RECLAIMED UPLAND VEGETATION COMMUNITY TRENDS ON SYNCRUDE'S MINE SITES

1980 TO 2019

Syncrude

ABSTRACT

This report summarizes up to 39 years of plant community development trends on Syncrude's reclaimed mine sites near Fort McMurray, Alberta. These trends are contrasted with a target condition, defined here as the Natural Range of Variability for species composition on older (60+ yrs) closed canopy forests having similar mesic and sub-mesic site conditions within 200 km of the mine sites.

The primary outcome of the study is that patterns of plant community change on reclaimed sites are consistent with Alberta's objectives for reclamation, which require increasing similarity between reclaimed and reference plant community structure over time. Evidence is also provided demonstrating the strong influence of a developing tree canopy on these patterns, where native forest-dependent species gain an increasing competitive advantage over time as compared to early arriving ruderal or weedy species. Overall, it is concluded that expected natural processes, consistent with conventional ecological theory, are leading to reclaimed sites demonstrating substantial convergence with locally common boreal forest ecosystems.

CITATION

Farnden C. 2021. Reclaimed upland vegetation community trends on Syncrude's mine sites. Syncrude Canada Ltd, Edmonton, Alberta. 61 p.

DISCLAIMER

ACKNOWLEDGEMENTS

In compiling this report there were meaningful contributions from a large number of colleagues both from within Syncrude and externally. The most valuable of these resulted from conversations in the field, discussing what we were seeing and often modifying our world views to fit new observations. Those learning experiences extend not just back through my most recent career in the oil sands, but also through previous decades in the forestry sector. To all of those I have worked with, and particularly to those who I consider valued mentors, I offer my heartfelt gratitude.

Special thanks are offered to Dr. Ellen Macdonald from the University of Alberta who reviewed an earlier version of this document and provided a large number of insightful and valuable comments.

This report is dedicated to those who came before: the reclamation professionals who were passionate about their vocations and found ways to do things better. In particular, Earl Anderson saw the need for a vegetation monitoring program spanning well beyond his own career, that would mostly inform future generations rather his own. We are all grateful for his foresight.

EXECUTIVE SUMMARY

This report describes long term development trends for understorey plant communities on Syncrude's reclaimed upland forest sites following open pit oil sands mining. The objective is to test whether the species composition of these reclaimed communities is converging with those of locally common boreal forest ecosystems. The approach used is consistent with the regulatory requirement for demonstrating *that "…reclaimed areas are progressing in the appropriate trajectories to achieve the targeted reclamation outcomes and end land use objectives"* (AER 2019). This requirement inherently recognizes that plant communities play a critical role in the ability of reclaimed lands to support a diversity of end land uses at mine closure. As such, the degree of similarity for plant community composition between reclaimed and locally common reference plant communities can be used as an indicator of reclamation success.

Reclaimed plant community trends are derived from a set of long term monitoring plots on Syncrude's Mildred Lake mine site, located roughly 35 km north of Fort McMurray Alberta. At each sampling location, the presence and % cover of every observed species is recorded on a periodic basis, typically every five years. Sample plots have predominantly mesic and sub-mesic soil moisture regimes, and are roughly comparable to ecosite classes b and d within the Boreal Mixedwood Ecological Area as described by Beckingham and Archibald (1996). More than 600 discrete community level observations have been made on these plots, spread over 182 independent plot locations with a monitoring period of up to 39 years.

The future target condition for reclaimed plant communities is assumed in this report to be the natural range of variability (NRV) for plant community composition as measured in older (60+ years) closed canopy forests having similar site (edaphic) conditions as the reclaimed sites, and falling within a similar regional climate. Data for this reference condition comes from 84 independent sampling locations extracted from ECOSYS, a provincial database primarily used for developing ecosystem classification systems in Alberta.

While trends over time are the main focus of this report, a convenient starting point for comparing reclaimed and reference sites is simply species occurrence. In the reference plots, 193 native boreal species were detected, which become the Target Species for this report. One hundred and thirteen (or 58%) of those Target Species have been detected so far in the reclaimed plots. For the 80 Target Species not yet detected, only 22 were relatively common (>5% occurrence) in the reference plots. A full listing of all species contributing to the report, along with their percent occurrence on both reclaimed and reference plots, is provided in the appendices.

A subset of the Target Species that are particularly prevalent on the surrounding landscape are labeled Characteristic Species. Measures of occurrence for Characteristic Species on reclaimed lands have been proposed by other initiatives (e.g. Alberta Environment 2010) as having a role in (i) reclamation certification and (ii) the identification of reclaimed ecosites. As such, it is expected that these species in particular will readily immigrate to and thrive on reclaimed sites. So far, 43 of 48 relevant Characteristic Species (or just under 90%) have been detected in the reclaimed plots.

Reclaimed areas also contain species that were not found in the reference plots. These include 100 that are native to the local boreal forest. Possible reasons for their absence from the reference plots include (i) the species are more typical of younger forest conditions having less developed tree canopies and/or (ii) they are more typical of alternate edaphic conditions (wetter or drier sites). The reclaimed plots also contain 99 species that are not native to local boreal forests. Most of these appear to be species that are adapted to rapidly invading denuded lands, and would typically be considered as ruderal species or "weeds". Others are agronomic or horticultural species (i.e. alfalfa, caragana and various grasses) that were intentionally introduced on reclaimed sites as cover crops to assist in erosion control or to facilitate development of desired soil properties.

There were no detections of *Prohibited Noxious Weeds* as defined under the Alberta Weed Act and associated Regulations.

Moving to results for the main objective of this study, trends of convergence with locally common boreal ecosystems are illustrated in summary Figure S-1 (with additional detail and metrics in the main report). For plot level species richness on the left side of this figure, it appears that there is a relatively slow start to the initial appearance of Target Species, particularly for older reclaimed areas. This is followed later by an increased rate of new Target Species detections. In the most recent assessments, the best performing plots from each reclamation decade have Target Species richness values equivalent to the mean value for the reference data.

An alternate metric for evaluating plant community convergence is provided on the right side of Figure S-1. The Bray-Curtis index contrasts species composition between pairs of plots, with values scaled between zero (no similarity) and 1 (perfect similarity). Key differences between these trends and those for Target Species richness include (i) the consideration of all species present (including weeds) and (ii) the weighting of species counts based on their abundance (% ground cover). Overall, the trends toward convergence appear to be muted using this index as compared to using Target Species richness. This is expected, given that index values are penalized for the presence of ruderal or weedy species that were not considered in the first set of trends. In order for a larger degree of apparent convergence to occur using this metric, there will need to be further reductions in the occurrence and cover of non-target species.

It has always been expected that reductions in non-target species will be triggered by tree canopy development. The trends of increasing similarity between reclaimed and reference communities noted earlier are entirely consistent with this expectation. Direct evidence for this effect is provided in Figure S-2. As trees on reclaimed sites continue to grow and expand their crowns, they have multiple effects on the understorey growing environment including reductions in light levels, decreased availability of water and nutrients, and dampening of micro-climatic extremes. These effects shift the competitive advantage in the understorey away from the early invading species that are favoured by open canopy

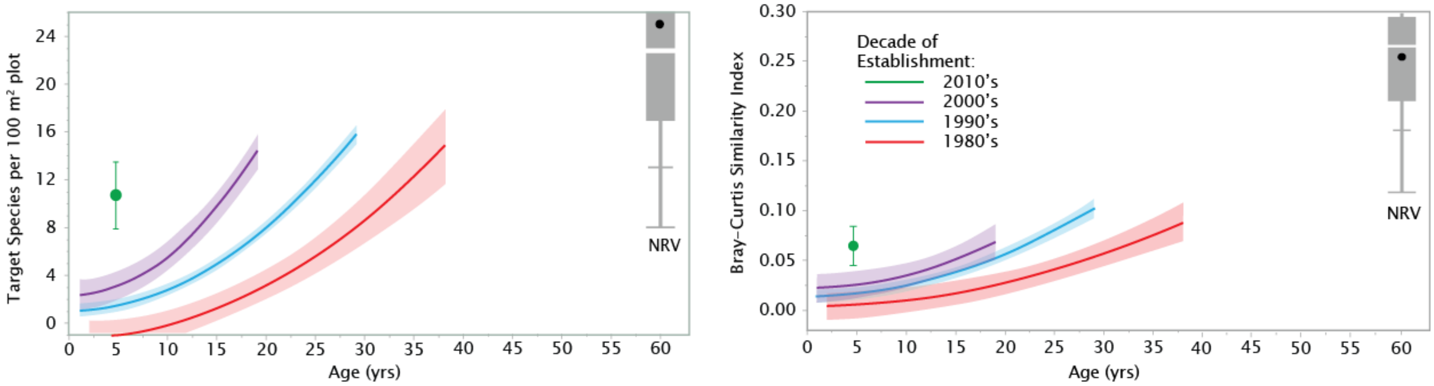

Figure S-1. Mean trends by age for (i) plot level Target Species richness on reclaimed sites in comparison to comparable values from d ecosite reference plots(left) and (ii) plot level similarity between reclaimed plots and reference plots on d ecosites using the Bray-Curtis index (right). Shaded bands indicated 95% confidence limits. Trends are parsed by decade of reclamation, with only the oldest reclaimed sites reaching the oldest observed ages. For reclamation from the 2010's, there is not yet sufficient data to interpret a trend, so only a single mean value is shown, arbitrarily positioned at the mid-point age of 5 years. In the box plot charts for the Natural Range of Variability (NRV) on right side of each chart, the black dot is the mean value for all reference plots, the white line through the grey box is the median, the upper and lower limits of the grey box are the 25th and 75th percentiles, the end of the lower wisker is the minimum value, and the line crossing the whisker partway along its length is the 10th

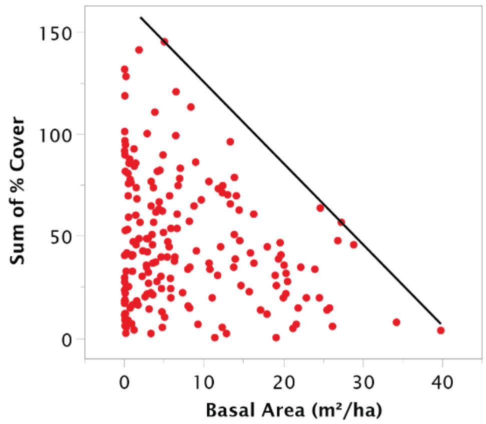

Figure S-2. Trends in summed cover for non-target species as related to basal area. Percent cover values are summed from species-level estimates and may exceed 100% due to vertical overlap. Basal area is the sum of the cross sectional area of all tree stems on a unit of land, and is a reasonable proxy in this case for canopy density or shading of the understorey (i.e. a higher basal area equates to (i) a greater number of trees, (ii) a similar number of larger trees or (iii) both).. The black boundary line estimates the maximum level of summed non-target species cover for a given level of tree cover.

conditions, to those species that are better adapted to compete effectively for resources under a closed forest canopy. The Target Species identified for this study are assumed to be well adapted to a closed forest canopy, which explains both the presence of those species in the reference data, and the increasing similarity of reclaimed sites to the reference NRV as the reclaimed sites move toward a closed canopy condition.

Two case studies which demonstrate variations in overall patterns of convergence are provided in Figure S-3. The photo in Figure S-3a illustrates a case where a relatively high degree of similarity has been achieved quite early, with 10 of 11 species that are easily identified in the photo being Target Species. The single non-target species, dandelion, is the only exotic species identified in the reference data set. Figure S-3b highlights a case of disparity in occurrence between reclaimed and reference sites for a particular species. Bunchberry occurs on a very high percentage of reference sites, often with high cover. On reclaimed sites, this species continues to be relatively uncommon, but patches that have

a b

Figure S-3. Examples of reclaimed understorey communities. The photo on the left highlights a community developing below a closed canopy jack pine stand, growing on 50 cm of peat-mineral mix soil over tailings sand, 24 years after establishment. Species readily evident in the photo include bristly black currant, fireweed, common red raspberry, wild strawberry, creamy vetchling, American purple-vetch, bearberry, Lindley's aster, knight's plume moss, blue-joint reedgrass and dandelion (see Appendix 1 for scientific names). The photo on the right is an 18-year old reclaimed site under an aspen canopy, again on tailings sand with a similar cover soil. The species with the red berries is bunchberry, a common species in local ecosystems that continues to be uncommon on reclaimed sites.

become established appear to be thriving and, at least in some cases, spreading rapidly. This species, like some mosses and lichens, appears to be a late arriver, and it is speculated here that it may require the modified growing environment under a tree canopy in order to gain a foothold on reclaimed sites.

Beyond these primary findings there are a number of other observations and conclusions for which supporting data and discussions are provided in the main body of the report:

1. The initial pace at which new Target Species are observed on reclaimed sites (species per year), and the rate of increase in that pace, appears to be higher for newer reclaimed areas than for older ones (as represented by "decade of reclamation"). While multiple explanations may exist, the most likely appears to be a move away from intensive agronomic approaches to reclamation, to practices which better conserve plant propagules in reclamation soils and better facilitate native species immigration.
2. With the possible (and speculated) exception of seeded and fertilized agronomic cover crops, there is no evidence that ruderal species and particularly noxious weeds are negatively impacting immigration rates of Target Species on reclaimed sites.
3. Consistent with many forested communities, there may be some degree of interruption to the rates of forest dependent species arrival and proliferation on reclaimed sites as they reach canopy closure and undergo a period of maximum canopy density. In many cases, the % cover of all understorey species will be temporarily reduced, including that for Target Species.
4. Trends observed in this study provide validation to the concept of using Characteristic Species counts, measured at a relatively early age, as an indicator of potential Target Species diversity within a Criteria and Indicators framework for reclamation certification. However, analyses to validate the proposed thresholds for this indicator have not yet been completed.
5. At the current stage of development, Characteristic Species and reclaimed community composition appear to be relatively weak indicators for discriminating between b and d ecosites. It is speculated that exposure to a wider range of seral stages (and possibly even disturbance cycles) over many decades will be required in order for Characteristic Species to become stronger indicators of ecosite on reclaimed lands.

While it is already shown in this report that upland reclaimed communities demonstrate promising patterns of convergence towards their reference counterparts, results presented here are only the beginning of possible learnings from Syncrude's long term vegetation monitoring program. None of these analyses can be considered as definitive, they simply add to the body of previous evidence. The degree to which further analyses using additional techniques or additional years of monitoring data will be needed depends on any one person or group's particular questions and degree of confidence in the existing evidence. Where the demands for refinement to this knowledge base justify the effort, there remains further potential to inform incremental improvements to reclamation practice and polices.

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1 INTRODUCTION

This report describes reclaimed plant communities and their patterns of change over time for upland landforms on Syncrude's mine sites near Fort McMurray, Alberta. The general intent is to evaluate the degree to which their species compositions are converging toward those typical of comparable growing sites on the surrounding un-mined landscape.

The expectation for mine site reclamation in Alberta, as expressed in Acts and Regulations, is to return the land to a condition of Equivalent Land Capability (Alberta 2014 a,b).This goal is interpreted through regulatory documents (i.e. operating approvals, guidelines and standards) that place a strong emphasis on reclaiming to a forested landscape that supports a diversity of end land uses comparable to (but not necessarily the same as) the pre-mining condition. The return of appropriate plant communities to reclaimed lands is a critical element supporting Equivalent Land Capability.

To this end, Syncrude is required to demonstrate through regulatory reporting that *"…reclaimed areas are progressing in the appropriate trajectories to achieve the targeted reclamation outcomes and end land use objectives" (AER 2019)*. As part of documenting and understanding such trends, the objectives of this study are to:

1. evaluate the degree to which discrete locations on reclaimed lands have understorey plant communities that are becoming increasingly similar over time to locations within regional reference landscapes,
2. make observations on the degree to which reclamation outcomes are predictable based on conventional ecological theory, and
3. discuss how observations of existing community trends can contribute to continuous improvement of reclamation practices.

The concept of convergence as expressed in regulatory documents is interpreted here as an explicit recognition of, and expectation for, ecological succession. In simple terms, ecological succession can be described as predictable patterns (and variability) of change in plant community composition that occur over time. For example, after a wildfire in the boreal forest, plant community composition will be noticeably different at years 10, 30, 50 and 70. While there can be considerable variability in the details, the gross patterns are well understood. Distinctly recognizable steps in this transition, characterized both by the plant community makeup and the associated environmental conditions, are called seral stages. There is no distinct end point to this process.

Seral stages have a substantial impact on the localized and periodic suitability of land relative to any given land use. For example, some wildlife species may be highly dependent for habitat on specific seral stages, while others may require different seral stages for different aspects of their habitat (i.e. winter thermal cover versus summer grazing). While it is the cumulative contributions of many plant communities and their seral stages over time and space that are truly meaningful for a full suite of land uses, this report is restricted to only the trends for individual sites as opposed to the larger landscape variability.

Except where otherwise noted, (i) the emphasis within this report is on understorey communities, or those that would typically be observed either in the absence of trees or beneath a canopy of trees, and (ii) tree species have been excluded from the analyses.

1.1 RECLAMATION OF UPLAND ECOSYSTEMS

The functional goal of reclamation practice is to set the initial conditions required to support desired patterns of plant community development based on natural processes. The active stages of reclamation include construction of post-mining landforms, reclamation soil placement and revegetation practices.

The process of oil sands mining completely disrupts pre-mining landscapes and the ecosystems that they support, and some important differences are evident in the mine closure topography and landform substrates. Where pre-mining landscapes are characterized by minimal surface relief, the closure landscape has a greater influence of hills and valleys with upland positions that are more isolated from the regional groundwater table. Prior to mining, upland ecosystems occur primarily on landforms consisting of Pleistocene glacial till, glaciolacustrine and glaciofluvial parent materials, while closure upland landforms are primarily constructed from Cretaceous (marine) sediments (overburden) and tailings sand.

Reclamation soils have two key roles: (i) they provide a suitable growing medium for the reclaimed ecosystems and (ii) they isolate the reclaimed ecosystem from any adverse physical properties or chemical constituents that may be present in the landform substrates. Surface soil materials generally consist of either peat (and potentially a portion of the underlying mineral soil) salvaged from native bogs and fens, or upland forest floor material (LFH) and a portion of the underlying A and potentially B horizons) salvaged from native upland forests. Subsoils material (where used) consists of mineral soil layers and parent material of suitable chemical quality that remain after surface soil salvage. Whenever possible, reclamation soils are salvaged and directly placed in a location that is ready for reclamation. However, storing reclamation soils in stockpiles for extended periods prior to their eventual placement is commonly necessary.

Revegetation of reclaimed lands relies on three processes: (i) the germination or sprouting of native species from propagules (seeds or vegetative fragments) contained in reclamation soils, (ii) the immigration of native species through natural processes such as seed transfer by wind or animals, and (iii) planting of particularly critical species. The focus of planting programs is on (i) trees, which have a profound effect on the growing conditions for other species and (ii) berry producing shrubs, which are particularly important for several end land uses following mine closure.

Reclamation practices are continuously improved over time. Some key historic changes include:

1. a transition from agronomic approaches to soil preparation, such as discing and harrowing, toward an approach that minimizes soil manipulation and disturbance and leaves a rough soil surface,
2. the elimination in most cases of broadcast fertilizer applications,
3. a transition from the use of mixed and persistent agronomic cover crops intended to control erosion, to non-persistent crops such as barley, and eventually to no cover crops at all in most cases,
4. the elimination of plantings of non-native trees and shrubs, and
5. a transition from monoculture or two-species tree planting prescriptions to an increased emphasis on multi-species mixes.

These changes, targeted at improved reclamation outcomes including development of native plant communities, have resulted from learnings based on both intensive research trials and observations by reclamation practitioners. Results from this report are expected to further inform this continuing evolution.

1.2 EXPECTATIONS FOR PLANT COMMUNITY DEVELOPMENT

In the absence of long term empirical studies for reclaimed plant community development, we can set the stage for expected patterns of community development following a simple conceptual model[1] (Figure 1). The starting condition for this model is a stand replacing disturbance, where all or a substantial portion of the tree canopy is removed, resulting in a near-ground growing environment for plant communities that is dramatically altered. The removal of the tree layer often provides increased access to resources for plants (particularly sunlight, but also soil moisture and nutrients), but also exposes understorey species to stresses not present under the protection of a canopy, to which some may not be physiologically or genetically adapted. Specific to the boreal forest context of this report are stresses associated with extremes of temperature and particularly growing season frosts, exposure to UV radiation and overall reductions in humidity but with increased variation.

In a general sense, early establishment of plant species after canopy removal is largely dictated by (i) biological legacies such as whole plants, plant fragments, or propagules that survive the disturbance, (ii) the presence of environmental conditions favourable to (or stresses unfavourable to) those survivors and propagules and (iii) early immigration of ruderal species, or those best suited to rapid colonization of disturbed land. The relative influence of these factors is not fixed, but can vary considerably. For example:

1. community membership in the early years will typically be a mix of multi-seral generalist species and ruderals; in cases where the legacy propagule bank is substantially depleted following a natural disturbance, or contains a substantial portion of species that are poorly adapted to the post-disturbance growing conditions, there will be greater opportunities for early seral specialists,
2. where the surviving propagule bank remains strong (as with most natural disturbances and following timber harvesting in the boreal forest), and the component species are capable of effectively competing under the post-disturbance environmental conditions, the legacy (late seral) species will start out with a relatively high presence and diversity, and may substantially dampen the initial peak of ruderal species,
3. some early successional species can be aggressive competitors on a site and may resist displacement by surviving native species or new colonists; many sod forming grass species are of particular note in this category, and
4. the availability of propagules for immigrating species (whether ruderal or forest-dependent) will be impacted by many factors including source distance and spread vectors.

The other major (and possibly greater) influence on the progression toward a late seral plant community is the re-development of a tree canopy. It is the trees and their moderating effect on the understorey growing environment that provide late seral specialist species with their greatest competitive advantage. As the tree canopy develops, it modifies the understorey growing environment including availability of resources and exposure to micro-climatic stresses. This in turn provides opportunities for species that are best adapted to the new conditions to get established and compete for resources.

The rate at which tree species re-establish after a disturbance has a major impact on the entire pattern and rate of understorey community development. Where trees become re-established early (such as with the suckering of trembling aspen following a wild fire), the trees may re-assert their influence quite early, such that stages A and B in Figure 1 might be completed in as little as a decade (Pinno et al 2001). Where the re-establishment of a tree canopy is delayed or sporadic, these stages can persist for many decades.

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[1] Synthesized from concepts and results described by Su et al. 2019, Dhar et al. 2018, Chang and HilleRisLambers (2016), Li et al. (2016), Pulsford et al. (2016), Valladeres et al. (2016), Shoo et al. (2015), Hart and Chen (2006, 2008), Brockenhoff et al. (2003), Kimmins (2003), Strong (2000), De Grandpré and Bergeron (1997), Oliver and Larson (1996), Stelfox (1995), De Grandpré et al. (1993), and Corns and La Roi (1976).

