Terrestrial Vascular Plant Monitoring Project for the Lower Athabasca (2012-2016)

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Preface

Terrestrial Vascular Plant Monitoring Project for the Lower Athabasca (2012 - 2016)

The following report is comprised of seven chapters related to the Terrestrial Vascular Plant Monitoring Project for the Lower Athabasca, formerly known as the Ecological Monitoring Committee for the Lower Athabasca (EMCLA) Rare Plants Project. This project is the result of a collaborative effort between Dr. Scott E. Nielsen (Applied Conservation Ecology Lab, Department of Renewable Resources, University of Alberta) and Monica Kohler and Dr. Dan Farr at the Alberta Biodiversity Monitoring Institute’s Application Center. The project began in 2012 and is ongoing.

This project was initially funded through the Ecological Monitoring Committee for the Lower Athabasca (EMCLA) (2012). Funding underwent several changes in governance, including the Joint Oil Sands Monitoring (JOSM) initiative from 2013-2014, and the Alberta Environmental Monitoring, Evaluation, and Reporting Agency (AEMERA) in 2015. Current funding is received through the Environmental Monitoring and Science Division (ESMD) of Alberta Environment and Parks, a division of the Government of Alberta. Ducks Unlimited provided in-kind support throughout the entirety of the project by making available their Enhanced Wetland Classification for the Lower Athabasca. Further funding was obtained through Natural Sciences Engineering and Research Council (NSERC) Canadian Graduate Scholarships – Masters (CGS-M) and Collaborative Research and Development grants and through the Alberta Conservation Association Grants in Biodiversity.

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Executive Summary

Preserving biodiversity in human-altered landscapes is a critical conservation issue. Developing science-based recommendations and evaluating tools for land managers are important parts of the conservation process and the means by which biodiversity and ecosystem function can be preserved in close proximity to human disturbance. Vascular plants are rarely afforded the conservation limelight, despite being a critical element of regional diversity and providing an array of ecosystem services. Rare species are also an important aspect of regional flora but patterns in their occurrence, methods to monitor them, and mitigation options to deal with human disturbances are often poorly understood.

The effects of oil and gas exploration and extraction in Alberta’s boreal forest are wide ranging and in many cases not well understood, yet land managers require information and data to make decisions at site to lease-scales and where possible mitigate their impacts. Provincial monitoring efforts by the Alberta Biodiversity Monitoring Program do not occur at the scale needed to provide information about regional diversity and rarity within Alberta’s oil and gas region.

In 2012 the Terrestrial Vascular Plant Monitoring Project for the Lower Athabasca, known at the time as the Ecological Monitoring Committee for the Lower Athabasca Rare Plants Project, was initiated to inform the status of rare vascular plant species, test protocols to improve sampling and monitoring, and develop models to assist with management of rare plants across the Lower Athabasca Planning Region (LAPR). Prior information on rare plants in the region was either too broad (i.e. ABMI 20-km grid) or too specific to individual parts of leases and developments (i.e. Pre-Disturbance Assessment). In the following report, we provide the findings of the past 5-years of research that addresses these challenges.

In the survey years of 2012 to 2015 a total of 602 Rarity and Diversity plots were completed, generating a comprehensive dataset consisting of 536 vascular plant species across regionally significant habitat types (Chapter 1). These plots were selected in early years using an iteration of a landscape model of rare vascular plant occurrence, which was updated in later years using plot-level data generated by this project. This model has since been used to guide regional conservation and land use planning efforts through the Biodiversity Management Framework, while providing significant additional value as a tool to guide regional and lease-level survey efforts (Chapter 2). A remote sensing-based application of Airborne Laser Scanning (ALS) data was then used to relate and predict vascular plant diversity within the core area of the LAPR and compared to current lease boundaries and caribou ranges (Chapter 3). We suggest that the rare plant and diversity models be used as planning tools to target surveys during environmental assessments and/or be used to avoid sensitive sites during construction and development.

Pre-disturbance assessments are an important element of the environmental assessment phase prior to lease development. These surveys provide locations of rare vascular plant species and allow oil and gas companies to mitigate for known populations of conservation concern. However, imperfect detection is rarely addressed and has the potential to generate false-absences, leading to possible population loss as a result of developments. Failure to detect species when they are present also affects the results of monitoring (attenuates trends) and research (increased Type II errors), yet little guidance is available on how to minimize detection
errors as it relates to survey protocols. For these reasons, we devote several chapters of this report to exploration of imperfect detection as it is critical to the issue of surveying and monitoring cryptic species like rare plants.

First, we conducted an analysis of pseudoturnover (change in species composition at a site between two observers) using a subsample of 67 plots where we had repeat survey data that was collected by well-trained, experienced observers. We consider our estimates to be comparatively low (average pseudoturnover of 15.4%) compared to what is reported in the literature with observed differences among functional groups being apparent with graminoids having the greatest variation in detection. Recognizing and understanding the presence of pseudoturnover in monitoring efforts, particularly in the oil and gas industry where monitoring sites are often visited by different observers, will lead to more reliable estimates of change in biodiversity (trend) over time (Chapter 4).

A second, experimental analysis of imperfect detection used decoy plants and detectability trails to understand how survey variables such as plot size, observer experience, and target species attributes of plant abundance and phenology influence detection. Results demonstrated that cryptic, low abundance vascular plants are detected far more poorly (0-35% success) than is currently recognized in plant surveys. Oil and gas related surveys which target rare species in large plot sizes are likely underestimating the occurrence of rare species demonstrating the need for carefully planned and documented (observer effort) surveys (Chapter 5).

Managing populations of rare species identified on lease areas after they have been successfully detected is a major challenge for industry and government. Translocation is a mitigative strategy used in the region to preserve species under the threat of destruction, but the execution of these projects is subject to limited planning, monitoring, and reporting. We tested the effectiveness of this mitigative tool for two rare peatland species observing high success rates over two monitoring years and a limited influence of recipient site characters, suggesting that monitoring, rather than recipient site location selection, may benefit most from increased resource allocation in future efforts (Chapter 6).

Finally, despite significant effort to locate and mitigate rare vascular plant species on lease areas, populations can be negatively impacted by the direct or indirect effects of development leading to extirpation of rare plant populations. Currently, the rate of extirpation due to oil and gas related factors is unknown. We conducted a remote sensing imagery- and field-based assessment of historic rare plant records from the Alberta Conservation Information Management System (ACIMS), including populations from multiple land-use types. More field sampling is proposed in 2017, but at present we estimate a 30% loss of populations with a trend for increased risk of extirpation when in closer proximity to disturbance. This emphasizes the need for on lease monitoring of plant populations and the communication of findings to ACIMS in the event of population loss (Chapter 7).

Overall, this work contributed to our knowledge and understanding of rare vascular plants in the Lower Athabasca Region of northeast Alberta, while providing tools and protocols that will increase the effectiveness of surveys, monitoring, management, and mitigation actions.
CHAPTER 1.0: Species richness, rare plant status, rare plant distribution, and sampling in the Lower Athabasca Region

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1.1 Summary

Between 2012 and 2015, vegetation surveys were conducted across 602 Rarity and Diversity plots in the Lower Athabasca Region of northeastern Alberta to evaluate vascular plant species richness and the distribution of rare species. A total of 536 species were detected with an average richness of 45.4 species per plot across 12 ecosite categories. Changes to Conservation Status Ranks by the Alberta Conservation Information Management System (ACIMS) affected the number and distribution of rare species (S1-S3) encountered during sampling, which decreased from 101 (18.8%) to 63 (11.8%) species between 2014 and 2015 following the re-evaluation of rankings. Data generated from this project have been used for landscape-level models of species rarity (Chapter 2), evaluation of the effectiveness of remote sensing metrics to predict species diversity (Chapter 3), and estimation of observer error relative to species richness across survey plots (Chapter 4).

1.2 Introduction

Across a four-year period between 2012 and 2015, vegetation surveys were conducted across a series of Rarity and Diversity plots as a component of a Rare Species Monitoring Project for the Lower Athabasca Region with the intent of evaluating plant species richness, distribution of rare species, and status of rare vascular plants in the oil sands region. Rarity status was determined based on the Subnational Conservation Status Rank scheme used by the Alberta Conservation Information Management System (ACIMS), a biodiversity data centre managed by the provincial government, generated using a NatureServe rank calculator (Master et al. 2012). Rare species were defined here as those with Conservation Status Ranks (S-Ranks) between S1 and S3 (Table 1.1), corresponding to status ranks for species that are “especially” to “somewhat vulnerable” to extirpation. At the project outset, the Status Ranks of species were based on those applied up through 2014. In 2015, species were re-evaluated by ACIMS with these updates having implications for our findings regarding the presence of rare species in the region.

The primary objectives of the Rarity and Diversity plot sampling project were to (1) collect new observations of rare vascular plants to further refine existing rare plant habitat models, (2) test a new rare plant monitoring protocol that complements those of the Alberta Biodiversity Monitoring Institute (ABMI) (ABMI 2010a; ABMI 2010b) and incorporates Alberta Native Plant Council (ANPC) guidelines (ANPC 2000; ANPC 2012) for rare vascular plant surveys, and (3) assess how changes to Conservation Status Ranks affect our understanding of the distribution of rare vascular plant species.
Table 1.1. Subnational Conservation Status Rank (S-Rank) definitions adapted from those used by the Alberta Conservation Information Management System (ACIMS 2017).

<table>
<thead>
<tr>
<th>Rank</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>Five or fewer occurrences for a taxon, or especially vulnerable to extirpation due to various factors</td>
</tr>
<tr>
<td>S2</td>
<td>Twenty or fewer occurrences for a taxon, or vulnerable to extirpation due to various factors</td>
</tr>
<tr>
<td>S2S3</td>
<td>Uncertainty between S2 and S3 status ranks for a taxon</td>
</tr>
<tr>
<td>S3</td>
<td>One hundred or fewer occurrences for a taxon, or somewhat vulnerable to extirpation due to various factors such as restricted range or relatively small population sizes</td>
</tr>
<tr>
<td>S3S4</td>
<td>Uncertainty between S3 and S4 status ranks for a taxon</td>
</tr>
<tr>
<td>S4</td>
<td>Apparently secure; taxon is uncommon but not rare, though potentially some concern due to various factors such as a decline in population sizes</td>
</tr>
<tr>
<td>S4S5</td>
<td>Uncertainty between S4 and S5 status ranks for a taxon</td>
</tr>
<tr>
<td>S5</td>
<td>Secure; taxon is common, widespread, and abundant</td>
</tr>
<tr>
<td>SU</td>
<td>Not ranked due to a lack of information or substantially conflicting information for a taxon, such as with species whose nativeness is unresolved</td>
</tr>
<tr>
<td>SNR</td>
<td>Not ranked because the conservation status has not yet been assessed for a taxon</td>
</tr>
<tr>
<td>SNA</td>
<td>Rank not applicable because a taxon is not a suitable target for conservation activities, such as with species that are introduced</td>
</tr>
</tbody>
</table>

1.3 Methods

Study area
The study area was defined as the Lower Athabasca Planning Region (LAPR), a 93,212 km$^2$ area in northeastern Alberta between 54°N and 60°N latitude (Figure 1.1). Elevation ranges from 202 m to 867 m a.s.l., with a mean annual temperature of -0.9°C and mean annual precipitation of about 438 mm (Zhang et al. 2014). The area represents one-quarter of the boreal forest region of the province and is characterized by a combination of coniferous, deciduous, and mixedwood upland stands dominated by species including aspen (Populus tremuloides), black spruce (Picea mariana), white spruce (Picea glauca), and jack pine (Pinus banksiana). Lowland areas consist of a variety of wetland types including fens, swamps, and bogs along with lakes and streams (Natural Regions Committee 2006; Zhang et al. 2014). Much of the area is occupied by undeveloped forests, however, a portion of the landscape is affected by anthropogenic disturbance related to oil sands development, forest harvesting, and agriculture.
Figure 8.1. Extent of the study area in northeastern Alberta and the location and number of Rarity and Diversity plots (n = 602) sampled per year between 2012 and 2015.

Site selection and plot establishment
Vascular plant surveys were conducted in 602 quarter-hectare plots (50 x 50 m) by 18 observers over four summer field seasons. Sites were selected in native terrestrial upland and lowland environments using a stratified random sampling design and model-based predicted locations of targeted rare plant species, compiled from the Ecological Monitoring Committee for the Lower Athabasca (EMCLA) database (see Appendix 1.1 for target species list). Disturbance and accessibility were also considered, as was expert advice regarding landscape features and habitats within the study region with a high probability of rare plant occurrence. No sites were located in open water ecosystems such as marshes or lakes.

Two plots were established per site, each of which was surveyed at least once by a single observer. Plots were separated by a maximum of 200 m, both to reduce travel time and allow for observers to work in close proximity. Where possible, the two plots per site were established in different ecosite types to promote broad representation in the dataset. Efforts were made to situate one of the plots in an area likely to support rare plants (e.g. open sand, rock faces, sites with unique landscape features, ephemeral habitats, transition zones, old growth forest, or jack pine stands).
To enable the examination of observer error within the dataset, 67 plots were surveyed by multiple individuals (Chapter 4.0). An additional eight plots were surveyed in both the spring and late summer of 2014 by the same individual to determine the influence of timing on species detection. See Appendix 1.2 for locations of all survey plots.

**Ecosite classification at survey plots**
Geographic coordinates of each plot centre were recorded with a handheld GPS unit. Ecosite type was determined based on 12 categories defined by the ABMI (Table 1.2) which reflect dominant vegetation community, structural stage, soil nutrients, and soil moisture level. Additional physical attribute data were collected including plot slope, aspect, dominant canopy species, soil pH, the percentage of the plot that was altered by human or natural disturbance, and percent bare ground or water.

<table>
<thead>
<tr>
<th>Ecosite Categories</th>
<th>Nutrient/Moisture Code</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>NT</td>
<td>Not Treed</td>
<td></td>
</tr>
<tr>
<td>PX</td>
<td>Poor/Xeric</td>
<td></td>
</tr>
<tr>
<td>PM</td>
<td>Poor/Mesic</td>
<td></td>
</tr>
<tr>
<td>PD</td>
<td>Bog (Poor/Hydric)</td>
<td></td>
</tr>
<tr>
<td>MX</td>
<td>Medium/Xeric</td>
<td></td>
</tr>
<tr>
<td>MM</td>
<td>Medium/Mesic</td>
<td></td>
</tr>
<tr>
<td>MG</td>
<td>Medium/Hygric</td>
<td></td>
</tr>
<tr>
<td>MD</td>
<td>Poor Fen (Medium/Hydric)</td>
<td></td>
</tr>
<tr>
<td>RG</td>
<td>Rich/Hygric</td>
<td></td>
</tr>
<tr>
<td>RD</td>
<td>Rich Fen (Rich/Hydric)</td>
<td></td>
</tr>
<tr>
<td>SD</td>
<td>Swamp</td>
<td></td>
</tr>
<tr>
<td>VD</td>
<td>Marsh (Very Rich/Hydric)</td>
<td></td>
</tr>
</tbody>
</table>

**Plant survey methods**
Observers performed time-unlimited surveys using belt transects to cover the entirety of each plot, beginning in one corner and walking in a pattern that mimicked parallel 50-m transects while scanning 1-2 m per side (Figure 1.2). Comprehensive species inventories were completed and data were entered in the field on paper datasheets in 2012 and using handheld tablets in subsequent years. Specimens that could not be identified in the field were collected and later reviewed by an expert botanist. Subspecies, varieties, and hybrids were included, and some records were classified at higher taxonomic levels (genus or family) if identification to species-level was not possible. Time of first encounter for each species detected was recorded (which became automatic following the introduction of tablets in 2013), along with the total survey time per plot. Surveys thus collected species presence-absence data, but not information regarding
abundance. When rare species were detected, however, Rare Plant Field Data Sheets provided by ACIMS were filled out with detailed descriptions of the location, population, and habitat.

Figure 1.9. Path of parallel belt transects used by observers for time-unlimited vascular plant surveys of the Rarity and Diversity plots (50 x 50 m) \((n = 602)\).

1.4 Results

*Vascular plant species richness*

Across the 602 Rarity and Diversity plots, a total of 27,320 observations of 536 plant species were recorded (see Appendix 1.3 for full species list). Average survey time per plot was 93 minutes and ranged from 20 to 290 minutes. Average species richness per ecosite category ranged from a minimum of 26.5 species (PD ecosite) to a maximum of 71.9 (SD), with an overall average of 45.4 and associated standard deviation of 21.5 (Table 1.3). Ecosite representation was not balanced due to a lack of availability in the study area, with few plots for some categories (VD) and many for others (RD). Species richness varied moderately among ecosite categories and was generally consistent within ecosite categories (Figure 1.3).

Table 1.3. Number of Rarity and Diversity plots \((n = 602)\) occurring in each of the 12 ecosite categories, mean vascular plant species richness per plot, and variation (standard deviation) per ecosite category.

<table>
<thead>
<tr>
<th>Ecosite</th>
<th>Number of Plots</th>
<th>Mean Species Richness ((\alpha))</th>
<th>Standard Deviation ((\alpha))</th>
</tr>
</thead>
<tbody>
<tr>
<td>NT - Not Treed</td>
<td>7</td>
<td>64.4</td>
<td>24.2</td>
</tr>
<tr>
<td>PX – Poor Xeric (poor, dry forests)</td>
<td>52</td>
<td>30.6</td>
<td>11.9</td>
</tr>
<tr>
<td>Ecosite</td>
<td>Species richness</td>
<td>Plant species richness</td>
<td>Conservation Status Ranks</td>
</tr>
<tr>
<td>---------</td>
<td>-----------------</td>
<td>------------------------</td>
<td>--------------------------</td>
</tr>
<tr>
<td>PM – Poor Mesic (moist conifer)</td>
<td>97</td>
<td>43.3</td>
<td>21.6</td>
</tr>
<tr>
<td>PD – Poor Hydric (bog)</td>
<td>39</td>
<td>26.5</td>
<td>14</td>
</tr>
<tr>
<td>MX – Medium Xeric (dry mixedwood)</td>
<td>35</td>
<td>44.1</td>
<td>15.4</td>
</tr>
<tr>
<td>MM – Medium Mesic (mesic mixedwood)</td>
<td>93</td>
<td>53.3</td>
<td>12</td>
</tr>
<tr>
<td>MG – Medium Hygric (moist mixedwood)</td>
<td>30</td>
<td>67</td>
<td>22.3</td>
</tr>
<tr>
<td>MD – Medium Hydric (poor fen)</td>
<td>88</td>
<td>36.5</td>
<td>20.2</td>
</tr>
<tr>
<td>RG – Rich Hygric (rich, moist forests)</td>
<td>25</td>
<td>61.2</td>
<td>19.1</td>
</tr>
<tr>
<td>RD – Rich Hydric (rich fen)</td>
<td>126</td>
<td>48.8</td>
<td>22.3</td>
</tr>
<tr>
<td>SD – “Swamp” Hydric (swamp)</td>
<td>8</td>
<td>71.9</td>
<td>20.5</td>
</tr>
<tr>
<td>VD – Very rich Hydric (marsh)</td>
<td>2</td>
<td>30</td>
<td>22.6</td>
</tr>
<tr>
<td>Total</td>
<td>602</td>
<td>45.4</td>
<td>21.5</td>
</tr>
</tbody>
</table>

Figure 1.10. Variation in vascular plant species richness for the 12 ecosite categories sampled across the Rarity and Diversity plots \( (n = 602) \).

**Plant species rarity and changes in Conservation Status Ranks**

The re-evaluation of Conservation Status Ranks by ACIMS in 2015 resulted in the rarity status of the 536 species encountered being downgraded (88 species), upgraded (20), or remaining
unchanged (416), with some previously unevaluated species receiving a new rank (12) (Table 1.4).

**Table 1.4.** Number of species for which Conservation Status Ranks were downgraded, upgraded, newly applied, or unchanged between 2014 and 2015 following the re-evaluation of Status Ranks by ACIMS.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Downgrade to Status Rank</strong></td>
<td></td>
<td>Total: 88</td>
</tr>
<tr>
<td>S1</td>
<td>S2</td>
<td>2</td>
</tr>
<tr>
<td>S1</td>
<td>S2S3</td>
<td>1</td>
</tr>
<tr>
<td>S1</td>
<td>S3</td>
<td>1</td>
</tr>
<tr>
<td>S2</td>
<td>S3</td>
<td>6</td>
</tr>
<tr>
<td>S2</td>
<td>S4</td>
<td>2</td>
</tr>
<tr>
<td>S3</td>
<td>S4</td>
<td>45</td>
</tr>
<tr>
<td>S3</td>
<td>S5</td>
<td>2</td>
</tr>
<tr>
<td>S3S4</td>
<td>S4</td>
<td>2</td>
</tr>
<tr>
<td>S3S4</td>
<td>S4S5</td>
<td>1</td>
</tr>
<tr>
<td>S4</td>
<td>S4S5</td>
<td>3</td>
</tr>
<tr>
<td>S4</td>
<td>S5</td>
<td>21</td>
</tr>
<tr>
<td>S4S5</td>
<td>S5</td>
<td>2</td>
</tr>
<tr>
<td><strong>Upgrade to Status Rank</strong></td>
<td></td>
<td>Total: 20</td>
</tr>
<tr>
<td>S5</td>
<td>S4</td>
<td>13</td>
</tr>
<tr>
<td>S4</td>
<td>S3</td>
<td>4</td>
</tr>
<tr>
<td>S3S4</td>
<td>S3</td>
<td>2</td>
</tr>
<tr>
<td>S3</td>
<td>S2</td>
<td>1</td>
</tr>
<tr>
<td><strong>Rank Newly Applied</strong></td>
<td></td>
<td>Total: 12</td>
</tr>
<tr>
<td>SNR</td>
<td>S3</td>
<td>3</td>
</tr>
<tr>
<td>SNR</td>
<td>S4</td>
<td>1</td>
</tr>
<tr>
<td>SNR</td>
<td>S5</td>
<td>6</td>
</tr>
<tr>
<td>SU</td>
<td>S1</td>
<td>1</td>
</tr>
<tr>
<td>SU</td>
<td>S2</td>
<td>1</td>
</tr>
<tr>
<td><strong>Rank Unchanged</strong></td>
<td></td>
<td>Total: 416</td>
</tr>
<tr>
<td>S1</td>
<td></td>
<td>2</td>
</tr>
<tr>
<td>S2</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>S3</td>
<td></td>
<td>38</td>
</tr>
<tr>
<td>S3S4</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>S4</td>
<td></td>
<td>60</td>
</tr>
<tr>
<td>S5</td>
<td></td>
<td>271</td>
</tr>
<tr>
<td>SNA</td>
<td></td>
<td>43</td>
</tr>
</tbody>
</table>
Based on the Status Ranks through 2014, 101 (18.8%) of the species detected were recognized as provincially rare (S1-S3); however, following the re-evaluation of rankings in 2015, this number decreased to 63 (11.7%) (Table 1.5). For the 2014 rankings, 54 plots (9.0%) were found to contain species determined to be especially vulnerable (S1) or vulnerable (S2), but this declined substantially to 33 plots (5.5%) when based on the 2015 rankings (Tables 1.6 & 1.7; Figure 1.4).

The majority of species detected were considered apparently secure (S3) or secure (>S3), which amounted to 380 and 430 species for the 2014 and 2015 rankings, respectively. A further 12 species were not ranked in 2014 (SNR or SU), but received ranks in 2015. Of the 536 species encountered, 493 were native to Alberta and the remaining 43 did not have an associated rank (SNA), as they were either exotic (41 species) or hybrids (2) and ACIMS does not assign ranks to these species.

Table 1.5. Number of vascular plant species \( (n = 536) \) detected in the Rarity and Diversity plots \( (n = 602) \) per Conservation Status Rank (S-Rank) for ranks used in 2014 and 2015.

<table>
<thead>
<tr>
<th>Conservation Status Rank (2014)</th>
<th>Number of Species Detected (%)</th>
<th>Conservation Status Rank (2015)</th>
<th>Number of Species Detected (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>6 (1)</td>
<td>S1</td>
<td>3 (0.6)</td>
</tr>
<tr>
<td>S2</td>
<td>9 (1.7)</td>
<td>S2</td>
<td>5 (0.9)</td>
</tr>
<tr>
<td>S2S3</td>
<td>-</td>
<td>S2S3</td>
<td>1 (0.2)</td>
</tr>
<tr>
<td>S3</td>
<td>86 (16)</td>
<td>S3</td>
<td>54 (10)</td>
</tr>
<tr>
<td>S3S4</td>
<td>6 (1.1)</td>
<td>S3S4</td>
<td>1 (0.2)</td>
</tr>
<tr>
<td>S4</td>
<td>88 (16.4)</td>
<td>S4</td>
<td>123 (22.9)</td>
</tr>
<tr>
<td>S4S5</td>
<td>2 (0.4)</td>
<td>S4S5</td>
<td>4 (0.7)</td>
</tr>
<tr>
<td>S5</td>
<td>284 (53)</td>
<td>S5</td>
<td>302 (56.3)</td>
</tr>
<tr>
<td>SNA</td>
<td>43 (8)</td>
<td>SNA</td>
<td>43 (8)</td>
</tr>
<tr>
<td>SNR</td>
<td>10 (1.8)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>SU</td>
<td>2 (0.4)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Total</td>
<td>536</td>
<td>Total</td>
<td>536</td>
</tr>
</tbody>
</table>

Table 1.6. Number of records for vascular plant species recognized as especially vulnerable (S1; \( n = 6 \)) or vulnerable (S2; \( n = 9 \)) based on the 2014 Conservation Status Ranks which were detected in the Rarity and Diversity plots \( (n = 54 \) of 602 total).

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Carex adusta</td>
<td>S1</td>
<td>S3</td>
<td>7</td>
</tr>
<tr>
<td>Carex hystericina</td>
<td>S1</td>
<td>S2</td>
<td>1</td>
</tr>
<tr>
<td>Lechea intermedia var. depauperata</td>
<td>S1</td>
<td>S1</td>
<td>1</td>
</tr>
<tr>
<td>Malaxis paludosa</td>
<td>S1</td>
<td>S2S3</td>
<td>9</td>
</tr>
<tr>
<td>Spiranthes lacera</td>
<td>S1</td>
<td>S2</td>
<td>3</td>
</tr>
</tbody>
</table>
Table 1.7. Number of records for vascular plant species recognized as especially vulnerable (S1; \( n = 3 \)) or vulnerable (S2; \( n = 5 \)) based on the 2015 Conservation Status Ranks which were detected in the Rarity and Diversity plots (\( n = 33 \) of 602 total).

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Carex hystericina</td>
<td>S1</td>
<td>S2</td>
<td>1</td>
</tr>
<tr>
<td>Lechea intermedia var. depauperata</td>
<td>S1</td>
<td>S1</td>
<td>1</td>
</tr>
<tr>
<td>Spiranthes lacera</td>
<td>S1</td>
<td>S2</td>
<td>3</td>
</tr>
<tr>
<td>Utricularia cornuta</td>
<td>S1</td>
<td>S1</td>
<td>1</td>
</tr>
<tr>
<td>Botrychium simplex</td>
<td>S2</td>
<td>S2</td>
<td>1</td>
</tr>
<tr>
<td>Cardamine dentata</td>
<td>S3</td>
<td>S2</td>
<td>3</td>
</tr>
<tr>
<td>Dichanthelium acuminatum</td>
<td>SU</td>
<td>S2</td>
<td>5</td>
</tr>
<tr>
<td>Leucophysalis grandiflora</td>
<td>SU</td>
<td>S1</td>
<td>18</td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td></td>
<td>33</td>
</tr>
</tbody>
</table>
Figure 1.4. Distribution of especially vulnerable (S1) and vulnerable (S2) vascular plant species across the Rarity and Diversity plots within the study area, based on the ranks that applied until 2014 ($n = 54$ of 602 total plots) and those used in 2015 ($n = 33$ plots).

Some ecosite types were found to support more rare species, although the particular categories with the highest average numbers of these changed between the 2014 and 2015 rankings (Tables 1.8 & 1.9; Figures 1.5 & 1.6). For the 2014 rankings, in descending order, the categories RD, SD, and MD had the three highest combined averages of S1-S3 species, but in 2015 these shifted to RD, MD, and PX. Conversely, the three categories with the lowest combined averages of rare species for the 2014 rankings in descending order were VD, PD, and MX, but these changed to PD, MG, and VD in 2015.

Table 1.8. Mean number of species of different Conservation Status Ranks (2014) present across the Rarity and Diversity plots ($n = 602$) per ecosite category.

