Revisiting reclaimed well pads in boreal forests: the role of time and changing criteria in the recovery of vegetation composition, forest structure, and plant traits

by

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Abstract

Boreal forests provide multiple ecological and economic services, including carbon storage, provision of wildlife habitat and recreational space, and timber supply. In Canada's western boreal forests, natural resource energy exploration and extraction results in substantial anthropogenic disturbances, including clearing forests for well pads. Well sites are decommissioned and then reclaimed: a process whereby disturbed land is to be set on a trajectory of ecological recovery. However, after meeting reclamation requirements sites are rarely monitored, resulting in uncertainty about long-term successional trajectories. Ecological succession of these post-reclaimed sites may be arrested, contributing to landscape fragmentation and its associated negative consequences. This includes loss of habitat and thus biodiversity, greater vulnerability to invasive species, and changes to ecosystem processes. To understand post-reclamation recovery, we collected data on vegetation at 25 well pads and adjacent reference boreal forests in north-west Alberta, Canada. Taxonomic (e.g., species occurrence), structural (e.g., basal area), functional (e.g., specific leaf area) and soil property (e.g., bulk density) data were used to assess the recovery trajectories of well pads of varying post-certification ages. Multivariate ordinations and analyses, generalized additive mixed models, and mixed effect models were used to quantify recovery patterns. Our analyses demonstrated that well pads of varying ages and criteria groups differed from adjacent reference forests. However, soil FH depth, leaf carbon, and diversity measures showed resilience. Overall, our data suggest that many well pads are not recovering even 44 years post-reclamation and that more time is needed to assess if recent changes to criteria are aiding recovery. Other factors may be influencing the trajectory of recovery in the understory plant community more than the time since post-reclamation. Results from this study can improve our understanding of post-disturbance successional dynamics and may help inform mitigation actions used to remove biological and environmental barriers limiting ecological recovery.

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Chapter 1. Introduction

1.0 Background

Since the last continental glaciation, northern vegetation, soils, geological material, water systems, and climate coevolved to create the boreal forest zone (Brandt et al. 2013). The boreal zone accounts for approximately 1.89 billion hectares globally found in cold, northern latitudes below the Arctic tundra and above temperate deciduous forests. Spanning large portions of mainly Russia, Canada, and Scandinavia, boreal forests (also called taiga forests) provide essential ecosystem services such as carbon sequestration, wildlife habitat, resource extraction, and recreation. Common terrestrial animal species include wood bison, elk, moose, beaver, woodland caribou, grizzly and black bears, and wolves. Tree genera *Abies, Larix, Picea, Pinus, Populus*, and *Betula* dominate the overstory. Shrubs, forbs, grasses, sedges, and bryophytes populate the understory plant community and provide the greatest amount of plant biodiversity in boreal forests, impacting plant community structure and biogeochemical processes (Nilsson and Wardle 2005; Hart and Chen 2006; Violle et al. 2007).

Boreal forests are typified by natural disturbances such as fire, insects, diseases, and their interactions. While boreal species are adapted to such disturbances, human activity has introduced other stressors that can have cumulative effects on boreal ecosystems. Anthropogenic disturbances, including clearcutting and oil and gas development (e.g., well pads, pipelines), have varying degrees of soil disturbance and impacts on the understory plant community. These disturbances have the potential to affect the future successional trajectory of the affected forest (Bergeron and Fenton 2012; Lupardus et al. 2019).

Forest successional trajectories are governed by the intensity and spatial distribution of the disturbance that causes them. Secondary succession (in which the seedbank remains intact) is prevalent in boreal forests following natural disturbance. Anthropogenic disturbance can resemble natural disturbances, resulting in similar successional paths. Clearcutting, for example, is comparable to stand-replacing fire in that it destroys the overstory canopy (but with less impact on forest floor flora, Burton et al. 2014). However, because oil and gas activities disturb both the overstory and understory plants as well as the soil, there are no true natural analogues. Understanding the variables and processes that influence succession is critical, especially as anthropogenic pressures like resource extraction increase (Oliver and Larson 1996; Messier and Puettmann 2011). Gaining a better knowledge of forest succession remains an important problem in ecology, as noted by Taylor et al. (2020), particularly considering the scale of anthropogenic disturbance in the boreal (Schneider et al. 2003). Therefore, understanding the succession of anthropogenic disturbance and its intensity is critical.

The forest understory can influence tree regeneration and succession (Lieffers et al. 1993; Royo and Carson 2006). Understory plants account for most of the boreal forest diversity and are critical to forest ecosystem health (Gilliam 2007). Understory plants (such as shrubs, forbs, graminoids, and non-vascular plants) provide food and shelter for insects, birds, and mammals. They also have an impact on the ecosystem functions, both above- and below-ground (Nilsson and Wardle 2005; Hart and Chen 2006). The understory can interact with living trees (George and Bazzaz 2003) by influencing tree regeneration and succession (Royo and Carson 2006). Below ground, the understory's quick growth and high turnover including of roots help with nitrogen cycling and leaf litter. Disturbances can affect access to water, nutrients, and light, which all have a significant impact on understory plant species and can impact the forest's future development because they can alter the competitive dynamics among plants. This raises a fundamental question: how do boreal forests recover after different types of disturbance, including anthropogenic disturbance? The full extent of the factors that influence plant communities in the understory is unknown. As a result, studying the response of boreal plant communities to anthropogenic disturbance requires an understanding of their dynamics (e.g., competition, resilience) and functioning.

The exploration and extraction of oil and gas resources often result in significant environmental disturbance, making it crucial to conduct research on reclamation methods to mitigate these impacts. The oil and gas industry contributes significantly to Alberta’s industrial footprint in the boreal forest through loss of forest and increased edge density, sometimes creating landscapes that have no historical equivalent (Pickell et al. 2015). A variety of operations by oil and gas companies disturb the land, such as seismic lines, pipelines, oil sands, processing plants, tailings ponds, and well pads (hereafter wells). In contrast to linear oil and gas disturbance (i.e., seismic lines, pipelines), drilling a well creates a ~100 × 100 m disturbance. A level surface is required before drilling can begin, so vegetation is cleared, surface soil is temporarily removed, and subsurface soils are leveled. The result is a bare well with removal of native vegetation and seed bank. Approximately 460,000 hectares of Alberta’s land are estimated to be affected by oil and gas wells (ABMI 2018; Janz et al. 2019) with approximately 41% in boreal forests.