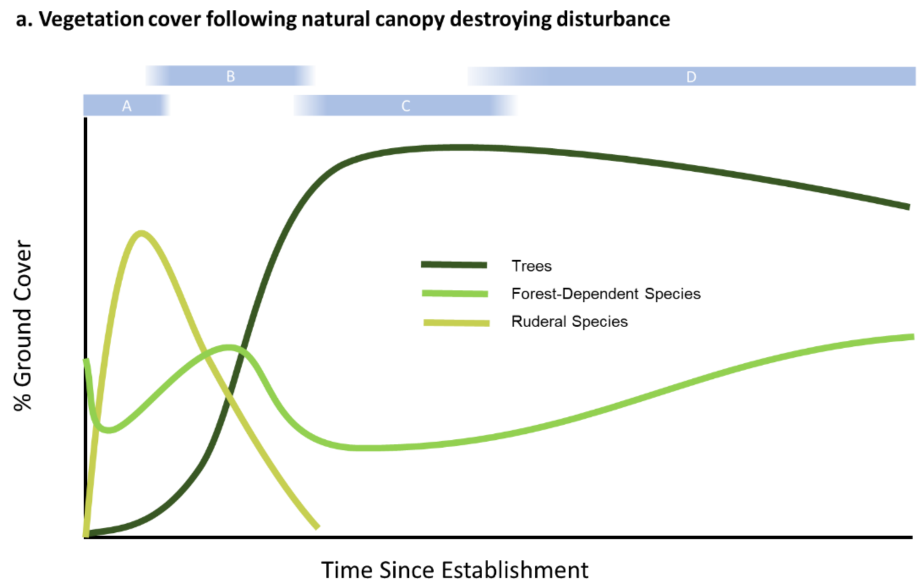

Figure 1. Conceptual model of community development following a stand-destroying disturbance. Plant community components include trees (dark green), ruderal or pioneer species (yellow-green) and late seral or forest-dependent species (light green). In the earliest years (stage A), ground cover is influenced by rapid colonization of ruderal or pioneer species, tempered by those legacy late seral species that are able to persist. As the tree cover starts to express influence (B), the growing environment is increasingly moderated by the developing forest canopy. Light levels at the ground reach their lowest levels at the earliest ages for which full tree canopy leaf area is achieved (C), a point at which the vigour and cover of even shade tolerant late seral understorey species will be impacted. As the canopy grows taller, a process of canopy opening starts to occur where individual trees increasingly abrade against each other in the wind and stochastic mortality of individual large trees opens gaps in the canopy. At this stage (D), light levels at the ground become both generally higher and increasingly varied, with understorey growing conditions favouring development of vegetation communities characteristic of later seral forests.

While the conceptual model in Figure 1 is focused on abundance (ground cover) of the different species groups, the preceding discussion demonstrates that they also provide context for understanding patterns in site-level species diversity. Following a natural disturbance such as fire, there is expected to be a decline in forest-dependent species richness immediately after the disturbance, as any species dependent on the protected environment under a tree canopy will have reduced vigour and may possibly disappear altogether. However, in many cases this may be only a small portion of the pre-disturbance community. The remaining legacy species have sufficient ecological tolerance to persist through stages A and B, possibly decline somewhat in stage C, and recover in stage D.

For reclamation following surface mining, the conceptual model in Figure 1 does not start immediately after the disturbance, but at a later date with the commencement of reclamation activities. Successful reclamation must mitigate the loss of those legacies, with a heavy reliance on natural processes for a wide diversity of species to return.

Throughout the remainder of this report, attempts will be made to contrast observed patterns of reclaimed plant community development with this conceptual model. It will be highlighted where observed patterns are consistent with these models, where they are different, and what those consistencies or inconsistencies imply for long term convergence of reclaimed plant communities with those from the target condition of locally common late seral ecosystems.

2 HISTORICAL ANALYSES OF RECLAIMED PLANT COMMUNITY TRENDS

For the mineable oil sands region of Alberta, there are several earlier studies of reclaimed community development trends upon which this study can build. Geographic Dynamics Corp (2006) used (i) data from the 2001 through 2004 re-measurements on the same monitoring program reported here (but with no trends based on earlier years of measurements) and (ii) a second set of data provided by Suncor Energy Inc from a nearby mine, to evaluate patterns of native species ingress. Other studies used data from the Long Term Plot Network (LTPN), a monitoring system that is in many ways comparable to that used in this study, but with fewer reclaimed plots having fewer re-measurements. Several analyses of vegetation community trends were completed as part of annual reporting for that program (including Paragon and AXYS 2008, Stantec 2009 and Stantec 2011), and two others were completed separately by other authors (Pinno and Hawkes 2015, Dhar et al 2020).

The analysis by Geographic Dynamics Corp (2006) concluded that ingress of native species was minimal within the first decade, with the exception of a relatively small group that were quite successful including aspen (*Populous tremuloides*), red raspberry (*Rubus idaeus*), Saskatoon berry (*Amelanchier alnifolia*), some willows (*Salix spp*), wild strawberry (*Fragaria virginiana*), asters and most native grasses. Notable absences included black spruce (*Picea mariana*), low-bush cranberry (*Viburnum edule*), twinflower (*Linnaea borealis*), bog cranberry (*Vaccinium vitis-idaea*), snowberry (*Symphorocarpos albus*), wild sarsaparilla (*Aralia nudicaulis*), wild lily-of-the-valley (*Maianthemum canadense*), tall lungwort (*Mertensia paniculata*), bishop's cap (*Mitella nuda*), dewberry (*Rubus pedatus*) and ferns. Shrub species in particular were noted as being slow to appear through ingress. Moving into the second decade, there was early evidence of a transition characterized by decreasing cover of ruderal species and an increasing presence of native species typically associated with locally common forested conditions.

The annual reporting studies cited above based on LTPN data used (i) comparisons of plant community richness, species or species group abundance, Shannon-Weiner diversity, and Shannon evenness, and (ii) non-metric scaling of Sørensen distances (indices of community dissimilarity) to draw conclusions on plant community convergence. In general, these analyses demonstrated:

1. increasing similarity between reclaimed and reference plots with increasing reclamation age,
2. a general increase in both richness and % cover for desirable species with increasing age on reclaimed land, and
3. inconsistent separation of plant community composition between dry and moist reclaimed sites.

Pinno and Hawkes (2015) binned discrete plot observations from the LTPN data by age class, site class (dry versus moist) and canopy species for parametric comparisons based on total species richness, non-native richness, percent cover by growth type (forbs, graminoids, shrubs and non-native species) and tree height. Key findings included:

1. few differences in plant community metrics between dry and moist reclaimed sites,
2. maximum total forb cover on reclaimed sites is reached at age 5, followed by a steady decline
3. non-native species cover decreases from 50 to 60% at age 5 to less than 20% at age 20,
4. a trend for increasing shrub cover with increasing reclamation age, but at a slow rate and with the highest levels achieved still being well below that for reference sites, and
5. better tree growth on moist as compared to dry reclaimed sites.

Dhar et al (2020) followed an approach similar to that of Pinno and Hawkes for parametric comparisons, but binned by combinations of reclaimed landform and soils rather than canopy species. They also added multivariate community composition analyses using non-metric multidimensional scaling (NMDS; distance metric not specified) and principle components analysis (PCA). Beyond general agreement with previous studies regarding richness and cover changes over time, this study found that:

1. species composition is quite different in the early stages than would be found following natural disturbances in boreal forests, yet still shows evidence of typical early successional progress (indicated by changes in predominance of plant functional types), and
2. community assembly on reclaimed sites appears to be driven primarily by random processes, suggesting that existing communities are not strongly inhibiting colonization by new species having equivalent tolerances to the current growing environment.

While all of these studies either explicitly or implicitly used a late seral vegetation community as a target condition for reclamation, there were no comprehensive models proposed for the transitions other than general allusions to ecological succession. Several of the papers explicitly recognized the expected impacts of a tree canopy and one discussed how canopy species may or may not have an influence. In a comparable study from a coal mine in western Alberta, Strong (2000) hypothesized that a dense overstorey is necessary "…to create forest communities that botanically approximate natural stands of a similar age".

In general, these preceding studies were limited in their interpretation of trends and screening of potential causal influences by the size of the reclaimed data set, and in particular by the low number of repeated measurements. Time (age) effects were heavily dependent on a chronosequence approach which assumes that discrete observations at different ages were otherwise equivalent. Missing is consideration of possible underlying changes over time in (i) reclamation practices or (ii) the frequency distributions of site factors such as landforms and landform substrates, not captured in the binning process, that may have had an impact on outcomes.

Beyond extensive monitoring programs, there are additional descriptions of plant community trends from controlled experiments studying reclamation practices:

- Multiple studies have investigated the impacts of reclamation soil materials and placement profiles:
    - Several authors (e.g. Mackenzie and Naeth 2010, Mackenzie et al 2012, Archibald 2014, Forsch 2014, Macdonald et al 2015, Melnik 2017, Jones and Landhäusser 2018) have investigated the beneficial impacts of directly placed forest floor material (top 10 to 30 cm of salvaged native soils) for vegetation community development. General findings as compared to other soil materials include superior colonization by desirable native species and an inhibitory effect on non-native species. Some studies noted that the initial flush of native species was partially lost from years 2 through 5 particularly on drier sites, but that the overall effect was still superior.
    - Rowland (2008) studied the impact of various reclamation soil profiles on vegetation development, and ranked prescriptions on their relative ability to support acceptable convergence using a chronosequence approach with similar assumptions and limitations to those from studies based on LTPN data above. Nutrient limitations were identified as a potential cause of less desirable community trends.
    - The intentional creation of surface roughness as a reclamation treatment (Lesko 1974, Melnik 2017) has demonstrated beneficial impacts for recruitment of native vegetation, presumably through both microsite diversification for in situ propagules and increased seed capture.
- Coarse woody debris as a surface treatment has been noted to have a beneficial impact on early native species recruitment by Brown and Naeth (2014), Pinno and Das Gupta (2018) and Forsch (2014), although the magnitude of responses have been variable.

In addressing the question of whether reclaimed plant communities are converging with those of locally common boreal ecosystems, the cumulative body of work suggests it to be the case. However, the evidence may be less than completely convincing because (i) the number of cases for older reclamation is small and (ii) there has been limited ability to explore the breadth of variation in reclaimed conditions, with sufficient replication, to draw truly meaningful and robust conclusions. It is intended that the data and results presented here, from a mostly independent data set, will add considerably to the cumulative body of evidence and understanding of vegetation community development patterns on reclaimed oil sands mines.

3 FRAMEWORK OF COMPARISON

There is no single description or metric of plant community composition that can be used to define a target condition: the natural conditions to which we wish to compare are themselves highly variable. Similarly, there is no simple or singular threshold for determining that reclaimed systems have achieved convergence with the desired target condition. The alternative approach used here is to identify a comparable reference target, or range of conditions from local landscapes that can be used as aspirational goals for reclamation. The reference landscape is then sampled to define the Natural Range of Variability (NRV). Where any one metric for a reclaimed site shows a consistent trend toward the range values from the reference plots, we can say there is an early pattern of convergence. Where one or more reclaimed plots cross one or more boundary conditions defining the NRV, we start to accumulate evidence for actually achieving convergence. We would gain increasing confidence in a claim of convergence with (i) an increasing portion of reclaimed sites entering the NRV for an increasing number of metrics and (ii) the mean values for any given metric on reclaimed sites approaching the mean value for the same metric from reference sites.

The concept of making comparisons to a natural range of variability is not new to this study, and for oil sands reclamation it was introduced in a preliminary fashion by GDC and FORRX (2008). However, those authors explored only a static concept of complete sameness, where the frequency distribution of values for a reclaimed site would be contrasted with a reference distribution from natural sites using a Chi squared test. The approach taken in this study is instead intended to explore trends toward that sameness. It does this by characterizing the frequency distribution for a reference condition and assessing whether there is a consistent pattern of increasing similarity between reclaimed sites and the reference distribution.

This report utilizes (i) three metrics of desirability for plant community composition based on species richness and cover, and (ii) an index based on plant community similarity. For each of these, an NRV is described based on the frequency distribution of values from the reference plots (or each possible pair of reference plots for similarity indices).

3.1 REFERENCE CONDITION AND DATA

As previously stated, the regulatory framework for mine site reclamation sets an expectation for achieving locally common boreal forest ecosystems where the plant community structure is converging with that of native sites. Building on this concept, the criteria selected for selecting reference sites included:

i. seral conditions typical of Stage D in the Figure 1 conceptual model (referred to in this report as “late seral” conditions)
ii. a similar regional climate based on the Central Mixedwoods ecological sub-region class (sensu Alberta Parks 2015),
iii. similar edaphic conditions based on “b” and “d” ecosite classes for the Boreal Mixedwood ecological area (sensu Beckingham and Archibald (1996)), and
iv. occurring within 200 km of the Syncrude mine site.

The definition of a reference condition for this project is more restrictive than for some other studies of reclaimed plant community trends (or recommendations for such). For example, GDC and FORRX Consulting (2008) combined data from a wider range of ecological sub-regions, with no distance restrictions and including all seral stages for their reference condition. Alternatively, Hawkes et al. (2012) and Alberta Environment (2010) envision a trend x trend comparison, where a progression of reclaimed seral stages is contrasted with a progression of reference seral stages. It is suggested here that a single (late) seral stage provides the most appropriate target for comparative purposes, given that:

i. the seral stage trajectories will inevitably be different for reclaimed and reference ecosystems given their very different starting conditions, and in particular the complete loss in biological legacies from the reclaimed sites,
ii. a single seral stage with restrictions on locality provides the most precise possible target condition, allowing for the easiest and most meaningful contrasts,
iii. late seral plant communities are widely considered to be those that best reflect the regional climate and are minimally influenced in their variability by a wide range of (often stochastic) environmental conditions associated with partial or complete forest canopy loss, and
iv. a late seral stage is a point at which the plant communities on sites with a wide range of starting conditions might be reasonably expected to demonstrate a high degree of convergence, barring adverse and undesirable environmental changes that would preclude such convergence.

In determining the ecosites for an appropriate reference condition for this study, it was recognized or assumed that the reclaimed sites being evaluated:

i. are almost exclusively on upland locations having soil moisture regimes not substantially enhanced by the presence of seepage water,
ii. have sufficient soil water storage capacity that they cannot be classed as having xeric or subxeric soil moisture regimes, and
iii. have been impacted by regulated soil placement practices that largely exclude the possibility of ‘very poor’ or ‘poor’ soil nutrient regimes.

The net impact of these factors is that all or a vast majority of monitored locations have soil moisture and soil nutrient regimes that exclude all but b or d ecosites as defined by Beckingham and Archibald (1996).

A data set from reference sites appropriate for comparison was extracted from the Government of Alberta’s Ecological Information System (ECOSYS) database. This system of ecological description plots, which is the basis for the ecosystem classification in the province, contains descriptions of plant community composition predominantly of a similar type[2] used in Syncrude’s monitoring system, allowing for direct comparisons. In screening for appropriate seral stages, it was assumed that plots must be labeled with at least one of:

i. a Successional Status of Maturing Seral, Old Seral, Mature Edaphic Climax or Mature Climatic Climax,
ii. a stand age of >= 60 years, or
iii. overstorey cover and height values consistent with old forest conditions.

The net result is a set of 84 reference plots, with 17 on (submesic) b ecosites and 67 on (mesic) d ecosites.

In applying the ECOSYS data for the purposes of this study, it is worth noting that:

- Each plot in the reference data set is assumed to be an independent sample or site.
- The plots were predominantly intended to be 100 m^2 in size, but a rigorous measurement of plot boundaries was not always applied. Some plots consist of a string of smaller sub-plots arranged along a transect, which have been combined here into a single tally.
- The degree of effort to tally a complete species list or to identify uncommon species or species that are notoriously difficult to identify (e.g. members of the genus Salix) is not documented and was likely inconsistent across a number of different sampling programs contributing to the data set. Geographics Dynamics Corp and FORRX Consulting (2008) noted evidence that tallies of lichen and moss species in particular appeared to vary in rigour within this data.

[2] Both the Syncrude monitoring plots and the sample plots used for ecosystem classification tally the presence and cover of all species within a fixed area plot. The Syncrude plots are most commonly 400 m^2 in area, while the ecosystem classification plots are more commonly 100 m^2 in area. An adjustment factor (described later) is applied to values for species richness to account for plot size differences.

- Sampling cannot be assumed to be completely random and this introduces the possibility of bias. It is suggested here that this risk is quite small, particularly given that data is being used for a general characterization of the NRV rather than being used to make precise statistical comparisons.

3.2 RECLAIMED DATA

Data describing reclaimed communities in this study have been collected as part of Syncrude's long term vegetation monitoring program (Syncrude 2021). This program uses of a set of 20 m x 20 m vegetation sample plots (one per site) which are periodically re-assessed to detect changes over time. Each plot combines a set of individual tree measurements following the traditions of forestry growth and yield studies along with a quadrat style assessment of plant community composition commonly used in ecological studies. For the growth and yield assessments, key metrics include species, stem diameter and height for every tree in the plot. For the total vegetation community there is a listing of % ground cover for each species present. For the majority of assessments, the vegetation community has been assessed over the entire 400 m^2 area of the plot.

The earliest plots were established in 1980, with more being added over time as newly reclaimed areas have been created. There are vegetation community records for 198 plot locations, but only 123 of those are currently considered as active; many have been lost to ongoing industrial activity or other disturbances such as highway construction. The intended re-assessment interval is 5 years, but the actual interval has been quite variable. As of 2019, the oldest surviving plot is 39 years old and has had 10 assessments of the tree measures and 8 for the plant community. This report utilizes data from 629 discrete plot assessments spanning up to 39 years.

As with the reference data, there are additional characteristics of the data set that should be recognized including:

- each plot in the reclaimed data set is considered to be an independent sample or site,
- the expertise of field crews has varied over time, with Syncrude staff and summer students completing most of the assessments prior to 2010, and specialized contract crews completing the assessments after 2010; this change may have impacted:
    - the expertise and degree of effort to tally a complete species list, and
    - the degree to which effort was made to accurately identify both *genus* and *species* of uncommon species or species that are notoriously difficult to identify (e.g. many mosses, sedges and willows)
- sampling is not completely random.

Reclaimed plots dominated by the occurrence of either Caragana (*Caragana arborescens*) or Siberian larch (*Larix sibirica*) were excluded from the reclaimed data set (16 of 198 plots). These introduced species are already accepted as having profound effects on plant community development, and any areas planted to these species must be excluded from the general conclusions of this report.

3.3 COMPARATIVE SPECIES CLASSES

The analysis framework for this report uses a set of interpretive species classes to facilitate meaningful comparisons between the community makeup of the reclaimed and reference plots. The classes used in this report reflect relative desirability of species as long term elements of reclaimed plant communities:

Target Species: Forest understorey species that are expected to occur locally in late seral ecosystems under similar edaphic conditions as the reclaimed plots. Target Species were defined as all species that both (i) occurred in the reference data set comprised of plots from late seral b and d ecosites and (ii) were identified as native species in the Alberta Conservation Information Management System (ACIMS) database. Given that there was only trivial occurrence of non-native species in the reference data, "Target Species" and membership in the "reference data" are essentially synonymous for understorey species.

Characteristic Species: A set of species expected to occur with high prominence and/or frequency for b and/or d ecosites as defined and listed by Alberta Environment (2010) in their Table 3-1. All non-tree Characteristic Species are also Target Species in this analysis, and where they are not specifically identified as separate classes in this report, all non-tree Characteristic Species are considered as part of the Target Species class. The concept of Characteristic Species as applied in this report originated with Beckingham and Archibald (1996) as descriptor species for various ecosystem classes (ecosites). The concept was later adapted for use in guidance documents for oil sands reclamation (e.g. Alberta Environment 2010), where a threshold count of Characteristic Species is proposed as a reclamation certification metric within a Criteria and Indicators framework.

Other Boreal Species: Species that are not included in “Target Species” but both (i) occur in forested plots from the Central Mixedwoods natural subregion in the ECOSYS database, and (ii) are classed as “Native” in the ACIMS database.

Other Native Species: Species classed as native to Alberta in the ACIMS database, but are not found in the full ECOSYS data for the Central Mixedwoods natural subregion.

Exotic Species: Species identified in the ACIMS database as not being native to Alberta.

These classes are used throughout the report. All species from both the reference and reclaimed data sets are listed in Appendix 1.

4 ANALYTICAL OVERVIEW

4.1 SPECIES OCCURRENCE AND COVER

The first section of results in this report addresses contrasts of species occurrence and cover. These measures on their own provide only a very limited understanding of plant community composition and convergence with reclamation targets, but provide important background perspective.

Evaluations of species occurrence (total listing or count of species present) start with a contrast of all species present in reclaimed plots versus those present in reference plots (Section 6.1). This contrast is intended to provide a perspective on (i) the degree to which the full diversity of Target Species is arriving on reclaimed sites and (ii) the degree to which non-target species continue to play a role on reclaimed sites. Subsequent evaluations then move to (i) trends over time for individual Characteristic Species (Section 6.2) and (ii) plot level trends in both Characteristic Species and Target Species richness (sections 6.3 and 6.4). The occurrence of individual Characteristic Species is intended to demonstrate which highly prevalent (and most indicative) species from the surrounding natural landscape are consistently appearing on reclaimed sites, and which of those are continuing to lag. The plot level richness trends are intended to provide a first level of insight into the degree to which community convergence (reclaimed to reference) is being achieved.

Beyond species occurrence, evaluations of percent ground cover (prominence) trends provide additional evidence of convergence (Section 6.5), where ideally we would observe the prominence of Target Species increasing relative to that of non-target species. This starts with a contrast of species groups (Target, Other Boreal, Exotic etc) by reclaimed age for the entire data set, then moves to plot level trends for Target Species cover. Given that there is a specific concern for continued occurrence of non-target species and particularly noxious weeds on reclaimed sites, the Target Species trends are followed by cover trends for non-native species (Section 6.6). A final sub-section (6.7) related to percent cover looks at the impacts of a tree canopy on understory vegetation. Here the focus is on determining the validity of assumptions related to understorey community dynamics as impacted by the transition from Stage B to Stage C in the Figure 1 conceptual model. This section, though brief, is considered as critical to highlighting the potential influence of a tree canopy on understorey communities, and particularly the assumed decline of ruderal species with replacement by late seral or Target Species.

4.2 INDICES OF SIMILARITY

Critical to making conclusions regarding reclaimed plant community convergence with the reference NRV are metrics of similarity or dissimilarity based on contrasts of community elements (species). Such techniques are often used in the field of ecology (Grieg-Smith 1983, Wolda 1981), and a wide range of techniques exist. With repeated comparisons over time, it can be illustrated whether or not there are recognizable and defensible patterns of increasing similarity (convergence) between the reclaimed and reference communities (Section 7.0).

The Bray-Curtis similarity index used in this report evaluates the overlap in species composition between pairs of plots, with the results scaled from zero (no similarity) to one (complete similarity). The amount of ground cover by species that is common to both plots will increase the index value, while any level of cover by species in one plot that exceeds the level in the other plot is considered to be a difference and will reduce the index value.

In early phases of this work, several different similarity indices were initially employed as the first step in a multivariate exploration of the data sets (i.e. non-metric multidimensional scaling). While these initial explorations demonstrated that there are many potential paths to follow using multivariate techniques in subsequent studies of these data sets for varying sets of questions, it was determined that such methods were beyond the scope of the current study. These early explorations also strongly indicated that employing multiple similarity indices did not add substantially to the weight of evidence, and only the Bray-Curtis index was retained.

4.3 EFFECTS OF ECOSITE ON COMMUNITY COMPOSITION

One of the expectations for plant community trends as expressed in Alberta Environment (2010) is that plant communities will be trending toward those of ecosystem classes or ecosites (sensu Beckingham and Archibald 1996) having similar edaphic conditions. While not a primary objective of this particular study, a preliminary evaluation of the degree to which this expectation is being met is addressed in Section 8.0.

4.4 ADDITIONAL OBSERVATIONS

In addition to the detailed analyses of plant community trends based on long term monitoring presented in Sections 6 through 8, some additional observations related to plant community outcomes are provided in Section 9. These are less quantitative in nature, and are intended to provide some further breadth to the total perspective on the diversity of reclamation outcomes. Section 9.1 provides some observations on plant community diversity at various scales that are not necessarily captured within the plot data. Section 9.2 presents outcomes resulting from distinct reclamation practices that, although obsolete, continue to have a large influence on their respective portions of the reclaimed landscape and are not aligned with current reclamation objectives. The specific practices in question relate to plantings of two exotic species: caragana and Siberian larch.