<table>
<thead>
<tr>
<th>Conservation Status Rank (2014)</th>
<th>Mean Number of Species Present Across Plots</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ecosite</td>
<td>S1</td>
</tr>
<tr>
<td>-------------------------------</td>
<td>-----</td>
</tr>
</tbody>
</table>

Table continued...
Table 1.9. Mean number of species of different Conservation Status Ranks (2015) present across the Rarity and Diversity plots ($n = 602$) per ecosite category.

<table>
<thead>
<tr>
<th>Conservation Status Rank (2015)</th>
</tr>
</thead>
</table>

**Figure 1.11.** Mean vascular plant species richness for the 12 ecosite categories sampled across the Rarity and Diversity plots ($n = 602$) and the number of species per Conservation Status Rank (2014).
Figure 1.6. Mean vascular plant species richness for the 12 ecosite categories sampled across the Rarity and Diversity plots ($n = 602$) and the number of species per Conservation Status Rank (2015).
1.5 Discussion

A total of 536 vascular plant species were detected in the 602 quarter-hectare Rarity and Diversity plots surveyed across the study area between 2012 and 2015. Average richness was 45.4 species per plot across all 12 ecosite categories, with SD (swamps) and PD (oligotrophic bogs) supporting the greatest and least diversity, respectively. Swamps, seasonally flooded wetlands with a mineral substrate, most often occur as small habitat patches in Alberta with microsites (hummocky micro-terrain) that promote species diversity. While peatlands include many, often diverse, types, oligotrophic bogs are characterized by exceptionally low nutrients, high acidity, and waterlogged organic substrate, conditions which limits the number of species capable of establishing and surviving in these habitats.

Changes to Conservation Status Ranks affected the number and distribution of rare species (S1-S3) encountered during sampling, which decreased from 101 to 63 species between 2014 and 2015 following the re-evaluation of rankings. We encountered at least one S3 species at nearly all sample plots, demonstrating the efficacy of model-directed adaptive sampled as applied here. Using the 2015 S-ranking we have located eight S1 or S2 species at 33 plots. The downgrading of Status Ranks for a large number of vascular plant species within the study area is likely in part an artefact of increased sampling effort over time by monitoring projects, such as what we have conducted here (all rare species detected in the project were submitted to ACIMS), and a large number of pre-disturbance assessments for oil sands developments that have led to a greater understanding of plant rarity in the region.
CHAPTER 2.0: Landscape patterns of rare vascular plants in the Lower Athabasca region of Alberta, Canada

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² Forest Management Branch, Forestry Division, Alberta Agriculture and Forestry
³ Application Center, Alberta Biodiversity Monitoring Institute
⁴ Environmental Monitoring and Science Division, Albert Environment and Parks
⁵ Department of Forest Resource Management, University of British Columbia

2.1 Summary

We used 602 quarter-hectare plots in the Lower Athabasca region to model rare vascular plant (S1 and S2 conservation status) habitat across the area based on landscape predictors of land cover (Ducks Unlimited-Enhanced Wetland Classification), LiDAR-derived vegetation structure, soils (pH), and terrain wetness. The LiDAR-derived and land cover variables were the most important predictors of rarity when considered individually for the 2012-2014 and 2015 conservation ranks respectively. Amount of rare plant habitat in the region was reduced substantially with changes in the new status rankings and shifted in its location. Generally, patterns of rarity went from fens being among the most important sites for encountering S1 and S2 ranked species to sandy, pine forests (Athabasca Plain) being the most important sites. Maps of rare plant habitat developed from this work are being used as an indicator for the Biodiversity Management Framework regional planning initiative for the Lower Athabasca. It also has value for lease-scale environmental assessments and mitigation planning, informing future monitoring programs and sites, and more generally in helping us understand the factors that promote or limit rare vascular plants within Alberta’s boreal forest.

2.2 Introduction

The oil sands region of Alberta represents a major source of environmental concern (Rooney, Bayley & Schindler 2012). Although significant efforts have been made toward reclamation of surface mineable oil sands and determining the responses of individual plant species to reclamation treatments (Renault et al. 1998; Purdy, Macdonald & Lieffers 2005; Price, McLaren & Rudolph 2010), much less is known about the effects of in situ oil sands developments where bitumen is extracted through sub-surface wells. In situ oil sands results in a much lower total anthropogenic footprint (~10-20% loss), but remaining native habitats are highly fragmented from the linear nature of disturbances (seismic lines, roads, pipelines, and transmission lines). Together with the permanent footprints, the loss and fragmentation of boreal forests is implicated in the declines of some key boreal species, with the most prominent being woodland caribou. Although much has now been done to understand the factors contributing to caribou decline (e.g. Dyer et al. 2002, Boutin et al. 2012), much less is known about other taxa, particularly non-
vertebrates, including plants. This has resulted in major knowledge gaps within the in-situ oil sands region on taxa such as plants, particularly for sensitive, rare vascular plants.

Sampling of rare vascular plants is difficult due to the fact that they are often cryptic in nature and isolated to specific, uncommon habitats. This has partly contributed to our current knowledge gaps, including information on where rare plants are most likely to occur and how they may respond to disturbances created by in situ oil extraction. Such information is needed for regional conservation assessments, effective land use planning, and for guiding mitigation activities and monitoring programs. Although vascular plant monitoring in Alberta is broadly addressed by the Alberta Biodiversity Monitoring Institute (ABMI) (Stadt et al. 2006), sampling intensity of the ABMI monitoring network is too diffuse (grid of 20 km) to be of value for assessments of local responses of species (Nielsen et al. 2009). It also suffers from low detection rates of rare species given a time-limited survey protocol and large plot size (1 ha) coupled with high observer errors (Zhang et al. 2014). In addition to these systematic monitoring sites, oil sands companies are required to perform Pre-Disturbance Assessment (PDAs) surveys prior to developing individual footprints. These PDAs include rare plant surveys that provide location data and general information for populations encountered on lease areas, but do not lend themselves to monitoring since sites are often later developed, survey effort is largely unknown, and information is not scaled past individual sites on oil sands leases. Complementary methods for rare plant surveys and regional monitoring of rare plants are needed to inform rarity of plants at regional scales, including the development of mapping tools that can be used by government and industry for the conservation and management of rare plant resources and environmental impact assessments at the scale at which projects (leases) occur.

Model-based sampling designs, where information from spatial models are used to guide sampling effort (locations), have been proposed as an alternative to the commonly employed random or systematic designs that dominate current monitoring and survey efforts (Guisan et al. 2006). In addition to creating efficiencies (up to 70% cost savings), model-based, adaptive designs provide up-to-date products that can be used to not only guide future sampling effort, but also provide critical information for making relevant management decisions related to the original monitoring objectives.

The purpose of this chapter is to develop, test, and apply an adaptive, model-based sampling design that defines rare plant habitat in the Lower Athabasca region of northeast Alberta. Specific to that goal, our objectives were two-fold: (1) identify the landscape factors that most affect presence of rare vascular plants in the Lower Athabasca; and (2) predict (map) rare plant habitat in the Lower Athabasca region. It is this region of Alberta that has the most extensive in situ oil sands operations thus requiring spatial tools to assist with land use decision-making, regional monitoring, and stewardship.

2.3 Methods

Study area, field plots, and definitions of rarity

We sampled rare vascular plants within the Lower Athabasca region in northeast Alberta, Canada over four years through the Rarity and Diversity plots for the Lower Athabasca project. Field methods are described in Zhang et al. (2014) and Chapter 1 of this report. Below we
summarize those methods and describe in more detail the allocation of sampling effort. Specifically, we used a stratified sampling approach to allocating field efforts based on the Ducks Unlimited Enhanced Wetland Classification (DU-EWC) and preliminary landscape models predicting rare plant locations that were periodically updated based on historic locations of rare plants (Alberta Conservation Information Management Systems [ACIMS] and industry Pre-disturbance assessments) and locations collected from the prior year’s surveys (Nielsen 2011). Stratification was thus adaptive to new information collected from field surveys (i.e., model-based iterative sampling). Model-based sampling designs are an alternative to static traditional fully random or stratified designs. With proper information guiding the adaptive sampling process, major cost savings (up to 70% over random) can be gained (Guisan et al. 2006).

Initially, S1, S2, and S3 plant population (sub-national rarity status ranks for Alberta assigned by ACIMS) locations were used to model potential landscape locations of rare plants within land cover types (Nielsen 2011). This was used to guide stratification from all known rare plant records. Later, landscape models used plot data from this research project on locations of where S1 and S2 plants were present. We excluded S3 ranked species as encounter rates of any S3 plant in a plot approached 100%, thus making their inclusion as a group in models meaningless. Landscape predictors included the DU-EWC land cover types and terrain and edaphic variables. The DU-EWC land cover classification scheme includes a number of detailed wetland classes (Table 2.1), such as graminoid rich fen. Separation of wetland types was desirable given the prevalence and importance of lowland land cover types in the Lower Athabasca. Initial model predictions of rarity were separated for each DU-EWC land cover type and sample sites within each ‘native’ land cover type was selected through randomization.

Table 2.1. List of Ducks Unlimited Enhanced Wetland Classification land cover types considered for models of rare plant occurrence in the Lower Athabasca region of northeast Alberta (source: Ducks Unlimited). Note that some classes (aquatic and anthropogenic were not listed or used in models).

<table>
<thead>
<tr>
<th>Class Name</th>
<th>Type of community</th>
<th>Soil Moisture</th>
<th>Hydro-dynamics</th>
<th>Nutrient Regime</th>
</tr>
</thead>
<tbody>
<tr>
<td>Emergent Marsh</td>
<td>Mineral Wetland</td>
<td>Very Hydric</td>
<td>Very Dynamic</td>
<td>Very Rich</td>
</tr>
<tr>
<td>Meadow Marsh</td>
<td>Mineral Wetland</td>
<td>Hydric</td>
<td>Very Dynamic</td>
<td>Very Rich</td>
</tr>
<tr>
<td>Graminoid Rich Fen</td>
<td>Peat Wetland</td>
<td>Hydric</td>
<td>Moving</td>
<td>Rich</td>
</tr>
<tr>
<td>Graminoid Poor Fen</td>
<td>Peat Wetland</td>
<td>Hydric</td>
<td>Slow Moving</td>
<td>Poor</td>
</tr>
<tr>
<td>Shrubby Rich Fen</td>
<td>Peat Wetland</td>
<td>Hydric</td>
<td>Moving</td>
<td>Rich</td>
</tr>
<tr>
<td>Shrubby Poor Fen</td>
<td>Peat Wetland</td>
<td>Sub Hydric</td>
<td>Slow Moving</td>
<td>Poor</td>
</tr>
<tr>
<td>Treed Rich Fen</td>
<td>Peat Wetland</td>
<td>Sub Hydric</td>
<td>Moving</td>
<td>Rich</td>
</tr>
<tr>
<td>Treed Poor Fen</td>
<td>Peat Wetland</td>
<td>Hydric</td>
<td>Slow Moving</td>
<td>Poor</td>
</tr>
<tr>
<td>Open Bog</td>
<td>Peat Wetland</td>
<td>Sub Hydric</td>
<td>Stagnant</td>
<td>Very Poor</td>
</tr>
<tr>
<td>Shrubby Bog</td>
<td>Peat Wetland</td>
<td>Sub Hydric</td>
<td>Stagnant</td>
<td>Very Poor</td>
</tr>
<tr>
<td>Treed Bog</td>
<td>Peat Wetland</td>
<td>Sub Hydric</td>
<td>Stagnant</td>
<td>Very Poor</td>
</tr>
<tr>
<td>Shrub Swamp</td>
<td>Mineral Wetland</td>
<td>Hydric</td>
<td>Dynamic</td>
<td>Rich</td>
</tr>
<tr>
<td>Hardwood Swamp</td>
<td>Mineral Wetland</td>
<td>Hydric</td>
<td>Dynamic</td>
<td>Rich</td>
</tr>
<tr>
<td>Mixedwood Swamp</td>
<td>Mineral Wetland</td>
<td>Hydric</td>
<td>Dynamic</td>
<td>Rich</td>
</tr>
<tr>
<td>Tamarack Swamp</td>
<td>Mineral Wetland</td>
<td>Hydric</td>
<td>Slow Moving</td>
<td>Medium</td>
</tr>
<tr>
<td>Conifer Swamp</td>
<td>Mineral Wetland</td>
<td>Sub Hydric</td>
<td>Stagnant</td>
<td>Medium</td>
</tr>
<tr>
<td>Upland Conifer</td>
<td>Upland</td>
<td>Mesic to Xeric</td>
<td>Upland</td>
<td>Upland</td>
</tr>
</tbody>
</table>
Sample sites were constrained to within a 2.5 km radius of roads with areas predicted to have greater chance of a rare plant being present emphasized. Anthropogenic habitats (clearcuts, agriculture, industry developments) and aquatic habitats dominated by open water were not considered in this study. Site randomization was done in ArcGIS using the Create Spatially Balanced Points tool where locations were spread across the available region and scaled so that more random locations were allocated within areas having higher probabilities rare plants (input inclusion probability raster). Random sites included oil sands leases, areas not currently leased, and provincial parks such as Lakeland and Sir Winston Churchill, but did not include the Cold Lake Air Weapons Range, remote areas such as the Birch Mountains, and the entire Canadian Shield north of Lake Athabasca which does not contain hydrocarbons and thus is not threatened from energy developments. In a few instances helicopter support was available and used to access a limited number of remote sites near Fort McMurray including plot locations on Stony Mountain and areas surrounding Gypsy Lake Wildland.

Because rare plants were more likely to occur in particular land cover types, such as fens, we sampled more locations of these land cover types, as well as some land cover types that dominated the region such as deciduous forest, but may have had some microsite or meso-terrain condition that would increase the likelihood of rare plants being present (Figure 2.1b). Emphasis on particular land cover types was determined based on initial queries describing known rare plant records by land cover type. Chapter 1 describes the number of plot locations by ecosites which relate to land cover types from the DU-EWC. All rare plant surveys were completed during the summer months (mid-June to mid-August). Plot size was 0.25 ha (50 x 50 m) with observers allowed to complete the plot without time constraints. Although the emphasis of this project was rare plants, we recorded the presence of all vascular plants within plots in order to fully describe assemblages and to provide more information on general plant biodiversity. Given the large plot size, no effort was made to estimate cover or abundance of common species.

All observers had previous experience with plant surveys with additional training provided in the herbarium (emphasis on S1-S3 plants in the region) and in the field. Unknown plants within plots were collected for later identification. Observers working in teams of two navigated to stratified plots using handheld GPS units. One observer established and surveyed the target plot based on the stratified random location, while the second observer established a paired plot within 200 m of the target plot and in a different land cover type to ensure independence among observations. The paired plot design among observers was used to satisfy safety protocols that limited observers from working no further than 200 m apart. The perimeter of each plot was delineated using 50 and 100 m transect tapes. Observers then surveyed their plot without assistance by walking the plot in ~2 m belt transects and stopping to record all new vascular plant species encountered and the time of observation. Rare plants (S1-S3) were flagged and after the completion of the survey returned to in order to fill in an ACIMS field data sheet describing the habitat, microhabitat, GPS coordinates, population size, and other attributes. Rare plant records were submitted annually to the Government of Alberta’s ACIMS program. In total, 602 unique sites were sampled over a 4-year period (2012-2015; Figure 2.1a) with 67 sites re-surveyed.
multiple times within the same day by a separate observer in order to evaluate observer error (see Chapter 4). For the purpose of this chapter, we use the first survey session at a site for those cases where the site was surveyed more than once.

**Figure 2.1.** (a.) Location of field plots and (b.) number of plots sampled per land cover type (dominant type within plot).

**Landscape predictors of rare plant habitats**

Environmental predictors of rarity included spatially-explicit variables representing soil conditions (soil pH) (Figure 2.2d), land cover from Ducks Unlimited Enhanced Wetland Classification (DU-EWC) (Figure 2.2b), terrain-derived moisture index (2.2c), and vegetation structure from airborne LiDAR sensors that measures variation in height and structure (Figure 2.2a). LiDAR-derived vegetation structure variables were available for most, but not all, parts of the study area (see Figure 2.2a) effectively representing crown lands outside of the Cold Lakes Air Weapons Range. Models using LiDAR data therefore also represent a subset of plots with a total of 469 plots available within areas having LiDAR data. LiDAR point cloud metrics were summarized for the region at the scale of the plot (50 m raster) using FUSION software (McGaughey 2016). LiDAR-derived variables used for models included canopy relief ratio (CRR), maximum canopy height (95th centile), and standard deviation in canopy height. An example land cover type for the region is shown in Figure 2.2b. The terrain-derived moisture index was estimated from a 50-m digital elevation model (DEM) using the Compound Topographic Index (CTI) method (Moore et al. 1993, Gessler et al. 1995). Although a smaller area was available for depth-to-water (DTW) from the Wet Area Mapping program, comparisons within that zone suggested that the CTI model from a lower resolution DEM performed as good or better than the more detailed DTW predictions and thus CTI was subsequently used in all models. Soil conditions were measured by soil pH based on Soil Landscapes of Canada version 3.2 (Soil Landscapes of Canada Working Group 2010). Although other soil variables were
available, they were either highly correlated with soil pH or did not correlate with rare plant locations. Climate variables were not used in models since the region is quite small relative to differences in climate and because the coldest parts of the study area (i.e. the Birch Mountains) were not sampled. All final predictor variables were scaled to a 50-m raster cell size to ensure that they matched the scale of plots and other rasters. Highly correlated variables \((r > |0.7|)\) were removed from analyses by choosing only one of the correlated variables, thus avoiding problems of multicollinearity.

**Figure 2.2.** Example landscape variables used to predict locations of rare vascular plants (S1 or S2 conservation status): (a.) LiDAR-derived canopy height (95\textsuperscript{th} centile; note that gray areas represent locations without LiDAR data); (b.) land cover (deciduous forest example; Ducks Unlimited); (c.) terrain wetness from 50 m DEM; and (d.) soil pH.

**Models of rare plant habitats**

We used logistic regression to model the probability of a S1 or S2 rare plant being present at a site based on landscape characteristics (predictors) in order to estimate rare plant habitats across the region. Models used the presence of any S1 or S2 plant within our 602 plots as the response variable (historic ACIMS locations were not used) and landscape variables as predictors. During the course of this study ACIMS reclassified the status of vascular plants in Alberta resulting in major changes to what we defined as rare plants (see Chapter 1 for a detailed review of changes). As a result, we developed two sets of models of plant rarity based on the two different periods of defined rarity. The first model represented 2012-2014 rankings and the second rankings for the 2015-current period. In both cases, all field plots (years of data) were used and differences only reflect the change in ranking of species.
Model selection was based on Akaike’s Information Criteria (AIC, Akaike 1974) where different sets of landscape predictors were used to compare support among candidate models (sets of variables). Given the large number of parameters and possible candidate models, parameters of the most supported model were inspected and where obvious weak responses were evident, variables were removed to be more parsimonious as confirmed by AIC scores. Final model parameters were then reported, including traditional statistics of model and parameter significance. Parameters were then used to predict rare plant habitat across the region using ArcGIS map calculator. Model performance and predictive accuracy of final selected models were based on percent deviance explained (pseudo-\(R^2\)) and Area-Under-the-Curve Receiver Operating Characteristic (AUC-ROC). Although ecological models often have poor explanatory power (2-5% \(r^2\), Møller & Jennions 2002), we considered models with pseudo-\(R^2 > 0.2\) as being reasonably explanatory. To confirm predictive accuracy of models, AUC-ROC values were ranked based on model training data with values < 0.7 are considered to represent poor model accuracy, values between 0.7–0.9 good model accuracy, and values > 0.9 high model accuracy (Swets 1988, Manel et al. 2001).

As airborne LiDAR data describing vegetation structure within the region were not available across the entire study area (Figure 2.2a), models were first developed for the area with LiDAR data and secondly for the remaining areas using more general landscape predictors. Final map predictions of rare plant habitat were then fused with the LiDAR-based predictions used wherever available and the more general model used where LiDAR data were not available. This fusion was done using the Conditional tool in ArcGIS.

2.4 Results

**Rare plant habitat (S1 & S2 ranking, 2012-2014)**

Of the 602 plots sampled, 47 had at least one S1 or S2 ranked vascular plant (39 plots when limited to the extent of available LiDAR data) using the 2012 to 2014 ACIMS rankings (Figure 2.4a). Occurrence of rare plants by land cover type (proportion) varied from 0 in marsh and open bog to 0.23 (more than 1 out of 5 plots) in graminoid-poor-fen (Figure 2.3). Other land cover types frequently occupied by S1 and S2 plants included tree-poor-fen, treed-rich-fen, shrub swamp, graminoid-rich-fen, and upland pine forests (Figure 2.3).
Figure 2.3. Encounter rate patterns of rare vascular plants (S1 or S2) within study plots based on dominant land cover type within the plot. Two rates are reported based on 2012-2014 ranking (‘2014 S1/S2’ light gray) and the most recent ranking (‘2015 S1/S2’ dark gray). See Chapter 1 for summary data by ecosite.
The most supported model predicting rare plant presence for the 2012-14 ranked S1 and S2 species included all individual landscape factors related to soils, terrain (wetness), land cover, and vegetation structure (Table 2.2). When considering individual (single) factors, vegetation structure from LiDAR was more supported (ΔAIC > 4) than any other factors, followed by land cover and terrain wetness. Soil pH was similar to the null model suggesting no support for that factor when considered individually. The most supported two-factor model included vegetation structure (LiDAR) and land cover. Interestingly, when considering 3 combined landscape factors, soil pH was added despite initially being neutral. The final adjusted global model that contained all 4 landscape factors had good model fit (pseudo-$R^2$ of 0.228) and model accuracy (ROC = 0.841; Table 2.2). Ranking of the importance of land cover types were similar to those described above (Figure 2.3). Overall, graminoid-poor-fens had the highest rate of rare plants (Table 2.2). Both soil pH and terrain wetness (CTI) had non-linear responses along their gradients with peak occurrence of rare plants at moderate levels of soil pH and wetness. Finally, for vegetation structure metrics the canopy relief ratio (CRR) was found to be positively related to rare plant occurrences, while vegetation height (95th centile) was negatively related to rare plant occurrences (Table 2.3). Parameters included in the model without LiDAR metrics were

**Figure 2.4.** Distribution of field plots with the presence of at least one vascular plant species ranked as a conservation status of S1 or S2 within the Lower Athabasca region based on (a.) 2012-2014 rankings or (b.) 2015 rankings.
similar to those with LiDAR variables (Table 2.3), while still maintaining reasonably good model fit (pseudo-R² = 0.182) and similar overall model accuracy (ROC = 0.812).

**Table 2.2.** Comparison of candidate models describing the presence of S1 or S2 rare plant within the Lower Athabasca region based on soils (S), terrain (T), land cover (L), and vegetation structure derived from airborne LiDAR data (V). AIC values in bold font represented the most-supported model (lower AIC is better) within the set of models tested (by single, two, three, and four-factor sets). Model complexity represented by number of parameters (K). ROC represents model predictive accuracy, while model fit (R²) was measured by percent deviance explained. The adjusted global model was the final model used for explanation and model prediction.

<table>
<thead>
<tr>
<th>Model</th>
<th>2012-14 ranking (S1 or S2)</th>
<th>2015 ranking (S1 or S2)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>AIC</td>
<td>K</td>
</tr>
<tr>
<td>Single factor models:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>S-Soils</td>
<td>270.48</td>
<td>3</td>
</tr>
<tr>
<td>T-Terrain wetness</td>
<td>266.79</td>
<td>3</td>
</tr>
<tr>
<td>L-Land cover</td>
<td>254.54</td>
<td>11</td>
</tr>
<tr>
<td>V-Vegetation structure</td>
<td><strong>250.60</strong></td>
<td>4</td>
</tr>
<tr>
<td>Null (constant)</td>
<td>270.65</td>
<td>1</td>
</tr>
<tr>
<td>Two-factor models:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>S+T</td>
<td>267.55</td>
<td>5</td>
</tr>
<tr>
<td>S+L</td>
<td>251.07</td>
<td>13</td>
</tr>
<tr>
<td>S+V</td>
<td>249.60</td>
<td>6</td>
</tr>
<tr>
<td>T+L</td>
<td>252.33</td>
<td>13</td>
</tr>
<tr>
<td>T+V</td>
<td>248.63</td>
<td>6</td>
</tr>
<tr>
<td>L+V</td>
<td><strong>247.09</strong></td>
<td>14</td>
</tr>
<tr>
<td>Three-factor models:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>S+T+L</td>
<td>249.89</td>
<td>15</td>
</tr>
<tr>
<td>S+T+V</td>
<td>248.69</td>
<td>8</td>
</tr>
<tr>
<td>S+L+V</td>
<td><strong>243.70</strong></td>
<td>16</td>
</tr>
<tr>
<td>T+L+V</td>
<td>244.56</td>
<td>16</td>
</tr>
<tr>
<td>Four-factor models:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Global (S+T+L+V)</td>
<td>242.62</td>
<td>18</td>
</tr>
<tr>
<td>Final model</td>
<td><strong>241.45</strong></td>
<td>17</td>
</tr>
</tbody>
</table>

**Table 2.3.** Logistic regression parameters for the most-supported (AIC) model (with and without LiDAR data) describing probability of a S1 or S2 vascular plant being present in the Lower Athabasca region of Alberta using 2012-2014 conservation status ranking. Land cover variables are in comparison to the reference category of deciduous forest.
Map predictions of the 2012-2014 S1 and S2 ranked vascular plant habitat showed patchy patterns of rare plant habitat throughout the region reflecting the value of a number of land cover types and other landscape factors (Figure 2.5a). Some notable sites included the southern parts of the Birch Mountains, the area around Winfred Lake east of Conklin, and Marguerite River Wildland along the Saskatchewan border east of Fort McKay.

**Rare plant habitat (S1 & S2 ranking, 2015-current)**

Of the 602 plots sampled, 31 had at least one S1 or S2 ranked vascular plant (27 plots when limited to the extent of available LiDAR data) as ranked by conservation status using 2015 rankings (Figure 2.4a). Noticeable changes in frequency of rare plant encounters were observed with the recent change in conservation status of plants. Proportion of occurrence by land cover varied from 0 for most land cover types to 0.16 in upland pine forests which nearly doubled in encounter rate of rare plants between 2014 and 2015 (Figure 2.3). This was largely due to previously unclassified species that were specialists to dry sandy plains (Athabasca Sand Plain) being ranked to S1 or S2 status with many sites on the sand plain now classified as having a conservation-ranked species. Other land cover types frequently occupied by S1 and S2 plants included tree-rich-fen, shrub-rich-fen, treed swamp, and upland conifer forests (Figure 2.3).
Figure 2.5. Predicted distribution of rare vascular plants (S1 or S2 conservation rank) within the Lower Athabasca of northeast Alberta, Canada based on landscape predictors and either (a.) 2012-2014 conservation status or (b.) 2015 conservation status. Prediction classes are based on model sensitivity, specificity, and optimal threshold classification probability (unlikely = sensitivity ≥ 0.9; low = sensitivity < 0.9, while being lower than the optimal threshold probability; moderate = sensitivity < 0.9, while being higher than the optimal threshold probability; high = specificity > 0.9 and higher than the optimal threshold probability).

The most supported model predicting rare plant presence for the 2015-ranked S1 and S2 species included all of individual landscape factors related to soils, terrain (wetness), land cover, and vegetation structure (Table 2.2). When considering individual (single) factors, land cover was much more supported than the other factors, followed by vegetation structure from LiDAR, soils, and terrain wetness. All single factors models were more supported than the null model. The most supported two- and three-factor models included vegetation structure (LiDAR) and land cover for the two-factor model with soil pH added for the three-factor model (similar to 2012-14
conservation status model). The final adjusted global model that contained these same 3 landscape factors with further simplification and overall good model fit (pseudo-$R^2$ of 0.347) and model accuracy (ROC = 0.891; Table 2.1). Soil pH had a non-linear response with peak occurrence of rare plants at moderate pH levels. Finally, for LiDAR-derived vegetation structure metrics the canopy relief ratio (CRR) was positively related to rare plant occurrences, while the standard deviation in canopy height was negatively related to rare plant occurrences (Table 2.3). Parameters included in the model without LiDAR metrics were similar to those with LiDAR variables, but with the upland conifer land cover type removed as there was less evidence for its inclusion once vegetation structure variables were removed (Table 2.4).