After a well is no longer operational and has been abandoned, subsequent reclamation practices then aim to mitigate the long-term impacts on structure and function from the wells by restoring soil, vegetation, and hydrology to an “equivalent land capability” (ESRD 2013) compared with the land pre-disturbance. The Alberta government first enacted regulations overseeing the reclamation of disturbed lands in 1963, with the introduction of the Surface Reclamation Act in response to landowner concerns about wells. Historically, reclamation involved planting agronomic seed mixes to prevent soil erosion with a "green is good" mindset (Powter et al., 2012), resulting in decreased biodiversity and ecosystem resiliency, with some sites remaining in an arrested or slowed successional state (Lupardus et al. 2019; Azeria et al. 2020). This difference between the well and the surrounding forest creates a fragmented environment, resulting in reduced connectivity, biodiversity loss for some species, increased edge effects, greater vulnerability to invasive species, and changes to ecosystem processes (Saunders et al. 1991; Fahrig 2003). In recent years (2010 update), reclamation requirements have shifted from "green is good" to incorporating native plant species and reestablishing woody vegetation on forest sites (ESRD 2013). Some well construction processes have also improved to create less extensive disturbance (for example, better drilling techniques). Since over ten years have passed after the well criteria were revised, we can begin to evaluate their success in recovering wells towards similar composition, structure, and function as benchmark mature reference forests (benchmark refers to the standard against which the success of well recovery can be measured, in this case, mature reference forests in the study area).

Key challenges in assessing plant recovery and reclamation success include: the lack of long-term monitoring and the limited consideration of taxonomic, structural, and functional data in combination. Additionally, there is often an absence of non-vascular species included in taxonomic and functional data collection for post-reclamation monitoring. This highlights the need for research on recovery of reclaimed wells that employs a more comprehensive approach. While vegetation composition and structure are important measures for determining ecological recovery, an understanding of the traits that define plant communities is essential for studying plant community dynamics and functioning in response to anthropogenic disturbance.

Plant functional traits are morpho-physio-phenological heritable features of a species that affect the performance of individuals via survival, growth, and/or reproduction (Violle et al. 2007; Garnier et al. 2015). These traits play a crucial role in shaping the diversity and productivity of terrestrial ecosystems. In recent years, functional traits have become a powerful way to assess ecosystem health and services (Lepš et al. 2011; Garnier et al. 2015). Matching plant characteristics to their function can further our understanding of specific adaptations species have developed for surviving in different environments. For example, species with slow growth rates and low seed production may outpace species with fast growth and high seed production when resources are limited because they can allocate more resources to stress tolerance instead of rapid growth and colonization features. Trait-based approaches also allow comparison of ecosystems over spatial and organizational scales (Dawson et al. 2021). The study of functional traits has become increasingly popular for determining a community's response to its environment and predicting how ecological systems will respond to disturbance (Lepš et al. 2011). For example, plant functional trait data have been used to evaluate plant community successional trajectories (Azeria et al. 2020), responses to climate change (Aubin et al. 2016), persistence of disturbance footprints (Dabros et al. 2022), and invasion resistance (Conti et al. 2018). Studying plant functional traits is important in the context of disturbance ecology, as understanding patterns in characteristics of disturbed plant communities can inform management strategies to restore ecosystem function and biodiversity (Aubin et al. 2024).

1.1 Thesis Objectives

There is a limited understanding of how oil and gas extraction followed by reclamation is affecting ecosystem composition, structure, and function over time. It remains unclear if and when plant communities on reclaimed well pads will return to pre-disturbance states. The aim of this study is to quantify the recovery of soil and plant communities following reclamation of oil and gas well sites in Alberta's boreal forests by asking: throughout the years since reclamation, is the plant community recovering towards the mature reference forest benchmark? In the following chapters, I will compare soil and vegetation properties between reclaimed well pads of varying ages and types of reclamation criteria and benchmark reference forests to determine their structural, taxonomic, and plant functional trait recovery over time. Trends will be assessed over years since certification and grouped into age-criteria groups based on well "age" (year sampled - certification year) and certification under pre- or post- 2010 criteria.

In Chapter 2 I quantify vascular and nonvascular species composition and plant community structure on well pads over time and grouped by age and criteria compared with reference benchmark forest and in relation to soil, diversity, and overstory variables.

In Chapter 3 I compare plant functional traits among well pads over time and grouped by age and criteria and reference benchmark forest. I also relate functional trait data to soil, diversity, and overstory variables.

In Chapter 4 I synthesize my findings from the two preceding chapters and offer final conclusions and broader applications of this study.

A better understanding of the recovery of plant communities and soil parameters from this study will inform the development of more effective reclamation practices and management of wells in the future as well as contributing to the study of cumulative effects of intensive oil and gas exploration in the study area. Additionally, this research will contribute to the broader understanding of succession in anthropogenically disturbed forests.

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Chapter 2. Compositional and structural recovery on reclaimed well pads of varying age and reclamation criteria towards mature reference benchmark forests

2.0 Introduction

Boreal forests (also called taiga) are the largest land biome accounting for 33% of Earth's forested area (Natural Resources Canada 2015). These forests are found in cold, northern latitudes below the Arctic tundra and above temperate deciduous forests. Boreal species are therefore adapted to long cold winters. Spanning large portions of mainly Russia, Canada, and Scandinavia, boreal forests provide essential ecosystem services. Ecologically, boreal forests provide carbon sequestration and wildlife habitat (Lemprière et al. 2013; Bradshaw and Warkentin 2015). Humans also benefit directly through recreation, medicinal herbs, and food sources (Uprety et al. 2012; Kujala et al. 2023). Economically, resource extraction is a major industry in boreal forests. Boreal forests produce approximately 33% of global wood and 25% of paper (Gauthier et al. 2015). However, such interactions between humans and boreal forests can lead to long-term forest quality decline. Habitat fragmentation, hydrologic discontinuity, reduced regeneration and productivity, and increased human access from industrial disturbances can cause extinction of endemic populations and impaired ecological function (Fahrig 2003; Nishi et al. 2013; Riva and Nielsen 2021). To preserve the services provided by boreal forests, it is important to study their response to disturbances, both natural and anthropogenic.