5 ANALYTICAL DETAILS

5.1 DATA PREPARATION

In preparation for analyses described in this report, a number of steps were required to prepare data from both the reclaimed and reference data sets:

- The data sets were screened for duplicate records.
- Where data records had species identified using a 7-letter (or other) code, all codes were updated to standardized codes based on scientific nomenclature used in the Alberta Conservation Information Management System (ACIMS), current as of September 2019.
- The data sets were screened for species codes with no known species match. Unknown codes were reconciled based on (i) searching online botanical databases for possible matches, and (ii) identifying high probability typographic errors. Two remaining cases of unknowns in the reclaimed data (out of 10,578 records) had one observation each with less than 1% cover, and were ignored.
- Observations within the reclaimed data of plants identified only to the genus or family level (1477 records), or to generic designations such as “grass” or “moss” (699 records) could not be used universally in all analyses. Key assumptions and implications include:
    - None of these records could be included in tallies of species richness or in calculations of similarity indices
    - For the calculation of percent cover by comparative species groups, these records were probabilistically assigned to comparative species groups based frequencies of occurrence for equivalent records identified to the species level. As an example, if there were 58 records having a species level ID within the genera *Elymus*, and 20 of those were for “Exotic” species and 38 were for “Target Species”, then 34.5% (20/58) of the records identified only to the genera *Elymus* would be assigned as “Exotic”, and 64.5% would be assigned as “Target”.
- As mentioned in Section 1, records of tree species were not including in most analyses. The exceptions are:
    - for listings and occurrence levels of “Characteristic Species”, where tree species are part of a pre-defined list, and
    - for analyses of tree cover impacts on understorey communities.

Initial data preparation and calculation of metrics such as similarity indices was completed within Microsoft Excel 2013..

5.2 REPRESENTATIONS OF THE NATURAL RANGE OF VARIABILITY FOR REFERENCE SITES

For any given measure of community composition or contrast of community similarity to other locations, discrete samples for the reference landscape can have a wide range of values. Rather than comparing just to a mean value for the reference conditions, it is useful to document the range of values that are observe, and the frequency of different values within that range. A common manner of displaying such information is using a bar chart of frequency values, but this is awkward for comparative purposes within this report. An alternate approach to displaying the same information is using box plots, which (i) are more compact, (ii) maintain most of the value of portraying the shape of a frequency distribution as do bar charts, and (iii) more explicitly display important statistics such as the mean, mode and selected quantiles (Figure 2).

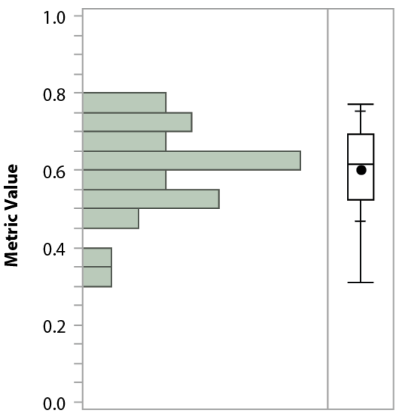

Figure 2. Portrayal of the NRV as box plots.
The bar chart on the left portrays a set of data value and their relative frequencies for an example NRV. The same range and relative frequencies of data values are portrayed in the box plot on the right of the figure, where 50% of the data values fall within the central box (the ends of which are the 25th and 75th percentiles), 80% of the data falls within the intermediate ticks along the whiskers on either side of the box (the 10th and 90th percentiles), and the ends of the whiskers represent the minimum and maximum values. Where the 10th or 90th percentiles are not shown, they are indistinguishable from the maximum or minimum values. The mean value is the black dot, and the median is the line bisecting the box.

5.3 STATISTICAL TESTS

All statistical tests and models in this analysis have been completed using JMP software (version 15) produced by SAS Institute Inc.

For the purposes of fitting mean trends over time to various measures of community development, a mixed modeling approach was employed to account for the lack of independence related to repeated measures on the same subject (plot). Time step was used for the repeated measure, plot was the random variable, and time plus categorical covariates were the fixed variable(s). All trend lines employ a curve fit based on a simple transformation of age squared. This approach universally provided better fit statistics than a straight line. Other more complex model forms provided better fits to specific cases, but none consistently. Where significant differences in slope were detected for subsets of the data, a Tukey multiple comparison approach was used to evaluate differences in slope between individual pairs of curves.

5.4 SAMPLE SIZE AND PLOT SIZE ADJUSTMENTS

In making comparisons between the reference and reclaimed data sets, it is important to recognize that the likelihood of detecting any given species in either of the data sets will increase with the number of plots. For this reason, the reclaimed data set (182 plots) is at an advantage for overall species detections as compared to the reference data set (84 plots). This would be a critical factor if direct statistical comparisons of numerical equivalence were being made, but that is not the case. Still, it is useful to the reader to understand the potential magnitude of this issue. For this reason, a resampling

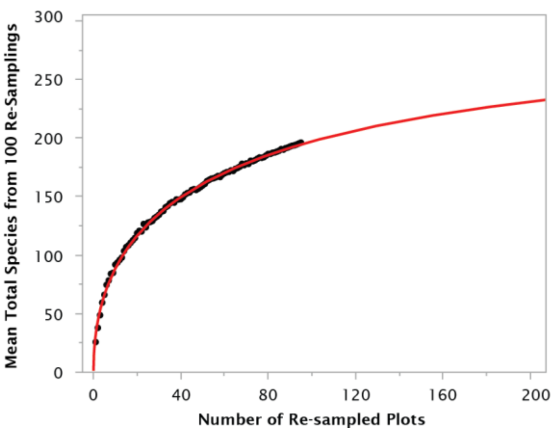

Figure 3. Prediction of total number of Target Species detections for the Reference data assuming a larger sample size closer to that for the reclaimed data set.
Black data points represent the mean number of total species detected in the reference data assuming 100 random sub-sets of size "x" from the total number of available plots. The red line is a non-linear fit using a Weibull model.

exercise was conducted where 100 random selections of *n* plots were made from the reference data set, for all values of n between 2 and 84 plots. For each value of n, the mean total number of species detected was determined, and a non-linear Weibull model was fit to the result (Figure 3). Extrapolating from this model, while somewhat of a stretch, provides an rough estimate of the total number of Target Species that might have been detected in the reference data if there had been the same number of plots as for the reclaimed data.

It is also recognized that the smaller plot sizes used in the reference data set (100 m^2 as compared to 400 m^2 in the reclaimed data) will have a similar effect on total species detections, but a comparable approach to estimating the magnitude of the effect is not available as it was for plot numbers. While plot size has almost double the impact on land area observed for detecting species than does plot numbers in this study, the overall impact is expected to be of similar or smaller magnitude, as the likelihood of detecting new species is larger on a completely independent sample area that on one that is immediately adjacent to an area already tallied. Overall, however, it is a near certainty that there would be at least some further additional species detections in the reference data if plot size were increased at the same time as plot numbers. However, it an absolute certainty that all of the additional detections would be for species with very low rates of occurrence in the reference data.

The same issue based on plot size also applies to plot level estimates of species richness (mean number of species per plot). Where plot level richness comparisons are being made between reclaimed and reference data, or where there are existing standards or expectations for species richness based on 100 m^2 plots, the larger plot used in the reclaimed data will overestimate reclamation accomplishments relative to those expectations. In order to correct for this effect, reclaimed plot measurements in 2017 through 2019 were assessed for both the full 20 x 20 m plot but also for each 10 x 10 m quadrant individually. Correlations between species counts for each 20 x 20 plot and a mean value for the corresponding quadrants allows for an unbiased adjustment of historical data from the larger plots to comparable values for the smaller plot size (Figure 4).

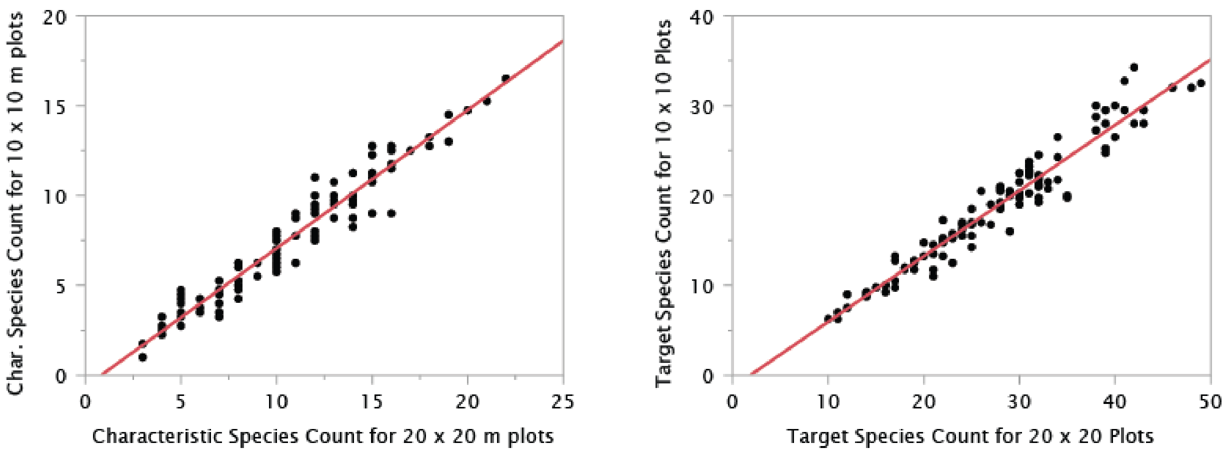

Figure 4. Correlation between species counts for 20 x 20 m plots versus those for 10 x 10 m quadrants of the same plots. Relationships are illustrated for Characteristic Species on the left and Target Species on the right. Each data point represents a single 20 x 20 plot, with the dependent (y) variable being the mean value for the four 10 x 10 quadrants within each plot.

5.5 SIMILARITY INDICES

Values for the Bray-Curtis similarity index were calculated using custom Visual Basic routines within the Microsoft Excel software package. All routines were validated using dummy data sets with known outcomes. Similarity values for each reclaimed plot represent the mean similarity to all reference plots. Similarity values for each reference plot, as used to generate the NRV, were determined by calculating the plot's similarity to all other reference plots, and again taking the mean.

$$\text{Bray Curtis Index} = \frac{2a}{b+c}$$

where:

a = the sum of all species level % cover that is common to both plots
b = the sum of all species level % cover values in plot 1
c = the sum of all species level % cover values in plot 2

5.7 APPROXIMATING RECLAIMED ECOSITE

Classifying ecosystems on reclaimed sites in the same manner as for reference sites, following the system outlined by Beckingham and Archibald (1996), is difficult and imprecise. Many of the required indicators used for making class determinations are either no longer present on a reclaimed site (i.e. soil diagnostic horizons) or have altered interpretive value. While ecosite classification can still be done, there is a much higher likelihood of classification error.

For the purposes of this study, we are interested in distinguishing between only two ecosites which make up the large majority of upland sites: b and d. Given that (i) the primary distinction between these two classes is moisture regime and (ii) groundwater and/or seepage is not a distinguishing factor for each of these classes, it is assumed that soil available water holding capacity (AWHC) is a reasonable proxy for distinguishing these classes (see Alberta Environment 2006 and Pojar et al. 1985 for precedents). AWHC for the top 100 cm was calculated following the methods of Saxton and Rawls (2006) using detailed data from soil pits at each plot, and a value of 90 mm/m was assumed as a class boundary. While this threshold has not been validated in any way for accurately representing the boundary between b and d ecosites, it should serve as a reasonable separator between relatively dry and relatively moist plots for the purposes of this study.

The range of AWHC values observed was 42 to 136 mm/m. Sites at the lower end of the range tend to occur on sites with coarse textured overburden or tailings sand landform substrates, and either coarse textured or very thin fine textured surface soils. Sites at the higher end of the range tend to have finer textured and thicker surface soils. High organic content (e.g. peat) in surface soils has the effect of increasing AWHC.

6 RESULTS - SPECIES RICHNESS AND COVER

6.1 TOTAL SPECIES OCCURRENCE

Across all plots and ages, 312 species were detected in the 182 reclaimed plots used in the study (Figure 5) as compared to 194 species in the 84 reference plots. However, adjusting for the smaller number of plots in the reference data (see Section 5.4) suggests that roughly 25 additional species might have been detected if the same number of plots had been available as in the reclaimed data. Some further number, likely of somewhat smaller magnitude, would have been detected if the reference plots had been the same larger size as the reclaimed plots.

In the reclaimed plots, 113 of the detected species are Target Species, or common to the reference plots. Another 144 are native to Alberta but are not typically found in older seral conditions for b and d ecosites. This suggests either (i) they are requisite early seral species that typically disappear in older forests, (ii) they are sufficiently uncommon in the reference ecosystems that they were never detected, (iii) they are off-site local species not typically found on b & d ecosites but instead under moister or drier edaphic conditions, or (iv) they are invaders from other climatic regions in the province.

A complete list of vegetation species observed in the study is provided in Appendix 1, including rates of occurrence for each species in both the reference data and the reclaimed plots. The vast majority of 80 Target Species from the reference data not yet detected in reclaimed areas are uncommon in the reference data, with 36 of them occurring in only

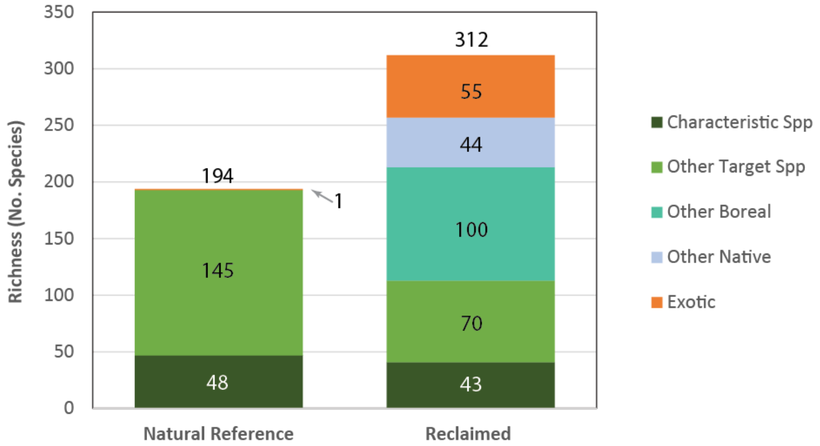

Figure 5. Comparison of total species richness for reclaimed and reference sites.
Reference data represents old seral conditions within 200 km of the Syncrude mine site on b & d ecosites (sensu Beckingham and Archibald 1996) and is drawn from a provincial database of ecosystem description plots. Richness values for each plant species class are listed within or beside each bar, with totals for each column at the top. For the Characteristic subset of Target Species in this tally, members of the genus Salix (willows) are considered independently where they are lumped together in Table 1. Note that species numbers in the reference data are likely underestimated relative to those in the reclaimed plots based on sampling differences (see discussion at start of this section).

a single reference plot and only 22 occurring on more than 5% of reference plots. Eighty four percent of species occurring in more than 10% of reference plots also occur in reclaimed plots. However, despite their simple presence in the reclaimed data set, there are some large differences in rates of occurrence between reference and reclaimed plots for particular species. Some notable examples include bunchberry (*Cornus canadensis*; 82% versus 4%), twinflower (*Linnaea borealis*; 79% versus 2%), lowbush cranberry (*Viburnum edule*; 79% versus 2%), dewberry (*Rubus pubescens*; 62% versus 8%) and bishop's cap (*Mitella nuda*; 56% versus 1%).

6.2 CHARACTERISTIC SPECIES – TRENDS IN OCCURRENCE

Characteristic Species are native plants from the natural landscape that have high values for frequency and/or prominence for a particular ecosite (GDC and FORRX 2008, Alberta Environment 2010). As listed in Table 1, there are 46 Characteristic Species defined for b and d ecosites together (differing slightly from the 48 reported in Figure 5 where willows were considered separately by species but have been grouped here following proposed regulatory reporting protocols), and 41 of those (89%) have been observed in Syncrude's vegetation monitoring plots under similar edaphic conditions.

For those species that are absent or uncommon in the reclaimed plots, it was initially suspected that they may be dependent on old seral forest conditions. However, there are no remarkable differences for these species in occurrence or mean cover between old seral forests as represented in the reference data versus in other plots from the same data source representing younger development stages (data not presented).

In evaluating and learning from these data, it is equally or of greater importance to look at temporal trends relative to age or stages of ecosystem development as opposed to just total numbers of species detected. Several such temporal patterns are evident in Table 1:

- Species such as wild red raspberry (*Rubus idaeus*; Figure 6a), wild strawberry (*Fragaria virginiana*; Figure 6b), common fireweed (*Chamerion angustifolium*) and the willows (*Salix spp*) appear early and in substantial numbers, and continue to have a high presence through all age classes. In general these are species that (i) are not particularly sensitive to seral stage and (ii) have reproductive strategies that allow them to rapidly invade reclaimed land. In many cases, it is assumed that these species arrive embedded in the reclamation soil either as banked seeds or vegetative fragments that can sprout. Many of these species also exhibit a wide ecological amplitude (they occur naturally across many ecosites), such that their propagules are likely to occur in directly placed reclamation soils regardless of the ecological source.

- Species such as wild lily-of-the-valley (*Maianthemum canadense*), common pink wintergreen (*Pyrola asarifolia*) and stair-step moss (*Hylocomium splendens*) start slowly, but increase in presence over time. Possible causes for this pattern may include:

    a. they do not typically have viable propagules contained in reclamation soils
    b. their seeds (or spores) are primarily spread by animals that are not attracted to the earliest stages of reclamation,
    c. they do not compete effectively with dense ruderal ground cover that is common in the first few years on many reclaimed sites,
    d. they require the modified environment that is provided by a forest canopy either for (i) the presence of suitable conditions for initial establishment, (ii) the elimination of competition from vigorous early seral vegetation, or (iii) protection from periodic micro-climate extremes such as growing season frosts, or
    e. any combination of the above.

- Species such as bunchberry (Figure 6c), common blueberry (*Vaccinium myrtilloides*), and *Ribes* spp. (Figure 6d) show up somewhat sporadically in a low percentage of plots. This pattern of occurrence suggests that they are inherently capable of colonizing reclaimed land regardless of seral stage, but that other factors are hampering their appearance. Possible (but untested) influences include:

Table 1. Occurrence of Characteristic Species for b and d ecosites by age class in reclaimed plots.

Dot size in the table is a visual representation of increasing frequency of occurrence in plots. Species with no dots in any age class have not been detected. All Salix species have been combined due to uncertainty in species identification. Frequency of occurrence in the reference plots is listed by ecosite in the two columns on the right. The number of plots having observations in each age class is represented by the value of n.

Percent of plots	
· <1% (non-zero)	● 5 to 25%
• 1 to 5%	⬤ > 25% of plots

			Age Class								Reference	
			0-4	5-9	10-14	15-19	20-24	25-30	30-34	35-39	b	d
		n=	200	134	126	127	64	38	8	8	17	67
Trees	*Abies balsamea*	balsam fir									12%	29%
	Betula papyrifera	white birch	•	•	•	●	●	●		●	41%	44%
	Picea glauca	white spruce	●	●	●	⬤	⬤	⬤	⬤	⬤	58%	82%
	Picea mariana	black spruce									47%	8%
	Pinus banksiana	jack pine	•		●	●	●	●	⬤	⬤	53%	6%
	Populus balsamifera	balsam poplar	•	●	●	⬤	⬤	⬤	⬤	⬤	6%	21%
	Populus tremuloides	aspen	●	●	⬤	⬤	⬤	⬤	⬤	⬤	100%	94%
Shrubs	*Alnus viridis*	green alder	•	•	•	•	•	•	●		29%	12%
	Amelanchier alnifolia	saskatoon	•	•	•	●	⬤	⬤	⬤	⬤	6%	26%
	Arctostaphylos uva-ursi	common bearberry	·	•	•	●	●	●	●	⬤	35%	4%
	Cornus canadensis	bunchberry	•		·		•	●		⬤	77%	82%
	Corylus cornuta	beaked hazelnut									0%	13%
	Linnaea borealis	twinflower					•		●	●	76%	78%
	Rhododendron groenlandicum	common Labrador tea	•	•	•	•	•				65%	21%
	Ribes americanum	wild black currant		·		·			●		0%	0%
	Ribes lacustre	bristly black currant	·	•	·	·	•	●	●	●	0%	10%
	Ribes oxyacanthoides	northern gooseberry	•	•	•	●	●	●		●	0%	7%
	Ribes triste	wild redcurrant	·	•		·	•	●			6%	22%
	Rosa acicularis	prickly rose	•	●	●	⬤	⬤	⬤	⬤	⬤	35%	68%
	Rubus idaeus	wild red raspberry	●	⬤	⬤	⬤	⬤	⬤	⬤	⬤	0%	24%
	Rubus pubescens	dewberry	●	●	•	•				●	18%	72%
	Salix species	willows	●	⬤	⬤	⬤	⬤	⬤	⬤	⬤	42%	21%
	Shepherdia canadensis	Canada buffaloberry	•	●	●	●	⬤	⬤	⬤	⬤	29%	28%
	Symphoricarpos albus	snowberry	•	·		•	•	•		●	6%	12%
	Vaccinium myrtilloides	common blueberry	•	·	•	•	•				94%	25%
	Vaccinium vitis-idaea	bog cranberry	•	·		·	•				1%	31%
	Viburnum edule	low-bush cranberry				·		•	●	●	41%	87%
Forbs	*Aralia nudicaulis*	wild sarsaparilla		·					●		35%	46%
	Chamerion angustifolium	common fireweed	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤	77%	72%
	Eurybia conspicua	showy aster	•	●	●	⬤	●	●	⬤		6%	12%
	Fragaria virginiana	wild strawberry	●	●	●	⬤	⬤	⬤	⬤	⬤	41%	49%
	Galium triflorum	sweet-scented bedstraw	•		•	•	•	●	●		0%	16%
	Lathyrus ochroleucus	cream-colored vetchling	●	●	●	●	⬤	●	⬤	●	59%	46%
	Maianthemum canadense	wild lily-of-the-valley	·		·	·	•	●	⬤	⬤	65%	54%
	Mertensia paniculata	tall lungwort	•	●	•	•	•	•			6%	50%
	Mitella nuda	bishops-cap	·				•				24%	63%
	Petasites frigidus	palmate-leaved coltsfoot	●	•	●	●	●	●			41%	68%
	Pyrola asarifolia	common pink wintergreen			•	●	⬤	●	⬤	⬤	18%	51%
Grasses	*Calamagrostis canadensis*	bluejoint	●	●	●	●	⬤	⬤	⬤	⬤	29%	59%
	Leymus innovatus	hairy wildrye	•	•	·	•	•	●	●		65%	34%
Mosses/ Lichens	*Cladonia mitis*	green/yellow reindeer lichen	·		·	•	•		●		59%	6%
	Cladonia rangiferina	grey reindeer lichen									6%	0%
	Cladonia stellaris	northern reindeer lichen									12%	1%
	Hylocomium splendens	stair-step moss				•	⬤	⬤	⬤	⬤	71%	76%
	Pleurozium schreberi	big red stem/Schreber's moss	·		·	●	●	•	⬤	⬤	88%	81%
	Ptilium crista-castrensis	knights plume moss					●	•	⬤	●	18%	41%

Figure 6. Characteristic species observed on Syncrude's reclaimed sites include the commonly occurring (a) strawberry (Fragaria virginiana) and (b) wild raspberry (Rubus idaeus), and the less frequent (c) bunchberry (Cornus canadensis) and (d) wild redcurrant (Ribes triste).

a. infrequent occurrences of suitable seedbed conditions, including physical factors influenced by weather such as soil surface temperature and moisture,
b. high levels of seed predation,
c. infrequent availability of appropriate transport vectors (i.e. animals),
d. infrequent seed crops (mast years),
e. seed transfer vectors best suited to spread over limited distances,
f. high but incomplete mortality related to competition by ruderal species, or any combination of these or other factors.

While there are five Characteristic Species from Table 1 that have not yet been tallied at all in the reclaimed monitoring plots, this does not necessarily mean they are totally absent from reclaimed lands. Two of the five (black spruce or *Picea mariana,* and grey reindeer lichen or *Cladonia rangiferina*) have been detected in sampling conducted for other purposes.