Table 2.4. Logistic regression parameters for the most-supported (AIC) model (with and without LiDAR data) describing probability of a S1 or S2 vascular plant being present in the Lower Athabasca region of Alberta using 2015 conservation status ranking. Land cover variables are in comparison to the reference category of deciduous forest and other unlisted native habitats.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Model w/ LiDAR variables</th>
<th>Model w/o LiDAR variables</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Coef.</td>
<td>SE</td>
</tr>
<tr>
<td>Soil pH</td>
<td>9.932</td>
<td>4.412</td>
</tr>
<tr>
<td>Soil pH$^2$</td>
<td>-1.069</td>
<td>0.478</td>
</tr>
<tr>
<td>L-Treed-rich-fen</td>
<td>2.650</td>
<td>1.345</td>
</tr>
<tr>
<td>L-Conifer swamp</td>
<td>4.130</td>
<td>1.244</td>
</tr>
<tr>
<td>L-Upland conifer</td>
<td>2.323</td>
<td>1.374</td>
</tr>
<tr>
<td>L-Upland pine</td>
<td>4.172</td>
<td>0.869</td>
</tr>
<tr>
<td>V-Canopy relief ratio</td>
<td>3.853</td>
<td>1.879</td>
</tr>
<tr>
<td>V-Canopy height (St.Dev.)</td>
<td>-0.796</td>
<td>0.252</td>
</tr>
<tr>
<td>Constant (intercept)</td>
<td>-27.41</td>
<td>9.957</td>
</tr>
</tbody>
</table>

Map predictions of 2015 S1 and S2 ranked vascular plant habitat showed distinct pattern with the Athabasca Sand Plain having the greatest likelihood of encountering rare plants (Figure 2.5b). Areas of treed-rich-fen, upland conifer, and conifer swamp were the other parts of the region showing distinct patterns of higher rare plant occurrences. The extent of these regions was much less than that of the 2012-14 predictions, where much more of the central and southern Lower Athabasca contained rare plant habitat (Figure 2.5a). This demonstrates the effect of the reclassification of the conservation status of plants with the key result being the emphasis the far northern sand plain and the de-emphasis of many of the fens and bogs common to the central parts of the study area.

2.5 Discussion

Rare plant (S1 & S2) habitat in the Lower Athabasca was modeled for the region for both the 2012-2014 conservation status period and the more recent 2015 to current period. Initial 2012-2014 models demonstrated significant areas of rare plant habitat throughout the in situ region, including high rates of encounter in most of the fens. In 2015 this pattern changed with changes in status ranking of species with more importance placed on drier sandy habitats such as the Athabasca Sand Plain in the north (area north of McClelland Lake). We suspect that a number of
species ranked as S1 and S2 on the sand plain are more common than current information provides (data and knowledge gaps). In fact, one species, *Leucophysalis grandiflora*, was given an S1-ranked status in 2015, yet found within 18 of 602 sites (see Chapter 1) suggesting that it is much more common than other available information suggests. Conversely, many sites where rarity was downgraded in models (e.g. fens) due to the collection of historic records of species associated with those habitats, may require further assessments and monitoring to ensure records associated with nearby disturbances haven’t resulted in their loss (see Chapter 7 on estimates of extirpation rates).

When considering landscape predictors of rarity, we found that not only was the Ducks Unlimited Enhanced Wetland Classification effective in predicting rarity, but so was LiDAR-derived vegetation structure metrics (Coops *et al*. 2007, 2016), particularly vegetation height (95th percentile) and the canopy relief ratio. In fact, when considered individually, the LiDAR-derived vegetation metrics were similar to better than land cover in predicting rare plant habitat. This suggests that remote-sensing based proxies of rare plant habitat may be used to not only predict current habitat, but also potentially used for monitoring change. More work is needed to validate these new relationships and to better understand mechanisms of those relationships. Regardless, some clear patterns and associations between land cover types (e.g. fens, pine forests) and vegetation structure provide a basis for understanding regional patterns in rarity. It should be noted that rare plants can occur in species-poor sites, like pine forests, and thus approaches to conserving the most diverse communities will not satisfy conservation of rare species and thus principles of complementary need to be considered.

Finally, model (map) outputs should be used for regional to local assessments planning. Currently, this product is being used in the Land Use Framework’s regional planning for the Lower Athabasca as a Biodiversity Management Framework (BMF) indicator. This suggests that oil sands operators should consider use of the rare plant habitat models developed here when doing lease-scale environmental assessments in order to guide site-level surveys and to identify approaches to plan developments that minimize their impact on important rare plant habitat.

**Implications for management and conservation**

Rare vascular plants of current conservation concern within northeast Alberta (S1 & S2 sub-national status) were found mostly within fens, especially treed-rich and shrub-rich fens, pine forests, treed swamps, upland conifer forests, and to a lesser degree deciduous forests. Particular care should be given to developments within these habitats and if disturbed, mitigation methods should be used to minimize their impacts. Where possible long-term monitoring of sites with populations of rare species should be considered (~5-year return frequency), particularly those in proximity to development. Map predictions of rare plant habitat should be considered within regional assessments, such as its use as an indicator in the Biodiversity Management Framework (which it currently is), for environmental impact assessments over large areas, such as in situ oil sands leases, and in regional conservation planning.
CHAPTER 3.0: Using airborne laser scanning to predict plant species richness and assess conservation threats in the oil sands region of Alberta’s boreal forest

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3.1 Summary

Timely and cost-effective monitoring of biodiversity across large areas is a major challenge, yet an important component of monitoring programs that inform policy and conservation strategies. Recent advances in Airborne Laser Scanning (ALS) provide new opportunities to simultaneously measure vegetation structure and terrain morphology at fine spatial scales. However, there is limited research on whether ALS metrics correlate with biodiversity measures. We used vascular plant data from 283 quarter-hectare (50 m × 50 m) plots from the boreal forest in northeast Alberta, Canada to evaluate the potential for ALS-derived metrics to explain species richness patterns for vascular plants, as well as for four growth forms: herbaceous (including forbs and graminoids) and woody plants. We found canopy height from ALS was the most consistent and important factor positively related to local patterns in vascular plant richness. Multivariate regression models of ALS-derived metrics explained 20% to 35% of the variation in species richness among vascular plant and the four subclasses. When considering the current distribution of in situ oil sands leases in the region, vascular plant richness inside of the leases are higher than those outside of the leases. Areas delineated for woodland caribou conservation had lower average plant richness suggesting that caribou conservation will do little to protect hotspots of plant diversity in Alberta’s boreal forest. Our results highlight the value of using fine-scale measures of ALS-derived vegetation structure to explain, predict, and potentially monitor local plant diversity for a high latitude forested ecosystem.

3.2 Introduction

Given recent and projected trends in climate change and human disturbance, biodiversity threats continue to be a major conservation concern (Sala et al. 2000; Thuiller 2007). Essential to understanding trends in biodiversity and subsequently prioritizing conservation efforts is the need to better understand environment-biodiversity relationships and to derive efficient methods for monitoring biodiversity change (Araújo & Rahbek 2006; Kreft & Jetz 2007). Taxonomic richness of species (alpha diversity), most often obtained directly from field surveys, is the most typical measure of biodiversity (Thuiller 2007). However, it is not practical to monitor biodiversity in this way across large regions. A major challenge in managing biodiversity conservation is therefore to link biodiversity measures at local scales to cost-effective monitoring across large areas (Mairota et al. 2015). Appropriate surrogates for direct assessments of
biodiversity are therefore needed. Advances in remote sensing technology have created opportunities for monitoring habitat and vegetation structure at local to global scales, leading to potentially better, more economical, and faster alternatives to field surveys (Pimm et al. 2015).

Habitat loss and climate change are currently considered the two most critical factors threatening biodiversity (Brooks et al. 2002, 2006; Thomas et al. 2004); both can be measured using remote sensing (Turner et al. 2003). Although some studies suggest that multispectral passive optical sensors can be used to predict biodiversity at large scales (Coops et al. 2008; John et al. 2008; Zhang et al. 2016), most passive spectrum-derived satellite indices do not consider vertical structure of vegetation, a key driver of biodiversity at local scales (MacArthur & MacArthur 1961; Bergen et al. 2009). Moreover, new insights and methods are needed to recognize that vegetation structure and species composition differ even in two adjoining sites sharing a consistent regional species pool (Cook et al. 2002). Indeed, the physical structure of vegetation has long been noted by scientists as key to explaining variation in species diversity, particularly for animals, in part because it relates to possible mechanisms of ecological complexity and niche partitioning (MacArthur & MacArthur 1961; Kalko & Handley 2001). However, metrics related to vertical distribution and stratification of vegetation have historically been feasible only through collection of field data (Baker & Wilson 2000), thus limiting their application to local case studies.

Recent advances in Airborne Laser Scanning (ALS) technology now provide opportunities for measuring and monitoring the structure and complexity of vegetation across larger areas. This includes measures of canopy cover, height class distribution of vegetation, and maximum canopy height (Bergen et al. 2009). These metrics of vegetation structure have been used to predict the richness of vertebrates, particularly for birds (e.g., Bradbury et al. 2005; Clawges et al. 2008; Coops et al. 2016), with little done to assess whether ALS metrics explain local patterns in plant diversity.

Plant diversity at local scales (i.e., community level) is known to be affected by many different factors, including biome-scale environmental conditions or regional-to-local factors of topography, environmental heterogeneity, vegetation type, and vegetation structure (e.g., Moser et al. 2005; Kreft & Jetz 2007; Fine 2015). Exploring the factors affecting plant diversity at the community level therefore requires understanding of both regional climate factors and local environmental variables, including those that can be measured by ALS. Here, we use 283 plant biodiversity plots from the boreal forest in northeast Alberta, Canada, an area undergoing rapid landscape change due to oil sands developments, to examine whether ALS vegetation and terrain measurements, in combination with other environmental variables, relate to patterns of plant species richness. Establishing this relationship will enable landscape-scale predictions of conservation values. The boreal forest is the largest terrestrial biome on the earth, playing a major role in global biodiversity conservation and ecosystem function (Melillo et al. 1993; Näsholm et al. 1998). The biome is, however, sensitive to global climate change and human disturbance (Sala et al. 2000; Larsson & Danell 2001). Understanding biodiversity patterns (e.g. biodiversity hotspots) in the boreal forest, as well as their relationships with local to regional factors, is one key step for managing biological conservation and monitoring change due to exogenous (e.g. climate change) and endogenous (e.g. habitat fragmentation) threats. This includes a better understanding the implications of exploration and extraction of oil in Alberta’s
oil sands (Rooney et al. 2012), the world’s largest oil reserve (Sherrington 2005), on plant biodiversity hotspots. And to explore whether the locations of plant biodiversity hotspots overlap with those of other major conservation objectives, in particular areas of woodland caribou (Rangifer tarandus) habitat, which represent the current focus of conservation initiatives in Canada’s boreal forest (Schneider et al. 2010).

3.3 Methods

Study area
The study area was located in the boreal forest of northeast Alberta, Canada ranging in latitude from 55.3° N to 57° N (Figure 3.1). This area is part of the Boreal Forest Natural Region, which includes the lower portion of the Athabasca River and Lake Athabasca (Natural Regions Committee 2006). Elevations in the area range from 231 m to 863 m a.s.l., with annual precipitation and mean annual temperatures ranging from 430 mm to 492 mm and from -1.2 °C to 0.3 °C, respectively. On the uplands, soils are typically Brunisols, while wetland areas are Mesisols, Organics, Gleysols, and Grey Luvisols. Forests in the area are comprised of a mosaic of deciduous, mixed wood, and coniferous stands, with upland stands dominated by Populus, Picea, and Pinus spp., while lowland areas are represented by fens, swamps, and bogs (Natural Regions Committee 2006; Zhang et al. 2014).

Plot data
Plot data were collected under the Terrestrial Vascular Plant Monitoring Project for the Lower Athabasca, formerly known as the Ecological Monitoring Committee for the Lower Athabasca (EMCLA) Rare Plants Project. Field surveys occurred in the summers of 2012 to 2015 with a plot size of 50 m × 50 m (0.25 ha). Vascular plants were identified to species in each plot and recorded as presence/absence data. Unknown specimens were collected and identified later in the lab to species. See Chapter 1 for detailed field methods. In total, 602 plots were completed, but only 283 plots overlapped with ALS data on both vegetation structure and topography-derived variables and thus were used in this study. Since the underlying drivers and assembly mechanisms of plant diversity may differ across growth forms (Mao et al. 2013), all plants were classified into five growth forms (subdivisions) based on records from Floras (http://www.efloras.org/). These subdivisions included (1) all vascular plants, (2) herbaceous plants (further separated to (3) forbs and (4) graminoids), and (5) woody plants. Only native species were considered in this paper. Non-native species were infrequently encountered and included only sparse cover of a small number of species (e.g. Taraxacum officinale in upland sites). It should be noted, however, that plots were not directly on human disturbances, such as vegetated well sites, pipelines, or clearcuts, but were in the region of where general forest disturbances create conditions of habitat fragmentation. We are not therefore testing here the direct effect of footprints from industrial practices on plant richness.
Airborne laser scanning metrics and environmental variables

Airborne laser scanning data were generated from aerial surveys conducted between 2005 and 2013. Point densities averaged 1.9 returns/m$^2$ with the data processed using the “area-based” technique (Reutebuch et al. 2005; Wulder et al. 2008). Specifically, ALS point clouds were processed with FUSION software (McGaughey 2016) to derive vegetation height and canopy metrics (Coops et al. 2016). Most generally, ALS data can be divided into three different forest vegetation attributes that relate to the horizontal and vertical vegetation structure: (1) canopy height at different percentiles; (2) percent of returns above a specified height of the ground to indicate vegetation cover at that height stratum; and (3) return proportion at specified height intervals or variability of return heights to indicate vertical structure (McGaughey 2016; Coops et al. 2016). Since many of these metrics are highly correlated with each other, we selected a suite of variables that we considered to have greater ecological meaning to biodiversity. Previous research has demonstrated that metrics based on first returns are more stable than those based on all returns (Goodwin et al. 2006; Næsset 2009; Bater et al. 2011). We considered the following nine LiDAR-derived variables: the 95th percentile of observed first return heights above ground to represent canopy height, percentage of first returns above 1.37 m (i.e. breast height) to represent percent canopy, percentage of first returns above mean height, proportion of first returns for the height strata of: below 0.15 m and between 0.15 m to 1.37 m, 1.37 m to 5 m, 5 m to 10 m, 10 m to 20 m, and 20 m to 30 m (Table 3.1). We used LiDAR-derived canopy height at the 95th percentile of observed heights to measure the maximum height of vegetation at a site rather than using maximum height measured by LiDAR. This reduced sampling bias from extreme
conditions (e.g., birds in flight, communication towers, etc.) or possible errors from LiDAR returns (Kane et al. 2010; Bolton et al. 2013).

Water availability at a site, soil moisture, and local disturbances caused by flood erosion are considered to be important factors shaping local biodiversity (Nilsson et al. 1999; Sala et al. 2000; Xiong et al. 2003). To indicate the effects of water availability on plant richness at each plot, depth to water (DTW) was estimated for the same study region using ‘Wet Areas Mapping’ (WAM) data derived from ALS point clouds (http://watershed.for.unb.ca). Depth to water is an index that indicates the vertical distance (elevation) to available water, thus indicating drier to wetter conditions of the soil (Murphy et al. 2007; White et al. 2012; Oltean et al. 2016). Terrain variability within a site is also a factor influencing local patterns in plant richness (Webb et al. 1999). We used terrain slope within plots to represent the effects of topography, including its effects on promoting environmental heterogeneity within a site. Terrain slope was calculated using ALS-derived digital terrain model (DTM).

Mean annual precipitation (MAP) and mean annual temperature (MAT) were used to account for the effect of broad-scale environmental variability in the size of the local species pool (Gaston 2000; Kreft & Jetz 2007). Mean annual precipitation and MAT were extracted for each plot using climate normals from Climate-AB data (http://tinyurl.com/ClimateAB). We note that because of the regional geographic extent of the study (the distance between the furthest plots is ca. 150 km) and the lack of mountainous terrain in the area, climate variables did not substantially vary across the region, but they did indicate general temperature and moisture gradients from colder and wetter to warmer and drier conditions.

**Relationships between plant biodiversity hotspots, oil sands, and woodland caribou**

Here we assess the threats to plant biodiversity hotspots from oil sands developments, as well as the effectiveness of woodland caribou (*Rangifer tarandus*) conservation in protecting these hotspots. To do this we compared plant species richness for sample plot locations inside active oil sands leases versus areas outside of active leases. Sites within lease boundaries represent natural forest conditions surrounding in-situ developments and are potentially impacted by edge or indirect effects, not by the footprint itself. Second, we compared sample plot locations of plant richness for areas inside versus outside of woodland caribou range, given that woodland caribou represent the main focus of conservation and restoration in Canada’s boreal forest (Schneider et al. 2010), but little is known about the effectiveness of caribou in conserving other taxa. To map locations of caribou habitat, we used caribou range maps from Alberta Environment and Parks (http://aep.alberta.ca). These ranges are utilized for recovery monitoring and conservation initiatives. Oil sands lease boundaries were current to 2013 and based on data from Alberta Environment and Parks (http://osip.alberta.ca). Surface mine leases were removed from comparisons and predictive maps of plant richness to account for the amount and severity of disturbance within mines. Active oil sands surface mining is devoid of vegetation and we considered these mining leases to have no conservation value in the near-term. Instead, we focus on in situ oil sands developments were bitumen is extracted from sub-surface wells and represent overall a larger combined footprint than that of the more well-known oil sands surface mines.
Statistical analysis

Natural logarithm, log_{10}, or square-root transformations were used to normalize ALS and environmental variables exhibiting highly skewed distributions, while a natural logarithm transformation was used to normalize species richness values. All statistical analyses were performed in R program (R Core Team 2015). Simultaneous Autoregressive (SAR) models with a spatial error model were used to account for spatial autocorrelation in plots (Kissling & Carl 2008). Richness of total vascular plants, herbaceous plants, forbs, graminoids, and woody plants were then regressed against ALS metrics and environmental variables using SAR models. SAR models were estimated using the package ‘spdep’ (Bivand et al. 2013; Bivand & Piras 2015). We first used spatial autoregressive one-predictor regression to examine the effects of individual factors (Table 3.1). We then used multivariate regression models of plant richness against different combinations of those variables to assess overall relationships. Pearson correlations among all ALS metrics and environmental variables were first examined to avoid multicollinearity in multivariate regression models. Where variable pairs had correlation coefficients |r|>0.7, the one with more ecological relevance and higher explanatory power for single factor regression models was kept (Dormann et al. 2013). After considering all variable correlations, the following uncorrelated variables were considered in models: mean annual precipitation (MAP), mean annual temperature (MAT), the 95th percentile of canopy height (CH), the proportion of first returns below 0.15 m (P0-0.15), proportion of first returns between 0.15 m to 1.37 m (P0.15-1.37), proportion of first returns between 1.37 m to 5 m (P1.37-5), proportion of first returns between 5 m to 10 m (P5-10), depth to water (DTW), and slope of plots (Slope). Since responses of biodiversity to environmental variables are not always linear (Gaston 2000), we assessed quadratic effects for all ALS metrics by comparing linear and quadratic effects of all investigated ALS metrics using Akaike Information Criterion (AIC) (Table 3.1 and Table 3.2). It has been proposed that if |ΔAIC| between two models is smaller than 2, both models could be considered as having similar support (Burnham & Anderson 1998; Mazerolle 2004). Thus, if the AIC of a regression model with a linear response was more than 2 AIC points larger than a quadratic response, the model with a quadratic function was used (Table 3.1 and Table 3.2). As a result, the quadratic of P5-10 was considered for explaining richness of vascular, herbaceous, woody, and forb plants, while P0.15-1.37 and P1.37-5 were considered for graminoids (Table 3.2; Figure 3.4). Since different combinations of canopy height and depth-to-water could potentially indicate types of forest habitats in this area, we also assessed the interactive effect canopy height and depth to water (i.e. CH×DTW). Finally, AIC was used to rank support among models with competing variable combinations. The model with the lowest AIC was considered the most parsimonious model and the results from this model were reported (Table 3.3). Akaike weights (w) were estimated for each variable based on the full set of models to compare relative importance of each variable (Burnham & Anderson 2002).

To assess the effects of locations of in situ oil sands leases and the effectiveness of woodland caribou conservation as an umbrella for areas of high plant biodiversity, we overlaid predicted plant species richness with oil sand leases (Figure 3.6) and woodland caribou ranges (Figure 3.7). We also directly compared plant richness from field data among treatment categories (leased vs. non-leased; caribou vs. no caribou) using t-tests of log_{10} transformed species richness (Figure 3.5).
3.4 Results

Results of univariate responses in plant richness

Total species richness of vascular plants was significantly positively associated with mean annual temperature (MAT), canopy height (CH), the percentages of LiDAR returns above 1.37 m (PR1.37), mean canopy height (PRmean), the slope of plots (Slope), and the return proportion within 10 m to 20 m (P10-20) and within 20 m to 30 m (P20-30) (Table 3.1; Figure 3.3). In contrast, mean annual precipitation (MAP) and the return proportion below 0.15 m were significantly negatively associated with total species richness (Table 3.1). Richness of vascular plants was not significantly related to depth to water (DTW), the return proportion within 0.15 m to 1.37 m (P0.15-1.37), within 1.37 m to 5 m (P1.37-5), and within 5 m to 10 m (P5-10) (Table 3.1).

Table 3.1. Simultaneous Autoregressive (SAR) univariate models of species richness for total vascular plants, herbaceous plants, woody plants, forbs, and graminoids against each explanatory variable*. The pseudo-r squares ($r^2$) of SAR models are given in the table, and symbols in brackets represent the trends of relationships between species richness and variables. The two most supported models (lowest AIC) for each growth form are shown in bold.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Vascular</th>
<th>*r^2</th>
<th>AIC</th>
<th>P* value</th>
<th>Herbs</th>
<th>*r^2</th>
<th>AIC</th>
<th>P* value</th>
<th>Woody</th>
<th>*r^2</th>
<th>AIC</th>
<th>P* value</th>
<th>Forbs</th>
<th>*r^2</th>
<th>AIC</th>
<th>P* value</th>
<th>Graminoids</th>
<th>*r^2</th>
<th>AIC</th>
<th>P* value</th>
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</thead>
<tbody>
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<td>MAP</td>
<td>0.155(-)</td>
<td>387.0</td>
<td>0.000</td>
<td>0.122(-)</td>
<td>539.0</td>
<td>0.000</td>
<td>0.098(-)</td>
<td>221.1</td>
<td>0.000</td>
<td>0.192(-)</td>
<td>568.3</td>
<td>0.000</td>
<td>0.025(-)</td>
<td>597.2</td>
<td>0.034</td>
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<td></td>
<td></td>
<td></td>
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</tr>
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<td>0.054(+)</td>
<td>418.9</td>
<td>0.000</td>
<td>0.063(+)</td>
<td>557.3</td>
<td>0.000</td>
<td>0.029(+)</td>
<td>241.8</td>
<td>0.000</td>
<td>0.083(+)</td>
<td>604.1</td>
<td>0.000</td>
<td>0.021(+)</td>
<td>598.6</td>
<td>0.099</td>
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<td></td>
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<td></td>
</tr>
<tr>
<td>CH</td>
<td>0.100(+)</td>
<td>404.9</td>
<td>0.000</td>
<td>0.069(+)</td>
<td>555.5</td>
<td>0.000</td>
<td>0.111(+)</td>
<td>216.8</td>
<td>0.000</td>
<td>0.149(+)</td>
<td>583.1</td>
<td>0.000</td>
<td>0.054(-)</td>
<td>588.6</td>
<td>0.000</td>
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<td>PR1.37</td>
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<td>420.4</td>
<td>0.001</td>
<td>0.028(+)</td>
<td>567.7</td>
<td>0.029</td>
<td>0.058(+)</td>
<td>233.3</td>
<td>0.000</td>
<td>0.090(+)</td>
<td>602.1</td>
<td>0.000</td>
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<td>0.000</td>
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<td>PRmean</td>
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<td>0.002</td>
<td>0.025(+)</td>
<td>568.6</td>
<td>0.051</td>
<td>0.053(+)</td>
<td>234.6</td>
<td>0.000</td>
<td>0.084(+)</td>
<td>603.9</td>
<td>0.000</td>
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<td>DTW</td>
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<td>0.012(+)</td>
<td>572.3</td>
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<td>0.033(+)</td>
<td>240.5</td>
<td>0.005</td>
<td>0.030(+)</td>
<td>620.2</td>
<td>0.011</td>
<td>0.112(+)</td>
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<td>0.005</td>
<td>0.021(+)</td>
<td>569.7</td>
<td>0.093</td>
<td>0.060(+)</td>
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<td>0.000</td>
<td>0.036(+)</td>
<td>618.3</td>
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<td>0.047(-)</td>
<td>562.2</td>
<td>0.001</td>
<td>0.072(-)</td>
<td>228.9</td>
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<td>0.135(-)</td>
<td>587.6</td>
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<td>0.009(-)</td>
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<td>571.9</td>
<td>0.477</td>
<td>0.007(+)</td>
<td>248.1</td>
<td>0.749</td>
<td>0.008(-)</td>
<td>626.4</td>
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<td>0.013(+)</td>
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<td>0.028(+)</td>
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<td>0.034(-)</td>
<td>594.7</td>
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</tr>
<tr>
<td>P10-20</td>
<td>0.068(+)</td>
<td>414.7</td>
<td>0.000</td>
<td>0.043(+)</td>
<td>563.2</td>
<td>0.002</td>
<td>0.077(+)</td>
<td>227.4</td>
<td>0.000</td>
<td>0.109(+)</td>
<td>596.2</td>
<td>0.000</td>
<td>0.068(-)</td>
<td>584.4</td>
<td>0.000</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P20-30</td>
<td>0.056(+)</td>
<td>418.3</td>
<td>0.000</td>
<td>0.050(+)</td>
<td>561.2</td>
<td>0.001</td>
<td>0.042(+)</td>
<td>238.0</td>
<td>0.001</td>
<td>0.095(+)</td>
<td>600.5</td>
<td>0.000</td>
<td>0.046(-)</td>
<td>591.2</td>
<td>0.001</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*CH, canopy height; PR1.37 and PRmean, the percentages of returns above 1.37 m and mean height, respectively; P0-0.15, P0.15-1.37, P1.37-5, P5-10, P10-20, P20-30 report the return proportion at 0 to 0.15 m, 0.15 to 1.37 m, 1.37 m to 5 m, 5 m to 10 m, 10 m to 20 m and 20 m to 30 m, respectively; MAP, mean annual precipitation; MAT, mean annual temperature; Slope, the terrain slope of the plot; DTW, the depth to water at the plot.

Of the variables assessed, MAP and CH were the two strongest predictors of species richness having the lowest AICs and individually explaining 15.5% and 10.0% (*pseudo* $r^2$) of the variation in vascular plant richness, respectively. MAP and CH were also the most supported predictors in single-regression models for herbaceous, forb, and woody plant richness, but not for richness of graminoids, which was better explained by DTW and the return proportion within 0.15 m to 1.37 m (P0.15-P1.37) (Table 3.1). Mean annual precipitation (MAP) was consistently negatively associated with richness of herbaceous, forb, graminoid, and woody plants, explaining 12.2%, 19.2%, 2.5%, and 9.8% of the variation, respectively. Canopy height was negatively
associated with richness of graminoids, but only explaining 5.4% of the variation (Table 3.1; Figure 3.3). Unlike total vascular, herbaceous, woody, and forb plant richness, plant richness of graminoids was negatively associated with depth to water explaining 11.2% of the variation (Table 3.1; Figure 3.3). Comparing results of linear and quadratic responses, quadratic relationships for P5-10 significantly explained richness of vascular, herbaceous, woody, and forb plants (Table 3.1; Table 3.2; Figure 3.4). In contrast, for graminoids, the quadratic terms were significant for P0.15-P1.37 and P1.37-5. The interactive effect of canopy height and depth to water (CH x DTW) explained 6.5% of the variation in richness of graminoids.

