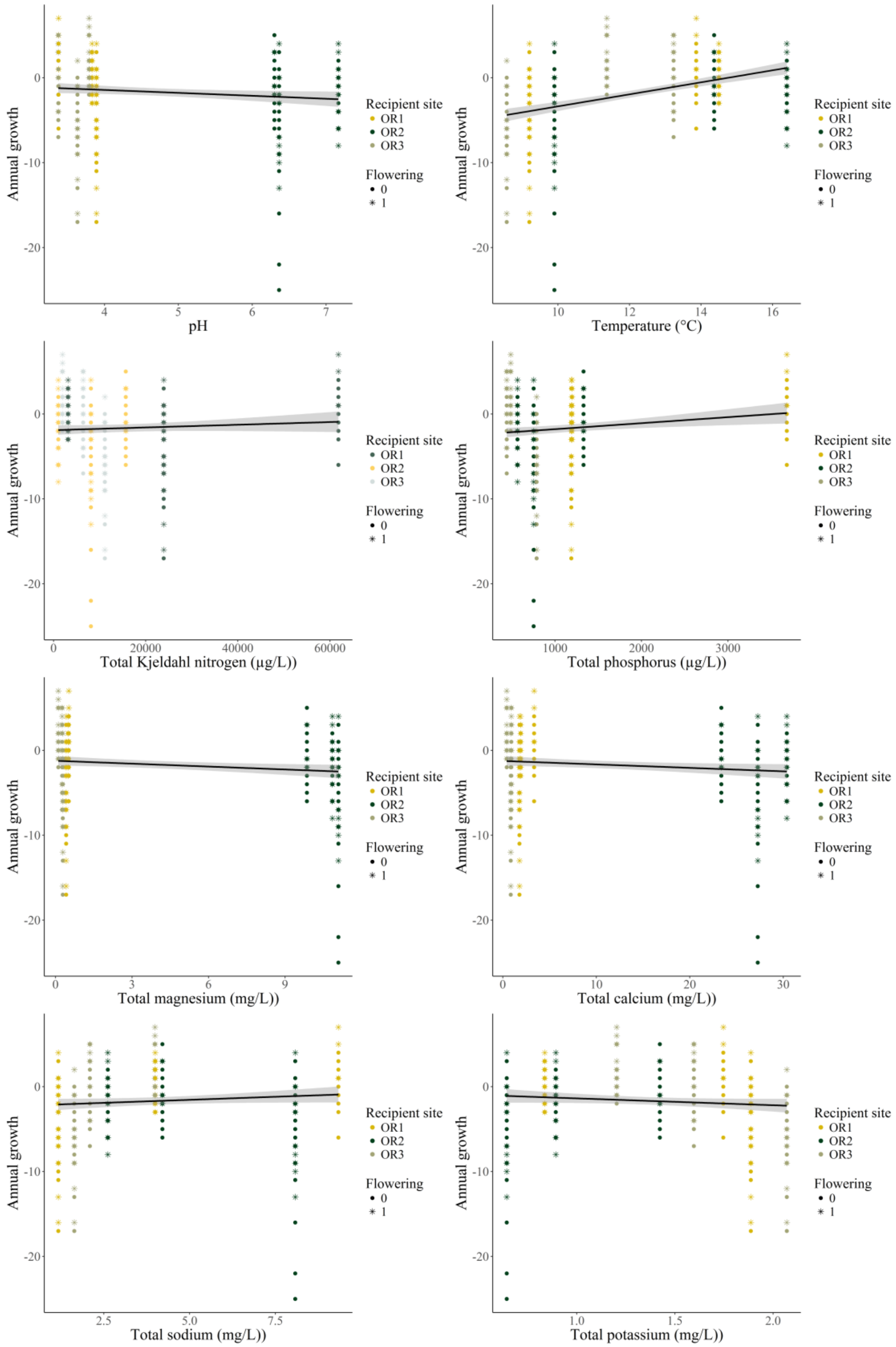


Figure A5.2. Scatter plots of annual growth and water chemistry (pH, temperature, and major nutrients) at recipient sites for *Carex oligosperma*. Linear regression lines are shown in black, with confidence intervals in shaded grey.