Expanding beyond Table 1, there are also three species of willow which are listed as Characteristic but were not observed in the reclaimed plots (pussy willow or *S. discolor,* Drummond's willow *or S. drummondiana* and myrtle-leaved willow *or S. myrtillifolia*). However, they also have a low or zero frequency of occurrence in the reference data for this study (1.2%, 0% and 0% respectively).

6.3 CHARACTERISTIC SPECIES - TRENDS IN PLOT-LEVEL RICHNESS

Moving from individual species to the community level, we can first look at plot level richness for Characteristic Species. A chart illustrating all plots at all ages is provided in Figure 7. Overall, there is a statistically significant trend of increasing richness of Characteristics Species over time, but with considerable variability in the data that can be accounted for using additional factors. Much of this variability has been accounted for in Figure 8, where the data from Figure 7 has been subdivided using a statistically significant (p<0.05) categorical variable representing decade of establishment. No other variables (including landform, soil material, soil depth, soil chemical properties, slope, aspect or ecosite) were able to explain comparable amounts of variability on a consistent basis. As a result this stratification of the data by decade of reclamation establishment has been carried through all subsequent analyses in the study. An interpretive discussion of this variable is provided later in Section 10.2.

Also in Figure 8, all reclaimed plots have been compared separately to reference plots for b and d ecosites (dry-poor versus moist rich site types). Overall, it appears that reclaimed sites are converging more closely with the target communities for d ecosites than for b ecosites.

Given that tallies of Characteristic Species have been recommended as one of several regulatory indicators for reclamation success and reclamation certification (Alberta Environment 2010, Table 5-3), the relevant minimum thresholds have also been added to Figure 8. While not a perfect basis for comparison here[3], it is notable that a relatively high number of plots have crossed the proposed certification threshold, with newer reclamation passing at an earlier age than older reclaimed sites. The consistent trends of increasing numbers of Characteristic Species being achieved beyond the intended assessment age range of 11 to 20 years lends support to the concept that an early assessment of Characteristic Species for the purposes of reclamation certification is defensible.

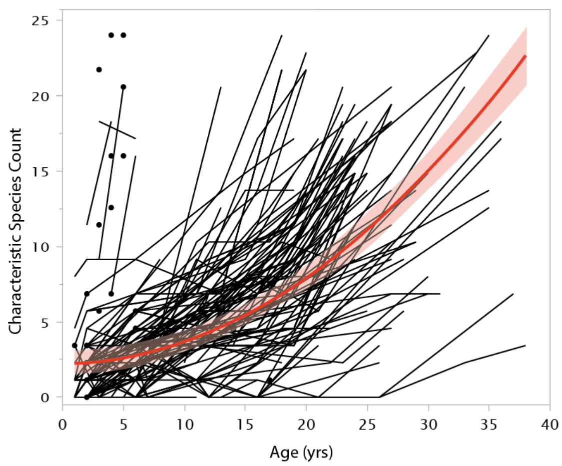

Figure 7. Trend in Characteristic Species richness (moist-rich site type) for all reclaimed plots. Each black line links all assessment ages for a single plot. Dots represent plots having a single measurement. The overall mean trend (red line with shaded confidence interval), based on the equation $y=a+bx^2$, is statistically significant (p <0.05).

[3] The regulatory thresholds proposed by Alberta Environment (2010) are drawn from a similar set of reference data (with some overlap) as that used in this study. As the source of an alternate reference metric, it should be recognized that the regulatory thresholds are based on plots from a wider range of seral conditions and a much larger geographic area (including plots from the Province of Saskatchewan). Beyond these differences, it is less than desirable as the source of a reference metric for this study given that a mean and standard deviation are insufficient information to generate even a crude approximation of the frequency distribution for individual sample values (the NRV). It is primarily provided here given that many readers will be already familiar with these values as a basis for comparison.

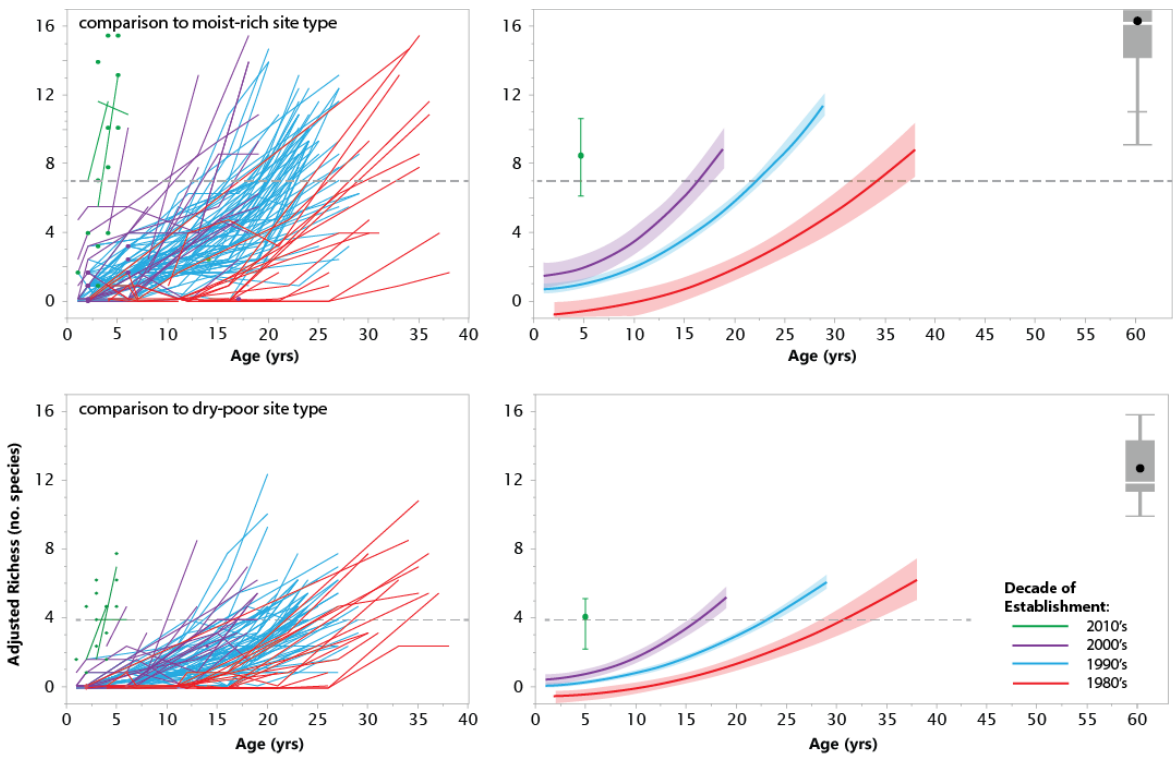

Figure 8. Trends in Characteristic Species richness per plot by reclamation age.
Data values have been scaled to estimate what would have been detected in smaller 100 m^2 sample plots comparable to those used for the reference data. Reclaimed plots are compared separately to targets and NRV's for moist d ecosites on the top and drier b ecosites on the bottom. Reclaimed data is colour coded by decade of reclamation establishment (1980's through 2010's). Solid lines on the left represent single plots over time and dots represent plots with a single observation. The box plots (defined in Section 5.2) on the right side of each chart represent the NRV from the reference data. The grey dashed horizontal lines represent the Characteristic Species minimum thresholds proposed for reclamation certification from Table 5-3 in Alberta Environment (2010), with values for dry-poor site types (encompassing b ecosites) applied to the bottom charts and those for moist-rich site types (encompassing d ecosites) in the top charts. Trend lines (solid lines on the right along with shaded 95% confidence intervals) for each decade of establishment, based on the equation $y=a+bx^2$, each had significantly different slopes based on a post-hoc Tukey multiple comparisons test. No trend line was determined for the youngest subset (2010's) where many plots still have only a single observation, but a mean value and confidence interval is provided instead.

In comparing temporal trends for richness in reclaimed plots versus the reference plots in Figure 8, it appears that the best performing reclaimed plots are just now approaching the lower bounds of the NRV for b ecosites, and have exceeded the lower bounds for d ecosites. While the mean values for reclaimed sites are still below the ecosite specific means for the reference plots, this is not unexpected coming from a starting condition where Characteristic Species were largely absent at year zero.

6.4 TARGET SPECIES – TRENDS IN PLOT-LEVEL RICHNESS

Where results for community richness in the previous section were restricted to Characteristic Species, or those species most commonly observed in natural forests, richness in this section considers all Target Species or the entire list of native species observed in the reference data set. Similar to Figure 8 for Characteristic Species, Figure 9 illustrates trends in cumulative achievement for richness (per plot) for all Target Species.

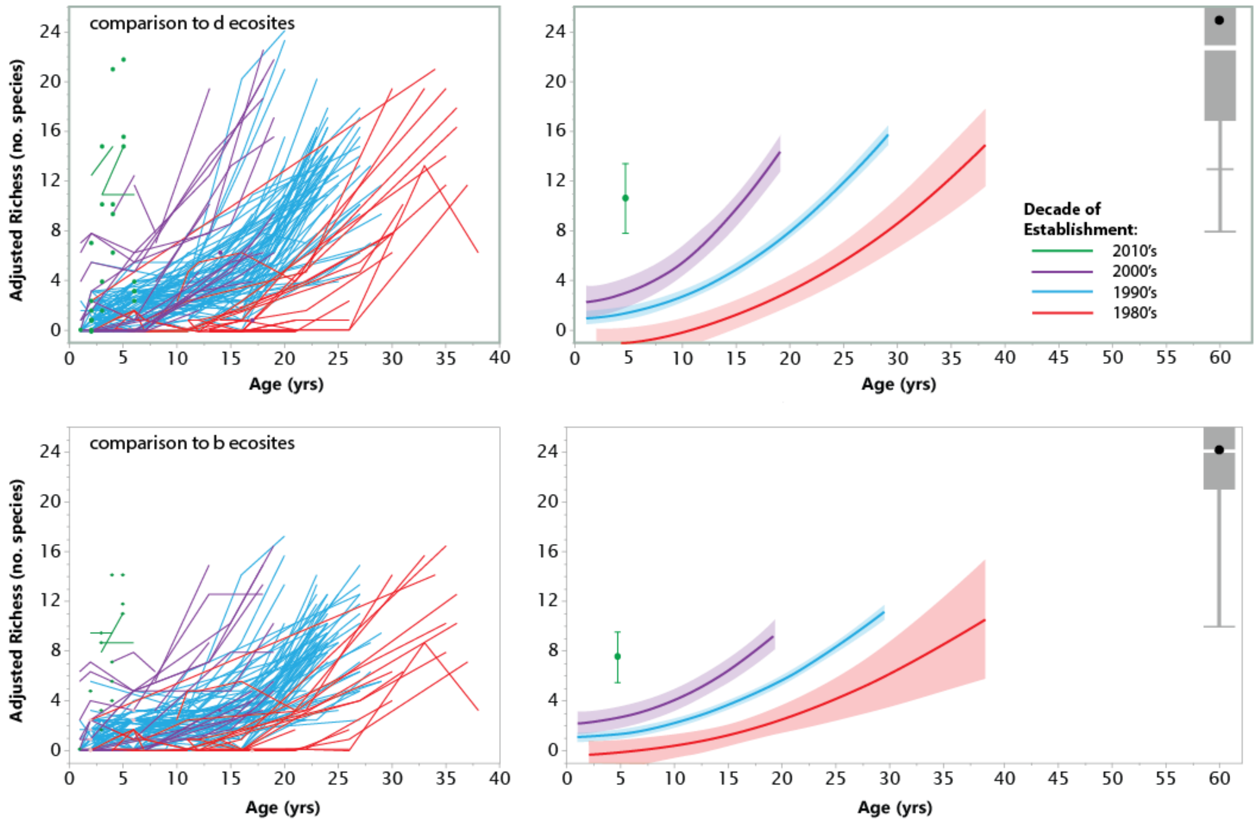

Figure 9. Trends in Target Species richness per plot by reclamation age.

Data values have been scaled to estimate what would have been detected in smaller 100 m^2 sample plots comparable to those used for the reference data. Reclaimed plots are compared separately to targets and NRV's for moist d ecosites on the top and drier b ecosites on the bottom. Reclaimed data is colour coded by decade of reclamation establishment (1980's through 2010's). Solid lines on the left represent individual plots over time, and dots represent plots with a single observation. The box plots (defined in Section 5.2) on the right side of each chart represent the NRV from the reference data. Trend lines (solid lines on the right along with shaded 95% confidence intervals) for each decade of establishment, based on the equation y=a+bx2, each had significantly different slopes based on a post-hoc Tukey multiple comparisons test. No trend line was determined for the youngest subset (2010's) where many plots still have only a single observation, but a mean value and confidence interval is provided instead.

As with the more restricted set of Characteristic Species, there are strong trends of increasing Target Species richness with increasing reclamation age. Visual extrapolation of the trend lines from Figure 9 suggests that most of the mean values for reclaimed Target Species richness could potentially match those for the NRV by 60 years of age, and it is clear that some individual plots are already closely approaching that condition. However, such extrapolations are risky. First, the equations used to fit the lines are simplistic approximations, and the slope of the fit lines at the highest limits of the age range have considerable uncertainty. It must also be recognized that extrapolation assumes that all factors affecting the rate will stay constant, which the conceptual model from Figure 1 suggests is not the case. As reclaimed sites approach Stage C in Figure 1, it is suspected that the increasing density of the tree canopy will favour the arrival of some Target Species, but may temporarily inhibit the arrival of others. In a few cases, the tree canopy in stage C may even exclude some Target Species that are already established (see also discussion in Section 6.7). Overall, however, the trends are encouraging.

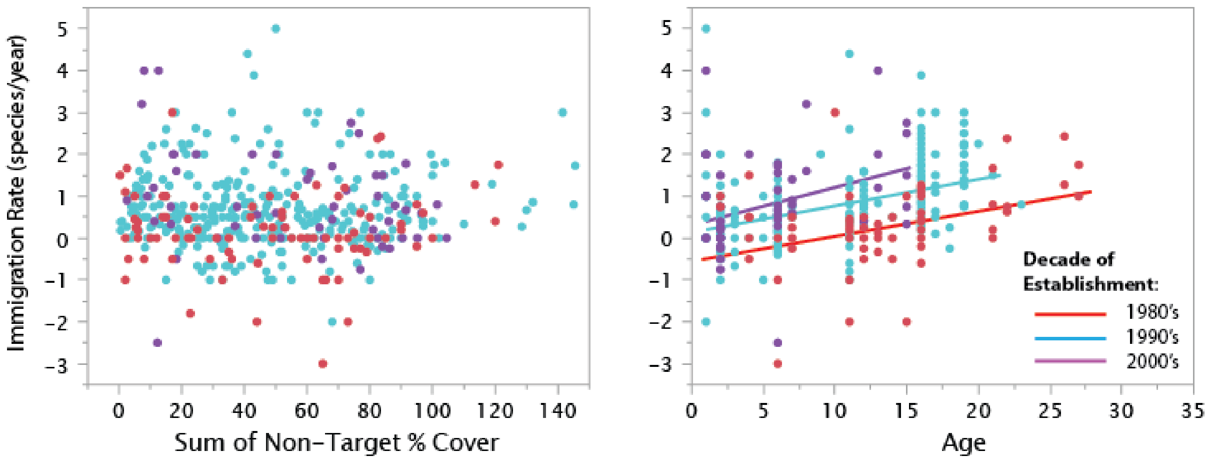

Figure 10. Trends in Target Species immigration by summed cover of competing (non-target) species (left) and reclamation age (right). For each interval between plot re-measurements, the mean rate of immigration is determined by calculating the difference in plot level Target Species richness at the beginning and end of the interval, which is then divided by the length of the interval. Summed cover and age are reported at the beginning of the interval. There was no significant trend for immigration rates by summed cover of non-Target Species. For the illustrated fits of trend lines by age, the positive slope is significant ($p < 0.05$), but there are no significant differences between the slopes by Decade of Establishment. The intercept for the 1980's Decade of Establishment is significantly different ($p < 0.05$) than for the other two decades.

As with Characteristic Species, there is evidence that rates of native species ingress have varied by decade of establishment. Mean rates of immigration for Target Species can be inferred from the trend lines in Figure 10. While it was initially expected that immigration of Target Species would be inhibited by competing vegetation form previously established ruderal species (represented in Figure 10 by the summed cover of non-target species), there is no evidence for such an effect. Instead there is a significant trend showing increased immigration rates with reclamation age, suggesting an acceleration of convergence with increasing age.

6.5 TARGET SPECIES - TRENDS IN PLOT LEVEL COVER

Summarizing all plots by age class and for all species groups (as previously defined in Section 3.4 and displayed in Figure 6), we can observe general trends in cover for species groups (Figure 11). Overall, it appears that cover of Target Species and Other Boreal Species is increasing over time. The same pattern is echoed in trends for Target Species cover by plot in Figure 12. These trends appear similar to those for richness, but are possibly somewhat dampened in their approach to mean values for the reference data. Where trends for richness suggest that reclaimed values are starting to merge with the NRV within 10 to 40 years after establishment, trends for percent cover suggest a longer time frame, and there are some plots that appear to have declining cover over time. However, it is important to recognize that:

- total understorey cover, including that of Target Species, is expected to decline during the 2 to 3 decades of maximum canopy shading following initial crown closure (stage C in Figure 1)
- these analyses consider only a portion of the entire plant community – or those species expected to occur in old seral conditions. Looking back at Figure 5, it is apparent that only one third of all species currently detected on reclaimed sites are Target Species (including Characteristic Species), and it not reasonable to expect Target Species cover on reclaimed plots to reach levels similar to those in the reference data until cover of ruderal species declines,
- estimates of cover are visual assessments with low precision, providing the possibility that a high estimate in one period followed by a low estimate in the following period will result in the temporary appearance of a negative trend (or the jagged appearance of some individual plot trend lines).

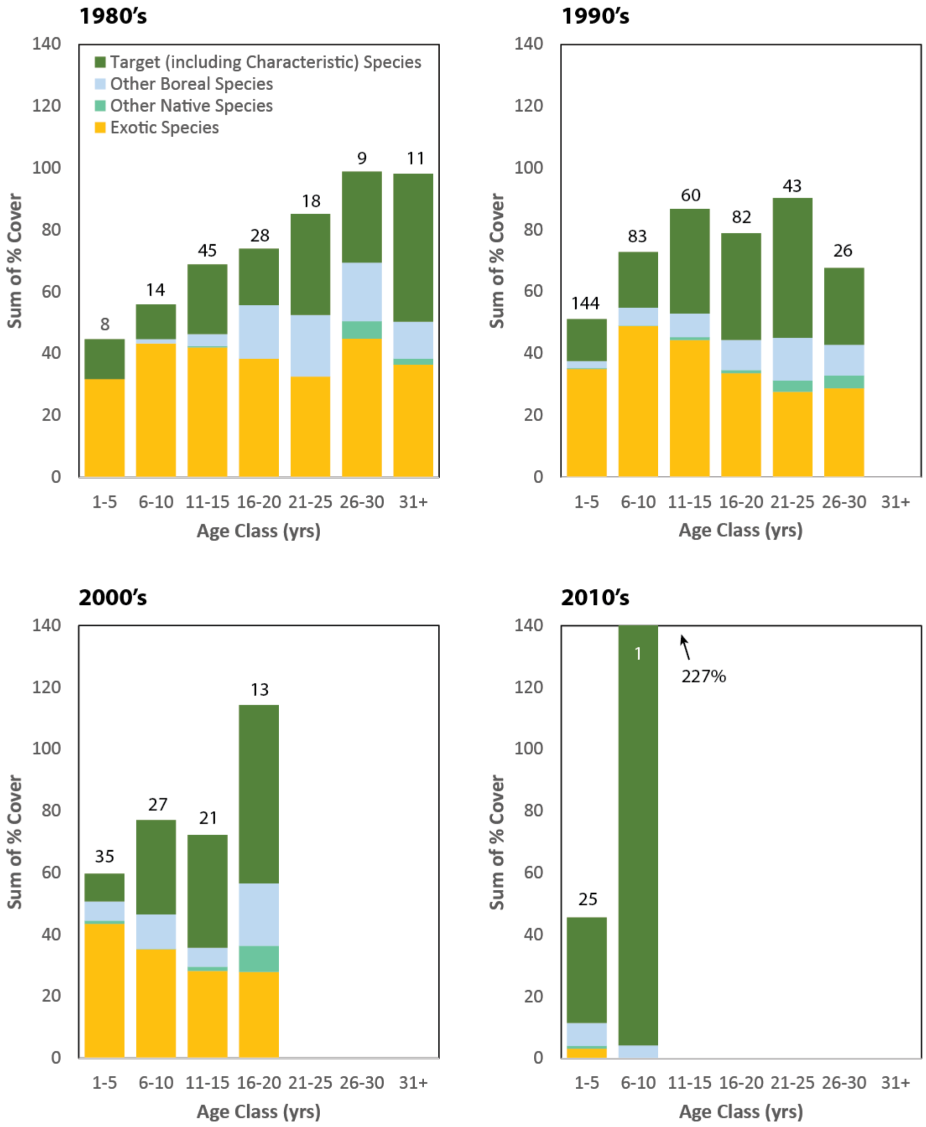

Figure 11. Changes in summed understorey percent cover by species groups (bar colours), age class and establishment decade. Percent cover values are summed from species-level estimates and may exceed 100% due to vertical overlaps. Numbers at the top of each bar indicate the number of plots being summarized. Only general trends should be inferred: different bars in the charts to not provide an exact apples-to-apples comparison as each one represents a different group of plots with no control over similarity of starting conditions.

Overall, the slower approach of Target Species cover values to the NRV envelope as compared to that for richness is not unexpected. While the arrival rate of native species appears to be unaffected by competing vegetation (Figure 10), the availability of growing space to expand their prevalence almost certainly will be.