Table 3.2. Simultaneous Autoregressive (SAR) models for quadratic regression for plant richness against selected ALS metrics and interactive effects of canopy height and water to depth (CH x DTW). The pseudo-\( r^2 \) square of SAR models are given in the table. ∆AIC is the difference in AIC value between these models and the corresponding linear SAR model (Table 3.1). If the AIC value of the linear SAR model was more than 2 points greater than the quadratic SAR model, the ∆AIC is shown in bold.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Vascular</th>
<th>Herbaceous</th>
<th>Woody</th>
<th>Forbs</th>
<th>Graminoids</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>( r^2 )</td>
<td>AIC</td>
<td>∆AIC</td>
<td>( r^2 )</td>
<td>AIC</td>
</tr>
<tr>
<td>CH</td>
<td>0.100</td>
<td>406.9</td>
<td>2.02</td>
<td>0.071</td>
<td>556.8</td>
</tr>
<tr>
<td>DTW</td>
<td>0.019</td>
<td>431.3</td>
<td>1.90</td>
<td>0.012</td>
<td>574.3</td>
</tr>
<tr>
<td>Slope</td>
<td>0.040</td>
<td>425.1</td>
<td>0.76</td>
<td>0.025</td>
<td>570.6</td>
</tr>
<tr>
<td>P0.15</td>
<td>0.074</td>
<td>414.9</td>
<td>1.82</td>
<td>0.047</td>
<td>564.0</td>
</tr>
<tr>
<td>P0.15-1.37</td>
<td>0.011</td>
<td>433.6</td>
<td>2.00</td>
<td>0.013</td>
<td>574.1</td>
</tr>
<tr>
<td>P1.37-5</td>
<td>0.011</td>
<td>433.5</td>
<td>2.01</td>
<td>0.014</td>
<td>573.7</td>
</tr>
<tr>
<td>P5-10</td>
<td>0.061</td>
<td>419.0</td>
<td>9.05</td>
<td>0.050</td>
<td>563.2</td>
</tr>
<tr>
<td>CH x DTW</td>
<td>0.015</td>
<td>430.4</td>
<td>-</td>
<td>0.020</td>
<td>570.1</td>
</tr>
</tbody>
</table>

*CH: canopy height; P0-0.15, P 0.15-1.37, P1.37-5, P5-10 represent the return proportion at 0 to 0.15 m, 0.15 to 1.37 m, 1.37 m to 5 m, 5 m to 10 m, respectively; Slope, the slope of the plot; DTW, the depth to water at the plot.

Results of multivariate simultaneous autoregressive models of plant richness
Simultaneous Autoregressive (SAR) multivariate models explained 19.5% to 35.0% (pseudo-\( r^2 \)) of species richness across the five growth forms of plants based on combinations of ALS vegetation metrics and other environmental variables (Table 3.3). In multivariate regression models, canopy height (CH) remained the most consistent and important variable overall explaining species richness for all growth forms except graminoids. Non-linear quadratic responses of P5-10 were also important for forbs, woody species, and total vascular plants. Depth to water was negatively associated with richness of total vascular, herbaceous, graminoid, and forb plants. Consistent with single-predictor regression models, MAP was an important predictor of plant richness and was included in the most supported models (lowest AIC values) for all growth forms, with MAP being inversely related to species richness. Predictions from multivariate SAR models suggested that plant richness was highest in the major river valleys for total vascular plants and herbaceous, woody, and forb species (Figure 3.2 a, b, c and e), while species richness of graminoids was highest in the flattest parts of the study area, representing fens (Figure 3.2 d).
Figure 3.2. Predicted species richness for northeast Alberta, Canada based on Simultaneous Autoregressive (SAR) models for richness of vascular (a), herbaceous (b), forb (c), graminoid (d), and woody plants (e). Results indicated that the areas associated with the highest plant richness were in or around river valleys, except for graminoids which peaked in richness in the flattest areas typified as being fens and bogs. Note, oil sands surface mine leases in the far north of the map were removed (shown in the maps in white).

Table 3.3. Simultaneous Autoregressive (SAR) multivariate models of richness of vascular plants, herbaceous plants, woody plants, forbs, and graminoids against combinations of explanatory variables*. The combinations with the lowest AIC (Akaike Information Criterion) were considered the most parsimonious models with the z-value for each coefficient given in the table. Pseudo $r^2$ of the most supported model for all five growth forms are reported. The Akaike weight ($w$) is based on a full model (combination of thirteen variables) and used to indicate the
relative importance of individual variables. The two highest values of \( w \) for each growth form are in bold.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Vascular</th>
<th>Herbaceous</th>
<th>Woody</th>
<th>Forbs</th>
<th>Graminoids</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>z-value</td>
<td>w</td>
<td>z-value</td>
<td>w</td>
<td>z-value</td>
</tr>
<tr>
<td>MAP</td>
<td>-6.44</td>
<td><strong>0.996</strong></td>
<td>-3.97</td>
<td><strong>0.982</strong></td>
<td>-8.61</td>
</tr>
<tr>
<td>P0.15-1.37</td>
<td>3.12</td>
<td><strong>0.899</strong></td>
<td>3.21</td>
<td><strong>0.960</strong></td>
<td>2.67</td>
</tr>
<tr>
<td>CH</td>
<td>3.15</td>
<td>0.859</td>
<td>3.40</td>
<td>0.951</td>
<td>2.90</td>
</tr>
<tr>
<td>DTW</td>
<td>-3.04</td>
<td>0.688</td>
<td>-3.49</td>
<td>0.931</td>
<td>0.469</td>
</tr>
<tr>
<td>Slope</td>
<td>0.344</td>
<td>0.291</td>
<td>2.10</td>
<td>0.794</td>
<td>0.279</td>
</tr>
<tr>
<td>P0.15</td>
<td>-2.42</td>
<td>0.833</td>
<td>-3.06</td>
<td>0.900</td>
<td>0.433</td>
</tr>
<tr>
<td>MAT</td>
<td>2.55</td>
<td>0.617</td>
<td>2.11</td>
<td>0.578</td>
<td>0.357</td>
</tr>
<tr>
<td>P1.37-5</td>
<td>-2.78</td>
<td>0.819</td>
<td>-2.76</td>
<td>0.904</td>
<td>-1.61</td>
</tr>
<tr>
<td>P5-10</td>
<td>1.65</td>
<td>0.386</td>
<td>0.353</td>
<td>2.05</td>
<td>0.418</td>
</tr>
<tr>
<td>P0.15-1.37(^2)</td>
<td>-4.94</td>
<td><strong>0.919</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P1.37-5(^2)</td>
<td>2.42</td>
<td>0.601</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P5-10(^2)</td>
<td>0.379</td>
<td>0.364</td>
<td>0.384</td>
<td>0.358</td>
<td></td>
</tr>
<tr>
<td>CH × DTW</td>
<td>0.479</td>
<td>0.390</td>
<td>-2.64</td>
<td>0.791</td>
<td>0.419</td>
</tr>
</tbody>
</table>

**Model performance**

<table>
<thead>
<tr>
<th></th>
<th>pseudo ( r^2 )</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.240</td>
</tr>
<tr>
<td></td>
<td>0.223</td>
</tr>
<tr>
<td></td>
<td>0.222</td>
</tr>
<tr>
<td></td>
<td>0.350</td>
</tr>
<tr>
<td></td>
<td>0.195</td>
</tr>
</tbody>
</table>

*CH, canopy height; P0-0.15, P0.15-1.37, P1.37-5 and P5-10 represent the return proportion at 0 to 0.15 m, 0.15 to 1.37 m, 1.37 m to 5 m and 5 m to 10 m, respectively; MAP, mean annual precipitation; MAT, mean annual temperature; Slope, the slope of the plot; DTW, the depth to water; CH × DTW, the interactive effects of CH and DTW.*
Figure 3.3. Scatter plots for richness of total vascular, woody, herbaceous, forb, and graminoid plants and canopy height, depth to water, slope, and mean annual precipitation. Richness, depth to water and slope were log-transformed. Lines are ordinary least squares linear regressions for relationships between those variables and plant richness for each of the five groups. VP, vascular plants; WP, woody plants; HP, herbaceous plants, and MAP, mean annual temperature. Units for canopy height, depth to water, slope and mean annual temperature are meter, meter (log-scaled), degree (log-scaled) and mm, respectively.
Figure 3.4. Non-linear relationships (i.e. quadratic) for richness of total vascular (a), herbaceous (b), forb (c) and woody (d) plants, and return proportion at 5 to 10 m (P5-10) and for richness of graminoids and return proportion at 0.15 to 1.37 m (P0.15-1.37, e) and 1.37 m to 5 m (P1.37-5, f). Richness data, P0.15-1.37 and P1.37-5 were log-transformed. Lines are quadratic regressions for relationships between those variables and plant richness. VP, vascular plants; WP, woody plants; and HP, herbaceous plants.

Relationships between plant biodiversity hotspots, oil sands, and woodland caribou
The t-tests demonstrated that log_{10}-scaled richness of vascular, woody, and forb plants were significantly different between plots inside and outside of caribou ranges ($p<0.05$) with caribou ranges having lower plant richness (Figure 3.5). Significant difference in plant richness were also found for plots inside versus outside of oil sands leases, except for richness of woody plants, with richness slightly higher inside leases than outside of leases (Figure 3.5). These results also reflected regional patterns based on predicted maps of plant richness, although substantial variation was evident among oil sands leases (Figure 3.6). As observed in predictive maps of multivariate SAR models, most caribou ranges occurred in areas with relatively low plant diversity. Thus, conservation of caribou ranges will not act as a surrogate to conserve those areas of highest vascular plant species richness (i.e. richness hotspots) (Figure 3.7).
Figure 3.5. Bar graphs for species richness of vascular plants, herbaceous plants, forbs, graminoids, and woody plants for field plots inside vs. outside of caribou ranges (A) and leased vs. non-leased in situ oil sands (B). Richness data were log$_{10}$-transformed. Different letters indicated that groups are significantly different ($p < 0.05$) from one another based on a t-test.
Figure 3.6. Location of in situ oil sands leases and predicted species richness for northeast Alberta, Canada according to: vascular (a), herbaceous (b), forb (c), graminoid (d), and woody (e) plant groups. Blue polygon lines represent boundaries of current in situ oil sands leases.
3.5 Discussion

Combining the effects of habitat-terrain characteristics (e.g. slope, depth to water), including climatic conditions, and the horizontal and vertical structure of vegetation, we examined patterns in local measures of plant richness in the boreal forest of northeast Alberta, Canada using ALS-derived vegetation structure measures, ALS-derived terrain measures, and climate. Our results demonstrate that climate, vertical structure of vegetation, and terrain-derived slope and depth to water explained (ca. 20% to 35%) local patterns in native plant species richness. However, the effects of local variables on plant diversity differed across plant growth forms (Table 3.1; Table
3.3; Figure 3.3). Overall, canopy height and mean annual precipitation (MAP) were the best predictors of vascular, herbaceous, forb, and woody plant richness (Table 3.1; Table 3.3).

Many hypotheses have been suggested to explain spatial patterns in plant richness at global and regional scales (Auerbach & Shmida 1987; Kreft & Jetz 2007; Fine 2015). Determinants of biodiversity may, however, change with spatial scale (Auerbach & Shmida 1987; Gaston 2000). Although our study plots were located within a relatively narrow geographic area of one ecosystem – the boreal forest (Figure 3.1), factors affecting large scale processes would be expected to influence regional species richness and thus richness of the plant community (Eriksson 1993). To examine this question further, we explored the effects of mean annual precipitation (MAP) and mean annual temperature (MAT) on the spatial variation in plant species richness of all vascular plants and the four growth forms of herbaceous, woody, forb, and graminoids. Our results demonstrated that, at a community level, these two variables significantly explained local variation in plant diversity (Table 3.1). For multivariate regression models, precipitation was consistently selected in models predicting richness of vascular plants and the four growth forms. MAP and MAT are considered two key factors in the water-energy hypothesis of global biodiversity patterns (Hawkins et al. 2003; Kreft & Jetz 2007). However, in our research, precipitation was negatively related to plant richness (Table 3.1), while temperature was positively related to plant richness across all growth forms. This supports previous studies suggesting that plant diversity in colder regions is primary limited by energy inputs where water availability is not a key limitation (Hawkins et al. 2003).

We measured local environmental conditions using ALS-derived measures of vegetation structure (height and cover) and terrain characteristics to assess their influence in explaining local patterns in plant diversity. Remote sensing data are usually linked to measures of productivity and canopy cover which are known to be related to species assembly and richness (Gillman & Wright 2006; John et al. 2008). We found that ALS metrics describing vegetation density (e.g. the percentage of returns above 1.37 m), and especially describing vegetation height, were useful predictors of plant diversity at local (community) scales (Table 3.1; Table 3.3). ALS-derived canopy cover has been previously demonstrated as a useful descriptor of vegetation structure (Coops et al. 2007; Smart et al. 2012). Our results further demonstrated that the effects of canopy on plant diversity differed among plant growth forms. The two related variables, i.e. the percentage of returns above 1.37 m (PR 1.37) and percentage of returns above mean height (PRmean), were positively associated with species richness of total vascular, forb, herbaceous, and woody plants, but negatively associated with species richness of graminoids. As would be expected, more open habitats had a higher capacity to maintain graminoid-rich communities, while older and more productive forests were more suitable for maintaining total vascular species and herbaceous, forb, and woody plants.

Overall, canopy height was one of most important factors associated with patterns of plant diversity (positively related) in both univariate and multivariate analyses (Table 3.1; Table 3.3). Canopy height may be a surrogate for structural complexity of vegetation (McElhinny et al. 2005) illustrating a positive association between structural complexity and plant diversity in the boreal forest. More complex structure is well accepted as one of the primary drivers of biodiversity (Wolf et al. 2012; St. Pierre & Kovalenko 2014; Loke & Todd 2016). However, most previous studies using LiDAR-derived vegetation structure relate to birds (e.g., Goetz et al.
with few studies focused on plant species richness where it is more difficult to argue for a direct increase in niche space. Light is a basic resource that limits plant growth (Craine & Dybzinski 2013) and plant communities with taller plant heights potentially provide more possible options for plants in competition for light (Falster & Westoby 2003). Our results showed ALS-derived canopy height was positively associated with species richness across plots for all growth forms, excluding graminoids, where canopy height was negatively associated with species richness (Table 3.1). These results may be due in part to the physiological adaption of plants to different types of habitats with species in the graminoid group represented by the families of Cyperaceae, Poaceae and Juncaceae, which are common to grasslands and peatlands (Edwards et al. 2010). In the case of the boreal forest of Alberta, open habitats are most often peatlands, including graminoid-dominated fens (Rooney et al. 2012). Other vertical measures of vegetation structure were associated with plant richness (Table 3.1). For example, species richness of graminoids was positively associated with the proportion of first returns below 15 cm (low ground layer; P0-0.15). In contrast, the relationship between richness of vascular, forb, and woody plants was negatively related to this same ground layer stratum (P0-0.15). These metrics depict characteristics of vertical stratification directly (Coops et al. 2007; Smart et al. 2012), which are related to the complexity of the canopy within the community. The physical structure of vegetation has been proposed as a key factor limiting diversity of ecosystems, particularly for birds that are dependent on forest structure (MacArthur & MacArthur 1961). Our results support the theory that vertical structure of vegetation is positively associated with plant diversity, not just birds (MacArthur & MacArthur 1961; Su & Bork 2007; Bergen et al. 2009).

Our study also demonstrated that hydrological conditions (represented by depth to water) and terrain slope in the boreal forest were associated with local plant diversity (Table 3.1; Table 3.3), which is supported by other studies (Webb et al. 1999; Sass et al. 2012). In our study, areas with steeper slopes had higher plant diversity for all growth forms except graminoids. Incised valleys and steeper terrain may therefore be potential hotspots for plant diversity in boreal forests. This supports hypotheses of species diversity-environmental heterogeneity where greater terrain variation results in more microsites and thus niches. Depth to water was negatively correlated with the richness of graminoid species (Table 3.1; Table 3.3), again indicating the specificity of many graminoid species to wet environments (i.e. fens).

Environmental and ALS measures used here represent only part of the factors associated with conditions affecting plant richness in the boreal forests. Plant diversity is also affected by other factors not measured in this study including land use, natural disturbances (e.g. fire), soil conditions, and species interactions (Perroni-Ventura et al. 2006; Kouba et al. 2015; Soliveres et al. 2015). Models that incorporate these variables may be more generalizable and have broader application to monitoring. Regardless, ALS-derived measures of vegetation structure show promise in directly measuring vegetation structure and thus indirectly monitoring plant biodiversity (e.g. Su & Bork 2007) across large (regional) scales. We suggest that measures of vegetation structure are more likely to relate to measures of plant diversity than data from multispectral passive optical sensors assessing horizontal features of sites (Krishnaswamy et al. 2009).
One of the most important goals for biodiversity monitoring is to conserve species from threats and set conservation priorities since biodiversity is unevenly distributed in space (Brooks et al. 2006; Freudenberger et al. 2013). In situ (non-mineable) oil sands leases were significantly different in plant biodiversity to non-leases for native habitats for all five groups except woody plants with plant richness typically being higher. However, when considering predictions across the region, many of oil sands leases are located in the areas with moderate to relatively lower vascular plant richness. In general, there was trend towards leases further north having higher total vascular plant richness (Figure 3.6). This suggests that the placement of oil sands leases within the landscape is not random with respect to the region’s plant biodiversity with some sites having greater conservation value and thus threats. In contrast to oil sands leases, vascular plant richness was lower inside woodland caribou ranges than outside of caribou ranges suggesting that the conservation of caribou, a threatened flagship species for Canada’s boreal forest (Weclaw & Hudson 2004; Festa-Bianchet et al. 2011; Moreau et al. 2012), will have little value for protecting hotspots of vascular plant biodiversity (Figure 3.7). Identifying this conservation gap is important for prioritizing future conservation efforts in Canada’s boreal forest that extend beyond a single, albeit charismatic, species.

**Implications for management**
Exploration of the utility of ALS-derived metrics is ongoing in the literature and research in this field is still developing, with datasets for the entirety of Alberta not yet complete. Based on findings from this investigation, we summarize the applicability of analyses using ALS-derived vegetation metrics for oil and gas related activities below. Determining which areas of a given lease may harbor the highest vascular plant diversity is important, and our findings suggest that these areas may not align with those considered of high value for other taxa (e.g. caribou). Our findings of a positive relationship between diversity and vertical vegetation structure and unique landforms (areas of topographic relief in the relatively flat boreal landscape) can inform lease-level summaries of expected diversity and assist in pre-survey planning stages for environmental assessments by highlighting target areas. Special attention should be paid toward structurally diverse and topographically variable areas. Local-scale spatial predictions of plant diversity may also prove effective for identifying where proposed developments (e.g. roads, well pads, processing plants) would have the least impact.
CHAPTER 4.0: Observer error in vascular plant surveys: evaluating pseudoturnover and the number of missed species

C. Denny¹, J. Dennett, and S.E. Nielsen¹

¹ Department of Renewable Resources, University of Alberta

4.1 Summary

Observer error related to imperfect detection of species is widely regarded as an important issue for vascular plant surveys. However, it is seldom formally estimated despite direct implications for assessing changes in biodiversity. Estimating the magnitude of error and understanding the factors affecting this will allow for more accurate conclusions to be drawn from survey data and facilitate improvements to sampling protocols. Inter-observer error can be estimated through the calculation of pseudoturnover and the number of species missed per plot, which are determined by comparing species lists between observer pairs. These two-error metrics were calculated for a subset of the Rarity and Diversity plots in the Lower Athabasca Region of Alberta (n = 67) and were related to site richness, ecosite type, and sampling effort, both for all species collectively and individual plant growth forms. Average pseudoturnover for all species was 15.4%, which is low compared to previous studies, and an average of 7.8 species were missed per plot. Observer error increased with species richness, and varied by growth form and ecosite type, but was not influenced by sampling effort. These findings indicate that the species richness of a given plot can inform estimates of the magnitude of observer error present, and suggest that plant functional groups should be assessed individually when examining the factors that affect observer error and how these can be addressed.

4.2 Introduction

Imperfect detection of species during vegetation sampling is a common source of error in monitoring programs (Nilsson & Nilsson 1985; Chen et al. 2013), and thus a challenge for inventorying and assessing changes in biodiversity. Observer errors can be classified as either those of omission (false-negatives), when a species present at a site is not detected, or commission (false-positives), when a species is detected but misidentified (Miller et al. 2011). Most often there is greater concern of omission errors, while commission is assumed to be less significant and more difficult to deal with than omission errors. Estimates of the magnitude of observer error and insight into the factors that potentially influence these errors, such as species richness, plant growth form, and ecosite type, enable more accurate inferences and improvements in sampling protocols (Morrison 2016). Although the issue of observer error is often acknowledged in the literature, it is seldom formally estimated despite its implications for research findings (Chen et al. 2013).

As observer detection accuracy is challenging to quantify because true species presence at a site is rarely known, observer error is instead typically evaluated in terms of precision by comparing the results of multiple observers (Morrison 2016). Inter-observer error involves differences in species detection among individual observers for a given survey site, which can arise due to both
omission and commission errors (Morrison 2016). Comparing species lists among observers allows for the calculation of pseudoturnover (Lynch & Johnson 1974) and the average number of species missed per site. Pseudoturnover refers to inter-observer error that suggests false changes in species assemblages and is based on unique species numbers and total site richness, with lower values indicating greater similarity in detection between observers (Nilsson and Nilsson 1985). A review of observer error in vegetation surveys found that mean pseudoturnover across studies was 10-30%, suggesting that most species lists produced by the sampling process are likely incomplete (Morrison 2016). The number of species missed represents only the unique species per observer for each site which were not detected by the other.

To assess observer error for the vascular plant sampling conducted at Rarity and Diversity plots established in the Lower Athabasca Region, a subset of sites surveyed by multiple observers was examined. Previous work by Zhang et al. (2014) evaluated observer error using an earlier version of this dataset, but additional sampling has since been conducted thus allowing for error to be estimated across a larger number of survey sites and observer pairs. Specifically, our objectives here were to (1) quantify percent pseudoturnover for all species collectively, as well as for individual growth forms (forbs, graminoids, shrubs, and trees), (2) determine the average number of species missed for all species and per growth form, and (3) evaluate relationships between these metrics and site richness, ecosite type, and sampling effort.

4.3 Methods

Repeat survey dataset
Plant species detection lists were calculated from 67 same-day repeat surveys that were randomly completed by 16 field technicians on a proportion of the 602 Rarity and Diversity plots sampled, with the goal of re-surveying at least 10% of all plots to assess observer accuracy (see Chapter 1 for sampling methodology). This subset included 63 plots that had been surveyed by two observers and four that had been surveyed by three observers, amounting to 75 pair-wise comparisons and thus 150 values for unique species per observer. In total, 11 ecosite categories were represented with only marsh (VD) unrepresented (Table 4.1).

Table 4.6. Number of Rarity and Diversity plots included in the repeat survey dataset (n = 67 plots) for each of the 11 ecosite categories.

<table>
<thead>
<tr>
<th>Ecosite</th>
<th>Number of Plots</th>
</tr>
</thead>
<tbody>
<tr>
<td>NT - Not Treed</td>
<td>1</td>
</tr>
<tr>
<td>PX – Poor Xeric (poor, dry forests)</td>
<td>3</td>
</tr>
<tr>
<td>PM – Poor Mesic (moist conifer)</td>
<td>9</td>
</tr>
<tr>
<td>PD – Poor Hydric (bog)</td>
<td>4</td>
</tr>
<tr>
<td>MX – Medium Xeric (dry mixedwood)</td>
<td>3</td>
</tr>
<tr>
<td>MM – Medium Mesic (mesic mixedwood)</td>
<td>20</td>
</tr>
<tr>
<td>MG – Medium Hygric (moist mixedwood)</td>
<td>7</td>
</tr>
<tr>
<td>MD – Medium Hydric (poor fen)</td>
<td>5</td>
</tr>
<tr>
<td>RG – Rich Hygric (rich, moist forests)</td>
<td>1</td>
</tr>
<tr>
<td>RD – Rich Hydric (rich fen)</td>
<td>11</td>
</tr>
</tbody>
</table>
Calculation of observer error metrics
Species lists were compared between observers for each repeat plot to determine total richness and number of unique species that had been detected per individual. Percent pseudoturnover (PT) was calculated following the approach of Nilsson & Nilsson (1985). For comparisons of species lists per plot, if observers’ A and B detect $S_{AA}$ and $S_{BB}$ unique species, respectively, and $S_A$ and $S_B$ species in total, pseudoturnover can be calculated as:

$$PT = \frac{S_{AA} + S_{BB}}{S_A + S_B} \times 100$$

The number of unique species per observer was averaged among plots, both for all species collectively and per growth form, to indicate the number of species missed by the other individual in the pair. Variation in sampling effort per observer pair was quantified as the difference in total plot survey time.

Model development
Linear regression models were estimated for pseudoturnover and number of species missed as a function of total species richness and species richness per growth form, as well as ecosite category. Log$_{10}$ transformations were used to normalize all data in pseudoturnover analyses, with a constant of one added to percent pseudoturnover beforehand to account for zero values. Analyses of the number of species missed had transformations for only some variables, based on model fit per growth form. The effect of sampling effort was assessed only within the pseudoturnover analysis that considered all species collectively. The ecosite reference category for comparisons was designated as MM (Viburnum edule/Shepherdia canadensis) where this variable was included in the models.

4.4 Results
Pseudoturnover
A total of 379 vascular plant species were detected in the subset of plots for which repeat surveys were conducted. Average survey time per plot was 90 minutes and ranged from 26 to 193 minutes. Average difference in survey time between observers per plot was 23 minutes and ranged from 0 to 109 minutes. Average percent pseudoturnover for all species collectively was 15.4%, with values ranging from 0% to 29.2% (Table 4.2). Growth forms differed in regards to mean pseudoturnover and the range of values observed. Graminoids had the highest mean pseudoturnover overall with an average of 20.4%, while shrubs had the lowest mean pseudoturnover at 12.9% (Table 4.2). Moderate variation in pseudoturnover was observed across and within ecosite categories (Figure 4.1).
Table 4.7. Number of species and percent pseudoturnover (n = 75 pair-wise comparisons) for all vascular plant species collectively (n = 379 species) and per growth form for the repeat survey dataset (n = 67 plots).

<table>
<thead>
<tr>
<th>Growth Form</th>
<th>Number of Species Overall</th>
<th>Mean Percent Pseudoturnover</th>
<th>Minimum Percent Pseudoturnover</th>
<th>Maximum Percent Pseudoturnover</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forb</td>
<td>212</td>
<td>15.9</td>
<td>0</td>
<td>44.4</td>
</tr>
<tr>
<td>Graminoid</td>
<td>96</td>
<td>20.4</td>
<td>0</td>
<td>81.8</td>
</tr>
<tr>
<td>Shrub</td>
<td>50</td>
<td>12.9</td>
<td>0</td>
<td>33.3</td>
</tr>
<tr>
<td>Tree</td>
<td>21</td>
<td>15</td>
<td>0</td>
<td>100</td>
</tr>
<tr>
<td>All Growth Forms</td>
<td>379</td>
<td>15.4</td>
<td>0</td>
<td>29.2</td>
</tr>
</tbody>
</table>

Figure 4.1. Variation in percent pseudoturnover (n = 75 pair-wise comparisons) for all vascular plant species collectively (n = 379 species) across the 11 ecosite categories included in the repeat survey dataset (n = 67 plots).
Models for pseudoturnover

Total species richness was positively related to pseudoturnover for all species \( (p = 0.002; R^2 = 0.113) \) \( (\alpha = 0.05) \) (Table 4.3; Figure 4.2). However, neither sampling effort nor ecosite type significantly affected pseudoturnover when considering all species.

Table 4.8. Summary of linear models examining relationships between percent pseudoturnover \( (n = 75 \) pair-wise comparisons) for all vascular plant species collectively \( (n = 379 \) species) and total richness, sampling effort, and ecosite category for the repeat survey dataset \( (n = 67 \) plots). Log\(_{10}\) transformations were applied to all continuous variables except sampling effort.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Coefficient</th>
<th>S.E.</th>
<th>( p )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Relationship with species richness (all growth forms): ( R^2 = 0.113 )</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.551</td>
<td>0.194</td>
<td>0.006</td>
</tr>
<tr>
<td>Richness</td>
<td>0.363</td>
<td>0.112</td>
<td>0.002</td>
</tr>
<tr>
<td>Relationship with species richness and sampling effort: ( R^2 = 0.110 )</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.523</td>
<td>0.198</td>
<td>0.010</td>
</tr>
<tr>
<td>Richness</td>
<td>0.392</td>
<td>0.118</td>
<td>0.001</td>
</tr>
<tr>
<td>Sampling effort</td>
<td>-0.001</td>
<td>0.001</td>
<td>0.405</td>
</tr>
<tr>
<td>Relationship with ecosite: ( R^2 = 0.029 )</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>1.116</td>
<td>0.046</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.226</td>
<td>0.132</td>
<td>0.090</td>
</tr>
<tr>
<td>PX</td>
<td>0.161</td>
<td>0.132</td>
<td>0.227</td>
</tr>
<tr>
<td>PM</td>
<td>0.089</td>
<td>0.085</td>
<td>0.295</td>
</tr>
<tr>
<td>PD</td>
<td>-0.141</td>
<td>0.098</td>
<td>0.157</td>
</tr>
<tr>
<td>MX</td>
<td>0.088</td>
<td>0.132</td>
<td>0.506</td>
</tr>
<tr>
<td>MG</td>
<td>0.133</td>
<td>0.093</td>
<td>0.158</td>
</tr>
<tr>
<td>MD</td>
<td>0.059</td>
<td>0.093</td>
<td>0.528</td>
</tr>
<tr>
<td>RG</td>
<td>0.235</td>
<td>0.219</td>
<td>0.286</td>
</tr>
<tr>
<td>RD</td>
<td>0.095</td>
<td>0.079</td>
<td>0.231</td>
</tr>
<tr>
<td>SD</td>
<td>0.162</td>
<td>0.132</td>
<td>0.221</td>
</tr>
</tbody>
</table>
Figure 4.12. Relationship between percent pseudoturnover \((n = 75 \text{ pair-wise comparisons})\) for all vascular plant species collectively \((n = 379 \text{ species})\) and total richness for the repeat survey dataset \((n = 67 \text{ plots})\). Axes were not log_{10}-transformed for legibility purposes; however, these variables were transformed in the linear model.