Vegetation of North American boreal forests have evolved along with natural disturbance for 12,000 years (Weber and Stocks 1998; Brandt et al. 2013). Boreal forests experience climatic (e.g., windthrow, frost, and ice damage), biological (e.g., insect outbreaks, disease, and herbivory), and pyrogenic (wildfires) disturbances. These natural disturbances play a crucial role in shaping the composition, abundance, and distribution of vegetation by modifying the surrounding environment and the availability of resources over space and time (Pickett and White 1985). Disturbance decreases the overall amount of living plant biomass (Reader 1991), while also providing surviving plants or propagules with more resources such as light, space, soil moisture, and nutrients (Canham and Marks 1985). This also creates an opportunity for new species to establish in the area (Grime 1973; Collins 1987). Depending on the magnitude, frequency, and intensity of the disturbance, boreal communities are reset to an earlier stage of ecological development, with a gradient of early successional possibilities (De Grandpré et al. 1993). In ecosystems that undergo frequent disturbance, like boreal forests, many species have evolved to survive dynamic conditions. Natural disturbances are acknowledged as vital for supporting species diversity and maintaining ecological processes; many boreal species (e.g., fire-adapted species like jack pine and lodgepole pine) even rely on disturbance for regeneration (Kneeshaw et al. 2011).

Successional trajectories of forests are determined by the intensity and spatial distribution of the disturbance that initiates them. In boreal forests, primary succession - where succession begins on bare ground - may occur (e.g., after oil sands mining, but it is rare). Secondary succession occurs after a disturbance agent selectively removes species from the community and in turn modifies the forest structure and composition depending on disturbance frequency, severity, and scale; it is the dominant type of succession in boreal forests. It can result from both natural and anthropogenic disturbance, with varying degrees of change in forest structure and function. Localized treefall may create minor forest changes that lead to similar species succeeding, whereas a stand replacing wildfire may burn and consume the overstory canopy, resetting the system and allowing pioneer species to restart growth. Clearcutting is like stand-replacing fire in that it removes the overstory canopy (albeit with reduced effects on the forest floor vegetation) (Burton et al. 2014). Oil and gas disturbances, such as well pads and surface mining, do not have clear natural equivalents since they affect the overstory and understory vegetation while also impacting soil. It is essential to understand the variables and processes that contribute to succession patterns, particularly with increasing anthropogenic pressures such as resource extraction (Oliver and Larson 1996; Messier and Puettmann 2011). Historically, research on succession has focused on one or a few components (e.g., only species composition) that impact forest succession, despite knowledge that multiple components influence succession (Meiners et al. 2015). Gaining a deeper understanding of forest succession remains a significant challenge in the field of ecology, as highlighted by (Taylor et al. 2020) especially given the magnitude of anthropogenic disturbance in the boreal (Schneider et al. 2003). Therefore, understanding the successional endpoints of anthropogenic disturbance and its intensity is crucial.

While disturbance is a natural part of boreal forest dynamics, the introduction of anthropogenic disturbances has led to novel conditions sometimes without historical equivalence. Larger scale anthropogenic disturbances, including clearcutting for timber, pulp, and paper extraction, and oil and gas development (e.g., well sites, seismic lines, pipelines), have varying degrees of soil disturbance and impacts on the forest vegetation. Many of these disturbances have been concentrated only in the past century and may overlap spatially or temporally with each other and/or natural disturbances. These disturbance dynamics may alter the future successional trajectory of the affected forest (Bergeron and Fenton 2012; Lupardus et al. 2019).

The type and magnitude of disturbance influences forest understory vegetation. Understory plants contain most of the boreal diversity and are essential for maintaining forest ecosystem health (Gilliam 2007). Understory plants, including shrubs, forbs, graminoids, and non-vascular plants, provide food and habitat for insects, birds, and mammals. They also affect the environment above- and below-ground (Nilsson and Wardle 2005; Hart and Chen 2006). They can interact with growing trees (George and Bazzaz 2003); for example, dense understories can alter tree regeneration and succession (Lieffers et al. 1993; Royo and Carson

2006). Below ground, rapid growth and high turnover of the understory contribute to nutrient cycling and leaf litter. Dynamics of these understory plant communities are shaped by a variety of factors and their interactions (Reich et al. 2012). Water, nutrients, and light largely determine the understory plant type. Anthropogenic disturbances can affect all three parameters. Disturbance of the understory will affect the future development of a forest. This leads to a critical question: will natural regeneration occur, or will reforestation efforts be necessary to establish tree and shrub growth in anthropogenically disturbed areas? Full comprehension of the factors that affect the plant communities in the understory remains unknown. Therefore, an understanding of boreal plant community dynamics and functioning is essential for studying their response to anthropogenic disturbance.

Oil and natural gas extraction is a significant anthropogenic disturbance in Canadian boreal forests, particularly in Alberta (Schneider et al. 2003). A variety of operations by oil and gas companies disturb the land, such as seismic lines, pipelines, oil sands, processing plants, tailings ponds, and well sites. There has been significant focus on the cumulative consequences of energy extraction in the mineable (surficial) Athabasca Oil Sands, but some argue that in-situ bitumen development may have considerably greater cumulative effects that are anticipated to grow significantly (Nishi et al. 2013). In contrast to linear oil and gas disturbance (i.e. seismic lines, pipelines), drilling a well creates a ~100 × 100 m disturbance. A level surface is required before drilling can begin, so vegetation is cleared, surface soil is temporarily removed, and subsurface soils are leveled. The result is a bare site lacking native vegetation and seed bank. Approximately 460,000 hectares of Alberta’s land are estimated to be affected by oil and gas wells (ABMI 2018; Janz et al. 2019) . Due to growing public awareness of the cumulative effects and industry contributions to habitat fragmentation and biodiversity loss, stakeholders have called for decreasing the total footprint of energy extraction (Powter et al. 2012).