6.6 TRENDS IN NON-TARGET VEGETATION COVER

Expanding beyond the limited case of Target Species from the previous section, trends in cover values for non-target community segments by age class (based on species groups from Section 3.3 and Figure 5) are also included in Figure 11. Here, the gains in Target Species cover over time are put into perspective with trends for the other groups. At first glance, it appears that non-boreal species native to other parts of Alberta are decreasing, and that cover of exotic species

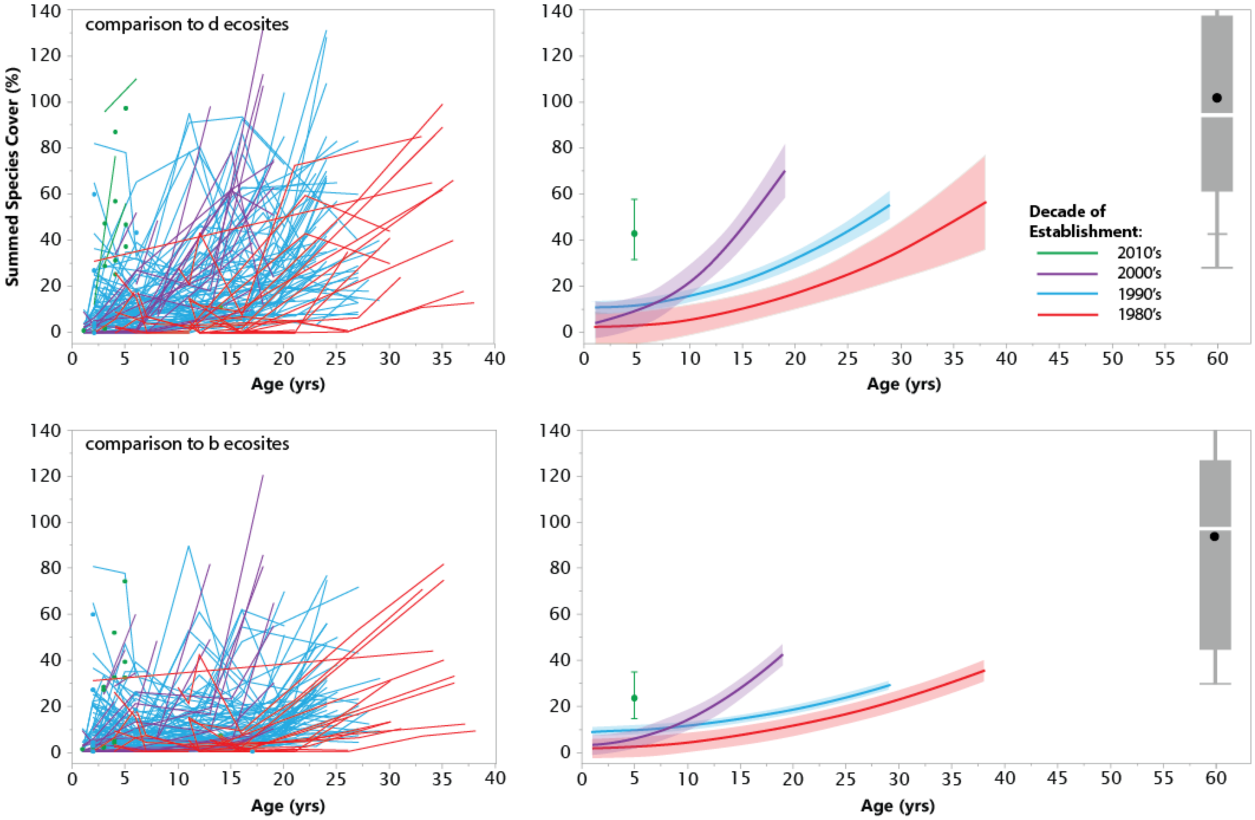

Figure 12. Trends in cover of understorey Target Species by reclamation age.
Percent cover values are summed by plot from species-level estimates, and may sum to greater than 100% due to vertical overlaps. Reclaimed data is colour coded by decade of reclamation establishment (1980's through 2010's). Solid lines on the left represent individual plots over time, and dots represent plots with a single observation. The box plots (defined in Section 5.2) on the right side of each chart represent the NRV from the reference data. Trend lines (solid lines on the right along with shaded 95% confidence intervals) for each decade of establishment, based on the equation $y=a+bx^2$, each had significantly different slopes based on a post-hoc Tukey multiple comparisons test. No trend line was determined for the youngest subset (2010's) where many plots still have only a single observation, but a mean value and confidence interval is provided instead.

is remaining relatively constant or even increasing slightly. However, in interpreting this trend it must be recognized that all of the data for the older age classes comes from plots where exotic species, and particularly grasses, were intentionally introduced as part of the reclamation prescriptions. They started with and are maintaining the highest levels of exotic cover. Plots from later reclamation that did not experience such introductions have not yet reached these older age classes. If the lower initial levels of exotic species in these later reclaimed areas is sustained over time, the mean cover levels for exotic species in older age classes will drop as these younger reclaimed areas are included in the class-specific mean values.

Moving beyond class-level summaries of cover for exotic species, trends in species-level cover are presented in Table 2. Of some concern is the long term (> 20 yrs) persistence of species such as caragana (*Caragana arborescens*), bird's-foot trefoil (*Lotus corniculatus*), alfalfa (*Medicago sativa*), the two sweet clovers (*Medicago fulcata* and *M. sativa*), perennial sow thistle (*Sonchus arvense*), dandelion (*Taraxicum officinale*), alsike clover (*Trifolium hybridum*), smooth brome (*Bromus inermis*), and timothy (*Phleum pratense*). As for % cover in Figure 11, some apparent trends from Table 2 may be deceptive, resulting from (i) each age class resulting from a different blend of practices, and (ii) not all plots having observations in all time periods – note in particular that the oldest two age classes have only 8 observations each. Also of note are the two exotic tree and shrub species (*Populus x* and *Caragana arborescens*) which appear to be increasing in occurrence but are actually (i) maintaining a fairly constant effect on a fixed area of land and (ii) impacting a declining

Table 2. Trends in exotic species frequency of occurrence in reclamation plots by age class.
Data were further parsed by period of establishment, with reclamation established prior to the year 2000 on the left (up to eight 5-year age classes), and more recent reclamation on the right (4 age classes). Species that were intentionally introduced as part of pre-1990 reclamation are indicated with an asterisk. Noxious weeds are indicated with a double asterisk.

Occurrence of Exotic Species by percent of plots
· <1% (non-zero) | • 1 to 5% | ● 5 to 25% | ⬤ > 25% of plots

		Age class (yrs), estab. 1980-1999								Age class (yrs), estab. 2000-2015			
		0-4	5-9	10-14	15-19	20-24	25-30	30-34	35-39	0-4	5-9	10-14	15-19
		200	134	126	127	64	38	8	8	200	134	126	127
Trees													
*Larix sibirica**	Siberian Larch			⬤	⬤	⬤	⬤	●	●				
*Populus x**	Hybrid poplar	·	·	·	·	●	●						
Shrubs													
*Caragana arborescens**	Common caragana					●	●	⬤	⬤				
Salix monticola	Mountain willow									·			
Forbs													
Artemisia absinthium	Wormwood	·											
Aquilegia canadensis	Canada columbine								·				
Astragalus cicer	Cicer milkvetch	·	•	●	•	•	•						
Axyris amaranthoides	Russian pigweed				•	•				•			
Capsella bursa-pastoris	Shepherds-purse				•								
Chenopodium album	Lambs -quarters	⬤	●	•	•		●	⬤	⬤	⬤			
*Cirsium arvense***	Canada thistle	•	•	•	•	●	⬤		●	●			
Cirsium vulgare	Bull thistle	·	•										
Crepis tectorum	Annual hawksbeard		•		•	●	●	●		•	•	●	●
Descurainia sophia	Flixweed		•	•	•					●			
Erodium cicutarium	Storks-bill	•											
Erucastrum gallicum	Dog mustard	●	•				●			⬤			
Fallopia convolvulus	Wild buckwheat								●				
Galeopsis tetrahit	Hemp-nettle	·	•		•	●	•			●	•	●	
Kochia scoparia	Kochia									●			
*Lotus corniculatus**	Birds-foot trefoil	●	⬤	⬤	⬤	⬤	⬤	●	⬤	⬤	•	⬤	⬤
Medicago falcata	Yellow lucerne		●	•		●				•			
*Medicago sativa**	Alfalfa	●	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤	●	⬤	⬤
Melilotus alba	White sweet clover	⬤	⬤	⬤	⬤	●	●	⬤		⬤	●	⬤	⬤
Melilotus officinalis	Yellow sweet clover	●	●	⬤	●	⬤	⬤	●	●	⬤	●	⬤	⬤
Persicaria maculosa	Ladys thumb	·								•			
Phlox divaricata	Wild blue phlox					·							
Plantago major	Common plantain								●	●	•		
Platanthera hyperborea	Northern green bog orchid						·		·				
Polygonum aviculare	Prostrate knotweed									·			
Rorippa islandica	Marsh yellow cress									·	·		
Rumex acetosella	Sheep sorrel		•										
Rumex salicifolius	Narrow-leaved dock		·	•	•	·							
Salsola tragus	Russian thistle	●	●	•						●			
Senecio vulgaris	Common groundsel	·	•	●		•				●			
*Silene latifolia***	White cocle	·											
Sinapis arvensis	Wild mustard	●			•					•			
*Sonchus arvensis***	Smooth perennial sow-thistle			●	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤
Sonchus asper	Annual sow thistle (forb)			•	●	•		●		●	●	⬤	
Stellaria media	Common chickweed			•						•			
*Tanacetum vulgare***	Common tansy		•		•	●							
Taraxacum officinale	Common dandelion	●	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤	⬤
Thlaspi arvense	Stinkweed									•			
Tragopogon dubius	Goat's beard						●						●
Trifolium hybridum	Alsike clover	●	●	●	⬤	⬤	⬤	⬤	⬤	●	⬤	⬤	●
Trifolium pratense	Red clover*	•	•	●	●	⬤	⬤		⬤	•		●	●
Trifolium repens	White clover	•			•	●	●			●		●	●
Urtica gracilenta	mountain nettle	•	·	•	•					•		·	
Vaccaria pyramidata	Cowherb									•		●	
Grasses													
*Agropyron cristatum**	Crested wheat grass					•	•	⬤	●	•			
Agrostis stolonifera	Red top			•	•	•		●		●	•	⬤	
*Bromus inermis**	Smooth brome				●	●	⬤	⬤	⬤	●	•	⬤	⬤
*Bromus tectorum***	Downy chess				•								
*Dactylis glomerata**	Orchard grass					•	●	●	⬤				
Elymus repens	Quack grass					●	●	●	⬤	•		●	●
Fagopyrum esculentum	Buckwheat									•			
*Hordeum vulgare**	Barley	●	•										
*Phleum pratense**	Timothy	·	●	•	•	⬤	●	⬤		●		●	⬤

proportion of reclaimed area over time as more and more reclamation occurs in which these species are absent. These species were only planted in the 1980's and very early 1990's, but occur in a substantial proportion of the small number of very old plots.

Five of the species listed in Table 2 are classed as noxious weeds under the Alberta Weed Act and its associated Regulation, but none are prohibited noxious weeds. Of these, only Canada thistle (*Cirsium arvense)* and smooth perennial sow-thistle (*Sonchus arvensis)* are relatively prevalent, and the former is much less prevalent in more recent reclamation. All noxious weeds are monitored and potentially treated as part of Syncrude's ongoing weed management program.

Overall trends for the grass species should be treated with caution, as there are indications that identification to the species level was particularly weak in the early years (roughly 1980 to 2000) for these plants. Roughly 1/3 of records for these species during this period were simply coded as "grass" or "Graminaceae".

6.7 TRENDS IN UNDERSTOREY COVER RELATED TO TREE COVER

While apparent trends from Figures 11 and 12 might suggest that reclamation practices are failing to limit or may even be favouring exotic species, it appears that this is largely an artifact of the oldest reclamation (and associated monitoring plots) being (i) the most impacted by intentional introduction of exotic species and (ii) having only a small number of plots on which to evaluate patterns for the oldest age classes. Given that the main driver of the predicted shift from ruderal to Target Species is expected to be the developing tree canopy, a measure of this effect is illustrated in Figure 13. Here, understorey cover for non-target vegetation is plotted against stand basal area. This is a standard metric from the forest sector that is highly correlated to canopy density and light penetration for early stages of even-aged stand development. The threshold of maximum cover evident in this chart suggests that non-target species are being strongly impacted by increased shading from a tree canopy. These observations are consistent with the expectations from the conceptual model in Figure 1. Target Species will also decline in a similar manner, but are expected to be somewhat more resilient to the low light and other modified environmental conditions.

Depending on canopy structure and light penetration, the ground cover maintained by understorey communities is quite variable. Examples in Figures 14 and 15 include (i) understorey communities having lower ground cover below canopies presumably with higher leaf area and light interception, and (ii) an understorey community of good vigour below a single species canopy of jack pine (*Pinus banksiana*). Based on subjective observations, the latter case appears to be

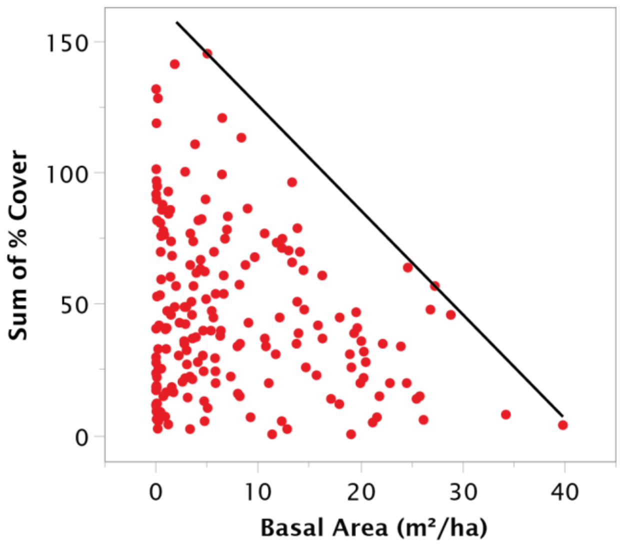

Figure 13. Trends in summed cover for non-Target species as related to basal area, a surrogate for canopy density. The boundary fit is estimated using quantile regression with a 99% quantile threshold ($p < 0.05$). Percent cover values are summed from species-level estimates and may exceed 100% due to vertical overlap.

Figure 14. Understorey vegetation conditions below uniformly closed tree canopies.
Plot 90-01-05 is shown on the left at age 28, with a white spruce basal area of 26 m^2/ha. Target and Other Boreal ground cover consists of 5% shrubs, 8% forbs, 4% grasses and 13% mosses and lichens. Despite low overall cover, there are 13 Target Species. Plot 94-02-24 is shown on the right with a primarily jack pine basal area of 23 m^2/ha at age 23. Willows and some aspen stems are filling gaps in the taller pine canopy. Target Species cover consists of 5% shrubs, 14% herbs, 4% grasses, 2% ferns and 7% mosses/lichens. There are 18 Target Species present, along with the exotic Taraxacum officinale (dandelion).

Figure 15. Forest understorey developing below a closed canopy jack pine stand, growing on 50 cm of peat-mineral mix soil over tailings sand, 24 years after establishment.
Target understorey species readily evident in the photo include bristly black currant (Ribes lacustre), fireweed (Chamerion angustifolium), raspberry (Rubus idaeus), wild strawberry (Fragaria virginiana), cream-coloured vetchling (Lathyrus ochroleucus), wild vetch (Vicia americana), bearberry (Arctostaphylos uva-ursi), Lindley's aster (Symphorotrichum ciliolatum), knight's plume moss (Ptlium crista-casristrensis) and bluejoint reedgrass (Calamagrostis canadensis). There is also a small amount of the exotic Taraxacum officinale (dandelion).

the general rule for single species canopies of pine and aspen, or anywhere that tree planting densities are either low enough or sufficiently variable to maintain a modest level of permanent canopy gaps. Single species stands having full occupancy of spruce, some combinations of tree species having overlapping crowns, and sites where tall willow species have grown in sufficient numbers to fill in gaps in the tree canopy are trending more towards the conditions illustrated in Figure 14. For reclaimed stands in the latter category, understorey light levels having been crudely measured using a smartphone app as low as 6% of that experienced under fully open conditions.

7 RESULTS - TRENDS IN COMMUNITY SIMILARITY

Based on the Bray-Curtis index of community similarity, trends in reclaimed plant community convergence towards reference site conditions are illustrated in Figure 16. Similar to earlier trends for Characteristic and Target Species richness and cover (Figures 8, 9 and 12), data has been colour coded by decade of establishment to provide further clarity on important influences.

Similar to other metrics, the illustrated relationships using the Bray-Curtis index suggest increasing similarity to reference conditions with increasing reclamation age. Unlike for previous metrics, not all subsets of the data based on decade of reclamation are statistically distinguishable to 95% confidence. Also unlike for previous metrics, there subjectively appears to be a lesser degree of approach (or convergence) to the reference conditions.

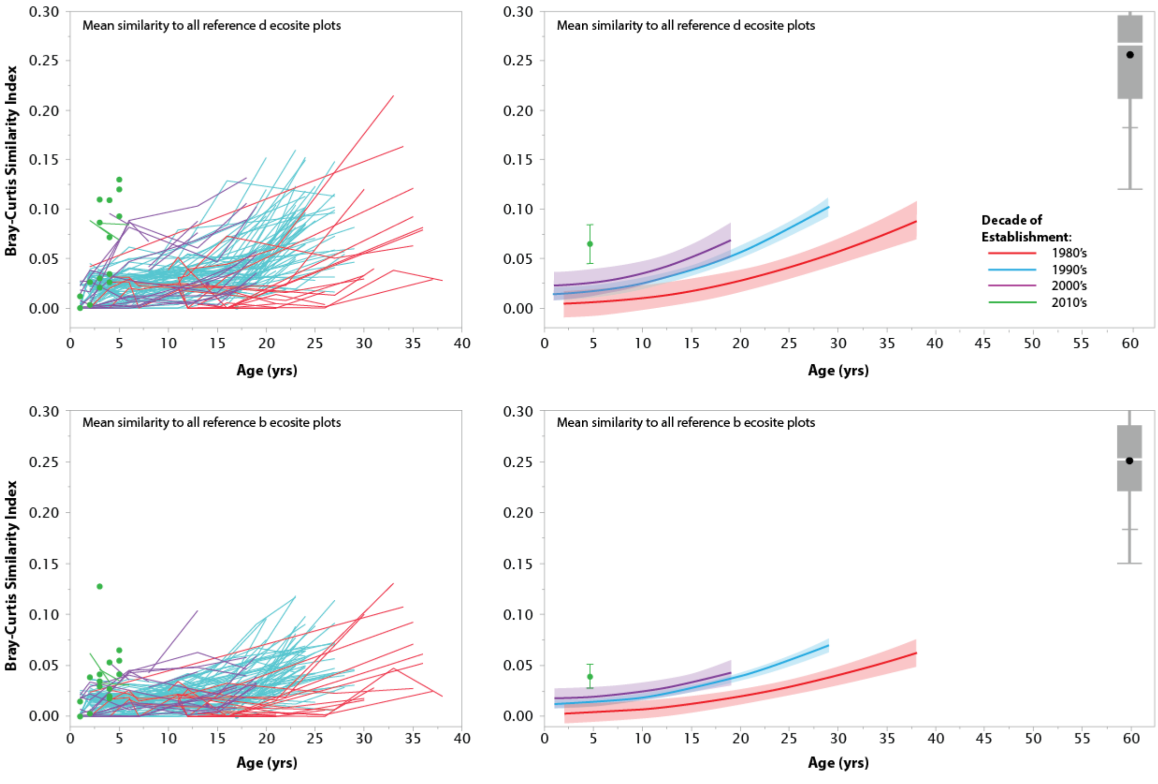

Figure 16. Trends in community similarity between reclaimed plots and reference plots based on the Bray-Curtis Similarity Index (reflecting species cover values).

Reclaimed data is colour coded by decade of reclamation establishment (1980's through 2010's). Solid lines on the left represent individual plots over time, and dots represent plots with a single observation. The box plots (defined in Section 5.2) on the right side of each chart represent the NRV from the reference data. Trend lines (solid lines on the right along with shaded 95% confidence intervals) are provided for each decade of establishment, based on the equation y=a+bx2. The slope for the 1980's decade is significantly different from that for the 1990's and 2000's based on a post-hoc Tukey multiple comparisons test. No trend line was determined for the youngest subset (2010's) where many plots still have only a single observation, but a mean value and confidence interval is provided instead.

8 RESULTS - EFFECTS OF ECOSITE ON COMMUNITY COMPOSITION

It was initially expected that moist reclaimed ecosites would trend more toward the plant communities for reference d ecosites and the communities for drier reclaimed ecosites would trend more toward the reference communities for b ecosites. A pair of preliminary tests for the effect of ecosite on plant community trends was completed by separating reclaimed plots into AWHC classes roughly approximating soil moisture regime distinctions between b (dry) and d (moist) ecosites (see Section 5.6 for methods and assumptions). The trends evaluated in Figures 8, 9,12 and 16 were then re-compiled for each of the sub-groups, with an example of the outcomes provided in Figure 17. The 1990's decade is used in the example given that it has the most balanced representation of dry versus moist sites. While results are illustrated only using the Bray-Curtis similarity index, similar degrees of overlap for dry versus moist site conditions were observed regardless of the community metric used.

Within the range of conditions represented by the plots in this study, there does not appear to be any detectible effect of moisture regime (or ecosite) on gross plant community trends. Drier and moister reclaimed sites are trending similarly toward either b or d ecosite reference plots.

It was further expected that moister reclaimed sites would achieve higher numbers of Characteristic Species than drier ones, and that stratifying plots by ecosite or moisture regime would provide results that would allow us to examine this expectation. However, the evidence for such a distinction is not clear; differences by dry versus moist sites were not significant for the 1980 and 1990 establishment decades, and for the later decades there are uncontrolled confounding factors other than moister regime (including reclamation soil materials) that preclude any confidence that significant differences are indeed the result of moisture regime. These other factors are not thoroughly addressed here, but are intended to be explored more completely in a companion study.

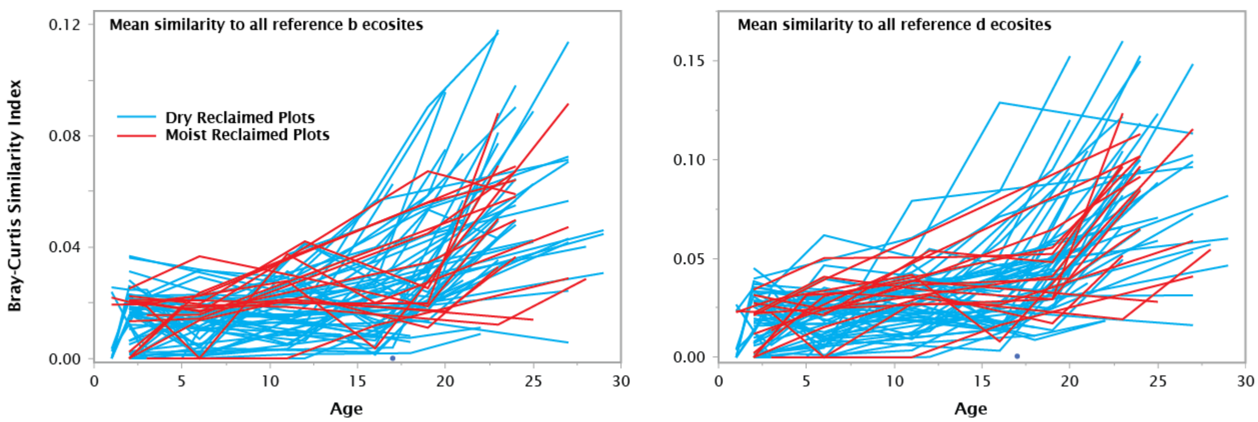

Figure 17. Differences in community similarity trends based on reclaimed ecosystem class (dry versus moist). Each line (or point) represents a single reclaimed plot, where y-values at each time step are the mean similarity to either (i) all b ecosite reference plots in the left hand chart or (ii) all d ecosite reference plots in the right hand chart. No significant differences (p<0.05) were found for models fit by reclaimed ecosystem class.

9 ADDITIONAL OBSERVATIONS

9.1 PLANT COMMUNITY DIVERSITY

There appears to be a common perception amongst those not frequently working on reclaimed lands that actual achievement of community diversity falls well short of that observed on natural landscapes. This report could potentially contribute to that perception if not carefully considered, as it portrays a large segment of the current inventory of reclaimed lands as being assigned to only two of twelve ecosites described by Beckingham and Archibald (1996) for the local Ecological Area. Placed in the appropriate context, however, the sampling program and associated reclaimed sites underlying this study are located on suitable terrain positions for only five of those ecosites (a through e). Of those, b and d were the most prevalent on the pre-mining footprint, occupying roughly 90% of the upland area. Examples of c and e ecosites likely occur within the reclaimed areas (and possibly the sample plots), but have not yet been explicitly identified. It is not expected that examples of a ecosites would occur, as these require very coarse textured, dry and nutrient poor soil conditions. These conditions did not occur within the Mildred Lake mine footprint prior to industrial disturbance, and with the exception of two research trials, have not been explicitly targeted in reclaimed areas.

While the analytical approaches employed in this study are not suited to quantifying the diversity of reclaimed communities, the outcomes should be useful to provide insights and guidance for such a study in the future. Additional observations on diversity are illustrated in Figures 18 and 19, based on field work for both monitoring and other projects in reclaimed areas. Reclamation practitioners typically plan for diversity at a relatively large scale, expecting that variability in terrain, soils and hydrology will occur within those units and in turn will influence community diversity.

9.2 CASE STUDIES OF NOVEL ECOSYSTEMS

To this point, analyses of community trends have been restricted to the majority of reclaimed plots where community trends are expected to be converging with local native forests. However, as mentioned previously there were two subsets of the data that were excluded as being noticeably different based on known causal factors. These are cases where a dominant vegetation species was intentionally introduced in the earliest years of reclamation, and has resulted in plant communities that are noticeably distinct.