Total richness was most strongly correlated with pseudoturnover in graminoids and trees with pseudoturnover positively related to species richness \((p < 0.001, R^2 = 0.187; \text{ and } p < 0.001, R^2 = 0.169, \text{ respectively})\), although relationships with individual growth form richness were more pronounced \((p < 0.001, R^2 = 0.297; \text{ and } p = < 0.001, R^2 = 0.277)\) (see Appendix 4.1 for individual growth form models). Species richness in both total and individual growth forms were weakly related to pseudoturnover for forbs and shrubs with their effects being positive and near-significant in most cases. No general relationships were apparent between ecosite and pseudoturnover for any of the growth forms, although certain ecosite categories had a significant effect in some instances.

Number of species missed

Individual observers missed an average of 7.8 species per plot, ranging from 0 to 31 total species, with forbs comprising the majority of species missed (Table 4.4). For context, among all Rarity and Diversity plots, overall average species richness was 45.4 and ranged from 26.5 to
71.9 species among ecosite categories (see Chapter 1). Moderately high variation in the number of species missed was observed across ecosite categories with inconsistency in the amount of variation per category (Figure 4.3).

**Table 9.4.** Number of species missed per plot \((n = 150\) values for unique species) for all vascular plant species collectively \((n = 379\) species) and per growth form for the repeat survey dataset \((n = 67\) plots).

<table>
<thead>
<tr>
<th>Growth Form</th>
<th>Average</th>
<th>Minimum</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forb</td>
<td>3.8</td>
<td>0.0</td>
<td>19.0</td>
</tr>
<tr>
<td>Graminoid</td>
<td>1.7</td>
<td>0.0</td>
<td>7.0</td>
</tr>
<tr>
<td>Shrub</td>
<td>1.5</td>
<td>0.0</td>
<td>6.0</td>
</tr>
<tr>
<td>Tree</td>
<td>0.8</td>
<td>0.0</td>
<td>5.0</td>
</tr>
<tr>
<td>All Growth Forms</td>
<td>7.8</td>
<td>0.0</td>
<td>31.0</td>
</tr>
</tbody>
</table>

**Figure 4.13.** Variation in the number of species missed per plot \((n = 150\) values for unique species) for all vascular plant species collectively \((n = 379\) species) across the 11 ecosite categories included in the repeat survey dataset \((n = 67\) plots).
**Models for the number of species missed**

Total richness demonstrated a strong and significant positive relationship with the number of species missed by a single observer for all species collectively \((p < 0.001, R^2 = 0.545)\) (Table 4.5; Figure 4.4). Ecosite was also strongly related to the number of species missed for all species, with certain categories having a significant effect on number of species missed \((R^2 = 0.406)\) (Table 4.5).

Total richness was most strongly related to the number of forb species that were missed, and was moderately related to that of the remaining three growth forms (Appendix 4.1). Individual growth form richness was strongly related to the numbers of forb and graminoid species missed, while relationships with those of shrubs and trees were moderate. Both total and individual growth form richness, however, had significant effects on the number of species missed for all growth forms. Ecosite had the strongest relationship with the number of forb species missed, and was moderately related to those of the other growth forms, with significant effects for certain categories.

**Table 4.10.** Summary of linear models examining relationships between the number of species missed per plot \((n = 150\) values for unique species) for all vascular plant species collectively \((n = 379\) species) and both total richness and ecosite category for the repeat survey dataset \((n = 67\) plots). Log_{10} transformations were applied to all continuous variables.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Coefficient</th>
<th>S.E.</th>
<th>(p)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Relationship with species richness (all growth forms): (R^2 = 0.545)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.893</td>
<td>0.112</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Richness</td>
<td>1.009</td>
<td>0.066</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td><strong>Relationship with ecosite: (R^2 = 0.406)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.819</td>
<td>0.035</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.300</td>
<td>0.071</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>PX</td>
<td>-0.078</td>
<td>0.112</td>
<td>0.490</td>
</tr>
<tr>
<td>PM</td>
<td>0.139</td>
<td>0.071</td>
<td>0.052</td>
</tr>
<tr>
<td>PD</td>
<td>-0.515</td>
<td>0.064</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>MX</td>
<td>0.010</td>
<td>0.112</td>
<td>0.931</td>
</tr>
<tr>
<td>MG</td>
<td>0.236</td>
<td>0.078</td>
<td>0.003</td>
</tr>
<tr>
<td>MD</td>
<td>-0.072</td>
<td>0.062</td>
<td>0.245</td>
</tr>
<tr>
<td>RG</td>
<td>0.278</td>
<td>0.188</td>
<td>0.141</td>
</tr>
<tr>
<td>RD</td>
<td>0.055</td>
<td>0.066</td>
<td>0.404</td>
</tr>
<tr>
<td>SD</td>
<td>0.244</td>
<td>0.112</td>
<td>0.031</td>
</tr>
</tbody>
</table>
Figure 4.4. Relationship between the number of species missed per plot ($n = 150$ values for unique species) for all vascular plant species collectively ($n = 379$ species) and total richness for the repeat survey dataset ($n = 67$ plots). Axes were not log$_{10}$-transformed for legibility purposes; however, these variables were transformed in the linear model.

4.5 Discussion

Effective management and conservation of biodiversity is predicated on the ability to detect ecological trends, which itself is contingent upon the recognition and minimization of error. Observer error during vascular plant sampling, represented here as percent pseudoturnover and the number of species missed per plot, generally increases with species richness. The magnitude of the error and the strength of relationships with richness, as well as ecosite, vary by plant growth form and depend on the error metric used. Observers missed an average of 7.8 species per plot. The observed average pseudoturnover of 15.4% for all species was low compared to previous studies, and was not influenced by differences in sampling effort between observers. It is likely that the time-unlimited sampling protocol allowed observers to survey to their saturation point and thus reduced observer error. Previous work based on a portion of this dataset by Zhang et al. (2014), which contrasted time-unlimited with a time-limited protocol, suggested that time limits may result in far higher discrepancies in species lists between observers.
Further, these findings indicate that the species richness of a given plot can inform estimates of the magnitude of observer error present, and suggest that plant functional groups should be assessed individually when examining the factors that affect this and how these can be addressed. Graminoids had the poorest repeatability between observers here, speaking to the need for careful training on the families Cyperaceae, Juncaceae, and Poaceae. Further, particular attention should be paid to this group during surveys.

**Implications for management**
The analysis of pseudoturnover across a large, multi-year study indicates that observer error occurs even among well-trained observer pairs with similar vegetation experience backgrounds. Observer error therefore cannot be ignored when interpreting the results of vegetation inventories. Reported absences of species of conservation concern should be interpreted with respect to measures of total site richness and the field sampling protocol used (time unlimited vs. limited). We encourage the use of time-unlimited protocols in vegetation surveys to reduce pseudoturnover. Smaller plot sizes would further decrease pseudoturnover, but would reduce the likelihood of encountering rarer microhabitats that would increase rare plant encounters. Finally, graminoids are a challenging group which may require extra consideration during surveys.
CHAPTER 5.0: Experimental detectability trials using decoy species

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2. Alberta Parks, Alberta Environment and Parks

5.1 Summary

Successfully detecting rare vascular plant populations during field surveys prior to oil and gas developments has direct implications for conservation of rare species. Industry cannot mitigate for populations of which they are unaware. Imperfect detection leads to underestimates of species presences on leases and thus decreases the reliability of survey data. The issue of imperfect detection has not been examined in detail within boreal environments where vegetation structure would be expected to influence detection rates. Here, we address this issue by using detectability trials with decoy plants, where species are targeted by volunteer observers unaware of their true presence or abundance in survey plots. Our findings indicate that the detectability of cryptic species is very low when abundance is low (0 - 35\%) and plot size is large (< 50\% in \textgeq 100 m\textsuperscript{2}). We suggest that future surveys in the oil and gas area consider species characteristics of target (rare) species, provide records of search effort, and limit plot size through alternative search methods.

5.2 Introduction

Ecological survey data are used to understand species presence and abundance across landscapes and to help guide conservation decisions. When survey data are inaccurate or biased, it affects our knowledge of species distribution, rarity, and conservation status, and conservation efforts. As with other taxa, detection of plants in surveys is imperfect (MacKenzie \textit{et al.} 2005; Morrison 2016). Factors demonstrated to influence plant species detectability include the observer, abundance, phenology, habitat attributes, and morphology (Chen \textit{et al.} 2009; Moore \textit{et al.} 2011; Alexander \textit{et al.} 2012; Garrard \textit{et al.} 2013; McCarthy \textit{et al.} 2013; Ng & Driscoll 2014; Morrison 2016). Work on plant detectability in forested systems is more limited, but studies thus far suggest low detectability (as low as 9\%) of target species in species rich forest plots in China (Chen \textit{et al.} 2009). Imperfect detection of rare species is of specific concern, as one key attribute of rarity is small population size (abundance), a trait shown to correlate with poor detection success.

In the oil sands area of Alberta, Pre-Disturbance Assessment (PDA) surveys are conducted prior to construction of in situ oil sands developments in order to locate populations of rare vascular plant species, which can then be managed through mitigation measures (see Chapter 6). Failure to detect rare species in areas which will undergo development could result in oil and gas-related losses in rare plant populations. Provincial Pre-Disturbance Assessment guidelines direct oil and gas companies and contractors to the Alberta Native Plant Council (ANPC) guidelines for survey methodology (Alberta Native Plant Council 2012; Alberta Energy Regulator 2014). These guidelines advise upon observer experience, pre-survey planning, and survey methodology, but
do not explicitly include recommendations or discussion around imperfect detection. Because plants are static during survey, appropriate effort (time and area covered) during a single visit at peak flowering can ensure detection success, rather than using costly repeat site visits over a single season (Bornand et al. 2014).

Presently, ANPC guidelines for rare plant surveys are not explicit regarding search area or effort (time). Although individuals may search a large geographic area in a single day on a typical oil and gas project, research has demonstrated a lack of repeatability between surveyors on plots of sizes ranging from 4 m² to 2500 m² (Leps & Hadincova 1992; Archaux, Bergès & Chevalier 2007; Zhang et al. 2014). Additionally, observer experience is expected to improve survey outcomes, although this character has not always been correlated with increased success (Moore et al. 2011; Alexander et al. 2012). In Alberta, it is recommended that observers have 4 months (1.5 to 2 field seasons) of taxonomic experience before commencing Pre-Disturbance Assessment (PDA) surveys (Alberta Native Plant Council 2012). Understanding how observers, survey attributes, and species characters interact to affect detection rates and incorporating measures to improve detection in survey guidelines will ensure reliable survey data and increase confidence in reported absences of rare or target species.

Here, we conducted controlled field trials in the manner of Moore et al. (2011), first used in Australia in an invasive species application. Populations of target species (decoys) that are not currently growing in the plot are planted prior to surveys, allowing for the manipulation of species-related factors (e.g. abundance, phenology) and determination of their influence on detectability. Results from the initial study showed that observer identity and plant abundance were the best predictors of observer success (Moore et al. 2011). The goal of our experimental decoy trials was to test the influence of plot size and observer experience (2015), abundance and distribution (patchiness) of target species, observer movement paths (2016), and species characteristics (both years).

5.3 Methods

Study site and decoy planting methods
Both experimental trials were located west of Edmonton, Alberta at Woodbend, a research area owned by the University of Alberta. Upland forest type across the property is predominantly mixedwood with moderate shrub cover, mainly Corylus cornuta (Beaked Hazelnut). While plots differed slightly in tree and shrub density, we considered them to have been effectively similar in structure. Plot boundaries were marked using wooden stakes and string or nylon rope to deter observers from leaving the plots during survey. Start locations were fixed and marked using large signs; observers were asked to meander survey plots beginning from the marked corner, but given no further directions on type of search effort. Decoy plants were planted using garden trowels at randomly determined locations within plots based on two random numbers representing the number of paces along the axes of the plot (i.e. first north/south, then east/west). Every effort was made to reduce disruption during planting. Excess soil was removed from the area and litter was sprinkled around the decoy plant. We watered and checked individuals regularly over both trials and replaced any specimens which were damaged (e.g. herbivory, chlorotic). We used two target species in each year, Symphiotrichum lanceolatum and Viola pedatifida (2015), and Allium cernuum and Petunia sp. (2016) (Figure 5.1).
Volunteer observers in both years were recruited through email and word of mouth. In 2015 we targeted individuals with specific years of vascular plant survey experience and time since their last survey. In 2016 we recruited individuals who had experience conducting targeted surveys, but did not require that these observers be experienced with vascular plants (e.g. we accepted individuals with experience surveying amphibians and bryophytes). Immediately prior to beginning their surveys, observers were shown example specimens of decoy species and told that neither, one, or both species may be present. They were able to revisit the example specimens throughout the day. We instructed observers to survey plots until they felt they had adequately surveyed the area and recorded the total time of survey, as well as the time at which they encountered any target species. Observers were not asked to make full species inventories, thus simulating targeted rare plant surveys. Ethics approval was granted for both trails through the University of Alberta Research Ethics Office (PRO00059103 in 2015 and PRO00064852 in 2016). Participants were debriefed once they had completed all surveys. At that time, study objectives and species presence within plots were disclosed.
**Effects of observer experience and plot size (2015 detectability trials)**

In the 2015 detectability trials, we focused on manipulating plot size and observer experience. We maintained species abundance in all plots at 1 individual/plot/species across the following five plot sizes: 1 m$^2$, 10 m$^2$, 100 m$^2$, 1000 m$^2$, and 2500 m$^2$ with three replicates per size ($n = 15$). Observers were categorized as: 1) **Expert botanist** with > 5 field seasons of rare plant and plant survey experience, 2) **Intermediate botanist** with 2-3 field seasons of general plant survey experience and had completed surveys within the preceding 4 months, and 3) **Intermediate observers** with > 2 field seasons of experience who had not completed a survey within the last 4 months (i.e. that field season). Group 2 (intermediate botanist) aligns with ANPC’s suggestion of 120 days of taxonomic experience for individuals conducting rare plant surveys. Sixteen recruited observers were asked to complete one replicate of each plot size if possible (a minimum of 5) and to complete additional plots if they were so inclined. The order in which plot sizes were completed and the replicate plot identity were randomized for each individual, although complete randomization was forgone at the end of the trial to ensure all plots had at least one observation in each observer experience category.

The two species targets (*Symphiotrichum lanceolatum* [Western willow aster] and *Viola pedatifida* [Crowfoot violet]) (Figure 5.1) were procured from Wild About Flowers, a native seed and plant nursery near Calgary, Alberta. Neither species was flowering at the time of the trial. We recorded the height and maximum width of each planted individual and the number of leaves in *V. pedatifida*. We did not count leaves in the aster as they were too numerous (>100 individual). We measured horizontal cover around each individual decoy plant using a range pole from a distance of 5 and 10 m in all four cardinal directions.

We used mixed-effect logistic regression models to relate detection success to the variables of interest, namely observer experience and plot size, and AIC model evaluation to rank support among candidate models (Burnham & Anderson 2002). Plot size was log transformed. To account for repeated measures in a plot across observers and observers across plots, we used a random effect on both observer and plot replicate. All analyses were completed in R (R Core Team 2015) using the package ‘lme4’ (Bates et al. 2015).

**Effects of population size and distribution (2016 detectability trials)**

In 2016 detectability trials, we maintained a constant plot size of 1000 m$^2$ and recruited 13 observers that had a background in targeted field surveys. In these trials, we manipulated abundance (1, 5, and 10 individuals) and distribution (clumped or diffuse) of two target species (*Allium cernuum* and *Petunia* sp.) across 15 plots using the design illustrated in Figure 5.2. Both species were in flower throughout the trial. To achieve the desired well-spaced arrangement of individuals within “diffuse” plots, we used the same random number pacing system described for the 2015 trials, however when a set of random numbers meant that an individual would be planted in close proximity (< 2 m) to another, we used the next number set to create a minimum distance between patches.
Figure 5.2. Study design used in a 2016 detectability trial. Closed circles indicate *Petunia* sp., open circles *Allium cernuum*. This design was replicated 3 times for a total of 15 experimental plots.

We asked participants to wear Columbus V990 GPS data loggers (Victory Technology Co., Ltd.) during surveys to generate location data suitable for analyzing movement paths. To relate detection success to movement patterns of observers, we measured observer movements as effective search paths in a GIS (ESRI 2015). Specifically, we created steps from GPS log 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). Total search area by each individual in each plot was then calculated as the proportion of each plot searched (total search area divided by plot size). We then used mixed-effect logistic regression models with AIC model evaluation to determine the relationship between species identity, abundance, arrangement, and observer movement metrics on success. To account for repeated measures, we used a random effect on observer and plot replicate. Models were built using the package ‘lme4’ (Bates et al. 2015) in R (R Core Team 2015).

5.4 Results

The influence of observer experience and plot size on detectability (2015 trials)

Sixteen volunteer observers completed 4 to 8 (although most often 5) surveys each, for a total of 83 surveys and 166 species-level observations. Overall, detection of both species was lower than anticipated, less than 50% in plots > 100 m² (10 x 10 meters) and declining rapidly with plot size (Figure 5.3). The more morphologically distinct *V. pedatifida* was found more frequently (57% success across all plots) than *S. lanceolatum* (47%), a more cryptic species that “blended” with similar *Asteraceae* species and *Galium boreale* within survey plots. In plots of 1000 m², the size
used in the 2016 trial, total success of *V. pedatifida* was 35%, as compared to 23% success in *S. lanceolatum*.

**Figure 5.3.** Total success in observing 2 target species across 5 plot sizes for 16 volunteer observers in a 2015 decoy plant detectability trial (n = 166).

Results of logistic regression demonstrated equivalent support for the top five ranked candidate models (ΔAIC < 2) (Table 5.1). All five models indicated that plot size was the major determinant of detection success, with target species having a weakly significant influence (*V. pedatifida* found more frequently), and an observed weak positive effect of height of plant. The lowest AIC ranked model parameters are summarized in Table 5.2. Observer experience level was not a significant factor in any candidate model.

**Table 5.1.** Results of logistic regression models of detection success for two species in the 2015 detectability trials (n observations = 166). Plot area was log transformed in all models. Aster was used as the reference category in the variable “Species”. Survey order refers to the order in which plots were completed by a given observer.
Table 5.2. Parameters and standardized coefficients with associated standard error values for the most supported AIC model of detection success (Table 5.1). Aster was used as the reference category for the variable “Species”. Plot area was log transformed, \( n = 166 \).

<table>
<thead>
<tr>
<th>Parameter (units)</th>
<th>Standardized coefficient</th>
<th>Standardized standard error</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>0</td>
<td>0</td>
<td>0.61</td>
</tr>
<tr>
<td>Plot area ((m^2))</td>
<td>-3.22</td>
<td>0.64</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Species (violet or aster)</td>
<td>2.09</td>
<td>1.01</td>
<td>0.04</td>
</tr>
<tr>
<td>Height (cm)</td>
<td>1.6</td>
<td>1.01</td>
<td>0.12</td>
</tr>
</tbody>
</table>

We built logistic regression models per species and observed differences in explanatory variables included in the best supported models. Observer experience and survey order were weakly significant in the best supported model for \( V.\ pedatifida \), however there was equivalent support for a model containing only plot size \((\Delta AIC = 2.1)\) (Tables 5.3 and 5.4). In contrast, for \( S.\ lanceolatum \) the most supported model contained species height and visibility (Tables 5.5 and 5.6).

Table 5.3. Results of AIC model comparison of candidate models relating the success of detecting \( Viola\ pedatifida \) to explanatory survey variables \((n = 83)\).

<table>
<thead>
<tr>
<th>Model</th>
<th>K</th>
<th>AIC</th>
<th>( \Delta AIC )</th>
</tr>
</thead>
<tbody>
<tr>
<td>success ~ plot area + survey order + experience level + (1</td>
<td>observer) + (1</td>
<td>plot)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ plot area + experience level + (1</td>
<td>observer) + (1</td>
<td>plot)</td>
<td>4</td>
</tr>
<tr>
<td>success ~ plot area + (1</td>
<td>observer) + (1</td>
<td>plot)</td>
<td>3</td>
</tr>
<tr>
<td>success ~ plot area + height + visibility + (1</td>
<td>observer) + (1</td>
<td>plot)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ plot area + height + (1</td>
<td>observer) + (1</td>
<td>plot)</td>
<td>4</td>
</tr>
<tr>
<td>success ~ plot area + height + leaf number + average width + (1</td>
<td>observer) + (1</td>
<td>plot)</td>
<td>6</td>
</tr>
<tr>
<td>success ~ (1</td>
<td>observer) + (1</td>
<td>plot)</td>
<td>2</td>
</tr>
</tbody>
</table>

Table 5.4. Parameters of the best-fitting model of \( Viola\ pedatifida \) detection success \((n = 83)\) as determined by AIC model evaluation (Table 5.3). Plot area was log transformed in all models. Survey order refers to the order in which plots were completed by a given observer. Expert observers (those with > 5 years of experience) were withheld as a reference category in the variable “experience level”.

<table>
<thead>
<tr>
<th>Parameter (units)</th>
<th>Standardized coefficient</th>
<th>Standardized standard error</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>0</td>
<td>0</td>
<td>0.61</td>
</tr>
<tr>
<td>Plot area ((m^2))</td>
<td>-3.22</td>
<td>0.64</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Species (violet or aster)</td>
<td>2.09</td>
<td>1.01</td>
<td>0.04</td>
</tr>
<tr>
<td>Height (cm)</td>
<td>1.6</td>
<td>1.01</td>
<td>0.12</td>
</tr>
</tbody>
</table>
Table 5.5. Results of AIC model comparison of candidate models relating the success of detecting *Symphiotrichum lanceolatum* (*n* = 83) to explanatory survey variables. Plot area was log transformed in all models. Survey order refers to the order in which plots were completed by a given observer. Expert observers (those with > 5 years of experience) were the reference category in the variable “experience level”.

<table>
<thead>
<tr>
<th>Parameter (units)</th>
<th>Standardized coefficient</th>
<th>Standardized standard error</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>0.00</td>
<td>0.00</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Plot area (m²)</td>
<td>-3.79</td>
<td>0.79</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Survey order</td>
<td>1.06</td>
<td>0.62</td>
<td>0.09</td>
</tr>
<tr>
<td>Intermediate observers w/ recent exp.</td>
<td>-1.54</td>
<td>0.79</td>
<td>0.05</td>
</tr>
<tr>
<td>Intermediate observers w/o recent exp.</td>
<td>-0.06</td>
<td>0.73</td>
<td>0.94</td>
</tr>
</tbody>
</table>

Table 5.6. Parameters of the best-fitting model of *Symphiotrichum lanceolatum* detection success (*n* = 83) as determined by AIC model evaluation (Table 5.5). Plot area was log transformed in all models.

<table>
<thead>
<tr>
<th>Parameter (units)</th>
<th>Standardized coefficient</th>
<th>Standardized standard error</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>0</td>
<td>0</td>
<td>0.043</td>
</tr>
<tr>
<td>Plot area (m²)</td>
<td>-3.36</td>
<td>0.963</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Visibility (proportion of range pole)</td>
<td>1.804</td>
<td>0.854</td>
<td>0.035</td>
</tr>
<tr>
<td>Plant height (cm)</td>
<td>1.713</td>
<td>0.807</td>
<td>0.034</td>
</tr>
</tbody>
</table>

Thus, we found limited support for the influence of observer experience in 2015. While we recorded variation in success between observers, these differences could not be attributed to their previous experience when considering both species (Table 5.1). Further, we did not observe any significant difference in effort (time) by experts as compared to intermediate groups with and without recent experience (Figure 5.4).
The influence of population size and distribution on detection success (2016 trials)
Thirteen observers completed between 3-5 surveys resulting in 53 total surveys with 106 observations of both species. We excluded one individual in movement analyses as their data logger malfunctioned and two plots from two other unique observers due to similar data logger failures. This left 12 individuals with 46 movement paths for analysis.

Detection success varied substantially between the showy (*Petunia* sp.) and cryptic (*A. cernuum*) species used in the trial (96% and 38%, respectively). Overall, the showy *Petunia* sp. was nearly perfectly detected and thus, given little variation among experimental treatments, not further considered. Diffusely arranged individuals of *A. cernuum* were 25 - 34% more likely to be detected than the same number planted in a clump with perfect detection failure for single individuals within plots (Table 5.7).

Table 5.7. Detection success of nodding onion (*Allium cernuum*) by 13 observers in 5 arrangement/abundance combinations across 15 experimental plots (n = 53) in 2016.
Results of logistic regression analyses of individual detections demonstrated a significant positive effect of abundance on detection with a weak trend of lower detection rates of clumped individuals. AIC values $< 2$ AIC points apart indicate equivalent support of the top 4 candidate models (Table 5.8). Thus, we suggest that abundance and arrangement of target species act together to influence success, but abundance is the more important predictor (arrangement was often only weakly significant). Overall, there was a weak positive relationship with survey order in all top models (see Table 5.9). The number of seasons completed by an individual observer was included in a single top candidate model, but was not significant. We also completed models independently for A. cernuum since this species had greater variation in detectability (see Appendix 5.1, Tables A5.1.1 and A5.1.2)

Table 5.8. Candidate models of detection success regressed against explanatory variables and ranked using AIC model evaluation ($n = 106$). Species abundance was log transformed in all models. No. of seasons refers to the number of seasons of vascular plant surveys conducted by an individual observer, and survey order is the order in which a given individual completed survey plots. Petunia sp. was withheld as the reference category in all models.

<table>
<thead>
<tr>
<th>Model</th>
<th>K</th>
<th>AIC</th>
<th>ΔAIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>success ~ species + abundance + survey order + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ species + arrangement + abundance + survey order + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>6</td>
</tr>
<tr>
<td>success ~ species + abundance + survey order + no. of seasons + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>6</td>
</tr>
<tr>
<td>success ~ species + arrangement * abundance + survey order + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>7</td>
</tr>
<tr>
<td>success ~ species + arrangement * abundance + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>6</td>
</tr>
<tr>
<td>success ~ species + arrangement + abundance + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ species + abundance + survey order + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ species + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>3</td>
</tr>
<tr>
<td>success ~ species + survey order + no. of seasons + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>2</td>
</tr>
</tbody>
</table>

Table 5.9. Parameters and standardized coefficients with associated standard error values for the most supported AIC model of detection success (Table 5.8) ($n = 106$). Species abundance was log transformed and survey order is the order in which an individual completed survey plots. Petunia sp. was used as the reference category for the variable “Species”.
Observers had quite variable backgrounds (plant surveys within Alberta, Canada, and internationally) and number of years of vascular plant survey experience (range = 0 – 14, median = 3). Observer identity or experience was not, however, related to detection success. Tortuosity and proportional search area did not differ among observers. We observed very uniform speeds across individuals (\( \bar{x} = 0.16 \text{ meters/second}, \ SE = 0.009 \)), although interestingly there was a trend in that the majority of *A. cernuum* detections occurred when \( \sim 30\% \) of the plot had been surveyed. Further search effort did not improve success rate suggesting a possible saturation effect for this species (Figure 5.5).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Standardized coefficient</th>
<th>Standardized standard error</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>0.00</td>
<td>0.00</td>
<td>0.68</td>
</tr>
<tr>
<td>Species</td>
<td>-5.35</td>
<td>0.69</td>
<td>0.03</td>
</tr>
<tr>
<td>Abundance</td>
<td>2.83</td>
<td>0.92</td>
<td>0.00</td>
</tr>
<tr>
<td>Survey order</td>
<td>1.50</td>
<td>1.20</td>
<td>&lt; 0.001</td>
</tr>
</tbody>
</table>

**Figure 5.5.** Detection success for 12 observers of *Allium cernuum* in 15 experimental plots, as compared to the proportion of the 1000 m\(^2\) plot covered by each individual’s buffered search path (\( n = 46 \)).