Towards decreasing this footprint, after a well is no longer operational and has been abandoned, subsequent reclamation practices then aim to mitigate damage from the wells by restoring soil, vegetation, and hydrology to an “equivalent land capability” (ESRD 2013) compared with the land pre-disturbance. Reclamation criteria assess sites for evidence of positive successional trajectories towards a target forest community (involves establishment of woody and herbaceous forest plant communities; ESRD 2013). The Alberta government first introduced legislation governing the reclamation of disturbed lands in 1963 with the enactment of the *Surface Reclamation Act* to answer concerns raised by landowners about wells (Powter et al. 2012). Historically, reclamation involved planting agronomic seed mixes to avoid soil erosion with a “green is good” mentality (Powter et al. 2012), resulting in decreased biodiversity and ecosystem resiliency with some sites remaining in an arrested or slowed successional state (Lupardus et al. 2019; Azeria et al. 2020). This contrast between the well and surrounding forest results in a fragmented landscape leading to decreased connectivity, biodiversity loss for some species, increased edge effects, higher vulnerability to invasive species, and changes to ecosystem processes (Saunders et al. 1991; Fahrig 2003). Research assessing ecosystem recovery on wells has also shown that wells have low soil quality (Janz et al. 2019); declines in native plant communities, plant cover, and plant species richness (Sylvain et al. 2019), and decreased functional diversity (Lupardus et al. 2020). Over 460,000 wells have been drilled in Alberta, with around 28% certified reclaimed or exempted (Well Status 2022). In more recent years (2010 update), reclamation requirements have moved beyond “green is good” towards using native plant species and reestablishing woody vegetation on forested lands (ESRD 2013). Some well construction practices have also changed to cause less intensive disturbance (e.g., advanced drilling techniques). Over 10 years have passed since the well criteria changed, giving us the opportunity to begin to assess its effectiveness in returning wells to a similar structure and function as the benchmark mature reference forest. Yet, reclaimed sites are rarely monitored post-certification to evaluate if successional trajectories are progressing towards mature forests (Lupardus et al. 2019).

In this study, I aim to evaluate the recovery of vegetation (including the understory and canopy) and soil properties of oil and gas wells in Alberta's boreal forests following their reclamation. We considered six potential pathways for recovery (Fig. 2-1, Macdonald et al. 2024) ranging from “No resilience” (well to reference similarity falls below reference to reference similarity) to “Resistance” (well to reference similarity curve overlaps with the lower confidence interval of the mean reference to reference similarity consistently over time). Intermediate resilience pathways included: “Lagged resilience” (declining similarity between wells and references (i.e., loss of resistance), followed by increasing similarity (resilience) with recovery at the year post-reclamation when the curve intersects the lower confidence interval of the reference to reference similarity), “Resilience” (low well to reference similarity in the recent years post-reclamation indicating a lack of resistance, followed by an increase, indicating subsequent resilience and recovery), “Temporary resilience” (low well to reference similarity in recent years post-reclamation followed by the curve intersecting the lower confidence interval of the reference to reference similarity then declining back below the reference to reference similarity range), “Declining resistance” (decrease in well to reference similarity over time, indicating a loss of resistance, with no evidence of subsequent resilience). In this study, I compared soil and vegetation properties between reclaimed wells and a set of mature reference benchmark forests to determine their plant community's structural and compositional recovery over time. Reclaimed wells were grouped by years since reclamation certification and categorized as either pre- or post- 2010 criteria. My main objectives included: i) quantifying compositional and structural measures of the understory and overstory plant community along with soil properties (i.e., pH, bulk density, and FH depth) on wells compared with benchmark reference forest, and ii) identifying how time since reclamation (and associated reclamation practices) affected similarity of understory community composition and forest structure and soil properties between wells and reference forest benchmark stands. I hypothesized that if age of post reclamation wells was important for recovery, then older reclaimed wells would have a community composition and structure and soil properties more like the forest benchmark because they have had more time to recover compared with the younger wells. However, I also hypothesized that if, in addition to time (age), reclamation practices in response to changes in criteria were important for recovery, then wells with younger reclamation dates (post-2010 criteria update) would have a community composition and structure and soil properties more like the forest benchmark than the older reclaimed wells reclaimed under ‘green is good’ practices because they were reclaimed under a more ecological framework (e.g., use of native species, minimum requirements of woody species, heterogeneous topography, and downed woody debris). Thus, it remains unclear if plant communities and associated well properties (e.g., soil characteristics) on reclaimed wells will return to a condition like their pre-disturbance state, and if so - how long will it take for recovery to occur.

2.1 Methods

2.1.1 Study Area

This study takes place in west-central Alberta in the Central Mixedwood Boreal Forest Natural Subregion (Albert et al. 2006) approximately 250 km northwest of Edmonton (near Fox Creek, AB; 54.4009° N, 116.8045° W; **Fig. 2-2**). The study area supplies resources for timber and oil and gas extraction, with a history of hydrocarbon exploration and development dating back to the 1960s. A high density of wells and associated infrastructure, including pipelines, roads, and access corridors characterize the area. This infrastructure can impact local hydrology, soil conditions, vegetation, and wildlife.

Common terrestrial animal species in this area include deer, black bears, cougars, moose, and coyotes. We sampled upland sites characterized by aspen (*Populus tremuloides* Michaux), mixedwood (mix of conifer and deciduous), and white spruce (*Picea glauca* (Moench) Voss) forests. Other common overstory species in the area include paper birch (*Betula papyrifera* Marshall), balsam poplar (*Populus balsamifera* Linnaeus), and balsam fir (*Abies balsamea* (Linnaeus) Miller). Common understory vegetation includes low bush cranberry (*Viburnum edule* (Michaux) Rafinesque), prickly rose (*Rosa acicularis* Lindley), green alder (*Alnus alnobetula* subsp. *crispa* (Aiton) Raus), bunchberry (*Cornus canadensis* Linnaeus), wild sarsaparilla (*Aralia nudicaulis* Linnaeus), dewberry (*Rubus pubescens* Rafinesque), and bluejoint grass (*Calamagrostis canadensis* (Michaux) Palisot de Beauvois) growing on Gray Luvisols of various textures. The subregion contains a mix of wetlands and upland forests with short, warm summers and long, cold winters (Albert et al. 2006). The highest and lowest mean monthly temperatures are 15.6°C (July) and -10°C (January) respectively (ACIS 2023) with an average annual temperature of 2.6°C (Fox Creek Junction weather station). Summer precipitation is highest with the greatest rainfall in July at 101 mm, and the average annual precipitation is 595 mm (Smerdon et al. 2019). Reference forest stands in the study area ranged from 28 to 141 years old with an average age of 74.5 likely due to frequent anthropogenic (e.g., clear cuts) and natural disturbances (e.g., stand-initiating fire).