The first of these cases is areas planted to the exotic species caragana (*Caragana arborescens*). Caragana is a leguminous tall shrub that is native to boreal and sub-boreal climates in Asia. It is notable for its ability to fix nitrogen in the soil, for its utility in erosion control and, in reclamation, for its ability to tolerate alkaline soil conditions (Favorite 2006). It was planted as an accepted practice in portions of Syncrudre’s reclaimed areas in the 1980’s. For locations with dense plantings of this species, the vast majority of co-planted trees (if any) have been outcompeted, apparently because they were never able to emerge from the taller caragana canopy. In the small handful of monitoring observations for this vegetation type in later years, values for both richness and cover of Target Species are consistently within the lower 20th percentile as compared to all other plots. Given the general lack of native tree species in these areas, coupled with ad hoc observations that caragana is readily reproducing in this environment, it seems unlikely that these areas will converge with locally common boreal ecosystems within the foreseeable future barring further human interventions.

Plot 99-02-06, SW Corner

Plot 99-02-06, NE Corner

Plot 99-02-05, SW Corner

Plot 99-02-05, NE Corner

Figure 18. Examples of understorey community variation within and between two independent plots in a single reclamation unit (June 2019) on an east-facing slope of the Southwest Sands Storage containment dike.
The plots are 230 m apart, and photo locations within each plot are roughly 25 m apart. This area was planted in 1999 to promote a tree cover of trembling aspen. The red-osier dogwood (Cornus stolonifera) evident in three of the photos was also planted.

The second case of continued dominance by an intentionally introduced species is areas planted to Siberian larch (*Larix sibirica*). Similar to caragana, this species was utilized in reclamation due to its reputation for promoting favourable soil characteristics under a wide range of challenging growing environments. Occasional plantings of this species continued on the Syncrude site through the 1990's. This tree species is native to boreal forests of northern Asia and Europe, and will often easily outgrow native tree species when planted in similar climatic conditions in boreal North America.

Siberian larch growing on reclaimed sites has the ability to develop very high relative leaf areas, resulting in a high degree of shading on the forest floor (Figure 20). In the first decade after reclamation, areas planted to this species hosted understorey plant communities that are similar in composition (richness and % cover) to those having native tree cover. However, the development of a closed forest canopy has occurred earlier and typically with a higher degree of shading than with native tree species. This has resulted in understorey conditions with low richness and cover of Target Species, typically below the 10th percentile of values for the larger data set. The heavy annual litter deposition from this species may also play a role in reduced understorey richness, particularly for moss and lichen species.

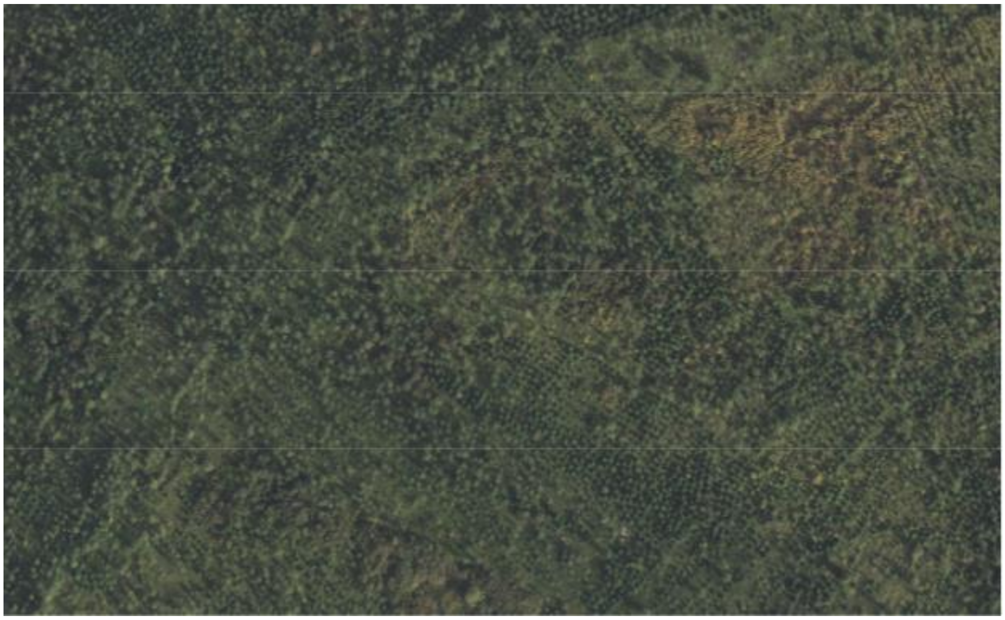

Figure 20. Aerial image (Sept. 2019) of a portion of the reclaimed W2 overburden landform, showing a spatial extent roughly 300 m east-west and 180 m north-south (5.4 ha).
This area was planted in 2005. The smaller crowned, darker and more sharply defined trees are white spruce (Picea glauca), and the larger crowned, lighter green and less distinct trees are aspen (Populous tremuloides). The trees are not yet large enough for the crowns to have coalesced to form a continuous canopy. Variations in patterns and texture within the image are generally consistent with variations in plant communities. The area that is more yellow-brown toned in the upper right corner is a local depression that is developing as a small complex of wetland communities

Figure 19. Understorey conditions below a closed canopy Siberian larch stand, aged 24, Mildred Lake mine site.
The live canopy in the foreground has risen above the top of the image, but can be seen in the background.

ADDITIONAL OBSERVATIONS

Over the longer term, it is expected that canopy densities for Siberian larch stands will be reduced, and light levels at the forest floor will increase. Referring back to the conceptual model in Figure 1, the observations of very low Target Species cover are assumed to be an extreme case of the patterns expected for stage C. As such, it is not unreasonable to predict that these ecosystems will re-establish a more vigorous and diverse understorey community as small gaps in the canopy become more prevalent or slightly larger following the transition from stage C to stage D.

10 DISCUSSION

10.1 PATTERNS AND RATES OF CONVERGENCE

While there is a clear pattern of convergence for reclaimed land community compositions with those for reference landscapes, some differences in patterns and rates do exist. For reclamation following surface mining, the conceptual model in Figure 1 does not start immediately after the disturbance, but at a later date with the commencement of reclamation activities. As such, some key factors impacting adherence to the model include:

1. Biological legacies from the previous forest have been completely eliminated.
2. The initial loss of biological legacies will be mitigated to some degree by (i) the importation of plant (and other species) propagules in the reclamation soil and (ii) the planting of trees and shrubs. The soils are native materials typically salvaged from a different location than where they are deployed, and may have been stored for varying lengths of time in stockpiles. In a minority of cases, soils can be salvaged from a donor site at the advancing front of the mine and directly placed on a reclamation site with similar edaphic conditions (soil moisture regime x soil nutrient regime). In these cases, a substantial portion of the legacy propagules typically survive and are suitable for the new location. In other cases of direct placement, reclamation soils must be shifted to different edaphic conditions, allowing only those legacy species that are ecological generalists to be suitable on the new site. This is often suspected to be the case where peat cover soils are used in oil sand mine reclamation: soils are salvaged from wetland or transitional wetland locations and placed on upland locations. For stockpiled soils, the original legacy propagules can be almost completely lost after as little as 16 months (Mackenzie and Naeth 2009, 2010, Mackenzie et al 2012, Naeth et al. 2013), leaving only those species that have colonized the surface of the stockpile as a substantially diluted (and possibly undesirable) propagule source.
3. The adjacency (or lack) of seed sources can have a large influence on immigration (both for ruderal or weedy species and forest-dependent native species).
4. Opportunities for colonization by various species, and particularly those that are not locally native, are heavily influenced by human activities. This can include the unintentional transport of seeds from far afield attached to vehicles or equipment, or the intentional introduction of non-native species believed to have beneficial effects. The latter case was quite common for oil sands mine reclamation in the 1980’s and 1990’s where exotic canopy species were planted or where temporary cover crops were seeded. Examples include the planting of caragana in the 1980’s, and Siberian larch in the 1980’s and 1990’s.
5. The chemical, physical, hydrological and micro-biotic properties of (i) the salvaged and sometimes stockpiled reclamation soils and (ii) the underlying landform substrates will differ to varying degrees from those supporting the pre-mining ecosystems. The degree to which locally common native species are sensitive to those changes (i.e. elevated pH, altered soil structural properties) may affect their competitive suitability on the reclaimed site.

Factors such as these have led to historic expressions of doubt regarding (i) acceptable development of reclaimed plant communities and (ii) suspicions that immigration of native species and associated patterns of ecological succession may be impeded or altered on reclaimed areas, such that end land uses dependent on those communities cannot be supported (e.g. Quideau et al 2013, Audet et al. 2015, intervenor submissions to Environmental Impact Assessment hearings as documented in AER 2019). Specific to the second point above, Audet et al. (2015) summarize and discuss a complex hierarchy of landscape, landform and local site requirements which, if not adequately replicated, could lead to novel ecosystems not conforming to the reclamation sub-goal of “*locally common boreal forest ecosystems*”. Other authors have discussed factors such as (i) aggressive invasion and subsequent persistence of non-native vegetation and particularly noxious weeds (e.g. introductory comments by Small et al. 2018), (ii) potential dependence on soil microbiotic communities and particularly mycorrhizal fungi that are presumed deficient in reclaimed soils (e.g. Visser and Danielson

1988), and (iii) reclamation soils being unacceptable growth media for some native species with a particular emphasis on chemical properties such as salinity or alkalinity (e.g. Calvo-Polanco et al. 2017).

Despite these theorized impediments, the empirical evidence presented here and in earlier studies of reclaimed community development suggests a more optimistic outlook. While reclaimed communities are not yet fully equivalent to the late seral reference conditions used in this report, such was never expected to be the case. Metrics for reclaimed upland plant communities are shown to have patterns of increasing values for plot level richness, abundance (% cover) and community similarity when contrasted with NRV's for the same metrics from reference old seral ecosystems. In other words, they are converging with their reference counterparts.

In comparing back to the conceptual model presented in Figure 1, this study confirms expectations of increasing Target Species cover concurrent with increases in richness through stages A and the earlier portions of stage B, where understorey cover peaks prior to substantial development of the tree canopy. The data also provides further insights on these patterns that are glossed over in the simplified conceptual model. In comparing overall trends by age for the various metrics, we can observe a number of cases where plots are experiencing earlier increases for Target Species richness than for Target Species cover. A key example includes the delayed move away from the x axis in Figure 13 (Target Species cover) as compared to Figure 10 (Target Species richness). This pattern is consistent with a process where Target Species are initially gaining a foothold but are delayed in expanding their cover, presumably due by competition from other species which currently occupy the available growing space. There are multiple possible mechanisms for the delayed increases in ground cover by these species, with two of the more likely being:

1. The newly established Target Species need time to develop sufficient root networks and leaf area in order to effectively compete for growing space. When first established, the new plants have minimal ability to produce photosynthates beyond what is needed to satisfy basic respiratory and maintenance demands, with little left over for substantial expansion of biomass.
2. The newly established vegetation requires its presumed advantage in shade tolerance in order to effectively compete with previously established vegetation, but that advantage does not come into effect until the developing tree canopy starts to lower light availability to ground vegetation.

Regardless of the reasons, the key point is that Target Species *are* becoming established, and they *are* eventually increasing in abundance.

Presumably, increases in tree cover will have disproportionate effects on various understorey species, and facilitate competitive advantages for forest dependent or Target Species. The trends illustrated in Figure 13, and the examples in Figure 15, highlight impacts of increasing tree cover on understorey cover in general, but particularly on ruderal or weedy species. Such patterns should be expected if we assume that the majority of non-target species are early seral specialists and are predominantly less shade tolerant than the forest dependent Target Species.

At the very highest levels of canopy cover being reached in a small subset of reclaimed stands, it is presumed that understorey light levels are declining to a point where all shrub and forb cover is being impacted to a high degree[4] (Figure 15). This condition falls within the expected limits of variability for the Stage C milestone in Figure 1. In these cases of high canopy shading, limited plot data from this study augmented by subjective observations suggest a general shift from forb and shrub cover to a developing moss layer, although many of the forb and shrub species maintain their presence at low cover levels in scattered sun flecks associated with small gaps in the canopy. For natural forests, there are more extreme cases where even the moss layer is extinguished: relatively rare examples exist in the boreal forest zone for some stands of white spruce, with common examples in temperate forests of British Columbia where stands of western hemlock (*Tsuga heterophylla*) and western redcedar (*Thuja plicata*) can be completely devoid of understorey vegetation.

[4] Other correlated factors such as increased litter fall may also contribute to this effect.

Much more common in boreal forests (and reclaimed areas) are conditions where the tree canopy that is more permeable to light, and where declines in total forb and shrub cover in Stage C are less pronounced.

Reverting our focus back to overall trends, there appears to be a general pattern of accelerating metric values toward the reference NRV with increasing reclamation age (Figures 8, 9, 12 and 16). For all metrics used in this study there are at least some sample plots that have developed to a condition falling within the NRV for the old seral reference samples, and in some cases the mean for all reclaimed plots falls within the NRV. While extrapolations of trends to future time periods must always be treated with caution, the current outlook is positive: it would require a major and abrupt change to current trends for reclaimed sites to not achieve convergence with reference values (where reclaimed mean value approximates reference mean value).

In building on earlier work, this study utilizes a mostly independent reclaimed data set with a larger number of sampling locations, a greater number of repeated measures on those locations (in a few cases spanning almost 40 years), and a wider diversity of community metrics. As a result, this study adds considerable confidence to conclusions from earlier studies suggesting that reclaimed communities are becoming increasingly similar to the desired reference or target conditions.

Beyond these primary observations from this study, there are some additional findings of narrower scope as highlighted below in Sections 10.2 through 10.7.

10.2 DECADAL VARIATIONS IN OUTCOMES

In analyzing plant community trends in this report, a considerable amount of variation in the data was explained using a categorical variable representing the decade in which each respective unit of land was reclaimed. This appears to suggest that growing environments of more recently reclaimed areas are more favourable to rapid development of target communities than were growing environments associated with earlier reclaimed areas. The overall effect might be considered as a lag, where in Figures 8, 9, 12 and 16 the trend lines for many individual plots track close to the x-axis for a longer period of time than for more recently reclaimed areas. For the earliest (1980's) reclamation, the lag period for individual plots appears to be as much as 25 years, dropping to near zero for reclaimed areas established since 2010.

The decades themselves are not necessarily meaningful, but are instead a crude but effective indicator that some other factors are in play, and that those factors are correlated to calendar year. While the correlative nature of this study cannot demonstrate cause and effect, there are several plausible (and not necessarily independent) explanations including:

1. alterations to the growing environment related to climate change,
2. changes in sampling methods that impact species detections, and
3. changes in reclamation practices over time,

While there is no solid basis for discounting climate change as the dominant factor, it seems unlikely. Year-to-year variability in weather related factors affecting both plant establishment and competitive relationships are expected to be of considerably greater magnitude than those related to either (i) overall shifts in climatic normals or (ii) shifts in the frequency of particularly favourable or unfavourable weather events. Also, there have been documented multi-year periods of warmer versus cooler, and wetter versus drier years during the 39 years of monitoring, but these do not correlate to the apparent decadal pattern of increased Target Species arrival.

The second factor relates to possible impacts of varying field crew expertise. University students hired on summer work terms were primarily used for data collection prior to 2010, followed later by specialist contract crews. Potential effects of the less experienced crews could include:

- lower species counts resulting from the grouping of species by genus, family or generic labels such as "grasses"
- lower rates of detection for small plants having lower frequency of occurrence

If one or both of these effects resulted in lower frequencies of detection, it has been postulated that this could be at least part of the cause for the observed decadal pattern, with an uptick in species detections once more experienced contract crews were employed. However:

- the grouping of species occurred primarily for grasses and mosses in the earliest years following reclamation, and evidence from later measurements on young plots by contract crews suggests that an insufficient number of Target Species would have been missed to have a meaningful impact on overall trends, and
- the isolation of slopes (new species per year) and changes in slope, for growth intervals entirely dependent on inexperienced crews as contrasted with those entirely dependent on more experienced crews, demonstrates that any underestimation of species richness by inexperienced crews cannot explain differences in trends by establishment decade (see Appendix B).

Similar to climate change, measurement issues cannot be completely discounted, but they do not appear to be a substantial influence.

The most plausible factor affecting the decadal pattern appears to be an evolution of reclamation practices from the 1980’s to the present. The first decades of oil sands mine reclamation followed much more of an agronomic paradigm than do current practices (Macyk and Drozdowski 2008). Early (1980’s) reclamation prescriptions typically included grading of the reclamation soil to a smooth surface, seeding with a grass mixture, harrowing, broadcast fertilizing and finally planting of trees. From a native species immigration perspective, repeated disturbances to the soil material would presumably have disturbed and possibly degraded the stored bank of soil propagules, and intensive soil manipulation may have created a surface condition that was unsuitable as a seed bed.

There is some documented evidence that the intentional seeding and fertilization of cover crops may have inhibited conservation and ingress of native species, based at least partially on the work of Hardy BBT (1990). That study found that seeding to agronomic grasses and legumes provided benefits for erosion reduction, but also strongly hindered the establishment of planted trees and shrubs. Consistent with lags for ingress of native species observable for early decades in Figures 8, 9, 12 and 16, Hardy BBT (1990) found natural establishment of native species to be minimal even after 15 years. Replacing the agronomic grasses with native grasses provided only a small benefit. Natural establishment of native species was greatest on areas not seeded at all, or seeded to an annual barley crop.

While the Hardy BBT (1990) results appear to contradict some of the competition related effects related to Target Species arrival noted here and by Dahr et al 2020, where cover of ruderal species does not appear to impact initial immigration of target species, there may be a valid explanation. A considerable number of the first native species to be observed on reclaimed sites arise from propagules embedded in reclamation soils (e.g. Melnik 2017). Beyond physical disturbance of and damage to those propagules, it seems plausible that competition from intentionally seeded and fertilized cover crops reaches its highest severity in the first year or two after seeding. This corresponds with the timing of native species emergence from soil legacies. If so, it is unlikely that the data used in this study would have had the temporal resolution to detect this effect: the first assessments would have been completed before the cover crop reached full development, and in most cases the second assessment would have occurred 3 to 5 years later.

Likely influenced by the Hardy BBT work and other comparable observations at the time, a shift in practices occurred in 1990, where grass mixes were predominantly replaced with barley as a temporary cover crop. The primary rationale for this change was based on the aforementioned recognition that grass mixes being used were competing with planted trees, often to the detriment of tree survival. The barley had a limited ability to reproduce in the local climate, and largely disappeared after the first growing season. While this change was primarily targeted at tree growth, there may have also been complementary effect on native plant emergence from soil propagules.

The remaining changes to practices were not so abrupt and evolved over a longer period. The use of broadcast fertilization waned through the 2000’s, as did the application of harrowing or other even more intensive tilling practices. In the early 2010’s the practice of “rough-and-loose” soil placement became prevalent, with the intended purpose of improving water penetration for erosion control, facilitating wind-blown seed capture, and creating nooks and cavities which would be ideal germination environments for the seeds of many species. It is also possible that these practices provide protection of captured seed from predation by birds and rodents. Some of these shifts were supported by targeted research programs, such as a trial which confirmed the suspected negative impacts of broadcast fertilizers on plant community composition (Sloan and Jacobs 2013). Many others changes have arisen simply through observation and adaptation by operational staff.

While all of the nuances of cause and effect have not been clarified, it seems plausible that this 40-year shift in practices has moved us away from unintentionally inhibiting the arrival of native species to new approaches that favour such immigration. Regardless of the causes, trends observed in the long term data on these sites imply that rates of establishment for native species are increasing over time for all variations on practices-of-the-day. For reclamation established in later time periods, Target Species appear to be arriving at a faster pace than in previous decades.

The changes in community development trends over time also raise a warning flag for interpreting earlier studies, and for researchers conducting further studies in the future. Previous studies (i.e. Rowland 2008, Pinno and Hawkes 2015, Dhar et al 2020) have employed a chronosequence approach to evaluating community development trends, where the older reclamation ages are represented exclusively by older reclaimed sites, and younger reclamation ages by a mixture of old and new reclamation. Such analyses implicitly depend on the assumption of statistical stationarity, or no underlying patterns of unmeasured effects. It is now evident that such stationarity does not exist, and it would be prudent to revisit the chronosequence-based analyses and outcomes of those earlier studies. While the general trends in community convergence for those studies will not likely be affected, some of the conclusions regarding causal influences might be.

10.3 EFFECTS OF ECOSITE ON DEVELOPING COMMUNITIES

While there is a general expectation that plant communities will reflect underlying edaphic conditions (soil moisture and nutrient gradients), the results here echo results of earlier studies (Stantec 2009 & 2011, Pinno and Hawkes 2015) suggesting that such distinctions are not yet evident on reclaimed sites. Within the range of soil moisture conditions that could be quantified for this study, there does not appear to be a detectible effect of moisture regime on the directionality of trends toward communities typical of either b or d ecosites. Additionally, the overall trend is for greater similarity to the reference communities for d ecosites than for b ecosites. While it is still expected that moisture regime will ultimately have a considerable impact on reclaimed floristic communities, such effects are not evident at this time. In comparing the composition differences for b and d ecosites in the reference data, some possible reasons emerge:

1. There is considerable overlap in species occurrence between b and d ecosites; of 193 native species observed in the reference data set, 93 were common to both, and another 96 occurred in ≤ 10% of the plots for the ecosite to which they were unique (Table A-1 in Appendix 1).
2. For those species that are sufficiently prevalent to be classed as Characteristic Species for either b or d ecosites, 22 of 52 are characteristic for both, and 47 of 52 can be found in both ecosites to varying degrees.
3. There are relatively few species which are typically present with high cover on reference b ecosites, and the lower cover levels of these species currently observed on reclaimed sites are closer to the cover levels typical for reference d ecosites (impacting particularly the Bray-Curtis index).
4. The species list for b ecosites is shorter than for d ecosites, and a higher proportion of these appear to have delayed or no appearance as of yet on reclaimed sites; of particular note are common blueberry (*Vaccinium myrtilloides*), green reindeer lichen (*Cladonia mitis*), and smooth cladonia (*Cladonia gracilis*).
5. The larger list of Characteristic Species for reference d ecosites suggests a higher likelihood of a match for reclaimed species that have arrived through stochastic processes not tightly associated with edaphic conditions.

Overall, there are few (if any) cases where the presence, absence or prevalence of a single species or even a careful selection of indicator species can definitively distinguish between b and d ecosites, even for the reference plots, and developers of field guides to ecosystem classification both in Alberta never expected that to be the case. Instead, reliance within those guides is placed on (i) the total collection of Characteristic Species coupled with (ii) a whole suite of non-biotic indictors.

In retrospect, given:

1. the early prominence on reclaimed sites of species that are not necessarily derived from biological legacies from the same location (Figures 5 and 11),
2. the initial establishment and (in many cases) persistence of native species from propagules arriving as a component of reclamation soils, which may or may not be sourced from equivalent ecosites, and
3. the fact that the developing communities have not yet experienced a full cycle of seral stage development and the associated suite of evolving environmental conditions (and particularly old seral understorey growing environments), such that the reproductive and competitive strategies for any one species that make it particularly suited to that location have not yet had a chance to be fully expressed,

it should not be surprising that insufficient divergence of community composition has yet occurred to clearly distinguish which reclaimed plots may trend more toward one late seral community composition than another over the longer term.