**5.5 Discussion**
Detection trials have two major advantages to uncontrolled field experiments. First, the truth is known, and thus each false absence can be accounted for. Second, variables of interest can be manipulated with regard to target species in ways which would otherwise be unfeasible. Here, we tested the influence of plot size and observer experience with constant target species abundance (2015) and the influence of observer movement and target species morphology, abundance, and arrangement (2016) on detection success. Understanding how detection success changes with survey variables allows for the development of improved survey guidelines (e.g. future iterations of ANPC survey guidelines) and best practices.

Together these trials have clearly demonstrated that probability of detecting cryptic species at low abundance (i.e. 1 individual/1000 m²) is very low overall (< 35%). The showy Petunia sp. used here demonstrates that consistently high detection rates (96%) can occur even at low abundance when the species is flowering and highly visible. However, a minority of boreal species bear flowers of this size or are as brightly coloured, suggesting that most species would go undetected when rare within plots and when not flowering. Many understory species in the boreal have low overall flowering rates and are most often encountered in their vegetative state. We observed perfect failure at detecting A. cernuum in 1000 m² plots, as compared to 35% in V. pedatifida and 23% in S. lanceolatum, despite A. cernuum being in flower at the time of survey with a distinctive (if slender) inflorescence. The larger size of the two vegetative species likely made them more detectable to observers. Thus, despite the advantages of distinct morphology and phenology, detection of cryptic species is likely far poorer than is currently recognized in plant studies and surveys using larger search areas.

As demonstrated in other work, detection success increased with target species abundance in our 2016 trial, a product of increased encounter rate between the observer and a larger number of individuals (Moore et al. 2011; McCarthy et al. 2013). Considering species arrangement, we recorded a 30% increase in detection success for clumps over single individuals of A. cernuum, presumably due to increased visibility of clustered individuals. However, clumps of 5 and 10 were detected at similar rates (~30%), suggesting that this visual advantage may not scale with clump size. These findings have applicability to the allocation of survey effort during targeted rare plant surveys. We suggest that surveys targeting species which are known to occur at high densities or in large, tufted growth forms (e.g. sedges such as Carex oligosperma and C. vulpinoidea) may require less effort than those targeting species which consistently occur at low densities (e.g. some Botrychium sp., and members of the Orchidaceae), and second that all reported absences of species should include a measure of survey effort (spatial scale and temporal sampling intensity).

Considering all four species targets and two trials, the neutral relationship between observer experience and detection success was surprising. Literature suggests that observer experience is often positively correlated with accuracy and success in detecting species (Morrison 2016) and socially, surveys completed by expert botanists regarded as more reliable. First, we suggest that targeted surveys are not subject to observer effects to the degree that complete site inventories may be. Complete knowledge of the flora would serve a considerable advantage in full species inventories, in both time expenditure and presumably, accuracy. It is also possible that trial search conditions differed from those in the field such that the advantage increased experience is expected to convey was negated. For example, many botanists use microsite associations when
searching for target species with which they are familiar. These associations were not present in this study due to random assignment of planting locations. However, the surveyed area in 2015 trials was often small (3 of 5 plot sizes ≤ 100 m²) and thus microsite associations can be considered irrelevant at this scale. We did not observe an advantage of using expert botanists in small plots when searching for a few target species that are first shown to the observer.

**Implications for management**

Below we summarize considerations that should be made during targeted surveys of rare plants. First, plot size should be limited where possible, possibly through search techniques that divide the total search area into sections with each section searched independently. This contrasts with meandering surveys of larger areas. It is noteworthy that 1000 m², the size used in 2016 trials, is 1/10th the size of the average wellpad footprint in the oil sands area, further highlighting the need for careful consideration of search area. Second, observer experience may not be as important as traditionally considered when hiring botanists for targeted surveys of one or few species. While experienced observers may lend an increased feeling of confidence to reported absences, particularly when searching for rare taxa, our results suggest that novice botanists can achieve very similar results in targeted searches. Finally, search effort should be documented and considered when evaluating reports of species absence, both in terms of time and of area searched in Pre-Disturbance Assessment surveys. Use of GPS data-loggers that track search paths should be considered when possible. Recording time to detection for target species will also allow for further understanding of patterns in detection in field surveys within the oil sands area.
CHAPTER 6.0: Evaluating translocation of rare species in peatlands as a mitigation technique

J. Dennett\textsuperscript{1} and S.E. Nielsen\textsuperscript{1}

\textsuperscript{1} Department of Renewable Resources, University of Alberta

6.1 Summary

Mitigative translocation is a conservation tool employed infrequently, but consistently by oil and gas companies in northeastern Alberta. Translocations are resource intensive projects which may fail to meet conservation goals due to shortcomings in planning, execution, or monitoring. We used experimental translocations in the oil sands region to evaluate this tool and to inform future mitigation efforts. Specifically, we focused on peatlands and two fen species, \textit{Sarracenia purpurea} and \textit{Carex oligosperma}. Transplanting occurred in the growing season of 2014 with follow up monitoring over a span of 3 years. Factors that were anticipated to influence transplant success were measured and include species composition and cover, and nutrient status. Results of monitoring in 2015 and 2016 indicate high transplant survival for both species and little variation between recipient sites despite differences in major nutrients. \textit{Sarracenia purpurea} transplants had consistent high survival and flowering rates over both years, while \textit{Carex oligosperma} transplants had reduced growth and survival between 2015 and 2016. Translocations are most often conducted under time and logistical constraints and may be most effective if employed on species which are known to have broad environmental tolerances.

Project status: Results current to 2016, final field monitoring and project completion in 2017.

6.2 Introduction

In situ oil sands developments in Alberta result in vegetation and topsoil disturbance that alters habitat for vascular plants. Mitigating the loss of populations of rare vascular plant species from human developments is a conservation priority. Mandatory Pre-disturbance Assessment (PDA) surveys are conducted prior to development on oil and gas lease areas to locate populations of concern (Alberta Energy Regulator 2014). Once rare species are identified, companies undertake mandatory or voluntary conservation measures to preserve these populations. Mitigative measures employed in the oil and gas industry include shifting the footprint to avoid direct loss of the population during construction, no action, seed collection, and translocating individuals. Recently distinguished in the literature from traditional translocation projects, mitigative translocation is the movement of plant material or animals which are at risk of imminent destruction due to development (Germano \textit{et al.} 2015).

This practice is used infrequently, but consistently in Alberta for vascular plants, receiving criticism as a conservation measure when the species ecology and determinants of success are poorly understood (Fahselt 2007; Maslovat 2009). Results from prior studies indicate a mixed success at best for re-introduction and augmentation projects (Fahselt 2007; Godefroid \textit{et al.} 2011; Primack & Drayton 2011; Lawrence & Kaye 2011; Drayton & Primack 2012; Clements
A widely recognized failing of traditional translocations is poor recipient site selection, presumably caused by a lack of understanding of species’ niches (Godefroid et al. 2011). Mitigative translocations conducted by oil and gas companies are limited in three main ways which differ from traditional projects. First, follow up monitoring is lacking, presumably through lack of allocation in resources and/or high turnover in the consulting industry leading to a loss of information around transplants. Second, public reporting of projects, even those which do receive monitoring, is rare. This reduces the ability to determine efficacy and understand the factors influencing successful translocations across species and projects. Finally, these projects are often more time sensitive than reintroduction or augmentation projects. This puts limitations on the pre-translocation planning process resulting in recipient sites being selected quickly, in some cases without consideration of methodology and knowledge of the ecology of the species being translocated.

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, habitat types which have not been the focus of translocation research in Canada (Clements 2013). In the oil sands region fens, groundwater fed peatlands, contain a greater number of rare species than other habitat types (Zhang et al. 2014; Chapter 2). Further, these habitats are more 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, Bayley & Schindler 2012; Raab & Bayley 2013). Given the conservation focus and knowledge gaps associated with this habitat, we chose to conduct experimental translocations for two rare peatland obligate species, *Sarracenia purpurea* and *Carex oligosperma*, in 2014. Our specific objectives were to determine overall survival and growth of these transplants and determine how recipient site characters may improve or reduce survival as they relate to characters at donor sites. In practice, oil and gas companies may have a limited time in which to select recipient sites and thus relating survival to recipient site characters can be used to direct future translocation efforts. Further, the success of our methodology can inform future guidelines and best practices for boreal plant translocations. This project has recorded 2 years of post-translocation data with the final year of monitoring to occur in 2017. Results presented here are based on the first 2 post-translocation field seasons where transplant survival, growth, and flowering were recorded.

### 6.3 Methods

**Donor and recipient study sites**

Donor populations were selected from known large (> 1000 individuals), healthy populations encountered during Ecological Monitoring Committee for the Lower Athabasca (EMCLA) Rare Plant Project surveys (now the Terrestrial Vascular Plant Monitoring Project for the Lower Athabasca Rarity and Diversity plots, see Chapter 1 for a detailed description). Three independent donor and recipient sites were selected for each species. Each focal species therefore has 6 experimental sites. No donor sites contained both focal species and no recipient sites had existing populations. Recipient sites were selected to vary in physical structure and vegetation composition from donor sites. All 6 *S. purpurea* sites are located in the vicinity of Conklin, Alberta. Three *C. oligosperma* sites were located near Fort Mackay, Alberta, while the other 3 were located near Conklin (Figure 6.1).
Figure 6.1. Location of 12 experimental translocation sites in northeast 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.

**Study design, removal, and planting methodology**

Translocations were conducted between late August and mid-September of 2014. We 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. 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 has a founder population of 50 individuals, from 3 different donor locations (Figure 6.2). This provided the minimum suggested founding population size of 50 individuals (Franklin 1980). In total, 210 transplants of each species were translocated. No transplants were moved between donor locations.
Figure 6.2. Study design schematic used in 2014 mitigative transplantations, where 210 transplants for each *Sarracenia purpurea* and *Carex oligosperma* were moved among three donor and recipient sites, respectively.

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 40 m². Replanting was conducted over a similarly sized area at all three recipient sites per species. A benefit to planting transplants in a small area is the increased likelihood of locating them in the future, noted to be a problem in previous work with *S. purpurea* (Linda Halsey, pers. comm.).

Prior to removal, each transplant was given an identification code with a metal washer attached to a loop of string and flagging tape to allow relocation in the peatland environment. A 0.25 m² quadrat (0.5 x 0.5 m PVC frame) was then placed around the transplant and percent cover was estimated for all surrounding species within the quadrat. Transplants were cut from the peat as small monoliths averaging 50 cm² with substrate attached. We removed healthy adult plants 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. We transported plants in coolers or tubs with icepacks between donor and recipient sites.

Planting in peat substrate was straightforward and only troublesome at sites with high root density. We 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 m²) were repeated when plants were translocated, giving two complete vegetation surveys for each transplant. This was also completed for donor site controls.
Spring relocation checks and water chemistry sampling
In early June of 2015 and 2016 spring relocation checks and water chemistry sampling was conducted at all experimental translocation sites. Spring checks consisted of re-marking all individuals with flagging tape overhead and replacing unique id tags. Due to corrosion of aluminum plated washers, plants were remarked in June 2016 using engraved metal ‘racetrack’ tags attached to 2 or 3 ft. pigtails inserted into the peat at each transplant. Each tag is engraved with the transplant’s identification code (visible in Figure 6.5).

Sampling of water chemistry at each site was used to determine the nutrient status of donor and recipient sites. For water sampling we laid out a transect in the orientation that water was expected to flow (e.g. perpendicular to open water or upland slopes) through the fen. 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. We then collected three water samples using piezometers inserted ~ 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 again until ~ 500 ml of water had been collected. Samples were not filtered in the field. The samples were then analyzed for the following: nitrite and nitrate (NO$_2^-$ and NO$_3^-$, respectively), total nitrogen (N), total kjeldahl nitrogen, total phosphorus (P), sodium (Na$^+$), potassium (K$^+$), calcium (Ca$^{2+}$), and magnesium (Mg$^{2+}$). This protocol will be repeated one last time in June, 2017. All analyses were conducted by the Biogeochemical Analytical Service Laboratory (BASL) at the University of Alberta.

Summer survival, growth, and flowering checks
All experimental translocation sites were revisited between late July and early August to conduct summer survival, growth, and flowering checks. By this time of year *Carex oligosperma* is fully mature but has not begun to shed perigynia and *Sarracenia purpurea* flowers are generally fully mature or beginning to senesce. For this project, we defined transplants to be deceased when no green stems were produced in *C. oligosperma* and all pitchers were completely brown (i.e. non-living tissue) in *S. purpurea*. Flowering and survival were recorded as binary variables. Growth in *S. purpurea* was determined by counting the number of living pitchers. Pitchers persist over at least one winter in this species and grow from the center of the plant, such that dead pitchers often form a ring around the exterior. Leaves in *C. oligosperma* senesce annually and new above ground material is produced each spring (Ryser & Kamminga 2009). For *C. oligosperma* we determined growth by first counting all stems and then measuring the tallest vegetative stem or culm (flowering stem). When transplants produced flowers, we recorded the average length, average width, number, and gender of spikes borne on culms using calipers.

Finally, at each donor site we measured 30 non-transplanted individuals of *C. oligosperma* to determine average height and spike measurements under normal growing conditions in 2015 and 2016. We conducted similar counts of pitchers and flowering rates at *S. purpurea* sites using a different set of 30 individuals in 2015 and 2016. As such, only the data for *C. oligosperma* will be considered here and used as a benchmark for transplant growth. Measurements of summer survival, growth, and flowering will be repeated one last time in 2017.
6.4 Results

Water chemistry
Differences in water chemistry were observed among recipient and donor sites for both species (Figures 6.3 and 6.4). SR3 and OD2 are located very close to gravel and paved roads, respectively, which corresponds to high sodium (not shown) and calcium levels from road salt and hardener applications.

![Figure 6.3. Water chemistry variables (Total kjeldahl nitrogen, total phosphorus, calcium, and pH) over 2 years of sampling at *Sarracenia purpurea* donor (SD1-3) and recipient (SR1-3) sites.](image-url)
Figure 6.4. Water chemistry variables (Total kjeldahl nitrogen, total phosphorus, calcium, and pH) over 2 years of sampling at *Carex oligosperma* donor (OD1-3) and recipient (OR1-3) sites.

**Survival, growth, and flowering**

Survival, growth, and flowering was high overall declining minimally between 2015 and 2016 (Table 6.1). *Sarracenia purpurea* transplants at recipient sites are virtually all extant (99% in 2015 and 98% in 2016), with an interesting case at recipient site SR3 where an individual believed dead in 2015 grew new leaf material in 2016. Transplants showed a net gain of pitchers between 2015 and 2016 at all recipient sites (Table 6.1). Anecdotally, transplants of *S. purpurea* appear robust at all recipient sites (Figure 6.5).
Figure 6.5. A transplanted *Sarracenia purpurea* at SR2, near Conklin, Alberta. This transplant contains a minimum of 7 individuals as this species produces only one flowering stalk per individual per year.

*Carex oligosperma* survival decreased from 94% in 2015 to 88% in 2016. Flowering rates for both focal species declined in 2016, although to a greater extent in *C. oligosperma* (Table 6.1). Comparison of average height between all *C. oligosperma* transplants at recipient sites and 90 un-transplanted individuals at donor sites confirmed field observations that transplants appear stunted (Figure 6.6). Average height of all transplants was 38.2 cm compared to controls (un-transplanted) averaging 83 cm in 2015. This disparity decreased slightly in 2016 to 52.9 cm and 68.9 cm, respectively.

Table 6.1. Survival, growth, and flowering counts at recipient sites of transplanted *Sarracenia purpurea* and *Carex oligosperma* over 2 years (percentages in brackets). Average change in pitchers is the difference in the count of pitchers per transplant between 2015 and 2016. Average change in stems is the difference in vegetative stems or culms per transplant between 2015 and 2016.
Table 6.2 reports control transplants where 20 individuals were immediately replanted at their donor site for evaluating the effects of transplanting. Survival, growth, and flowering trends were similar among these individuals and translocated (founder) populations at recipient sites for *S. purpurea* (Table 6.1).

**Table 6.2.** Survival, growth, and flowering counts at recipient sites of control transplants of *Sarracenia purpurea* at donor sites. Average change in pitchers is the difference in the count of pitchers per transplant between 2015 and 2016.

<table>
<thead>
<tr>
<th><em>Sarracenia purpurea</em></th>
<th>2015</th>
<th>2016</th>
<th>av. Δ pitchers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Recipient 1</td>
<td>Living (50)</td>
<td>Flowering (21)</td>
<td>Relocated (50)</td>
</tr>
<tr>
<td>Recipient 2</td>
<td>Living (49)</td>
<td>Flowering (25)</td>
<td>Relocated (49)</td>
</tr>
<tr>
<td>Recipient 3</td>
<td>Living (44)</td>
<td>Flowering (18)</td>
<td>Relocated (45)</td>
</tr>
<tr>
<td>Total</td>
<td>Living (143)</td>
<td>Flowering (64)</td>
<td>Relocated (144)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><em>Carex oligosperma</em></th>
<th>2015</th>
<th>2016</th>
<th>av. Δ stems</th>
</tr>
</thead>
<tbody>
<tr>
<td>Recipient 1</td>
<td>Living (44)</td>
<td>Flowering (17)</td>
<td>Relocated (46)</td>
</tr>
<tr>
<td>Recipient 2</td>
<td>Living (45)</td>
<td>Flowering (18)</td>
<td>Relocated (50)</td>
</tr>
<tr>
<td>Recipient 3</td>
<td>Living (46)</td>
<td>Flowering (15)</td>
<td>Relocated (47)</td>
</tr>
<tr>
<td>Total</td>
<td>Living (135)</td>
<td>Flowering (50)</td>
<td>Relocated (143)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><em>Sarracenia purpurea</em></th>
<th>2015</th>
<th>2016</th>
<th>av. Δ pitchers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Donor 1</td>
<td>Living (20)</td>
<td>Flowering (9)</td>
<td>Relocated (20)</td>
</tr>
<tr>
<td>Donor 2</td>
<td>Living (18)</td>
<td>Flowering (7)</td>
<td>Relocated (18)</td>
</tr>
<tr>
<td>Donor 3</td>
<td>Living (20)</td>
<td>Flowering (5)</td>
<td>Relocated (20)</td>
</tr>
<tr>
<td>Total</td>
<td>Living (58)</td>
<td>Flowering (21)</td>
<td>Relocated (58)</td>
</tr>
</tbody>
</table>
Figure 6.6. The inflorescence of a transplanted *Carex oligosperma* in 2016. Aside from shorter stature, transplants sometimes developed fewer perigynia and more male flowers than usually observed in this species.

**Failure to relocate transplants**

As can be seen in Tables 6.1 and 6.2, not all transplants were successfully relocated at recipient sites. For *C. oligosperma*, 4 individuals were never relocated at recipient sites, and 7 were only relocated in one monitoring year. Relocation of *S. purpurea* was similar, with 1 transplant never relocated and 9 only relocated in one monitoring year. Relocation of *S. purpurea* at donor sites was similar, with 1 individual never relocated and 6 only found in one monitoring year.

The identification of control transplants at *C. oligosperma* donor sites was unexpectedly complicated by the species’ rhizomatous growth form. This sedge forms dense mats and tends to dominate wetlands where it is found. We were unable to determine if shoots in the region of the original transplant tag originated from the transplant or from neighbouring individuals. Further, rapid peat growth at one donor site (OD3) resulted in the burial of original transplant markers after a single season. Due to these factors, data collected on control transplants will not be used.

**6.5 Discussion of 2015 and 2016 results**

Mitigative translocations are being conducted at high costs under time and logistical constraints in the oil sands region of Alberta. These projects are rarely publicly reported with regional success rates largely unknown. Evaluating this conservation practice for peatland species provides an opportunity to consider the efficacy of this approach and determine factors which may increase success. Here, we conducted experimental transplants of two rare peatland species with the objective of determining if environmental factors at recipient sites influence survival and growth. Monitoring of these transplants will conclude in the summer of 2017 for 3-years post-transplant. To date we have observed high transplant survival and low variability in growth and flowering among recipient sites, despite demonstrated variation in major nutrient levels and field measured variation in community composition and vegetation structure. This suggests that the environmental tolerances of these two species may be broader than the conditions under
which they established at their sites of origin, and that these rare species are more dispersal than environmentally limited. Production of a second generation at recipient sites would confirm the ability to regenerate under these conditions, another dimension of site suitability, although this project considers success to be transplant survival, rather than reproduction (Godefroid et al. 2011).

The initial high survival of transplants demonstrated here is supported by examples from the literature (Drayton & Primack 2000; Godefroid et al. 2011; Cypher 2014), although initial success may not always correlate with long term success (Drayton & Primack 2012). For instance, we are anticipating further declines in survival for *C. oligosperma* in 2017 based on evidence of decreased stem production from 2015 to 2016 and two years of stunted growth that would potentially reduce stored carbohydrates. Reduction in flowering rates between 2015 and 2016 may indicate weakened plants, but could also be the result of individuals not flowering in consecutive years, a trait which is not uncommon in understory vascular plants. *Carex oligosperma* 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* and *Carex magellanica* (Laberge et al. 2015). Although the use of seed in translocation often yields poor results (Godefroid et al. 2011), these findings may indicate a lack of amenability of *C. oligosperma* to movement from its natal site.

*Sarracenia purpurea* transplants show very little variation in flowering and growth rates among recipient sites and between recipient and donor sites. This species is relatively well studied and has demonstrated a fairly broad range in habitat tolerances (e.g. acidic to alkaline fens) in the eastern United States (Karberg & Gale 2013) and a tendency toward rapid colonization and growth when introduced to Swiss peatlands (Parisod, Trippi & Galland 2005). It is possible that over a longer timeframe then what is considered here *S. purpurea* may form self-sustaining populations at recipient sites. We suggest that our documented success to date with this species serve to reinforce the idea of restricting mitigative translocations to species whose ecology is relatively well understood or which have demonstrated success across environmental gradients in other research. Use of species whose tolerances are unknown may be best approached on an experimental basis, rather than considered as an active conservation strategy. This may be the most effective use of resources available for mitigative translocations.
CHAPTER 7.0: Persistence of historic rare vascular plant populations in the oil sands region of Alberta

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7.1 Summary

Rare vascular plant species are of management and conservation priority due to increased susceptibility to extirpation. Related decision making processes rely on understanding which species are rare and where their populations occur. In Alberta, the Alberta Conservation Information Management System (ACIMS) manages species-level spatial data and provides the ranks used to define rarity at the provincial level. However, a proportion of the population records maintained by ACIMS were obtained through surveys for oil and gas-related projects conducted prior to disturbance. If populations are extirpated due to construction of associated infrastructure, rarity ranks may be misapplied and the effect of energy development on species persistence may not be properly understood. We completed remote sensing-based assessments and field visits for historic ACIMS rare plant populations in the oil sands area to determine the prevalence of disturbance footprint across populations and the rate of extirpation. The majority of populations in the region are located within 500 m of footprint, but small-scale disturbances such as seismic lines tend to be the most prevalent footprint type. Field observations indicated approximately 30% of historic populations had been extirpated and that the likelihood of persistence declined with increasing proximity to disturbance. These findings suggest revisitation surveys in disturbed landscapes such as the oil sands area should be encouraged to both improve the accuracy of the provincial rare plant database and to understand how oil and gas-related activities may threaten plant populations.

Project status: Field and remote sensing work completed in 2016 is summarized here. Additional field site visits are planned for 2017.

7.2 Introduction

The maintenance of rare vascular plant species at provincial and national scales is both culturally and ecologically significant. The ability to create and achieve conservation and management goals for rare species requires accurate categorization of rarity and conservation statuses. In turn, rarity ranking schemes rely on accurate data representing the location and status of populations for each species (Rabinowitz, Cairns & Dillon 1986; Hartley & Kunin 2003; Master et al. 2012). Extirpations of historic recorded populations can introduce bias into conservation rankings if these records are considered in rarity assessments. Specifically, inclusion of extirpated records can result in species appearing prevalent on the landscape and cause inaccurate rarity ranks to be
applied. In Alberta, rare element occurrences of species (populations) are maintained by the Alberta Conservation Information Management System (ACIMS). Records of rare species, or those of conservation concern, are submitted by the public to the provincial government, most often by amateur botanists or those employed by consulting, government, or research agencies.

ACIMS uses NatureServe methods to assign sub-national ranks (S-Ranks) to all native vascular plant 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, population size, habitat specificity, and population trends. Although the calculator is comprehensive, arguably the majority of species have substantial data gaps for these attributes. As such, ranking is often based primarily on two 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; Lorna Allen, pers. comm.). At a sub-national level, this provides species or community level ranks of S1-S5, with S1 being especially vulnerable to extirpation and S5 being secure. Additional ranks indicate cases where species are unable to be assessed due to extinction, provincial extirpation, lack of taxonomic resolution, or insufficient data (e.g. SU). Uncertainty is expressed through combined ranks (e.g. S1S2).

Population records are often collected and submitted to ACIMS by consultants as part of Pre-disturbance Assessment (PDA) rare plant surveys conducted on oil and gas leases (Alberta Native Plant Council 2012; Alberta Energy Regulator 2014). While submission to ACIMS is recommended, only the PDA survey itself is mandatory (Alberta Energy Regulator 2014). Submitted records correspond to proposed development projects that may result in imminent direct or indirect disturbance to identified populations of conservation concern. Changes in land use have been identified as the primary cause of extirpation of local populations (Fagan, Kennedy & Unmack 2005; Pergl et al. 2012; Gerke, Farnsworth & Brumback 2014). For instance, a revisitation study for 63 historic populations of a single species in Switzerland observed that 24% of extirpations were associated with increased levels of agricultural disturbance and fragmentation (Lienert, Fischer & Diemer 2002). This raises concerns regarding the use of records associated with Pre-disturbance Assessments to inform provincial rankings of rarity and conservation status, as including populations (element occurrences) at high risk of extirpation may artificially inflate record numbers and thus result in status ranks being more secure than true conditions. Presently, we do not know the extent of footprint in proximity to historic vascular plant records or the regional rate of population extirpation related to oil and gas development.

Here, we addressed this knowledge gap in two parts. First, we used a remote-sensing imagery-based approach to quantify the amount and type of footprint in proximity to 188 ACIMS rare vascular plant records within the oils sands area. Second, we visited a subset of 40 populations during peak flowering periods within the oil sands area of northeast Alberta in 2016. At each site we identified whether historic rare plant populations were indeed still present to better understand whether oil sands developments affected the persistence of known rare plant populations. We plan to visit an additional 20 sites in 2017 to increase sample size and to better understand regional patterns of population loss.
7.3 Methods

Study area
Our study area was defined by the boundaries of the provincial oil sands area (OSA) and associated surface mineable area (SMA) (Figure 7.1). The oil sands area covers roughly one-fifth of the province (21% or 140,000 km²), encompassing all three major provincial oil sands deposits and nine natural sub-regions, and predominately consists of boreal mixedwood and other boreal sub-region types. Within the oil sands area, the surface mineable area occupies only 4,800 km² (3.4% of the OSA) of land surrounding the urban areas of Fort McMurray and Fort McKay, Alberta. The surface mineable area contains bitumen deposits which can be extracted via conventional methods (i.e. surface mining) and encompasses all provincial surface mining operations. Oil extraction activities in the oil sands area are comprised of in-situ oil sands developments that typically use steam assisted gravity drainage (SAGD) or other solvents to extract bitumen via wells. We considered these two areas separately in analyses as footprints of these oil sands developments are vastly different (Rooney, Bayley & Schindler 2012)
Figure 7.1. Study area and locations of rare plant populations in the oil sands and surface mineable areas \((n = 188)\) considered in a revisitation project.