2.1.2 Site selection and sampling design

This observational study was conducted during the summer of 2023 (early June - early August) using a chronosequence of reclaimed wells that captured a range of ages post-certification. The study encompasses two distinct sets of reclamation criteria, reflecting changes in certification standards over time. These different criteria are accounted for in our analysis. Further details on how these groupings were managed are provided in the statistical analysis section below. Potential sites were identified using data from AbaData (Abacus Datagraphics

Ltd.'s AbaData Oil and Gas Map Software. Accessed May 2023). I selected wells of homogeneous disturbance with flat or nearly flat topography accessible by road and less than ~30 minutes of walking to help with accessibility of sites. Each well was paired with an upland adjacent reference forest that had not been disturbed by oil and gas and was also uninterrupted by road, pipeline, etc. This reference forest plot was used as a benchmark with which to compare the recovery of the wells. While this study did not measure paired sites with respect to time since disturbance, ideally, it would be beneficial to track changes over time with paired ages. However, due to logistical constraints and disturbance history of the sites, this longitudinal approach was not feasible. Sites were visited to confirm that there was no additional disturbance that was not captured by AbaData and were excluded when there had been subsequent major disturbance in the area. As there had been recent fires in the area and associated road closures, any burned areas were excluded from site exploration. From the number of potential (n= 64) sites, I sampled 25 sites (well + adjacent reference forest benchmark plot pairs), which were in a ~50 km^2 range (Fig. 2-2); their reclamation certification dates ranged from 1985-2016 (38-7 years before data collection). A main goal in site selection was to measure a range of ages, but many wells in the study area had similar certification years (e.g., high concentration of wells with 1998 certification dates). Selection of wells reclaimed under the new criteria was also limited in the area. Ideally, each age criteria group would have an equal sample size, but this wasn’t feasible due to the unbalanced nature of well ages. We maximized sampling of plants during the peak of the growing season, so field work was completed by early August.

Each site (n=25) had a circular well plot and adjacent benchmark forest reference plot (n=24, two close wells shared one reference); each plot had 25 m transects in each cardinal direction from the center point (n=4) and associated quadrants (e.g., NE quadrant). Sample points 5 m and 10 m from the center along each transect were used for soil core and canopy cover data collection (Fig. 2-3). Contact points every 1.5 m along each transect were used for compositional vegetation (vascular and non-vascular) and height strata data collection (Fig. 2-4a). Data collection protocols are described below.

2.1.3 Data collection and processing

The first phase of data collection began in mid-June with the first visit to each site (pair of well and forest plot). At each study site, a magnetic locator was used to locate and flag the center of the well (well bore). This marked the center of the study plot (if not found, an estimate based on GPS coordinates, equal distances to the four well edges, or area of heaviest disturbance was used). We then established four 25 m transects oriented to each cardinal direction using 30 m and 50 m tapes and marked 5 m and 10 m from the center using pigtails and flagging tape.

Adjacent reference forest benchmark plots were established following the same procedure with a corner that was closest in proximity to the wellsite at least 35 m from the well corner to avoid edge effects and reference center approximately 60 m from the well edge (Fig. 2-3). The chosen adjacent forest plot was ideally homogeneous (i.e., in site type, species present, topography, etc.); however, if there was a visible age difference, evidence of other disturbances, or change in topography, transects were adjusted to attempt a mostly homogeneous representation of the reference forest. If the reference forest on one side of the well appeared older than the other, the older forest was most often chosen to keep the average age of the reference forests in a similar range (avoiding large variation in benchmark conditions as the goal was to have mature forests as the benchmark to compare with the sites).

2.1.3.1 Vegetation

To assess the re-establishment of a forest canopy on the well, canopy cover was estimated at 5 m and 10 m sample points using a convex spherical densiometer. The spherical densiometer was taped so that 17 crosshairs were visible (Strickler 1959). We placed the spherical densiometer on a tripod approximately at breast height (1.3 meters from ground), and noted how many cross hairs were covered with foliage or branches (i.e., canopy), with the spherical densiometer pointing in each cardinal direction.

In each quadrant, five-minute moss surveys were conducted (total of 20 minutes per plot). A small colony of each moss species was collected in brown paper bags which were labeled by site and plot for future identification. Moss identification contributed to species richness data and served as indicators for habitat type (e.g., certain moss species are typical of boreal forests or disturbed sites).

To quantify tree and shrub re-establishment and structural complexity, we measured the diameter of a subset of trees in each plot. If all four quadrants were mostly homogeneous in terms of tree/shrub cover, diameter at breast height (DBH) for all stems greater than 5 cm DBH in one randomly chosen quadrant were measured and recorded along with the species and notes of any tree damage. If there were differences among quadrants (e.g., two quadrants were treed and two quadrants were bare), DBH for trees in two quadrants were taken to capture the variation (i.e., one treed and one bare).