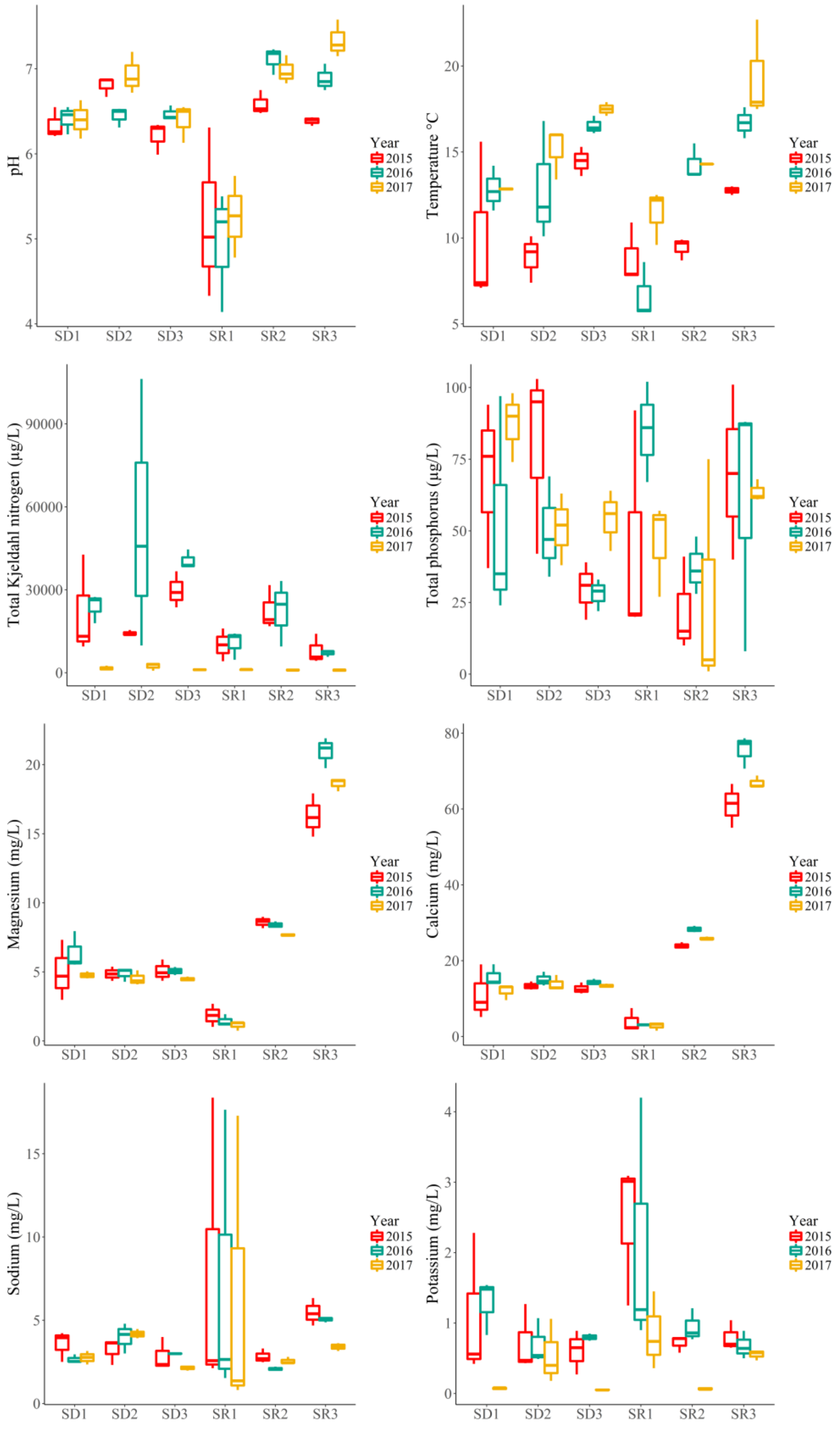


Figure A5.3. Boxplots of water chemistry (pH, temperature, and major nutrients) at donor and recipient sites for *Sarracenia purpurea* over three years of sampling.