**Assessing historic rare plant populations using remote sensing imagery**

Locations of rare plant populations were obtained from the publicly available ACIMS database (ACIMS 2016). ACIMS tracks the conservation status of both individual vascular plant species and species communities and refers to these records as element occurrences. Here, we focused on element occurrences of single species (populations) that were contained within the oil sands area boundary, which amounted to 188 records of 47 unique species (Figure 7.1). ACIMS records are represented digitally in a GIS by polygons of varying size based on population extent and/or spatial accuracy of the original field observation. Median polygon size was 2,600 m², with populations falling both within and outside of oil sands leases. Publicly available provincial oil sands lease boundaries, current to 2013, were examined to determine whether records occurred within lease areas (Government of Alberta, 2016).
We used three metrics to evaluate the human footprint in proximity of historic rare plant populations. First, we determined the number of records for which footprint occurred within the original polygon boundaries, using the Alberta Biodiversity Monitoring Institute Human Footprint Mapping Layer (2012) (ABMI 2016) (footprint frequency). Second, we buffered the centroid of each record by radii of 100 m, 500 m, and 1 km and estimated the proportion of footprint in each buffer class using the ABMI layer (footprint proportion). We compared the proportion of footprint within these buffer classes using a Wilcoxon Rank Sum test. Finally, we exported ACIMS polygons to Google Earth and visually examined each record using the most current and clear imagery available (2008 - 2016) (DigitalGlobe 2016). Based on the visual extent of disturbance, we categorized records as having high, moderate, or low footprint (footprint severity). A high footprint was associated with polygons that were entirely disturbed by anthropogenic activities (e.g. Figure 7.2A and 7.2B). Moderate records were those with substantial amounts of disturbance, but also intact habitat remaining within the original polygon (e.g. Figure 7.2C). Records classified as having a low footprint were either undisturbed or had little disturbance within the original polygon. This could include minor vegetated (early seral) disturbances such as exploratory seismic lines, or small scale disturbances adjacent to the polygon (e.g. Figure 7.2D, 7.2E, and 7.2F).
Figure 7.2. Examples of footprint types associated with provincial records of rare vascular plant populations in the oil sands area of Alberta. A) Population likely lost to conventional surface mining, B) Population likely lost to commercial in-situ, C) Developing in-situ likely impacting population, D) Developing in-situ adjacent to population, E) Exploratory seismic development adjacent to population, F) Undisturbed population. All maps created from Google Earth version 7.1.7.2026, imagery from DigitalGlobe 2016.

Assessing status of historic rare plant populations in the field

We visited 40 ACIMS populations representing 19 species within the oil sands area between June and August of 2016 (Figure 7.1). Site locations and target species are provided in Appendix 7.1. Sites were stratified based on logistical constraints and chosen to encompass a range of habitat and disturbance types in both terrestrial and aquatic habitats. Two observers with survey experience were trained using specimens from the University of Alberta herbarium (ALTA) prior to conducting surveys. The observers visited each site during the expected flowering period to increase detectability (Moore et al. 2011). The centers of the original ACIMS polygons were used as the plot centers for all rare plant searches. At terrestrial sites \( (n = 32) \), surveyors searched a circular plot with a radius of 50 m around the record center (maximum search area of 7,850
m²). For sites that had been cleared of forest cover and are maintained as disturbed ground (i.e. wellpad surface), the cleared area was given a precursory scan and the search radius was established around the edge of the feature, if possible (e.g. radius began from the vegetated edge of a wellpad). For aquatic open water sites \((n = 8)\), 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. To address concerns regarding detectability of cryptic and/or low abundance populations, both observers wore GPS data loggers during surveys to track search paths and recorded the total search time (effort) for all sites (see Appendix 7.2 for results and discussion of survey time analysis). Transect tapes and a handheld GPS were used to ensure that the search radius was adhered to and the total search area was covered. These protocols will be used in all 2017 surveys.

For each site surveyed we assigned a broad habitat type based on four categories. These included (1) aquatic (i.e. open water), (2) lowland (i.e. a singular fen), (3) upland (a general category consisting largely of mixedwood stands), and (4) anthropogenic-altered. Field sites included two wetland margin sites (beaver pond edge and river margin) and a single lowland fen connected to a large lake. These three sites were included in our aquatic habitat category as they were highly hydrologically regulated, resulting in three final habitats (aquatic, upland, and anthropogenic-altered). We classified a site as anthropogenic-altered where the soils had been modified by human activity such that they were no longer in a natural state (e.g. vegetated gravel berms surrounding wellpads). It should be noted that disturbance was present across all habitat types, with the anthropogenic-altered classification only assigned to those modified to the extent described.

**Statistical analysis of field data**

Persistence of rare plant populations across all sites was assessed using logistic regression. To assess the effect of oil sands footprint on persistence of rare plant populations surveyed in the field, we considered 3 different measures of disturbance: (1) minimum distance to nearest disturbance from the reported population boundary, (2) type of nearest disturbance, and (3) proportion of total mapped human footprint within 100 m, 500 m, and 1 km. All continuous variables were log transformed. We also considered record age, reported initial population size, and habitat type as explanatory variables. Given that detectability of rare plants cannot be assumed to be perfect (MacKenzie, Nichols & Lachman 2002; McCarthy et al. 2013), reported rates of persistence are likely underestimated, although we do not expect bias in detectability based on covariates tested here. All analyses were conducted in R (R Core Team 2015) using the package ‘lme4’ (Bates et al. 2015) with a series of candidate models compared using Akaike Information Criteria (AIC) (Burnham & Anderson 2002).

**7.4 Results**

**Oil sands footprint in proximity to historic rare plant populations**

Across the region, 38% and 48% of recorded populations occurred on oil and gas lease areas within the oil sands area (45 of 119) and surface mineable area (33 of 69 populations), respectively. In estimating footprint frequency, we expected to see a greater number of
undisturbed populations outside of lease areas. Instead, we observed similar frequency between both on and off lease areas and between the oil sands and surface mineable areas (Table 7.1).

Table 7.1. Proportion of all ACIMS recorded rare vascular plant records ($n = 188$) within the surface mineable and oil sands areas in Alberta with human footprint within the originally reported polygon, reported by footprint type and whether the record occurred on or off an oil and gas lease area. More than one footprint type could occur within a given polygon.

<table>
<thead>
<tr>
<th>Population type</th>
<th>Oil sands area Number of populations (%)</th>
<th>Surface mineable area Number of populations (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Populations on lease</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mine Site</td>
<td>-</td>
<td>5 (15%)</td>
</tr>
<tr>
<td>Seismic</td>
<td>16 (36%)</td>
<td>6 (18%)</td>
</tr>
<tr>
<td>Pipeline/Transmission line</td>
<td>8 (18%)</td>
<td>3 (9%)</td>
</tr>
<tr>
<td>Wellpads</td>
<td>12 (27%)</td>
<td>7 (21%)</td>
</tr>
<tr>
<td>Forestry</td>
<td>1 (2%)</td>
<td>2 (6%)</td>
</tr>
<tr>
<td>Industrial infrastructure</td>
<td>5 (11%)</td>
<td>4 (12%)</td>
</tr>
<tr>
<td>Roads</td>
<td>2 (4%)</td>
<td>2 (6%)</td>
</tr>
<tr>
<td>Agriculture</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>None</td>
<td>25 (56%)</td>
<td>16 (48%)</td>
</tr>
<tr>
<td><strong>Populations off lease</strong></td>
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<td></td>
</tr>
<tr>
<td>Mine Site</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Seismic</td>
<td>23 (31%)</td>
<td>9 (24%)</td>
</tr>
<tr>
<td>Pipeline/Transmission line</td>
<td>10 (14%)</td>
<td>2 (5%)</td>
</tr>
<tr>
<td>Wellpads</td>
<td>12 (16%)</td>
<td>8 (22%)</td>
</tr>
<tr>
<td>Forestry</td>
<td>4 (5%)</td>
<td>-</td>
</tr>
<tr>
<td>Industrial infrastructure</td>
<td>9 (12%)</td>
<td>2 (5%)</td>
</tr>
<tr>
<td>Roads</td>
<td>12 (16%)</td>
<td>1 (5%)</td>
</tr>
<tr>
<td>Agriculture</td>
<td>12 (16%)</td>
<td>-</td>
</tr>
<tr>
<td>None</td>
<td>38 (51%)</td>
<td>21 (57%)</td>
</tr>
</tbody>
</table>

Results of footprint proportion estimates indicate that, in both regions, ~ 65% of rare plant populations have disturbance within 100 m of the record centroid. This figure increases to ~ 90% at distances of 500 m and 1 km. Wilcoxon tests suggested a significantly higher proportion of footprint in proximity to records on lease when compared to off lease areas and no significant difference between the surface mineable and oil sands area at all buffer distances (Table 7.2).

Table 7.2. Results of Wilcoxon Rank Sum tests on the proportion of footprint across 3 buffer sizes compared for on and off lease areas and the surface mineable and oil sands areas ($n = 188$).
Visually classified amounts of human footprint (footprint severity) for the 188 rare plant records across both regions included 36 populations (19%) with moderate or high footprints (Appendix 7.3). Seven populations (4%) were entirely dominated by footprint within the polygon and surrounding area (5 within the surface mineable area, 2 within the oil sands area; Figure 7.2A) suggesting the loss of those populations. Level of footprint affecting 9 populations (5%) could not be determined as the polygon area was so large as to encompass multiple land cover types and disturbances. Thus, visually, the majority of populations in the oil sands area had no, low, or indirect (adjacent) footprint (Figure 7.2E, 7.2F).

**Field surveys of population persistence**

Rare plant populations were relocated at 27 of 40 sites (68%). Three sites are suspected to have been misidentification of species in the original records based on similar species found at the site (see Appendix 7.4 for details). These records were removed from statistical analysis. Thus, the minimum estimated rate of persistence of rare plants in the study area was 73%. Of the 10 presumed extirpations, 4 populations were located within oil sands leases resulting in an on lease persistence of 67%, versus off lease persistence of 76% across the entire region (Table 7.3). Of the sites visited thus far, 17 were located within the surface mineable area and 12 of these are currently persisting (70%). Of the 20 surveyed sites within the oil sands area, 15 are persisting (65%).

**Table 7.3.** Number of rare plant records (populations) detected in the summer 2016 by location on or off oil sands leases ($n = 37$).

<table>
<thead>
<tr>
<th></th>
<th>Off lease</th>
<th>On lease</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Extirpated</td>
<td>6</td>
<td>4</td>
<td>10</td>
</tr>
<tr>
<td>Persisting</td>
<td>19</td>
<td>8</td>
<td>27</td>
</tr>
<tr>
<td>Total</td>
<td>25</td>
<td>12</td>
<td>37</td>
</tr>
<tr>
<td>% persisting</td>
<td>76%</td>
<td>67%</td>
<td>73%</td>
</tr>
</tbody>
</table>

The 19-species targeted in habitats ranging from open water wetlands to dry, sandy uplands, and the rate of persistence varied among these habitats. All populations persisted in aquatic-related habitats (open water wetlands, a single lowland, and two riparian margins, $n = 11$), 67% persisted in upland sites ($n = 18$), and finally 50% persisted in anthropogenic-altered sites ($n = 8$). Where species were persisting, we observed small population sizes ($\leq 30$ individuals) at 14 of the 27 sites (52%). Persistence by species and population size where encountered is reported in Table 7.4.

**Table 7.4.** Persistence across populations of 19 rare vascular plant species at 37 field sites within the oil sands area.
### Species Ranking

<table>
<thead>
<tr>
<th>Species</th>
<th>S-Rank</th>
<th>No. records</th>
<th>% persisting</th>
<th>average pop^*n where persisting</th>
<th>Pop^*n range</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Botrychium crenulatum</em></td>
<td>S3</td>
<td>1</td>
<td>0</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Carex oligosperma</em></td>
<td>S3</td>
<td>1</td>
<td>100</td>
<td>1000</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Carex vulpinoidea</em></td>
<td>S3</td>
<td>2</td>
<td>50</td>
<td>10</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Cyripedium acaule</em></td>
<td>S3</td>
<td>7</td>
<td>71</td>
<td>37</td>
<td>4-150</td>
</tr>
<tr>
<td><em>Dryopteris cristata</em></td>
<td>S3</td>
<td>3</td>
<td>100</td>
<td>36</td>
<td>15-80</td>
</tr>
<tr>
<td><em>Gratiola neglecta</em></td>
<td>S3</td>
<td>2</td>
<td>50</td>
<td>3</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Houstonia longifolia</em></td>
<td>S3</td>
<td>1</td>
<td>100</td>
<td>1</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Isoetes echinospora</em></td>
<td>S2</td>
<td>2</td>
<td>100</td>
<td>17</td>
<td>5-30</td>
</tr>
<tr>
<td><em>Lactuca biennis</em></td>
<td>S3</td>
<td>4</td>
<td>50</td>
<td>1.5</td>
<td>1-2</td>
</tr>
<tr>
<td><em>Lathyrus palustris</em></td>
<td>S1</td>
<td>1</td>
<td>100</td>
<td>100</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Liparis loeselii</em></td>
<td>S2</td>
<td>1</td>
<td>100</td>
<td>40</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Najas flexilis</em></td>
<td>S3</td>
<td>1</td>
<td>100</td>
<td>100</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Nymphaea leibergii</em></td>
<td>S2</td>
<td>4</td>
<td>100</td>
<td>62</td>
<td>50-100</td>
</tr>
<tr>
<td><em>Nymphaea tetragona</em></td>
<td>S2</td>
<td>1</td>
<td>100</td>
<td>75</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Phegopteris connectilis</em></td>
<td>S3</td>
<td>1</td>
<td>100</td>
<td>75</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Polygaloides paucifolia</em></td>
<td>S2</td>
<td>1</td>
<td>100</td>
<td>1000</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Potentilla bimundorum</em></td>
<td>S2</td>
<td>1</td>
<td>0</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Sceptridium oneidense</em></td>
<td>S1</td>
<td>2</td>
<td>50</td>
<td>20</td>
<td>n/a</td>
</tr>
<tr>
<td><em>Spiranthes lacera</em></td>
<td>S2</td>
<td>1</td>
<td>0</td>
<td>n/a</td>
<td>n/a</td>
</tr>
</tbody>
</table>

**Statistical analysis of field survey data**

Results of logistic regression with AIC model comparison suggest that population persistence is equally well-explained by and negatively related to increasing proximity to footprint and habitat type (ΔAIC < 2), but not related to record age, footprint type or proportion (Table 7.5). Model fit as evaluated by AIC was virtually identical across all buffer sizes (footprint proportion), hence, only the 100 m model is shown (Table 7.5). However, coefficient estimates (β) and standard error values in the habitat model suggest weak fit (Table 7.6); thus, we consider distance to footprint to be the best candidate model. Footprint metrics and other explanatory variables were too highly correlated to be included within the same model, therefore all models contained a single variable. Reported initial population size was not available for all records and therefore could not be used in model comparison, however, examination of the data suggested no relationship with persistence.

**Table 7.5.** Ranking of candidate models comparing support for factors predicting the persistence of rare plant records in the oil sands region using field site data (n = 37). Both distance to closest disturbance and proportion of disturbed area within 100 m were log transformed.

<table>
<thead>
<tr>
<th>Model</th>
<th>K</th>
<th>AIC</th>
<th>ΔAIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Presence ~ distance to closest footprint</td>
<td>1</td>
<td>38.48</td>
<td>0</td>
</tr>
<tr>
<td>Presence ~ broad habitat class</td>
<td>1</td>
<td>40.05</td>
<td>1.57</td>
</tr>
<tr>
<td>Presence ~ proportion of footprint within 100 m</td>
<td>1</td>
<td>44.28</td>
<td>5.8</td>
</tr>
</tbody>
</table>
Table 7.6. Model parameters for the two most-supported models based on AIC comparison ($\Delta$AIC < 2, Table 7.5). Distance to disturbance reflects the minimum distance to the closest footprint type from the record polygon and was log transformed. Upland habitat was withheld as the reference category for habitat class.

<table>
<thead>
<tr>
<th>Model and parameters</th>
<th>$\beta$ value</th>
<th>Std. error</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Distance to footprint model</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.056</td>
<td>0.502</td>
<td>0.911</td>
</tr>
<tr>
<td>Distance to disturbance</td>
<td>1.23</td>
<td>0.497</td>
<td>0.013</td>
</tr>
<tr>
<td><strong>Habitat class model</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.693</td>
<td>0.5</td>
<td>0.166</td>
</tr>
<tr>
<td>Habitat: Aquatic</td>
<td>All sites persisting</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habitat: Anthropogenic-altered</td>
<td>-0.693</td>
<td>0.866</td>
<td>0.423</td>
</tr>
</tbody>
</table>

7.5 Discussion

Our remote sensing imagery-based estimation of the frequency, proportion, and intensity of footprint around 188 rare vascular plant populations in the oil sands and surface mineable regions suggests that, while footprint is common in close proximity to records, it is most often of low intensity. Populations on lease areas have higher proportions of surrounding footprint than those off lease, and populations within the more heavily developed surface mineable region do not differ from those within the larger oil sands area. Field visits to a subset of these sites ($n = 37$) recorded a minimum persistence rate of 73%. Persistence was best explained by distance to nearest footprint, with populations further from disturbance being more likely to persist. Our results suggest that proximity is more indicative of persistence than footprint type or total amount. Although sites on oil and gas lease areas were suspected to be at greater risk of extirpation, we did not observe large discrepancies in persistence between on and off lease populations (67 and 76%, respectively). Model results suggested weak support for lower persistence on anthropogenic-altered sites when compared to intact upland sites and we observed persistence rates in these habitats of 50 and 67%, respectively. Together, results of imagery-based analysis suggesting close proximity of footprint to records and field visits suggesting a negative effect of proximity to disturbance indicate the need for future revisitation efforts.

We relocated species growing in environments where we had expected them to be extirpated (e.g. Carex vulpinoidea growing on gravel berms surrounding wellpads). It is important to note that populations persisting on sites with highly modified soils or those where local hydrology may be affected by past or future construction may not persist into the future. Population recruitment, health, and species longevity are beyond the scope of this project. Long-term monitoring would be required to fully understand the dynamics of extirpation of rare plant populations in the oil sands region. As far as we know, this is the first attempt at a dedicated re-
inventory of ACIMS records within the oil sands region. Field visits in 2017 will be highly valuable for the continued updating of the ACIMS provincial dataset and understanding the influence of large- and small-scale footprint on rare vascular plant populations.

**Implications for management**

Human disturbance poses a threat to vascular plant species due to the potential alteration or destruction of habitat. We advocate for the development of revisitation standards for assessing the status of rare plant populations within oil and gas leases. In the case of surveying for small populations in the field, search efforts need to be intensive enough to ensure adequate species detection; here, a maximum of 2 person hours were necessary to encounter a single individual plant. If our initial findings from these surveys are representative of the condition of rare plant populations across the region tracked by ACIMS, it is presumable that some ranks may be misapplied in the future if historical records are not verified prior to inclusion. Revisitation of populations suspected to be extirpated based on mapped proximity to human footprints should be prioritized, but we do not consider this alone to be a reliable proxy. As extirpation occurred even on sites with little disturbance, some degree of stochasticity is apparent. These findings indicate that rare plant populations located in altered habitats may not persist into the future, and emphasize the need for further study to assess this potential trend and its possible effects on conservation status of plants in the region.
Literature Cited


## APPENDIX 1.1: Target rare vascular plant species list compiled from the EMCLA database for the Rarity and Diversity plot surveys in the Lower Athabasca Region.

<table>
<thead>
<tr>
<th>Vascular species</th>
<th>Conservation Status Rank (2014)</th>
<th>Flowering time</th>
<th>Highest detectability</th>
<th>Previous collections by month (%)</th>
<th>Identified vegetatively</th>
<th>Habitat</th>
<th>Site description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Carex houghtoniana</td>
<td>S3S4</td>
<td>June-July</td>
<td>June-July</td>
<td>17</td>
<td>42</td>
<td>dry</td>
<td>Fire beneficial; dry acidic sandy soils; often with pine</td>
</tr>
<tr>
<td>Carex supina</td>
<td></td>
<td></td>
<td>July</td>
<td>100</td>
<td>dry</td>
<td>Dry sandy gravelly habitats, eroding slopes</td>
<td></td>
</tr>
<tr>
<td>Carex umbellata</td>
<td>S2</td>
<td>April-July</td>
<td>July</td>
<td>89</td>
<td>0</td>
<td>Dry-mesic</td>
<td>Sandy habitats in the boreal, especially disturbed areas, open woods particularly pine</td>
</tr>
<tr>
<td>Potentilla multifida</td>
<td>S1</td>
<td>July</td>
<td>July</td>
<td>73</td>
<td>9</td>
<td>dry</td>
<td>Sandy areas, often in slightly disturbed areas</td>
</tr>
<tr>
<td>Spiranthes lacera</td>
<td>S1</td>
<td>mid-July to August</td>
<td>mid-July to August</td>
<td>43</td>
<td>57</td>
<td>dry</td>
<td>Dry woodlands and grasslands; often with <em>Vaccinium myrtilloides</em></td>
</tr>
<tr>
<td>Stellaria arenicola</td>
<td>S1</td>
<td>July to August</td>
<td>Summer</td>
<td>0</td>
<td>22</td>
<td>dry</td>
<td>Sandy areas only</td>
</tr>
<tr>
<td>Tanacetum bipinnatum huronense</td>
<td>S2</td>
<td>May-July</td>
<td>Summer</td>
<td></td>
<td></td>
<td>dry</td>
<td>Gravely or sandy areas.</td>
</tr>
<tr>
<td>Carex backii</td>
<td>S3</td>
<td>May-July</td>
<td>Early</td>
<td>38</td>
<td>25</td>
<td>both</td>
<td>Dry (to moist) shady woods. Elsewhere in riparian woodland.</td>
</tr>
<tr>
<td>Species</td>
<td>Season</td>
<td>Blooming Period (months)</td>
<td>Abundance</td>
<td>Bloom Duration</td>
<td>Habitat Characteristics</td>
<td></td>
<td></td>
</tr>
<tr>
<td>-------------------------------</td>
<td>-----------------</td>
<td>--------------------------</td>
<td>-----------</td>
<td>----------------</td>
<td>-------------------------------------------------------------------------------------------------------------</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Chrysosplenium tetrandrum</em></td>
<td>Summer</td>
<td>May-July</td>
<td></td>
<td>May-July</td>
<td>both rock crevices, wet conifer forests</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Artemisia tilesii spp. elatior</em></td>
<td>July-Sept; fruits late summer and fall</td>
<td>Summer</td>
<td>17</td>
<td>50</td>
<td>both woodlands, river flats and alpine slopes</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Cypripedium acaule</em></td>
<td>S3</td>
<td>Late June and July</td>
<td>June-July</td>
<td>23</td>
<td>16 both Wetlands, woods, and overgrown sand dunes; deceptive orchid- poor pollination</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Malaxis paludosa</em></td>
<td>S1</td>
<td>June-August</td>
<td>Summer</td>
<td>40</td>
<td>60 both wet bogs, in sphagnum moss</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Cardamine pratensis</em></td>
<td>S3</td>
<td>May-June</td>
<td>Summer</td>
<td>11</td>
<td>0 both wet along creeks, in swamps; high water table</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Carex capitata</em></td>
<td>S3</td>
<td>June-August</td>
<td>Summer</td>
<td>42</td>
<td>32 both wet areas, calcareous fens</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Carex oligosperma</em></td>
<td>S3?</td>
<td>Late June and July</td>
<td>Summer</td>
<td>21</td>
<td>56 both wet meadows and bogs</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Carex retrorsa</em></td>
<td>S3</td>
<td>May-September</td>
<td>Late spring to early fall</td>
<td>45</td>
<td>27 both swamps and wet meadows</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Chrysosplenium iowense</em></td>
<td>S3?</td>
<td>May-July</td>
<td>May-July</td>
<td></td>
<td>wet both shady moist to wet stream banks and marshes in montane areas</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Drosera linearis</em></td>
<td>S3</td>
<td>mid June to early July</td>
<td>Summer</td>
<td></td>
<td>wet both marl fens, either in shallow water or on soil hummocks</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Eupatorium</em></td>
<td>S1S2</td>
<td>Late July to early July</td>
<td>Summer</td>
<td></td>
<td>wet both wet to moist meadows and open</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Species</td>
<td>Season</td>
<td>Bloom Period</td>
<td>Moisture</td>
<td>Environment</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>-------------------------------</td>
<td>--------</td>
<td>-------------------------------</td>
<td>----------------</td>
<td>-------------------------------------------------</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Hypericum majus</em></td>
<td>Summer</td>
<td>Late June to September</td>
<td>wet</td>
<td>wet sites in the boreal forest</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Juncus brevicaudatus</em></td>
<td>Summer</td>
<td>July to August (fruits)</td>
<td>wet</td>
<td>very moist to wet substrate; lake shores and marshes</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Sarracenia purpurea</em></td>
<td>Summer</td>
<td>spring flower; pitcher in late spring/summer</td>
<td>wet</td>
<td>Bogs, fens, wet meadows</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Carex heleonastes</em></td>
<td>Summer</td>
<td>June - August</td>
<td>wet</td>
<td>Wet open calcareous sites on fens and marshes. Also in bogs, muskegs, lake shores, swamps, wet sandy roadsides, seeps</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Panicum acuminatum</em></td>
<td>SU</td>
<td></td>
<td>wet</td>
<td>Moist sandy soils at woodland edges, marshy places, around hot springs</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Lycopodiella inundata</em></td>
<td>S2</td>
<td></td>
<td>wet</td>
<td>Sphagnum bogs; elsewhere on sand shores and in marshes and other wet sites</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
APPENDIX 11.2: Location of Rarity and Diversity plots.

Table A1.2. Plot ID, location, and date of field surveys conducted in the Rarity and Diversity plots (n = 602) in the Lower Athabasca Region between 2012 and 2015. Plot identification codes with A2 or B2 indicate those surveyed twice by the same observer in different seasons of the same year (i.e. early and late summer visits) (n = 8).

<table>
<thead>
<tr>
<th>Plot ID</th>
<th>Year</th>
<th>Field Date</th>
<th>Easting (UTM NAD83 Z12)</th>
<th>Northing (UTM NAD83 Z12)</th>
<th>Ecosite</th>
<th>Number of Surveys</th>
</tr>
</thead>
<tbody>
<tr>
<td>009A</td>
<td>2013</td>
<td>7/28/2013</td>
<td>499329</td>
<td>6007464</td>
<td>RD</td>
<td>1</td>
</tr>
<tr>
<td>009B</td>
<td>2013</td>
<td>7/28/2013</td>
<td>499328</td>
<td>6007404</td>
<td>RD</td>
<td>1</td>
</tr>
<tr>
<td>1003A</td>
<td>2013</td>
<td>07/04/2013</td>
<td>467162</td>
<td>6368784</td>
<td>MX</td>
<td>2</td>
</tr>
<tr>
<td>1003B</td>
<td>2013</td>
<td>07/04/2013</td>
<td>467066</td>
<td>6368884</td>
<td>MX</td>
<td>1</td>
</tr>
<tr>
<td>1008A</td>
<td>2013</td>
<td>07/08/2013</td>
<td>465774</td>
<td>6366192</td>
<td>MM</td>
<td>1</td>
</tr>
<tr>
<td>1008B</td>
<td>2013</td>
<td>07/08/2013</td>
<td>465890</td>
<td>6367561</td>
<td>MM</td>
<td>1</td>
</tr>
<tr>
<td>1015A</td>
<td>2014</td>
<td>7/25/2014</td>
<td>465960</td>
<td>6368996</td>
<td>RD</td>
<td>1</td>
</tr>
<tr>
<td>1015B</td>
<td>2014</td>
<td>7/25/2014</td>
<td>466013</td>
<td>6368829</td>
<td>PM</td>
<td>1</td>
</tr>
<tr>
<td>101A</td>
<td>2012</td>
<td>8/27/2012</td>
<td>478082</td>
<td>6146446</td>
<td>MD</td>
<td>1</td>
</tr>
<tr>
<td>101B</td>
<td>2012</td>
<td>8/27/2012</td>
<td>478390</td>
<td>6146441</td>
<td>MD</td>
<td>1</td>
</tr>
<tr>
<td>102A</td>
<td>2013</td>
<td>08/10/2013</td>
<td>517010</td>
<td>6166402</td>
<td>PM</td>
<td>1</td>
</tr>
<tr>
<td>102B</td>
<td>2013</td>
<td>08/10/2013</td>
<td>516891</td>
<td>6166255</td>
<td>PM</td>
<td>1</td>
</tr>
<tr>
<td>1039A</td>
<td>2014</td>
<td>7/24/2014</td>
<td>470681</td>
<td>6369862</td>
<td>PX</td>
<td>1</td>
</tr>
<tr>
<td>1039B</td>
<td>2014</td>
<td>7/24/2014</td>
<td>470734</td>
<td>6369691</td>
<td>MX</td>
<td>1</td>
</tr>
<tr>
<td>1044A</td>
<td>2013</td>
<td>8/26/2013</td>
<td>489240</td>
<td>6387969</td>
<td>RD</td>
<td>1</td>
</tr>
<tr>
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APPENDIX 1.3: Complete list of vascular plant species detected at the Rarity and Diversity plots in the Lower Athabasca Region between 2012 and 2015.

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APPENDIX 4.1: Individual growth form models for pseudoturnover and the number of species missed.

Models for pseudoturnover per growth form

Table A4.1. Summary of linear models examining relationships between pseudoturnover for forbs and total richness, forb richness, and ecosite category for the repeat survey dataset. Log₁₀ transformations were applied to all continuous variables.