In all reference forest plots, and in wells where trees were present, tree heights, DBH, and tree cores were taken for four representative (ideally tallest/oldest) trees to give an approximation of tree age and vertical structure. Tree species were recorded, and DBH was measured using a DBH tape wrapped around the tree at 1.3 m height. Tree height was measured using a vertex hypsometer. The hypsometer and puck were calibrated before first use each day (and after changing locations) by setting the hypsometer to CALIBRATE and pressing ON while standing 10 m from the puck. After calibration, one person would stand in front of the selected tree with the puck held at 1.3 m. Another person would stand as far as possible (ideally the height of the tree) from the puck. Holding the hypsometer up to their eye and aiming it at the puck, the ON button was held down until a beep was heard and red crosshair blinked. Once the crosshair was blinking, the hypsometer was aimed at the top of the tree. The height reported by the hypsometer was recorded on the corresponding datasheet. To provide an estimate of tree age, tree cores were obtained using an increment borer. We first inserted the borer auger (bit) into the handle, then placed the auger tip against the tree at 1.3 m. We turned the auger clockwise until the initial 2-3 cm had been penetrated. We then turned the handle with both hands, pushing into the tree until reaching approximately the center of the tree (visual assessment using extractor length outside the tree). We inserted the extractor into the borer auger (bit), turned the auger counterclockwise, and pulled the extractor and tree core from the auger (Haglöf Sweden 2012). When trees were not round, cores were obtained from the narrow width of trees. Tree rings were counted on-site to get an approximate age. On the well, if trees greater than 5 cm DBH were present, the heights and DBH were taken but cores were only collected if DBH was large enough (> 20 cm DBH) to support use of the increment borer.

In late July, during the peak of the growing season in the area, plant species occurrence and vertical structure data were collected using a contact-point method modified for forest systems (Aubin et al. 2008). Species present within 15 cm radius circular quadrats (“points’’) were recorded every 1.5 m for 30 points along the North-South and East-West transects of the well and reference 50 × 50 m study plots for a total of 60 points (Fig. 2-4a). A center pole (wooden dowel) with vertical markings every 50 cm was placed every 1.5 m along the transects of the study plot, and a 15 cm ruler was used to create the outer boundary for the contact point revolving around the center (Fig. 2-4b). Any photosynthetic plant parts present (leaves and green stems) within the 15 cm boundary were counted using six-digit species codes for vascular and non-vascular plants (first three letters of genus and first three letters of species, e.g., *Viburnum edule* was labeled VIBEDU) and given an occurrence value of 1 in each height strata where it was present. If a species could not be identified with supplemental use of field guide, photos of the plant were taken and voucher specimens were collected; labeled with site name, well/reference, specimen number; and placed in a plant press to be identified in the evening. For some species, analysis occurred at the genus level (see Appendix Table A-1). Species data were then processed into occurrence data by strata and occurrence combining among strata (aggregated by point). Aggregated occurrence data (number of points containing a given species) were then used to create percent occurrence data for each species in the study plot.

2.1.3.2 Soil

Soil FH (organic layer excluding Litter(L) layer, including Fibric and Humic layers) depth (mm) data and soil cores were also collected at 5 m and 10 m sample points. The magnetic locator was used at each sample point to ensure there was no metal debris or pipes where we collected soil cores. Then, vegetation was removed by hand from the area where the soil core sampler was placed. We used a double-cylinder, drop-hammer soil core sampler to collect soil cores from the top layer of soil (~top 20 cm). We resampled from an adjacent surface if large coarse fragments or large roots/rocks were present in the original core. Once driven into the ground, the sampler was pushed in a circular motion to detach it. Then the core was removed, the ends trimmed if necessary, and depths measured and recorded on the corresponding datasheet. A ruler was used to measure core depth (to mm precision) from the surface to the bottom of the hole and FH depth (separation with mineral layers determined visually and tactilely). Cores were stored in plastic bags with labels (site and plot, the date, sample point, core depth, FH depth) for transport. After samples were back at the lab, Soil samples were later heated to 105° C for 24 hours and weighed, then divided by the core volume (core depth (radius 2.54 cm * pi)^2) to get bulk density values (g-cm^{-3}). Bulk density of the plot was calculated as the average of each set of 8 cores. To prepare soil cores for pH measurements, they were then ground and sieved using a 2 mm sieve to remove any rocks. Soil pH was measured using the procedures outlined by (Kalra and Maynard 1991) and 10 g of soil were mixed with 20 mL of 0.01 M $CaCl_2$ solution. After the solution was absorbed, the mixture was stirred with a glass rod for approximately 10 seconds for five repetitions over a 30-minute period. The suspension was left to rest for 30 minutes, then the pH was measured and recorded using a digital pH reader.

2.1.4 Statistical Analyses

Prior to conducting the statistical analyses, I first examined the range of ages and their distribution across different reclamation criteria and time. Age was calculated as the difference between 2023 (the sampling year) and the reclamation year. I used analyses that could account for non-linear responses after disturbance due to the overlap of years post-certification and the 2010 criteria change. To address these potential confounding effects, I classified the wells into categorical age-criteria groups. This classification was based on approximately 10-year ranges, with the understanding that only the "young" category adhered to the new criteria introduced in 2010. The resulting age-criteria groups (Table 2-1) are labeled accordingly throughout the paper. Despite this categorical approach, we still wanted to capture recovery over time, so a Generalized Additive Mixed Model (GAMM) approach was used to account for non-linear recovery patterns, which is discussed in subsequent sections.

All statistical analyses were performed using R statistical software (version 4.2.2, 2022-10-31, RStudio Team 2020). Data were analyzed and visualized using univariate and multivariate techniques to assess differences in species composition, structure and soil between wells and reference plots, comparing wells of different post-certification ages.

To assess species richness and diversity within the study sites, I utilized the *vegan* package in R. Richness was calculated by summing the total number of species present (i.e., non-zero values) within each plot from the species occurrence matrix. For diversity, I calculated both Shannon's Diversity Index and the inverse Simpson's Diversity Index using the diversity function (Magurran 2004). Shannon's Diversity Index (H') was calculated as:

$$H' = -\Sigma p_i ln(p_i)$$

where p_i is the proportion of individuals belonging to species i in the plot. Inverse Simpson's Diversity Index ($1/D$) was computed as:

$$1/D = \frac{1}{\Sigma {p_i}^2}$$

Both indices provide a measure of alpha diversity, with Shannon's Index emphasizing richness and evenness, while the inverse Simpson's Index is more sensitive to common species.