This issue also brings into question the strong reliance placed on Characteristic Species for the purpose of classifying reclaimed land to ecosite as is anticipated in Alberta Environment (2010). For the non-mined landscape, authors including Strong (2004), Timberline (2006) and GDC and FORRX (2008) have noted minimal variation in species presence for local boreal plant communities through natural cycles of seral stages. This suggests a strong legacy effect for species composition, where the species most common to old seral stages will persist in younger stages after disturbances such as fire. The recognizable community composition for discrete ecosites is not then simply a function of physical edaphic conditions, but the cumulative effect of species level adaptations to edaphic conditions and seral stage environments over multiple disturbance cycles. If this is true, it seems unreasonable to expect a comparable degree of interpretive value for Characteristic Species on reclaimed sites as for un-mined landscapes, where on reclaimed sites they have experienced only the earliest portions of a single seral stage sequence.

10.4 VALIDATION OF CHARACTERISTIC SPECIES THRESHOLDS FOR CERTIFICATION

While the original delineation of Characteristic Species was intended for the identification of ecosystem classes (such as ecosites as described by Beckingham and Archibald 1996), an alternate use for the regulatory process of reclamation certification was suggested in Alberta Environment (2010). It was proposed mean plot level counts of Characteristic Species, taken at a relatively early age, would be a meaningful indicator of longer term achievements for species richness. Alberta Environment (2010) develops this further by proposing a set of minimum threshold values for Characteristic Species counts as a pass/fail test.

A key assumption in this approach to evaluating reclamation performance is that Target Species richness will reliably continue to increase after the initial (and early) assessment. In order to validate this assumption Alberta Environment (2010) suggests that repeated measures monitoring would be needed to quantify trends for continued immigration of Characteristic Species over time. While Syncrude's monitoring methodology is different from that suggested by Alberta Environment (2010), the results provide exactly the evidence that was envisioned in that document: that (i) the occurrence of Target and particularly Characteristic Species increases with increasing reclamation age, and (ii) there is minimal risk that low numbers of species assessed at an early age will remain static at those low levels.

10.5 RISKS RELATED TO EXOTIC SPECIES

For the majority of reclaimed land, the evidence from this study suggests that exotic species and particularly noxious weeds are not having a meaningful inhibitory effect on native plant community development (see section 9.2 for exceptions). These findings are consistent with those of Dhar et al (2020) who studied similar data and trends based on community assembly theory. While not conclusive, there is reasonably strong evidence provided by this study that a fully developed tree canopy (or other overhead shade such as tall shrubs) will provide effective control of weeds on reclaimed lands.

In the short term, if stronger evidence is required for the effective control of weeds by a forest canopy before a larger number of monitoring plots reach crown closure, it is recommended that a more extensive program of temporary sample plots would be an appropriate approach rather than establishment of additional long term monitoring plots. There are several hundred hectares of reclaimed land with available that have reached crown closure, with many variation in canopy species composition. These sites could be characterized for the presence/absence of weeds relative to the canopy conditions that exist.

10.6 IMPLICATIONS FOR OPERATIONAL PRACTICES

While the observed species composition of reclaimed ecosystems remains different from that of old seral reference sites, there are far more native species present than can be attributed to artificial regeneration practices (planting) alone. A key assumption of reclamation practice is that most Target Species will arrive on reclaimed land through natural processes, and to a large degree this study validates that assumption. Syncrude has only ever planted less than 30 species in total, and less than 20 commonly. This means that roughly 90 late seral Target Species and 100 other native boreal species have arrived completely through natural processes.

While historical changes to practices appear to have brought significant improvements to reclamation outcomes over the past four decades, there may still be opportunity for further adjustments. It could be argued that Characteristic Species which continue to be absent or rare on reclaimed sites (Table 3) would be the highest priority, if any, for any future research. Of these, the quickest gains would likely be for the tree species, and particularly black spruce (*Picea mariana*), for which nursery and outplanting best practices should be well known. However, for balsam fir a reliable source of seed conforming to Alberta's genetic conservation guidelines (Alberta 2016) may continue to be troublesome. This is because groves of large trees capable of producing cones are not common in the vicinity of the mine sites.

Table 3. Comments on state of knowledge for Characteristic Species associated with b and d ecosites that are uncommon on Syncrude's reclaimed areas

	Species	b	d	Comments
Tree	*Abies balsamea*		✓	Unreliable or unknown seed supply
	Picea mariana		✓	Nursery production and planting well understood.
Shrub	*Corylus cornuta*		✓	Collection of mature seeds has been problematic due to predation by squirrels; use of vegetative cuttings being investigated.
	Rubus pubescens		✓	Minimal propagation experience from horticulture; seed germination inconsistent in oil sands trials; seed may be difficult to collect in any quantity
	Symphoricarpos albus		✓	Active research ongoing into nursery and outplanting best practices; seed germination inconsistent
	Vaccinium myrtilloides	✓		Active research ongoing into nursery and outplanting best practices
	Vaccinium vitis-idaea	✓		Preliminary research underway
	Viburnum edule		✓	Active research ongoing into nursery and outplantng best practices; seed germination and seedling growth inconsistent
Forb	*Aralia nudicaulis*	✓	✓	Some propagation experience from horticulture; uncertain seed supply
	Cornus canadensis	✓	✓	Preliminary research underway; preliminary monitoring for vegetative spread of established patches
	Mitella nuda		✓	Limited research exists into propagation; uncertain seed supply
Lichen	*Cladonia mitis*	✓		Local research suggests technical feasibility of transplantation through spreading of fragments.
	Cladonia rangiferina	✓		
	Cladonia stellaris	✓		

Many of the shrub species that still have minimal occurrence on reclaimed sites are already the subject of active research by oil sands operators, with the objective of developing reliable best practices for propagation. Less active is work on the forb and lichen species, although lichens in general are believed to commonly regenerate from windblown fragments as has been demonstrated in the oil sands for *Cladonia mitis* (Duncan 2004). The berry producing shrubs have historically been given research priority due to their importance as food sources for wildlife and, in many cases, due to their importance as traditional foods for local First Nations.

Beyond artificial regeneration practices (nursery production and planting), potential may also exist to further facilitate natural immigration rates. For example:

- Hardy BBT (1990) noted varying effects of surface soil organic content on ruderal species invasion,
- The reclaimed data set used in this report offers potential to provide further insights, including the effects of specific reclamation practices on occurrence of individual Target Species; this might help to determine if the occurrence of any one or a group of species could potentially be enhanced through selective application of certain practices, with a particular emphasis on reclamation soil materials and placement options.

While the previous portions of this section focus on human interventions, long term monitoring data such as that supporting this report can also point out where it is simply appropriate to wait. Even where some species are uncommon on reclaimed sites, examples can be found where they do exist and are subjectively growing quite well. This in turn implies that these species can do well on reclaimed sites, but their typical vectors of reproduction result in slow rates of spread. Examples include Labrador tea (*Rhododendron groenlandicum*), gooseberries (*Ribes* spp), baneberry (*Actaea rubra*), bunchberry (*Cornus canadensis*), wild sarsaparilla (*Aralia nudicaullis*) and many common forest floor mosses and lichens. For some of these species, the most appropriate strategy beyond current practices may be patience.

11 CONCLUSIONS

The objectives of this study were to:

1. evaluate the degree to which discrete locations on reclaimed lands have understorey plant communities that are becoming increasingly similar over time to locations within the adjacent un-mined landscapes,
2. make observations on the degree to which reclamation outcomes are predictable based on conventional ecological theory, and
3. discuss how observations of existing community trends can contribute to continuous improvement of reclamation practices.

The primary conclusion of the study is that patterns of plant community change for reclaimed sites are consistent with Alberta's objectives for reclamation, which require increasing similarity between reclaimed and reference ecosystem structure over time. In particular, reclaimed plant communities appear to have accelerating rates of development toward similarity with the selected reference conditions within the period of monitoring, although in many cases these rates will likely be interrupted for one or more decades by the onset of closed forest canopy conditions. While the initial composition after reclamation establishment is dominated by a large number of species not common to boreal forests, these do not appear to be inhibiting the arrival of Target Species which are increasing in diversity and cover while the early ruderal species are decreasing. It is clear that full convergence has not yet been reached and will not be for several decades. However, for all metrics employed in this study there are at least some sample plots that have entered the NRV for old seral reference samples, and in some cases the mean for all reclaimed plots has entered the NRV.

Following on the second objective of the study, reclaimed plant communities appear to be following reasonably predictable patterns of change consistent with theoretical models. While reclaimed floristics in the first decade or two after mining are quite different from that found after stand replacing disturbances on other local landscapes, this is entirely expected given the complete removal of biological legacies imposed by surface mining. Despite large differences in starting conditions when compared to un-mined sites, reclaimed sites are still demonstrating a steady progression to replace those legacies with new ingress of species typical found in locally common ecosystems. Also as expected, the development of a tree canopy is demonstrated to have a strong and desirable influence on plant community trends.

These outcomes related to the first two objectives of the study do not (and were not intended to) directly support the regulatory requirement of either setting or meeting certification thresholds. Instead, they (i) provide confidence that theoretical models of plant community development are applicable to reclaimed lands and that reclaimed plant communities are not developing in a remarkably novel manner, and (ii) help validate indicators and measures that are used directly for certification. From a regulatory perspective, this gives us confidence that relatively early assessments of plant communities as part of a certification process are viable, with a low risk of long term deviations from expected community development patterns.

The primary conclusion related to the third objective looks not to the future but to the past. While not conclusive, perceived increases in the rate of plant community development over the last four decades are consistent with expectations based on evolving reclamation practices during the same period.

Beyond these primary findings there are a number of other observations and conclusions for which supporting data and discussions are provided in the report:

1. There appears to be a lag period before Target Species begin to arrive in substantial numbers. This lag is most apparent for the earliest reclamation and has largely disappeared for the most recent reclamation. While multiple explanations may exist, the most likely appears to be a move away from intensive agronomic approaches for

reclamation, to those which better conserve plant propagules in reclamation soils and better facilitate native species immigration.

2. The observations from point 1 above indicate that chronosequence approaches to analyzing reclaimed plant community trends may not be appropriate for the oil sands region of northern Alberta. This approach requires an assumption of statistical stationarity, which for community analysis means that all case studies that are linked into a composite time sequence can be assumed to have acceptably similar starting conditions. This is not the case for reclamation monitoring data where different time periods are associated with different initial conditions.
3. With the possible (and speculated) exception of seeded and fertilized agronomic crops, there is no evidence that ruderal species and particularly noxious weeds are negatively impacting the initial arrival of Target Species on reclaimed sites.
4. Consistent with many forested communities, there may be some degree of interruption to the rates of forest dependent species arrival and proliferation on reclaimed sites as sites reach canopy closure. In extreme cases, the % cover of all understorey species will be temporarily reduced, including that for Target Species.
5. The trends observed in this study provide evidence to validate the concept of using Characteristic Species, measured at a relatively early age, as an indicator of potential Target Species diversity within a Criteria and Indicators I framework for reclamation certification. However, analyses to validate the proposed thresholds for this indicator have not yet been completed.
6. At the current stage of development, Characteristic Species and reclaimed community composition appear to be relatively weak indicators for discriminating between b and d ecosites. It is speculated that exposure to a wider range of seral stages (and possibly even disturbance cycles) will be required in order for Characteristic Species to become stronger indicators of ecosite on reclaimed lands.

While it is already shown in this report that upland reclaimed communities demonstrate promising patterns of convergence towards their reference counterparts, results presented here are not the end of possible learnings from Syncrude’s long term vegetation monitoring program. None of these analyses can be considered as definitive, they simply add to the body of previously existing evidence. The degree to which further analyses using additional techniques or additional years of monitoring data will be needed depends on any one person or group’s particular questions and degree of confidence in the existing evidence. Where the demands for refinement to this knowledge base justify the effort, there remains further potential to inform incremental improvements to reclamation practice and policy.

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APPENDIX A: SPECIES OCCURRENCE TABLES

Table A-1. List of species occurring on b and d ecosites in the reference data set (old seral communities within 200 km of Syncrude's mine site). The list is sorted by vegetation layer (trees, shrubs, forbs, grasses/sedges, mosses, liverworts and lichens); within layers species are sorted by overall mean frequency of occurrence by plot (proportion of plots the species was found in) within the reference data. Nomenclature follows that used in the Alberta Conservation Information Management System (ACIMS) as of September 2019.

Latin Name	Common Name	Occurrence - Reference		Occurrence - Reclaimed
		b	d	
Trees				
Populus tremuloides	trembling aspen	1.000	0.941	0.615
Picea glauca	white spruce	0.588	0.824	0.495
Betula papyrifera	paper birch	0.412	0.441	0.110
Abies balsamea	balsam fir	0.118	0.294	0.000
Populus balsamifera	balsam poplar	0.059	0.206	0.410
Picea mariana	black spruce	0.471	0.088	0.000
Pinus banksiana	jack pine	0.529	0.059	0.195
Larix laricina	tamarack	0.000	0.015	0.025
Shrubs				
Linnaea borealis	twinflower	0.765	0.779	0.020
Viburnum edule	low-bush cranberry	0.412	0.868	0.020
Rosa acicularis	prickly rose	0.353	0.676	0.560
Rubus pubescens	dewberry	0.176	0.721	0.090
Vaccinium vitis-idaea	bog cranberry	1.000	0.309	0.025
Vaccinium myrtilloides	velvetleaf blueberry	0.941	0.250	0.055
Rosa woodsii	woods' rose	0.588	0.250	0.045
Rhododendron groenlandicum	common labrador-tea	0.647	0.206	0.065
Shepherdia canadensis	Canada buffaloberry	0.294	0.279	0.310
Salix bebbiana	Bebb's willow	0.412	0.206	0.095
Amelanchier alnifolia	Saskatoon	0.059	0.265	0.325
Lonicera involucrata	bracted honeysuckle	0.118	0.250	0.000
Ribes triste	wild red currant	0.059	0.221	0.035
Rubus idaeus	common red raspberry	0.000	0.235	0.655
Alnus viridis	green alder	0.294	0.118	0.065
Alnus incana	speckled alder	0.059	0.132	0.005
Arctostaphylos uva-ursi	bearberry	0.353	0.044	0.155
Cornus stolonifera	red-osier dogwood	0.000	0.132	0.495
Symphoricarpos albus	snowberry	0.059	0.118	0.035
Ribes lacustre	bristly black currant	0.000	0.103	0.060
Ribes oxyacanthoides	Canadian gooseberry	0.000	0.074	0.110
Corylus cornuta	beaked hazelnut	0.000	0.059	0.000
Prunus pensylvanica	pin cherry	0.176	0.015	0.010
Symphoricarpos occidentalis	northern snowberry	0.000	0.029	0.035
Vaccinium caespitosum	dwarf huckleberry	0.059	0.015	0.000
Betula glandulosa	tundra dwarf birch	0.059	0.000	0.010

Latin Name	Common Name	Occurrence - Reference		Occurrence - Reclaimed
		b	d	
Ribes glandulosum	skunk currant	0.000	0.015	0.065
Ribes hudsonianum	northern black currant	0.000	0.015	0.030
Salix discolor	pussy willow	0.000	0.015	0.000
Salix glauca	smooth willow	0.000	0.015	0.100
Salix pyrifolia	balsam willow	0.059	0.000	0.010
Salix scouleriana	Scouler's willow	0.000	0.015	0.005
Salix serissima	autumn willow	0.000	0.015	0.070
Forbs				
Cornus canadensis	bunchberry	0.765	0.824	0.040
Chamerion angustifolium	fireweed	0.765	0.721	0.850
Petasites frigidus	coltsfoot	0.412	0.676	0.145
Maianthemum canadense	wild lily-of-the-valley	0.647	0.544	0.100
Mitella nuda	bishop's-cap	0.235	0.632	0.010
Lathyrus ochroleucus	creamy vetchling	0.588	0.456	0.350
Fragaria virginiana	wild strawberry	0.412	0.485	0.540
Lysimachia latifolia	northern starflower	0.471	0.456	0.085
Pyrola asarifolia	pink wintergreen	0.176	0.515	0.195
Aralia nudicaulis	wild sarsaparilla	0.353	0.456	0.010
Mertensia paniculata	tall bluebells	0.059	0.500	0.085
Equisetum sylvaticum	woodland horsetail	0.118	0.426	0.075
Galium boreale	northern bedstraw	0.118	0.412	0.240
Orthilia secunda	one-sided wintergreen	0.412	0.324	0.015
Lycopodium annotinum	stiff clubmoss	0.176	0.309	0.005
Vicia americana	American purple vetch	0.176	0.265	0.610
Achillea millefolium	common yarrow	0.176	0.235	0.710
Viola renifolia	kidneyleaf white violet	0.118	0.250	0.010
Geocaulon lividum	northern comandra	0.294	0.162	0.000
Symphyotrichum ciliolatum	Lindley's aster	0.059	0.221	0.510
Actaea rubra	baneberry	0.000	0.176	0.010
Equisetum pratense	meadow horsetail	0.235	0.118	0.115
Galium triflorum	sweet-scent bedstraw	0.000	0.162	0.060
Goodyera repens	dwarf rattlesnake-plantain	0.059	0.132	0.040
Equisetum arvense	common horsetail	0.059	0.118	0.650
Eurybia conspicua	showy aster	0.059	0.118	0.315
Lonicera dioica	twining honeysuckle	0.059	0.059	0.030
Pedicularis labradorica	Labrador lousewort	0.235	0.015	0.000
Platanthera orbiculata	lesser roundleaf orchid	0.059	0.044	0.000
Viola canadensis	Canada violet	0.059	0.044	0.020
Pyrola chlorantha	green-flower wintergreen	0.059	0.029	0.010
Spiranthes romanzoffiana	hooded ladies'-tresses	0.000	0.044	0.005
Streptopus amplexifolius	clasping twisted-stalk	0.059	0.029	0.000
Delphinium glaucum	tall larkspur	0.000	0.029	0.000
Hedysarum boreale	boreal sweet-vetch	0.059	0.015	0.000
Lathyrus venosus	purple peavine	0.000	0.029	0.105
Saxifraga tricuspidata	prickly saxifrage	0.118	0.000	0.000
Thalictrum venulosum	veined meadowrue	0.000	0.029	0.010
Achillea alpina	Siberian yarrow	0.000	0.015	0.080
Anemone canadensis	Canada anemone	0.000	0.015	0.020
Apocynum androsaemifolium	spreading dogbane	0.000	0.015	0.000
Astragalus americanus	American milkvetch	0.000	0.015	0.000

Latin Name	Common Name	Occurrence - Reference		Occurrence - Reclaimed
		b	d	
Campanula rotundifolia	American harebell	0.059	0.000	0.020
Cirsium hookerianum	Hooker's thistle	0.059	0.000	0.000
Cypripedium acaule	pink lady's-slipper	0.000	0.015	0.000
Equisetum scirpoides	dwarf scouring-rush	0.000	0.015	0.000
Helenium autumnale	common sneezeweed	0.000	0.015	0.000
Heracleum maximum	cow-parsnip	0.000	0.015	0.005
Hieracium umbellatum	narrow-leaved hawkweed	0.000	0.015	0.165
Lilium philadelphicum	wood lily	0.059	0.000	0.020
Microseris nutans	nodding silverpuffs	0.059	0.000	0.000
Moneses uniflora	one-flowered wintergreen	0.000	0.015	0.000
Monotropa uniflora	indian-pipe	0.059	0.000	0.000
Platanthera obtusata	small northern bog orchid	0.000	0.015	0.000
Thalictrum occidentale	western meadowrue	0.000	0.015	0.000
Tiarella trifoliata	lace foamflower	0.000	0.015	0.000
Urtica dioica	stinging nettle	0.000	0.015	0.205
Viola adunca	sand violet	0.000	0.015	0.035
Ferns				
Gymnocarpium dryopteris	northern oak fern	0.000	0.029	0.000
Matteuccia struthiopteris	ostrich fern	0.000	0.015	0.000
Grasses/Sedges				
Calamagrostis canadensis	blue-joint reedgrass	0.294	0.588	0.400
Leymus innovatus	hairy wildrye	0.647	0.338	0.080
Schizachne purpurascens	purple oat	0.000	0.059	0.015
Melampyrum lineare	cow-wheat	0.059	0.015	0.000
Carex foenea	bronze sedge	0.000	0.015	0.025
Cinna latifolia	slender wood reedgrass	0.000	0.015	0.040
Elymus trachycaulus	slender wild rye	0.000	0.015	0.230
Oryzopsis asperifolia	white-grained mountain-ricegrass	0.059	0.000	0.005
Mosses				
Pleurozium schreberi	red-stemmed feather moss	0.882	0.809	0.160
Hylocomium splendens	stairstep moss	0.706	0.765	0.235
Ptilium crista-castrensis	knight's plume moss	0.176	0.412	0.050
Dicranum polysetum	wavy-leaved broom moss	0.235	0.147	0.000
Diphasiastrum complanatum	trailing clubmoss	0.353	0.088	0.000
Evernia mesomorpha	boreal oakmoss lichen	0.118	0.118	0.005
Plagiomnium cuspidatum	woodsy leafy moss	0.059	0.103	0.120
Pylaisiella polyantha	many-flowered pylaisia moss	0.059	0.103	0.100
Eurhynchiastrum pulchellum	elegant beaked moss	0.059	0.088	0.210
Brachythecium salebrosum	golden ragged moss	0.000	0.088	0.045
Lycopodium dendroideum	treelike clubmoss	0.059	0.074	0.005
Pohlia nutans	common nodding moss	0.000	0.088	0.370
Dicranum fuscescens	dusky fork moss	0.000	0.074	0.000
Orthotrichum obtusifolium	blunt-leaved bristle moss	0.000	0.074	0.000
Sanionia uncinata	sickle moss	0.000	0.074	0.175
Lycopodium lagopus	one-cone ground-pine	0.118	0.029	0.000
Mnium spinulosum	red-mouthed leafy moss	0.000	0.059	0.000
Polytrichum juniperinum	juniper haircap moss	0.000	0.059	0.060
Tomentypnum nitens	golden fuzzy fen moss	0.000	0.059	0.075
Aulacomnium palustre	ribbed bog moss	0.059	0.029	0.110
Ceratodon purpureus	red roof moss	0.000	0.044	0.280

Latin Name	Common Name	Occurrence - Reference		Occurrence - Reclaimed
		b	d	
Oncophorus wahlenbergii	Wahlenberg's spur moss	0.000	0.044	0.000
Orthotrichum speciosum	showy bristle moss	0.000	0.044	0.000
Plagiomnium drummondii	Drummond's leafy moss	0.000	0.044	0.000
Campylophyllum hispidulum	common fine wet moss	0.000	0.029	0.000
Haplocladium microphyllum	tiny-leaved haplocladium moss	0.000	0.029	0.000
Polytrichum commune	common hair cap moss	0.118	0.000	0.000
Amblystegium serpens	amblystegium moss	0.000	0.015	0.000
Brachythecium rivulare	waterside feather moss	0.059	0.000	0.000
Dicranum fragilifolium	fragile-leaved broom moss	0.000	0.015	0.000
Plagiomnium ciliare	wavy-leaf moss	0.000	0.015	0.000
Polytrichum strictum	bog haircap moss	0.000	0.015	0.005
Sciuro-hypnum starkei	starks ragged moss	0.000	0.015	0.000
Liverworts				
Ptilidium pulcherrimum	naugehyde liverwort	0.000	0.074	0.000
Jamesoniella autumnalis	Jameson's liverwort	0.000	0.015	0.000
Lepidozia reptans	creeping fingerwort	0.000	0.015	0.000
Lophocolea minor	lesser crestwort	0.000	0.015	0.000
Lichens				
Peltigera aphthosa	common freckle pelt	0.647	0.309	0.005
Hypogymnia physodes	monk's-hood lichen	0.471	0.265	0.000
Arthonia patellulata	aspen comma	0.353	0.235	0.000
Usnea alpina	subalpine beard lichen	0.294	0.191	0.000
Cladonia mitis	green reindeer lichen	0.588	0.059	0.030
Parmelia sulcata	hammered shield lichen	0.176	0.132	0.005
Tuckermannopsis platyphylla	broad wrinkle lichen	0.294	0.103	0.000
Usnea hirta	bristly beard lichen	0.176	0.132	0.005
Bryoria glabra	shiny horsehair lichen	0.294	0.088	0.000
Peltigera polydactylon	many-fruited pelt lichen	0.059	0.103	0.000
Vulpicida pinastri	powdered sunshine lichen	0.176	0.074	0.000
Cladonia chlorophaea	mealy pixie-cup lichen	0.000	0.103	0.035
Cladonia fimbriata	trumpet lichen	0.059	0.088	0.000
Caloplaca holocarpa	firedot lichen	0.235	0.029	0.000
Cladonia coniocraea	common powderhorn lichen	0.000	0.088	0.065
Peltigera canina	dog lichen	0.059	0.074	0.000
Usnea lapponica	powdered beard lichen	0.059	0.074	0.005
Bryoria fuscescens	pale-footed horsehair lichen	0.000	0.074	0.000
Cladonia gracilis	smooth cladonia	0.235	0.015	0.020
Ramalina pollinaria	powdery twig lichen	0.000	0.074	0.000
Cladonia coccifera	madame pixie lichen	0.118	0.015	0.000
Cladonia pyxidata	pebbled pixie-cup lichen	0.059	0.029	0.045
Cladonia stellaris	star-tipped reindeer lichen	0.118	0.015	0.000
Alectoria sarmentosa	common witch's hair lichen	0.059	0.015	0.000
Cladonia cervicornis	browned pixie-cup lichen	0.000	0.029	0.000
Cladonia cornuta	bighorn pixie lichen	0.000	0.029	0.015
Cladonia deformis	lesser sulphur-cup lichen	0.118	0.000	0.000
Cladonia multiformis	sieve lichen	0.000	0.029	0.065
Cladonia squamosa	dragon cladonia	0.118	0.000	0.000
Parmeliopsis hyperopta	gray starburst lichen	0.059	0.015	0.000
Peltigera rufescens	felt lichen	0.000	0.029	0.000
Peltigera scabrosa	greater toad pelt lichen	0.000	0.029	0.000