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<td>0.102</td>
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<td>0.289</td>
</tr>
<tr>
<td>SD</td>
<td>0.204</td>
<td>0.159</td>
<td>0.206</td>
</tr>
</tbody>
</table>
Table A4.2. Summary of linear models examining relationships between pseudoturnover for graminoids and total richness, graminoid richness, and ecosite category for the repeat survey dataset. Log_{10} transformations were applied to all continuous variables.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Beta Coefficient</th>
<th>S.E.</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Relationship with species richness (all growth forms): $R^2 = 0.187$</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.664</td>
<td>0.432</td>
<td>0.128</td>
</tr>
<tr>
<td>Richness</td>
<td>1.057</td>
<td>0.249</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Relationship with graminoid richness: $R^2 = 0.297$</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.372</td>
<td>0.146</td>
<td>0.0132</td>
</tr>
<tr>
<td>Graminoid richness</td>
<td>0.884</td>
<td>0.156</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Relationship with ecosite: $R^2 = 0.048$</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>1.123</td>
<td>0.105</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.061</td>
<td>0.302</td>
<td>0.839</td>
</tr>
<tr>
<td>PX</td>
<td>0.352</td>
<td>0.302</td>
<td>0.248</td>
</tr>
<tr>
<td>PM</td>
<td>0.099</td>
<td>0.194</td>
<td>0.611</td>
</tr>
<tr>
<td>PD</td>
<td>-0.436</td>
<td>0.226</td>
<td>0.058</td>
</tr>
<tr>
<td>MX</td>
<td>-0.221</td>
<td>0.302</td>
<td>0.466</td>
</tr>
<tr>
<td>MG</td>
<td>0.277</td>
<td>0.213</td>
<td>0.198</td>
</tr>
<tr>
<td>MD</td>
<td>-0.223</td>
<td>0.213</td>
<td>0.299</td>
</tr>
<tr>
<td>RG</td>
<td>0.474</td>
<td>0.502</td>
<td>0.349</td>
</tr>
<tr>
<td>RD</td>
<td>0.199</td>
<td>0.181</td>
<td>0.275</td>
</tr>
<tr>
<td>SD</td>
<td>0.165</td>
<td>0.302</td>
<td>0.588</td>
</tr>
</tbody>
</table>
Table A4.3. Summary of linear models examining relationships between pseudoturnover for shrubs and total richness, shrub richness, and ecosite category for the repeat survey dataset. Log_{10} transformations were applied to all continuous variables.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Beta Coefficient</th>
<th>S.E.</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Relationship with species richness (all growth forms): R² = 0.036</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.506</td>
<td>0.288</td>
<td>0.083</td>
</tr>
<tr>
<td>Richness</td>
<td>0.321</td>
<td>0.166</td>
<td>0.057</td>
</tr>
<tr>
<td><strong>Relationship with shrub richness: R² = 0.026</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.693</td>
<td>0.214</td>
<td>0.00184</td>
</tr>
<tr>
<td>Shrub richness</td>
<td>0.333</td>
<td>0.192</td>
<td>0.087</td>
</tr>
<tr>
<td><strong>Relationship with ecosite: R² = 0.028</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>1.061</td>
<td>0.065</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.263</td>
<td>0.187</td>
<td>0.164</td>
</tr>
<tr>
<td>PX</td>
<td>0.020</td>
<td>0.187</td>
<td>0.913</td>
</tr>
<tr>
<td>PM</td>
<td>0.013</td>
<td>0.120</td>
<td>0.912</td>
</tr>
<tr>
<td>PD</td>
<td>-0.205</td>
<td>0.140</td>
<td>0.148</td>
</tr>
<tr>
<td>MX</td>
<td>-0.405</td>
<td>0.187</td>
<td>0.034</td>
</tr>
<tr>
<td>MG</td>
<td>0.115</td>
<td>0.132</td>
<td>0.387</td>
</tr>
<tr>
<td>MD</td>
<td>-0.024</td>
<td>0.132</td>
<td>0.857</td>
</tr>
<tr>
<td>RG</td>
<td>0.101</td>
<td>0.310</td>
<td>0.746</td>
</tr>
<tr>
<td>RD</td>
<td>0.018</td>
<td>0.112</td>
<td>0.876</td>
</tr>
<tr>
<td>SD</td>
<td>0.133</td>
<td>0.187</td>
<td>0.479</td>
</tr>
</tbody>
</table>
Table A4.4. Summary of linear models examining relationships between pseudoturnover for trees and total richness, tree richness, and ecosite category for the repeat survey dataset. Log_{10} transformations were applied to all continuous variables.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Beta Coefficient</th>
<th>S.E.</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Relationship with species richness (all growth forms): $R^2 = 0.169$</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-1.424</td>
<td>0.571</td>
<td>0.015</td>
</tr>
<tr>
<td>Richness</td>
<td>1.317</td>
<td>0.329</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

| Relationship with tree richness: $R^2 = 0.277$ |
| Intercept      | -0.173           | 0.198| 0.387 |
| Tree richness  | 1.414            | 0.261| <0.001|

| Relationship with ecosite: $R^2 = 0.052$ |
| Intercept      | 0.736            | 0.137| <0.001|
| NT             | 0.570            | 0.394| 0.153 |
| PX             | 0.194            | 0.394| 0.624 |
| PM             | 0.431            | 0.253| 0.094 |
| PD             | -0.287           | 0.295| 0.334 |
| MX             | 0.371            | 0.394| 0.350 |
| MG             | 0.300            | 0.278| 0.285 |
| MD             | -0.278           | 0.278| 0.321 |
| RG             | 0.735            | 0.655| 0.266 |
| RD             | -0.034           | 0.237| 0.886 |
| SD             | 0.637            | 0.394| 0.111 |
Figure A4.1. Variation in pseudoturnover per growth form across the 11 ecosite categories included in the repeat survey dataset.
Figure A4.2. Relationships between pseudoturnover per growth form and total richness for the repeat survey dataset. Axes were not log_{10}-transformed for legibility purposes; however, some variables were transformed in the linear models.
Figure A4.3. Relationships between pseudoturnover per growth form and individual growth form richness for the repeat survey dataset. Axes were not log$_{10}$-transformed for legibility purposes; however, some variables were transformed in the linear models.
Models for the number of species missed per growth form

Table A4.5. Summary of linear models examining relationships between the number of forb species missed and total richness, forb richness, and ecosite category for the repeat survey dataset. Log_{10} transformations were applied to all continuous variables.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Beta Coefficient</th>
<th>S.E.</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Relationship with species richness (all growth forms): R² = 0.411</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.800</td>
<td>0.118</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Richness</td>
<td>0.807</td>
<td>0.069</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Relationship with forb richness: R² = 0.394</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.332</td>
<td>0.081</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Forb richness</td>
<td>0.659</td>
<td>0.058</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Relationship with ecosite: R² = 0.296</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.573</td>
<td>0.035</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.235</td>
<td>0.071</td>
<td>0.001</td>
</tr>
<tr>
<td>PX</td>
<td>-0.147</td>
<td>0.112</td>
<td>0.193</td>
</tr>
<tr>
<td>PM</td>
<td>0.084</td>
<td>0.071</td>
<td>0.238</td>
</tr>
<tr>
<td>PD</td>
<td>-0.402</td>
<td>0.064</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>MX</td>
<td>0.054</td>
<td>0.112</td>
<td>0.632</td>
</tr>
<tr>
<td>MG</td>
<td>0.195</td>
<td>0.078</td>
<td>0.013</td>
</tr>
<tr>
<td>MD</td>
<td>-0.023</td>
<td>0.062</td>
<td>0.710</td>
</tr>
<tr>
<td>RG</td>
<td>0.334</td>
<td>0.188</td>
<td>0.078</td>
</tr>
<tr>
<td>RD</td>
<td>-0.032</td>
<td>0.066</td>
<td>0.624</td>
</tr>
<tr>
<td>SD</td>
<td>0.257</td>
<td>0.112</td>
<td>0.023</td>
</tr>
</tbody>
</table>
Table A4.6. Summary of linear models examining relationships between the number of graminoid species missed and total richness, graminoid richness, and ecosite category for the repeat survey dataset. A log_{10} transformation was applied only to the number of species missed.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Beta Coefficient</th>
<th>S.E.</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Relationship with species richness (all growth forms): R² = 0.26</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.011</td>
<td>0.041</td>
<td>0.782</td>
</tr>
<tr>
<td>Richness</td>
<td>0.006</td>
<td>0.001</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Relationship with graminoid richness: R² = 0.422</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.052</td>
<td>0.027</td>
<td>0.052</td>
</tr>
<tr>
<td>Graminoid richness</td>
<td>0.029</td>
<td>0.002</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Relationship with ecosite: R² = 0.204</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.275</td>
<td>0.032</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.260</td>
<td>0.065</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>PX</td>
<td>0.117</td>
<td>0.102</td>
<td>0.255</td>
</tr>
<tr>
<td>PM</td>
<td>0.136</td>
<td>0.065</td>
<td>0.037</td>
</tr>
<tr>
<td>PD</td>
<td>-0.181</td>
<td>0.058</td>
<td>0.002</td>
</tr>
<tr>
<td>MX</td>
<td>-0.066</td>
<td>0.102</td>
<td>0.523</td>
</tr>
<tr>
<td>MG</td>
<td>0.194</td>
<td>0.071</td>
<td>0.007</td>
</tr>
<tr>
<td>MD</td>
<td>-0.044</td>
<td>0.057</td>
<td>0.435</td>
</tr>
<tr>
<td>RG</td>
<td>0.225</td>
<td>0.172</td>
<td>0.191</td>
</tr>
<tr>
<td>RD</td>
<td>0.183</td>
<td>0.060</td>
<td>0.003</td>
</tr>
<tr>
<td>SD</td>
<td>0.083</td>
<td>0.102</td>
<td>0.419</td>
</tr>
</tbody>
</table>
**Table A4.7.** Summary of linear models examining relationships between the number of shrub species missed and total richness, shrub richness, and ecosite category for the repeat survey dataset. No log\(_{10}\) transformations were applied to any variables.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Beta Coefficient</th>
<th>S.E.</th>
<th>(p)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Relationship with species richness (all growth forms): (R^2 = 0.180)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.003</td>
<td>0.246</td>
<td>0.989</td>
</tr>
<tr>
<td>Richness</td>
<td>0.027</td>
<td>0.004</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td><strong>Relationship with shrub richness: (R^2 = 0.185)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.318</td>
<td>0.288</td>
<td>0.270</td>
</tr>
<tr>
<td>Shrub richness</td>
<td>0.137</td>
<td>0.020</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td><strong>Relationship with ecosite: (R^2 = 0.136)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>1.750</td>
<td>0.191</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.917</td>
<td>0.387</td>
<td>0.019</td>
</tr>
<tr>
<td>PX</td>
<td>-0.917</td>
<td>0.614</td>
<td>0.137</td>
</tr>
<tr>
<td>PM</td>
<td>-0.250</td>
<td>0.387</td>
<td>0.519</td>
</tr>
<tr>
<td>PD</td>
<td>-1.292</td>
<td>0.349</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>MX</td>
<td>-1.250</td>
<td>0.614</td>
<td>0.043</td>
</tr>
<tr>
<td>MG</td>
<td>0.536</td>
<td>0.427</td>
<td>0.211</td>
</tr>
<tr>
<td>MD</td>
<td>-0.789</td>
<td>0.339</td>
<td>0.021</td>
</tr>
<tr>
<td>RG</td>
<td>0.750</td>
<td>1.028</td>
<td>0.467</td>
</tr>
<tr>
<td>RD</td>
<td>-0.341</td>
<td>0.360</td>
<td>0.344</td>
</tr>
<tr>
<td>SD</td>
<td>0.417</td>
<td>0.614</td>
<td>0.498</td>
</tr>
</tbody>
</table>
Table A4.8. Summary of linear models examining relationships between the number of tree species missed and total richness, tree richness, and ecosite category for the repeat survey dataset. A $\log_{10}$ transformation was applied only to the number of species missed.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Beta Coefficient</th>
<th>S.E.</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Relationship with species richness (all growth forms): $R^2 = 0.166$</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.041</td>
<td>0.037</td>
<td>0.264</td>
</tr>
<tr>
<td>Richness</td>
<td>0.004</td>
<td>0.001</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td><strong>Relationship with tree richness: $R^2 = 0.220$</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.044</td>
<td>0.032</td>
<td>0.169</td>
</tr>
<tr>
<td>Tree richness</td>
<td>0.036</td>
<td>0.005</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td><strong>Relationship with ecosite: $R^2 = 0.135$</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>0.138</td>
<td>0.028</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>NT</td>
<td>0.195</td>
<td>0.057</td>
<td>0.001</td>
</tr>
<tr>
<td>PX</td>
<td>-0.008</td>
<td>0.090</td>
<td>0.927</td>
</tr>
<tr>
<td>PM</td>
<td>0.162</td>
<td>0.057</td>
<td>0.005</td>
</tr>
<tr>
<td>PD</td>
<td>-0.093</td>
<td>0.051</td>
<td>0.071</td>
</tr>
<tr>
<td>MX</td>
<td>0.172</td>
<td>0.090</td>
<td>0.059</td>
</tr>
<tr>
<td>MG</td>
<td>0.081</td>
<td>0.063</td>
<td>0.200</td>
</tr>
<tr>
<td>MD</td>
<td>-0.057</td>
<td>0.050</td>
<td>0.255</td>
</tr>
<tr>
<td>RG</td>
<td>0.212</td>
<td>0.151</td>
<td>0.164</td>
</tr>
<tr>
<td>RD</td>
<td>0.009</td>
<td>0.053</td>
<td>0.862</td>
</tr>
<tr>
<td>SD</td>
<td>0.188</td>
<td>0.090</td>
<td>0.039</td>
</tr>
</tbody>
</table>
Figure A4.4. Variation in the number of species missed per growth form across the 11 ecosite categories included in the repeat survey dataset.
Figure A4.5. Relationships between the number of species missed per growth form and total richness for the repeat survey dataset. Axes were not log$_{10}$-transformed for legibility purposes; however, some variables were transformed in the linear models.
Figure A4.6. Relationships between the number of species missed per growth form and individual growth form richness for the repeat survey dataset. Axes were not log_{10}-transformed for legibility purposes; however, some variables were transformed in the linear models.
APPENDIX 5.1 Single species models (*Allium cernuum*).

**Table A5.1.1.** Results of AIC model comparison of candidate models relating the success of detecting *Allium cernuum* (*n* = 53) to explanatory survey variables. Abundance was log transformed in all models. Survey order refers to the order in which plots were completed by a given observer.

<table>
<thead>
<tr>
<th>Model</th>
<th>K</th>
<th>AIC</th>
<th>ΔAIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>success ~ abundance + survey order + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>4</td>
</tr>
<tr>
<td>success ~ abundance + arrangement + survey order + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ abundance + arrangement + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>4</td>
</tr>
<tr>
<td>success ~ abundance * arrangement + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>5</td>
</tr>
<tr>
<td>success ~ (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>2</td>
</tr>
<tr>
<td>success ~ survey order + (1</td>
<td>plot) + (1</td>
<td>observer)</td>
<td>3</td>
</tr>
</tbody>
</table>

**Table A5.1.2.** Parameters of the best-fitting model of *Allium cernuum* detection success (*n* = 53) as determined by AIC model evaluation (Table A5.1.1). Abundance was log transformed in all models.

<table>
<thead>
<tr>
<th>Parameter (units)</th>
<th>Standardized coefficient</th>
<th>Standardized standard error</th>
<th>p-value</th>
</tr>
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<tbody>
<tr>
<td>Intercept</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Abundance</td>
<td>2.30</td>
<td>0.93</td>
<td>0.01</td>
</tr>
<tr>
<td>Survey order</td>
<td>1.32</td>
<td>0.71</td>
<td>0.06</td>
</tr>
</tbody>
</table>
## APPENDIX 7.1 Location of 40 historic rare plant populations visited in the field in 2016 to determine persistence.

**Table A7.1.1.** Field locations surveyed in 2016 \((n = 40)\). EO_ID is the element occurrence ID assigned by ACIMS to recorded populations, we used these ID’s for field visits.

<table>
<thead>
<tr>
<th>EO_ID</th>
<th>Target species</th>
<th>S-Rank</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Habitat Class</th>
<th>Date of survey(s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>7307</td>
<td><em>Lactuca biennis</em></td>
<td>S3</td>
<td>54.586400</td>
<td>-110.453260</td>
<td>Upland</td>
<td>7/11/2016</td>
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<tr>
<td>9346</td>
<td><em>Nymphaea leibergii</em></td>
<td>S2</td>
<td>57.432360</td>
<td>-111.613460</td>
<td>Aquatic</td>
<td>8/21/2016</td>
</tr>
<tr>
<td>9347</td>
<td><em>Nymphaea leibergii</em></td>
<td>S2</td>
<td>57.419750</td>
<td>-111.559520</td>
<td>Aquatic</td>
<td>8/25/2016</td>
</tr>
<tr>
<td>9348</td>
<td><em>Nymphaea leibergii</em></td>
<td>S2</td>
<td>57.419210</td>
<td>-111.554990</td>
<td>Aquatic</td>
<td>8/25/2016</td>
</tr>
<tr>
<td>9349</td>
<td><em>Nymphaea leibergii</em></td>
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<td>-111.548700</td>
<td>Aquatic</td>
<td>8/25/2016</td>
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<tr>
<td>9568</td>
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<td>S2</td>
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<td>6/25/2016</td>
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<tr>
<td>10145</td>
<td><em>Potentilla bimundorum</em></td>
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<td>-111.634628</td>
<td>Disturbed</td>
<td>7/25/2016</td>
</tr>
<tr>
<td>12310</td>
<td><em>Carex vulpinoidea</em></td>
<td>S3</td>
<td>54.599339</td>
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<td>Disturbed</td>
<td>8, 13/07/2016</td>
</tr>
<tr>
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<td><em>Carex vulpinoidea</em></td>
<td>S3</td>
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<td>-110.503670</td>
<td>Disturbed</td>
<td>7/7/2016</td>
</tr>
<tr>
<td>12919</td>
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<td>S3</td>
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<td>-111.601275</td>
<td>Upland</td>
<td>8/20/2016</td>
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<tr>
<td>12921</td>
<td><em>Cypripedium acaule</em></td>
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<td>Upland</td>
<td>6/26/2016</td>
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<tr>
<td>13052</td>
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<td>Upland</td>
<td>7/20/2016</td>
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<tr>
<td>14087</td>
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<td>Aquatic</td>
<td>8/14/2016</td>
</tr>
<tr>
<td>15698</td>
<td><em>Sceptridium oneidense</em></td>
<td>S1</td>
<td>57.065161</td>
<td>-111.876823</td>
<td>Upland</td>
<td>26/07/2016</td>
</tr>
<tr>
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<td>7, 11/07/2016</td>
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<tr>
<td>1609</td>
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<td>8/22/2016</td>
</tr>
<tr>
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<td>8/26/2016</td>
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<tr>
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<td>8/26/2016</td>
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<tr>
<td>16998</td>
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<td>8/26/2016</td>
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<tr>
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<td>S3</td>
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<td>7/22/2016</td>
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</tr>
<tr>
<td>18593</td>
<td><em>Isoetes echinospora</em></td>
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<td>55.946318</td>
<td>-112.028961</td>
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<td>8/13/2016</td>
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<tr>
<td>18899</td>
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<td>6,13/07/2016</td>
</tr>
<tr>
<td>18901</td>
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<tr>
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<tr>
<td>19178</td>
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<tr>
<td>21514</td>
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<td>7/27/2016</td>
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<tr>
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<td>6/26/2016</td>
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<tr>
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<tr>
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<td>8/25/2016</td>
</tr>
<tr>
<td>21809</td>
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<td>56.434471</td>
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<td>8/24/2016</td>
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<tr>
<td>Site</td>
<td>Species</td>
<td>Status</td>
<td>Latitude</td>
<td>Longitude</td>
<td>Location</td>
<td>Date</td>
</tr>
<tr>
<td>------</td>
<td>--------------------------</td>
<td>--------</td>
<td>--------------</td>
<td>--------------</td>
<td>-----------</td>
<td>------------</td>
</tr>
<tr>
<td>21810</td>
<td><em>Phegopteris connectilis</em></td>
<td>S3</td>
<td>56.429970</td>
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<td>Upland</td>
<td>6/25/2016</td>
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<tr>
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<td>7/17/2016</td>
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<tr>
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<td>6/25/2016</td>
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<tr>
<td>22585</td>
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<td>7/22/2016</td>
</tr>
<tr>
<td>24360</td>
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<td>S3</td>
<td>53.742860</td>
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<td>7/14/2016</td>
</tr>
<tr>
<td>24362</td>
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<td>S3</td>
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<td>-110.760607</td>
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<td>8/6/2016</td>
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<tr>
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<td>10/12/2016</td>
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<td>24414</td>
<td><em>Houstonia longifolia</em></td>
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<td>8/6/2016</td>
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<tr>
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<td>Upland</td>
<td>8/20/2016</td>
</tr>
<tr>
<td>24443</td>
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<td>56.484033</td>
<td>-111.301773</td>
<td>Disturbed</td>
<td>8/28/2016</td>
</tr>
</tbody>
</table>

* Five *Najas flexilis* sites included in the ACIMS database occur around the shores of Ethel Lake, outside of Cold Lake, Alberta with no physical boundaries between sites (i.e. effectively one population). We report their locations here but have treated them as one population in all analyses.
APPENDIX 7.2 Analysis of survey time and target species abundance in rare plant population revisitation surveys.

Imperfect detection of organisms during surveys, particularly rare species, has gained significant attention in recent years (MacKenzie et al. 2005; Chen et al. 2009; Alexander et al. 2012; McCarthy et al. 2013). Population size, a factor shown to influence detectability (Alexander et al. 2012; McCarthy et al. 2013), varied widely among sites and target species at the 37 field sites discussed in Chapter 7. To better understand potential advantages to surveying large populations and the difference in effort expended to detect small vs. large populations in the field, we compared survey effort (time) and the population size of detected target species using linear regression. Both variables were log transformed prior to analysis to normalize variables.

Surveyors searched the target area exhaustively using time unlimited surveys at all 37 field sites. A maximum of 21 person hours occurred at one site, although in many cases ($n = 13$) the target species were detected shortly after starting surveys (‘detection upon arrival’). Median total search time when species were encountered was 0 minutes (range: 0-120, $\bar{x} = 26$), when species were absent median search time was 360 minutes (range: 10-1260, $\bar{x} = 401$). Where population sizes were small (< 30 individuals), total search effort required to detect species was at maximum 2 person hours, however up to 10.5 person hours were expended at non-detection sites to achieve reasonable confidence of absence. All populations > 30 individuals were detected upon arrival. Supporting our expectation, survey effort (time) was significantly negatively related to population size ($r^2 = 0.62, p = < 0.001$).

Our findings have practical application for future revisitation surveys. Logistically, surveys to confirm absence may require multiple days and target species which occur at low abundance require significant search time (e.g. 2 hours for a 50-m radius circle). Surveys reporting extirpations of small populations (as determined by the reported initial population size) must be accompanied by a metric of search effort to ensure confidence in findings. These data are now requested by ACIMS with public submissions; however, we suggest this be adopted as a standard by industry, consultants, and researchers.
APPENDIX 7.3. Visually classified amounts of human footprint (footprint severity) for 188 provincial rare plant records.

Table A7.3.1. Amount of footprint potentially impacting historic rare plant populations based on visual examination using Google Earth imagery within the oil sands area, Alberta ($n = 119$).

<table>
<thead>
<tr>
<th>Amount of human footprint</th>
<th>Number (%)</th>
<th>Species</th>
</tr>
</thead>
</table>
| High                      | 6 (5%)     | *Cardamine parviflora*  
|                           |            | *Gratiola neglecta*    |
|                           |            | *Polygaloides paucifolia* |
|                           |            | *Potentilla bimundorum* |
| Moderate                  | 15 (13%)   | *Blysmopsis rufa*       |
|                           |            | *Botrychium hesperium*   |
|                           |            | *Campanula aparainoides* |
|                           |            | *Carex vulpinoidea*     |
|                           |            | *Cypripedium acaule*    |
|                           |            | *Houstonia longifolia*  |
|                           |            | *Lactuca biennis*        |
|                           |            | *Malaxis paludosa*      |
|                           |            | *Plantago maritima*      |
|                           |            | *Potentilla bimundorum*  |
|                           |            | *Spiranthes lacera*     |
| Low                       | 90 (76%)   | *Arctagrostis latifolia ssp. arundinacea* |
|                           |            | *Arethusa bulbosa*      |
|                           |            | *Astragalus bodinii*     |
|                           |            | *Botrychium crenulatum*  |
|                           |            | *Botrychium matricariifolium* |
|                           |            | *Botrychium michiganense* |
|                           |            | *Carex oligosperma*     |
|                           |            | *Carex vulpinoidea*     |
|                           |            | *Cypripedium acaule*    |
|                           |            | *Dryopteris cristata*   |
|                           |            | *Elodea canadensis*     |
|                           |            | *Eutrema salsugineum*   |
|                           |            | *Gymnocarpium jessoense* |
|                           |            | *Houstonia longifolia*  |
|                           |            | *Isoetes echinospora*   |
|                           |            | *Lactuca biennis*       |
|                           |            | *Leucophysalis grandiflora* |
Malaxis paludosa  
Najas flexilis  
Nymphaea tetragona  
Pellaea glabella ssp. simplex  
Phegopteris connectilis  
Piptatherum canadense  
Polygaloides paucifolia  
Salix sitchensis  
Scirpus pallidus  
Spiranthes lacera  
Utricularia cornuta  
Utricularia ochroleuca

Table A7.3.2 Amount of footprint potentially impacting historic rare plant populations based on visual examination using Google Earth imagery within the surface mineable area, Alberta (n = 69).

<table>
<thead>
<tr>
<th>Amount of human footprint</th>
<th>Number (%)</th>
<th>Species</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>6 (9%)</td>
<td>Campanula aparinoides</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cypripedium acaule</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Malaxis paludosa</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Nymphaea tetragona</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Potentilla bimundorum</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Sceptridium oneidense</td>
</tr>
<tr>
<td>Moderate</td>
<td>9 (13%)</td>
<td>Cypripedium acaule</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Leucophysalis grandiflora</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Sceptridium oneidense</td>
</tr>
<tr>
<td>Low</td>
<td>53 (77%)</td>
<td>Campanula aparinoides</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Carex oligosperma</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cypripedium acaule</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cystopteris montana</td>
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<tr>
<td></td>
<td></td>
<td>Dryopteris cristata</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Elodea canadensis</td>
</tr>
<tr>
<td>ND</td>
<td>1 (1%)</td>
<td><em>Polygaloides paucifolia</em></td>
</tr>
<tr>
<td>-----</td>
<td>-------</td>
<td>--------------------------</td>
</tr>
</tbody>
</table>

*Epilobium halleananum*

*Gentianopsis detonsa ssp. raupii*

*Lathyrus palustris*

*Liparis loeselii*

*Malaxis paludosa*

*Nymphaea leibergii*

*Nymphaea tetragona*

*Plantago maritima*

*Sparganium glomeratum*

*Spartina pectinata*

*Spiranthes lacera*
APPENDIX 7.4 Presumed misidentification of three ACIMS recorded populations in the oil sands area.

Three of our 40 surveyed populations (7%) are strongly suspected to be misidentifications in the original records (false positives). These populations were recorded and submitted as *Dryopteris cristata* (S3, $n = 2$) and *Spiranthes lacera* (S2, $n = 1$). In the case of the two *D. cristata* records, a similar common species, *Dryopteris carthusiana*, was found in abundance at both survey locations. Both locations were searched for 6 person hours by our observers. *Dryopteris cristata* is distinguished from its congener by being mostly bipinnate, rather than bipinnate to tripinnate, and mostly lacking spinulose tips to the pinnules, key characters which can be challenging to recognize. We suspect that young individuals of *D. carthusiana* could have been mistaken for *D. cristata*. In the case of *S. lacera*, an orchid identified by its singular row of spirally arranged white flowers, site conditions did not match those associated with this species. *Spiranthes lacera* is almost exclusively found in dry, sandy, Jack pine-dominated forests in our region; site conditions at this location were moist mixedwood, dominated by *Picea glauca* and *Populus balsamifera*, with a mossy substrate. Four meters from the original record centroid we encountered a single *Goodyera repens*, a superficially similar species which, despite key differences, shares a white, loosely spiralled floral arrangement with *S. lacera*. We consider this to have been a misidentification based on the superficial similarity of this species, the location of the individual relative to the record centroid, and the habitat.