Principal component analysis (PCA) and non-metric multidimensional scaling (NMDS) ordination were used to observe variation in plant community composition data (PCA and NMDS) and Pearson correlations between plant community composition data and environmental variables (NMDS) such as diversity variables (e.g., Shannon diversity), structural variables (e.g., basal area), and soil variables (e.g., soil pH). I chose NMDS because it is free from assumptions of normality, dimensionality, linearity, and the shape of species-response curves to gradients (Kruskal, 1964) and is often used for biological community analysis (McCune et al. 2002; Oksanen 2011; Legendre and Legendre 2012; Borcard et al. 2018). In NMDS, the relationships between samples are determined by ranked dissimilarities in species composition (Legendre and Legendre 2012). The final scores reflect relative measures of multidimensional compositional differences, where the number of dimensions corresponds to the number of species in the data matrix, minus one. I utilized the *metaMDS* function from the R package *vegan* to perform several runs, aiming for a stable ordination configuration (Oksanen 2010). The 'metaMDS' function automatically applies a square root transformation, centers the data, performs principal component rotation, and uses expanded scores with Wisconsin double standardization. The final model, which minimized the difference between the ordination and Bray-Curtis distances, was chosen as the best solution.

The R package *Adonis* (from *vegan* package) was used to run permutational analysis of variance (perMANOVA) of community composition based on age group using Bray-Curtis distance. The perMANOVA tested significant differences in the community composition among wells in different age-criteria groups and between wells and reference forest. A permutation test was used to generate the test statistic for the null hypothesis that there are no differences among age-criteria groups. The predictor variable was set as age-criteria group with site (well-reference pair location) as a random variable. Pairwise comparisons were then performed among age-criteria groups using 9999 permutations and using the Holm method for P value adjustment.

We used an Indicator Species Analysis (ISA) to find species associated with age-criteria groups. We used the *multipatt*, *strassoc*, and *A.g* functions from the *indicspecies* package to determine species associations with each group. Aspecificity (A) and sensitivity (B) of the species indicate affiliation. For each species present at a given plot, quantity A represents the likelihood of plot association with the plot-group combination (Murtaugh 1996). The frequency of the species at the sites in the plot-group combination is indicated by Quantity B. Based on species occurrence scores and the frequency of all species within a given group, ISA determines indicator values for each species. In comparison to all other sites, the coefficient of determination (R^2) evaluates the positive or negative connection of species for plots in comparison to all other plots. Species that were found at only one location were left out. All significant species (alpha=0.5) were included in the final output; however, only species with $p < 0.001$ and $R^2 > 0.7$ were considered strong indicator species.

To compare environmental variables over time on wells, I used a mixed effect model for each variable grouped by age-criteria group with site ID (well and reference shared location) as a random variable. The model identified significant differences among age-criteria groups (which includes the benchmark forests) and accounted for spatial pairing to get a more reliable signal. Each model was then tested for normal distribution using the Shapiro-Wilk test. Post-hoc tests including either Tukey test (if data were normal) or pairwise comparisons with Bonferroni adjustment (if data didn't meet normality assumption) were used to identify which groups were different, with differences shown using significance letters applied to interval plots.

Height strata data from the contact-point method were visualized using an occurrence bar chart grouped by plant life form with height class on the vertical axis. This allowed visualization of vertical diversification between plot types and well age groups. Canopy cover values were summed for each sample point, divided by 68, and multiplied by 100 to get an estimate of canopy cover (%). Sample points were then averaged for each well and reference. Basal area (in square meters per hectare) for each plot was calculated as below.

$$Tree\ BA\ (m^2)\ = (\frac{DBH}{200})^2\ * \pi$$

$$Tree\ BA\ (\frac{m^2}{ha}) = \frac{\Sigma(all\ Tree\ BA)}{(\pi \times 25m^2/4\ ) * \#\ of\ quadrants}$$

Generalized Additive Mixed Models (GAMMs) using Euclidean (continuous data without zeroes) or Bray-Curtis (data containing zeroes and composition data) similarity indices were used to construct recovery curves of key variables (e.g., community composition, soil bulk density, basal area) over time towards the benchmark adjacent forest range. Similarity values range from 0 (complete dissimilarity) to 1 (complete similarity). GAMMs are well suited for mixed-effect models with random effects (e.g., in this study: variability of observations among wells due to different reclamation practices) and non-linear relationships between predictor and response variables. For example, ecological recovery patterns may show lags or variable resilience, which is best seen in a non-linear model. GAMMs were performed using the gamm function from the *gamm4* package (Wood et al. 2017) version 0.2-6. I used GAMMs to analyze the relationship between the age post-certification for each well and the reference similarity compared to all other references. The predictor variable was years post-reclamation. To avoid overfitting, I selected the "*k*" value (either 3, 4, or 5 out of a maximum of n-1=24) based on AIC outputs for models run under each scenario. The GAMM model output, including the 95% confidence interval around the smoother, was generated using the "predict" function.

I calculated similarity (1 - Gower's distance) between reclaimed wells and reference plots over years post-reclamation, which was visualized using the *ggplot2* package (Wickham et al. 2016). In these visualizations, each point on the GAMM plot represents the similarity of a well at a given age to each benchmark forest reference plot. Additionally, the GAMM model was plotted with a smoother and gray shading showing 95% confidence interval of the correlation. Mean similarity among benchmark reference forests was computed by comparing each benchmark reference to all other references using a custom function in R. On the right side of the plot, the mean similarity of these benchmark reference sites is displayed in red with its confidence interval for comparison. Graphs were analyzed to determine the "full recovery" point, defined as when the GAMM curve crossed the lower confidence interval of the reference vs. reference similarity.

2.2 Results

2.2.1 Understory community composition

The species occurrence matrix included 156 vascular and non-vascular plants, including 121 vascular species, 11 vascular genera that could not be identified to species (e.g., *Salix*, *Carex*, etc.) and 24 bryophyte species.

Principal Component Analysis (PCA) of compositional data showed principal component 1 explaining 39.13% of the variance in understory composition data (Fig. 2-5). This component appears to represent the separation between reference and wells. Principal component 2 accounts for 10.6% of the composition variation and does not appear to align with well age. Species vectors with high positive loadings on the first axis (pointing towards wells) include mainly graminoids and forbs with a higher prevalence of introduced species than vectors with negative loading (towards the reference). Species vectors with negative loadings along the first axis (pointing towards references) include native forbs, subshrubs, and shrubs as well as mosses typical of mixedwood boreal forests (e.g., *Pleurozium schreberi*, *Hylocomium splendens*, and *Ptilium crista-castrensis*). I chose the top 30 species (highest occurrence) to be represented as vectors. A greater number of vectors have negative loadings (pointing towards reference plots) than positive loadings (pointing towards well plots).