Latin Name	Common Name	Occurrence - Reference		Occurrence - Reclaimed
		b	d	
Cladonia botrytes	wooden soldiers lichen	0.000	0.015	0.040
Cladonia cenotea	powdered funnel lichen	0.000	0.015	0.000
Cladonia ecmocyna	frosted cladonia	0.000	0.015	0.000
Cladonia rangiferina	gray reindeer lichen	0.059	0.000	0.000
Lecanora impudens		0.000	0.015	0.000
Melanelixia albertana	powder-rimmed camouflage lichen	0.059	0.000	0.005
Nephroma parile	powdery kidney lichen	0.000	0.015	0.000
Parmeliopsis ambigua	green starburst lichen	0.000	0.015	0.000
Peltigera elisabethae	concentric pelt lichen	0.000	0.015	0.000
Peltigera horizontalis	flat-fruited pelt lichen	0.000	0.015	0.000
Peltigera lepidophora	scaly pelt lichen	0.000	0.015	0.000
Ramalina dilacerata	punctured ribbon lichen	0.000	0.015	0.000
Ramalina sinensis	fan ramalina	0.000	0.015	0.000
Ramalina thrausta	angel's-hair lichen	0.000	0.015	0.000
Usnea cavernosa	pitted beard lichen	0.000	0.015	0.000
Xanthomendoza fallax	hooded sunburst lichen	0.000	0.015	0.000

Table A-2. List of species occurring only in the reclaimed data set. The list is sorted by vegetation layer (trees, shrubs, forbs, grasses/sedges, mosses, liverworts and lichens); within layers species are sorted by frequency of occurrence by plot (proportion of plots the species was found in). Also provided is a species group classification indicating whether the species is typically found on (i) sites in the boreal forests of northern Alberta other than those typified by the reference data, (ii) other locations in Alberta, or (iii) are considered as species that are not native to Alberta based on notations in the Alberta Conservation Information Management System (ACIMS). Nomenclature follows that used in ACIMS as of September 2019

Latin Name	Common Name	Occurrence in Reclaimed Plots	Species Class
Trees			
Larix siberica	Siberian larch	0.095	Exotic
Populus x	hybrid poplar	0.010	Exotic
Shrubs			
Salix candida	hoary willow	0.035	Other Boreal
Salix pseudomonticola	false mountain willow	0.005	Other Boreal
Salix exigua	narrow-leaf willow	0.045	Other Boreal
Rubus chamaemorus	cloudberry	0.020	Other Boreal
Caragana arborescens	common caragana	0.055	Exotic
Betula pumila	dwarf birch	0.035	Other Boreal
Prunus virginiana	choke cherry	0.010	Other Boreal
Rubus arcticus	dwarf raspberry	0.025	Other Boreal
Salix planifolia	flat-leaved willow	0.035	Other Boreal
Forbs			
Cirsium vulgare	bull thistle	0.010	Exotic
Comarum palustris	marsh cinquefoil	0.005	Other Boreal
Equisetum hyemale	common scouring-rush	0.005	Other Boreal
Erigeron canadensis	horseweed	0.025	Other Native
Symphyotrichum puniceum	purple-stemmed aster	0.104	Other Boreal
Stachys pilosa	marsh hedge-nettle	0.030	Other Boreal
Silene latifolia	white cockle, bladder campion	0.005	Exotic
Ranunculus macounii	Macoun's buttercup	0.010	Other Boreal
Rorippa islandica	marsh yellow cress	0.015	Exotic
Persicaria maculosa	lady's-thumb	0.010	Exotic
Chenopodium album	lamb's-quarters	0.368	Exotic
Descurainia sophia	flixweed	0.035	Exotic
Melilotus officinalis	yellow sweet-clover	0.393	Exotic
Capsella bursa-pastoris	shepherd's-purse	0.005	Exotic
Kochia scoparia	summer-cypress	0.015	Exotic
Thlaspi arvense	stinkweed	0.005	Exotic
Polygonum aviculare	prostrate knotweed	0.010	Exotic
Rumex acetosella	sheep sorrel	0.005	Exotic
Rumex occidentalis	western dock	0.025	Other Boreal
Sibbaldia tridentata	three-toothed cinquefoil	0.020	Other Boreal
Lysimachia thyrsiflora	tufted loosestrife	0.010	Other Boreal
Lotus corniculatus	bird's-foot trefoil	0.532	Exotic
Lepidium densiflorum	common pepper-grass	0.015	Other Native
Fallopia convolvulus	wild buckwheat	0.005	Exotic
Solidago lepida	elegant goldenrod	0.264	Other Boreal
Medicago sativa	alfalfa	0.731	Exotic
Medicago falcata	yellow lucerne	0.090	Exotic
Symphyotrichum boreale	marsh aster	0.005	Other Boreal
Moehringia lateriflora	blunt-leaved sandwort	0.104	Other Boreal
Arabis eschscholtziana	Eschscholtz's rockcress	0.010	Other Native
Aquilegia brevistyla	blue columbine	0.025	Other Boreal
Potentilla pensylvanica	prairie cinquefoil	0.010	Other Boreal
Taraxacum officinale	common dandelion	0.781	Exotic
Circaea alpina	small enchanter's nightshade	0.005	Other Boreal
Sisyrinchium montanum	common blue-eyed grass	0.005	Other Boreal
Erigeron acris	northern daisy fleabane	0.010	Other Native
Oxytropis monticola	late yellow locoweed	0.005	Other Native

Latin Name	Common Name	Occurrence in Reclaimed Plots	Species Class
Corydalis aurea	golden corydalis	0.030	Other Native
Oxytropis deflexa	reflexed locoweed	0.010	Other Boreal
Primula incana	mealy primrose	0.005	Other Native
Eleocharis palustris	creeping spike-rush	0.015	Other Boreal
Stellaria crassifolia	fleshy stitchwort	0.005	Other Boreal
Dracocephalum parviflorum	American dragonhead	0.045	Other Native
Erigeron ochroleucus	buff fleabane	0.005	Other Native
Castilleja raupii	purple paintbrush	0.015	Other Boreal
Comandra umbellata	common comandra	0.010	Other Boreal
Oxytropis campestris	northern locoweed	0.005	Other Native
Ranunculus abortivus	small-flowered buttercup	0.005	Other Native
Senecio vulgaris	common groundsel	0.070	Exotic
Tanacetum vulgare	common tansy	0.025	Exotic
Axyris amaranthoides	Russian pigweed	0.015	Exotic
Chenopodium capitatum	strawberry blite	0.035	Other Native
Plantago major	common plantain	0.030	Exotic
Aquilegia canadensis	Canada columbine	0.005	Exotic
Castilleja miniata	common red paintbrush	0.060	Other Boreal
Packera paupercula	balsam groundsel	0.075	Other Boreal
Erysimum cheiranthoides	wormseed mustard	0.065	Other Boreal
Parnassia palustris	northern grass-of-parnassus	0.035	Other Boreal
Heterotheca villosa	golden aster	0.005	Other Boreal
Symphyotrichum lanceolatum	western willow aster	0.005	Other Boreal
Malaxis monophyllos	white adder's-mouth	0.035	Other Native
Boechera collinsii	Collins' rockcress	0.005	Other Native
Phlox divaricata	wild blue phlox	0.015	Exotic
Scutellaria galericulata	marsh skullcap	0.065	Other Boreal
Maianthemum stellatum	star-flowered Solomon's-seal	0.045	Other Boreal
Antennaria neglecta	broad-leaved everlasting	0.020	Other Boreal
Platanthera huronensis	northern green bog orchid	0.010	Other Boreal
Erigeron philadelphicus	Philadelphia fleabane	0.005	Other Boreal
Astragalus canadensis	Canadian milkvetch	0.308	Other Native
Tephroseris palustris	marsh ragwort	0.005	Other Native
Trifolium hybridum	alsike clover	0.358	Exotic
Rumex salicifolius	narrow-leaved dock	0.015	Exotic
Tragopogon dubius	common goat's-beard	0.015	Exotic
Potentilla norvegica	rough cinquefoil	0.413	Other Boreal
Stellaria longifolia	long-leaved chickweed	0.164	Other Boreal
Potentilla gracilis	graceful cinquefoil	0.055	Other Boreal
Geum macrophyllum	large-leaved yellow avens	0.040	Other Boreal
Gentianella amarella	felwort	0.030	Other Boreal
Arnica chamissonis	leafy arnica	0.020	Other Boreal
Rhinanthus minor	yellow rattle	0.020	Other Boreal
Solidago simplex	sticky goldenrod	0.005	Other Boreal
Geranium bicknellii	Bicknell's geranium	0.060	Other Native
Melilotus alba	white sweet-clover	0.682	Exotic
Sonchus arvensis	perennial sow-thistle	0.542	Exotic
Cirsium arvense	creeping thistle	0.129	Exotic
Erodium cicutarium	common storksbill	0.015	Exotic
Artemisia absinthium	absinthe wormwood	0.005	Exotic
Fragaria vesca	woodland strawberry	0.224	Other Boreal
Mentha arvensis	wild mint	0.060	Other Boreal
Persicaria lapathifolia	pale persicaria	0.035	Other Boreal
Epilobium ciliatum	northern willowherb	0.025	Other Boreal
Geum aleppicum	yellow avens	0.025	Other Boreal
Dasiphora fruticosa	shrubby cinquefoil	0.020	Other Boreal
Chamaedaphne calyculata	leatherleaf	0.015	Other Boreal
Rorippa palustris	marsh yellow cress	0.010	Other Boreal
Viola nephrophylla	bog violet	0.010	Other Boreal

Latin Name	Common Name	Occurrence in Reclaimed Plots	Species Class
Geum rivale	purple avens	0.005	Other Boreal
Maianthemum trifolium	three-leaved Solomon's-seal	0.005	Other Boreal
Oxytropis sericea	early yellow locoweed	0.005	Other Boreal
Thermopsis rhombifolia	golden bean	0.134	Other Native
Plantago eriopoda	saline plantain	0.005	Other Native
Barbarea orthoceras	American winter cress	0.005	Other Native
Senecio eremophilus	cut-leaved ragwort	0.005	Other Native
Trifolium pratense	red clover	0.229	Exotic
Salsola tragus	Russian-thistle	0.139	Exotic
Erucastrum gallicum	dog mustard	0.104	Exotic
Sonchus asper	prickly annual sow-thistle	0.100	Exotic
Crepis tectorum	annual hawk's-beard	0.095	Exotic
Trifolium repens	white clover	0.080	Exotic
Galeopsis tetrahit	hemp-nettle	0.070	Exotic
Astragalus cicer	cicer milk vetch	0.060	Exotic
Sinapis arvensis	wild mustard	0.045	Exotic
Stellaria media	common chickweed	0.010	Exotic
Vaccaria pyramidata	cow cockle	0.010	Exotic
Ferns			
Botrychium lunaria	moonwort	0.040	Other Native
Botrychium virginianum	Virginia grape fern	0.035	Other Boreal
Grasses/Sedges			
Phalaris arundinacea	reed canary grass	0.020	Other Boreal
Carex foenea	silvery-flowered sedge	0.025	Other Native
Fagopyrum esculentum	common buckwheat	0.005	Exotic
Carex brunnescens	brownish sedge	0.015	Other Boreal
Bromus inermis	smooth brome	0.284	Exotic
Festuca rubra	red fescue	0.040	Other Boreal
Carex bebbii	Bebb's sedge	0.020	Other Boreal
Carex canescens	hoary sedge	0.015	Other Native
Agrostis scabra	rough hair grass	0.075	Other Boreal
Anthoxanthum hirtum	sweet grass	0.015	Other Native
Eriophorum angustifolium	narrowleaf cotton-grass	0.005	Other Boreal
Bromus ciliatus	fringed brome	0.060	Other Boreal
Elymus repens	quackgrass	0.134	Exotic
Carex concinna	beautiful sedge	0.035	Other Boreal
Agrostis stolonifera	redtop	0.085	Exotic
Carex utriculata	small bottle sedge	0.045	Other Boreal
Deschampsia cespitosa	tufted hair grass	0.040	Other Boreal
Calamagrostis stricta	narrow reed grass	0.005	Other Boreal
Poa palustris	fowl bluegrass	0.114	Other Boreal
Koeleria macrantha	June grass	0.015	Other Boreal
Carex atherodes	awned sedge	0.070	Other Boreal
Elymus canadensis	Canada wildrye	0.015	Other Native
Hordeum jubatum	foxtail barley	0.438	Other Boreal
Bromus tectorum	downy brome	0.005	Exotic
Poa pratensis	Kentucky bluegrass	0.308	Other Boreal
Carex siccata	hay sedge	0.119	Other Boreal
Carex aurea	golden sedge	0.020	Other Boreal
Calamagrostis purpurascens	purple reed grass	0.015	Other Boreal
Elymus lanceolatus	northern wheat grass	0.010	Other Boreal
Carex deweyana	Dewey's sedge	0.159	Other Native
Juncus balticus	wire rush	0.005	Other Boreal
Typha latifolia	common cattail	0.005	Other Boreal
Carex viridula	green sedge	0.040	Other Native
Phleum pratense	timothy	0.209	Exotic
Hordeum vulgare	cultivated barley	0.085	Exotic
Carex aquatilis	water sedge	0.040	Other Boreal
Carex vaginata	sheathed sedge	0.005	Other Boreal

Latin Name	Common Name	Occurrence in Reclaimed Plots	Species Class
Glyceria borealis	northern manna grass	0.005	Other Native
Dactylis glomerata	orchard grass	0.055	Exotic
Agropyron cristatum	crested wheatgrass	0.025	Exotic
Mosses/Lichens			
Cladonia verticillata	ladder lichen	0.005	Other Native
Thuidium delicatulum	moss	0.045	Other Native
Syntrichia ruralis	hairy screw moss	0.114	Other Native
Cladonia borealis	boreal pixie-cup	0.005	Other Native
Peltigera didactyla	alternating dog lichen	0.005	Other Boreal
Peltigera neopolydactyla	carpet pelt lichen	0.035	Other Native
Peltigera malacea	veinless pelt lichen	0.030	Other Boreal
Marchantia polymorpha	green-tongue liverwort	0.005	Other Boreal
Cladonia turgida	crazy-scale lichen	0.005	Other Native
Baeomyces rufus	brown beret lichen	0.010	Other Native
Candelaria concolor	lemon lichen	0.005	Other Boreal
Tuckermannopsis americana	fringed wrinkle lichen	0.005	Other Boreal
Thuidium recognitum	moss	0.164	Other Boreal
Leptobryum pyriforme	moss	0.045	Other Boreal
Dicranum undulatum	wavy dicranum moss	0.005	Other Boreal
Cratoneuron filicinum	moss	0.050	Other Boreal
Polytrichum piliferum	awned hair-cap moss	0.010	Other Boreal
Hypnum revolutum	moss	0.025	Other Boreal
Funaria hygrometrica	cord moss	0.010	Other Native
Climacium dendroides	moss	0.015	Other Boreal
Rhytidium rugosum	pipecleaner moss	0.015	Other Boreal
Cladonia cariosa	split-peg lichen	0.025	Other Native
Abietinella abietina	wiry fern moss	0.338	Other Boreal
Cladonia bellidiflora	floral pixie	0.010	Other Native
Cladonia crispata	organ-pipe lichen	0.010	Other Native
Cladonia cristatella	British soliders lichen	0.010	Other Native
Flavopunctelia flaventior	speckled greenshield lichen	0.010	Other Native

APPENDIX B: TESTING FOR POSSIBLE EFFECTS OF MEASUREMENT ERRORS

In order to test for potential effects of varied field crew expertise as described in Section 10.2, an analysis was completed to screen for changes in species detection rates at the transition point of 2010, where contract crews having more training and experience were employed as opposed to the earlier use of students hired on summer work terms. Concern was raised that prior to the transition, an underestimation of species presence may have occurred, and that this may have artificially created the appearance of some of the apparent trends found in this report. In particular,

1. many of the trends suggest an increased rate of new species detections in the last 10 to fifteen years, and the start of this period is roughly correlated to the 2010 transition year, and
2. any underestimation of species may have influenced the older plots more than the younger plots (there was a longer period of monitoring by less experienced crews for the older plots), resulting in the appearance of artificial differences between decades of establishment.

The effect being evaluated is illustrated conceptually in Figure B-1, where the difference between the two line colours represents any existing underestimation of species richness prior to 2010. Any errors that did exist are assumed to be corrected in the first assessment after 2010, where the more experienced crews would presumably detect a higher number of species on a consistent basis.

The general approach taken here for evaluating for potential effects of systemic measurement error is to isolate and contrast measurement intervals that are unaffected by the transition period. These are intervals for which both the start and end assessments were completed either by 1) the less experienced crews or (ii) the more experienced crews. In generating comparative statistics based on these intervals, intervals are treated as subsampling within plots as the subjects.

In testing for the first the first of these postulated outcomes listed above, the null hypothesis is that slopes (rates of new species arrival) are relatively constant over time, with the alternate hypothesis being that the mean slope for later intervals is greater than for earlier intervals. Looking at summary data provided in Table B-1, the mean slopes for the post-2010

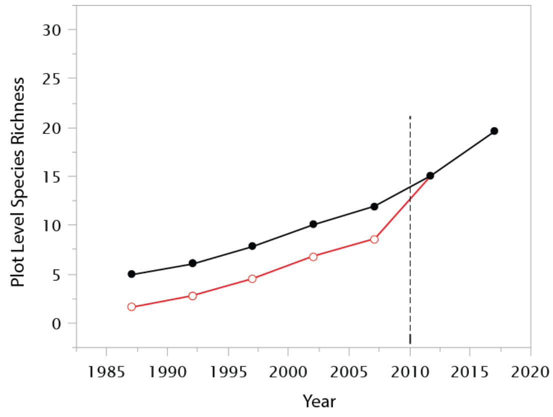

Figure B-1. Conceptual diagram illustrating differences in species detection that may have occurred between inexperienced and experienced field crews (red and black lines respectively). At the end of the transition interval spanning 2010, it is expected that the experienced crews would pick up any species missed in earlier measurements by the inexperienced crews, resulting in a steeper slope for the measurement interval spanning the transition year.

Table B-1. Differences in Target Species richness and rates of change (slope) by decade of establishment, parsed for pre-2010 and post-2010 assessments. Slopes are calculated only for measurement intervals that do not span the 2010 transition year. Decades for which an insufficient number of plots have had a full interval measured do not have a value for slope.

	Pre-2010 Assessments			Post-2010 Assessments		
Decade of Establishment	Mean Assessment Age	Mean Richness	Mean Slope	Mean Assessment Age	Mean Richness	Mean Slope
1980’s	13.2 +/- 1.4	1.8 +- 0.5	-0.04 +/- 0.17	33.3 +/- 1.7	10.8+/- 3.4	-
1990’s	6.6 +/- 0.6	3.1 +/- 0.3	0.41 +/- 0.10	21.9 +/- 0.7	10.2 +/- 0.8	1.40 +/- 0.25
2000’s	3.4 +/- 0.8	3.5 +/- 1.2	0.48 +/- 0.40	14.0 +/- 1.4	8.9 +/- 1.9	1.34 +/- 0.88
2010’s	-	-	-	3.4 +/- 0.6	11.2 +/- 3.0	-
		Overall mean	0.32 +/- 0.1		Overall mean	1.32 +/- 0.30

Period are significantly greater than for the pre-2019 period for both (i) the 1990’s decade of establishment (ii) all decades of establishment lumped together. There are insufficient intervals to detect significant differences for other establishment decades on their own. We can also look at just the post-2010 slopes (where presumably we have the greatest confidence in the measurements), and use them to project backward from the mean post-2010 richness values. Even using the lower confidence limit for slope, assuming a constant rate of new species arrival would require a negative value (-14.8) for species richness as a starting condition for the 1990’s decade of reclamation. This is not a possible scenario, so the rates of arrival observed for the post-2010 period cannot have been sustained since year zero, and must have increased over that of earlier periods.

We can also use just the post-2010 assessment intervals to judge the second postulated outcome of measurement bias as listed above. In this case, the null hypothesis (no difference in trends by decade of establishment) would require that the overall trend could be plotted using a chronosequence approach using the post-2010 data on its own. The results for such a chart are displayed in Figure B-2. The trend implied by the chronosequence is close to a flat line, but the mean slopes suggested by the same data, parsed by decade of establishment, are inconsistent with a flat line. The null hypothesis must therefor be rejected: there must be trend differences by decade of establishment.

These analyses provide evidence that (i) an apparent acceleration in the arrival of native species on reclaimed sites and (ii) apparent differences in trends by decade of establishment cannot be alternatively explained by systematic measurement errors, where inexperienced field crews were underestimating species richness prior to 2010. They cannot, discount that such systematic errors may have occurred, just that they could not have had the specific postulated effects on the analyses and conclusions in this report. If systematic underestimations did occur, it seems the most likely effect is that the closer-to-flat portions of the trends in Figures 8, 9, 12 and 16 are slightly lower than they otherwise should be, but the overall trends would be otherwise very similar.

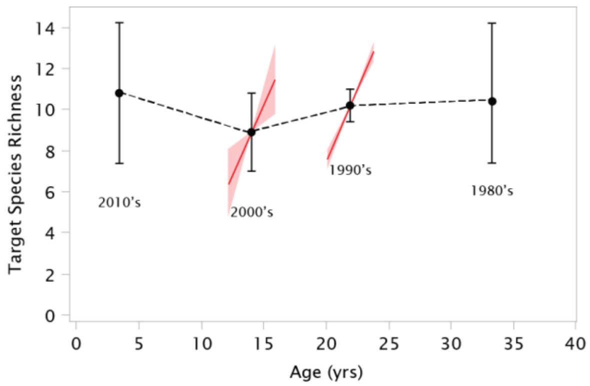

Figure B-2. Illustration of chronosequence for characteristic species richness by age using only post-2010 measurements and assuming no trend differences by decade of establishment. Black dots represent the mean post-2010 value for each establishment decade, along with associated 95% confidence intervals. For the two decades of establishment for which sufficient full intervals occurred during this period from which a slope could be calculated, the mean slope (red line) and 95% confidence interval for that slope (shaded red) are also shown.