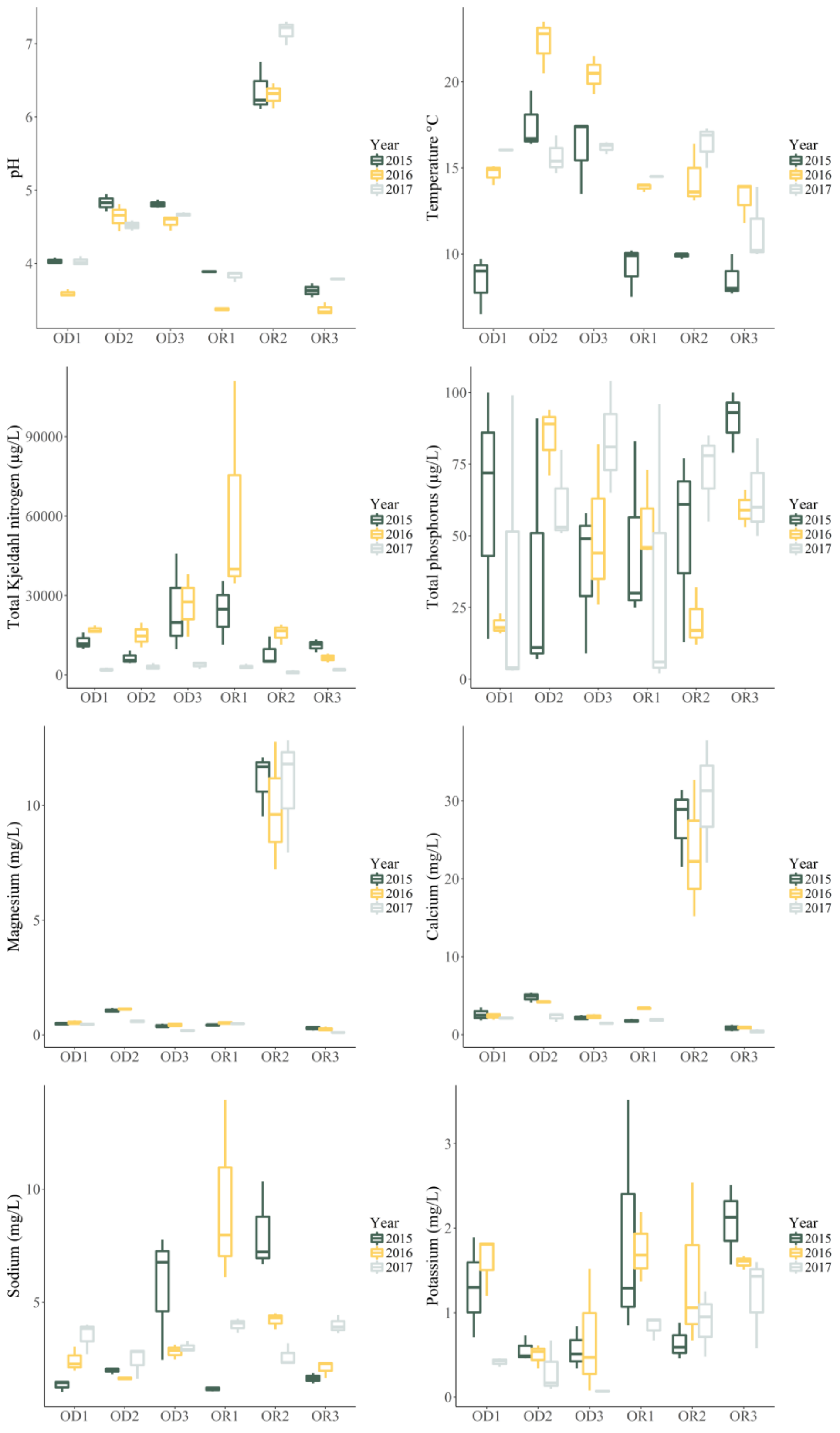


Figure A5.4. Boxplots of water chemistry (pH, temperature, and major nutrients) at donor and recipient sites for *Carex oligosperma* over three years of sampling.