The 2-dimensional NMDS for compositional data also showed a clear distinction between references and wells, with overlapping confidence interval ellipses among well age-criteria groups (Fig. 2-6). The best solution was repeated once in 20 tries (max 100). The final NMDS 2-dimensional solution converged on the 9th attempt and had a final stress of 0.12. I overlaid soil, composition, and structural vectors on the NMDS, which showed higher soil compaction and soil pH on wells than reference forests with greater structural and compositional diversity in references (Fig. 2-7).

While the perMANOVA showed a significant ($P<0.001$) difference in understory community composition among age-criteria groups, post-hoc pairwise perMANOVA comparisons revealed significant differences between wells and the reference benchmark forests rather than among well groups.

Given the significant difference between wells and reference, I first identified indicators for all well categories combined and the references. Indicator species analysis of the understory species data matrix collected via contact-point method identified 39 species correlated with reference plots and 27 species correlated with wells (Table 2-2). Overall, aspecificity (A) ranged from 0.71 to 1.00 and sensitivity (B) from 0.17 to 1.00. Top indicators for reference plots were *Cornus canadensis* ($A=0.982$, $B=1.000$, $R^2=0.991$, $P=0.001$; native shrub) and *Linnaea borealis* ($A=1.000$, $B=0.958$, $R^2=0.979$, $P=0.001$; native forb/subshrub). Top vascular plant indicators for wells were Small Graminoid -representing *Deschampsia cespitosa, Carex deweyana, Carex* *aurea*- (A=0.941, B=1.000, R^2=0.970, P=0.001; native graminoid), *Taraxacum officinale* (A=0.943, B=0.960, R^2=0.951, P=0.001; introduced ruderal forb), and *Trifolium hybridum* (A=0.947, B=0.880, R^2=0.913, P=0.001; introduced ruderal forb).

I then did an indicator species analysis which separated the wells by criteria (Table 2-3). This ISA revealed distinct differences in species composition between new criteria wells, old criteria wells, and reference forests. For wells subject to new reclamation criteria, 16 species were identified as strong indicators, with *Symphyotrichum puniceum* and various moss species being particularly notable. *Symphyotrichum puniceum* (SYMPUN) exhibited excellent specificity (A=0.6385) and perfect sensitivity (B=1.000), resulting in a strong indicator value (stat=0.799, P=0.005). Mosses, such as *Bracythecium spp.* (BRACMOSS) and *Aulacomnium palustre* (AULPAL), also served as strong markers, with indicator values of 0.752 (P=0.004) and 0.682 (P=0.029), respectively. These species characterize early-successional plant communities dominated by mosses, shrubs, and pioneer forbs. In contrast, wells reclaimed under old criteria were associated with seven top indicator species, including *Trifolium hybridum* (TRIHYB) and *Vicia americana* (VICAME). Both species demonstrated high specificity (A=0.722 and 0.698, respectively) and sensitivity (B=0.950 for both), with strong indicator values (stat=0.828, P=0.001 for TRIHYB; stat=0.814, P=0.001 for VICAME). In reference forests, 28 indicator species were identified, with *Linnaea borealis* (LINBOR) and *Cornus canadensis* (CORCAN) being the most reliable indicators. *Linnaea borealis* and *Cornus canadensis* exhibited high specificity (A=1.000 and 0.943, respectively) and near-perfect sensitivity (B=0.958 and 1.000, respectively), resulting in exceptionally high indicator values (stat=0.979, P=0.001 for LINBOR; stat=0.971, P=0.001 for CORCAN).

2.2.2 Plant community structure

Vertical structure was observed via distribution of plant life forms of understory flora according to the recorded height strata in all wells vs reference forest and wells of different age-criteria groups. Between well and reference forest plots, reference forests had a greater proportion of trees, shrubs, and subshrubs in the 0-100 cm height classes compared to wells which were mainly composed of forbs, graminoids, and moss (Fig. 2-8). Reference plots had greater vertical structural spread with more plants in the strata above 100 cm height. Differences are not very visible when comparing wells of different age-criteria classes, although young and mid-young age groups have higher occurrences of plants above 50 cm (Fig. 2-9). Wells within the old age group have the lowest proportion of woody species.

Proportion of occurrence data for vertical structure revealed that all wells had lower woody vegetation (i.e., trees and shrubs) in the lower strata (1.8% (0-50 cm strata) - 25.5% (50-100 cm strata) than reference forests (18.9% (0-50 cm strata) - 62.6% (50-100 cm strata); Fig. 2-10). Young reclaimed wells had a higher percentage (42.4% - 55.2%) of trees in strata 150 cm and higher than older wells (16% - 33.3%), showing greater similarity in upper strata composition to reference forest sites. In the 50-150 cm stratum, Mid-Old and Old wells had greater percentages (25.5% (50-100 strata) - 90.7% (100-150 strata) of woody vegetation than younger wells (15% (50-100 strata) - 46.5% (100-150 strata). In the 0-50 cm strata, all wells had similar percentages of plant life forms (dominance of forbs and graminoids).

2.2.3 Ecological recovery over time

Out of six possible temporal patterns of similarity between wells and benchmark reference forests, I observed resilience (four out of nine GAMMs, 44%) or no resilience (five out of nine GAMMs, 56%). Here “resilience” represents a significant temporal trend with low reclaimed well vs reference similarity in young sites, indicating a lack of resistance, followed by a subsequent increase, indicating resilience and recovery. “No resilience” refers to a temporal trend in which the well vs reference similarity falls below that of the reference vs reference similarity. For the 56% of curves that showed no resilience, there was no recovery observed within the maximum time post-reclamation for which we collected data (44 years). For the cases that did not show resilience, the difference in similarity between reclaimed wells and reference benchmark plots (as a percent of mean similarity among references) varied from 6.4% to 77.5%, with a mean of 35.6% and median of 33.1% (Table 2-4). For the 44% of curves that exhibited resilience, recovery occurred between 20- and 44-years post-reclamation, with a difference in similarity ranging from 0.3% to 9.5%.

2.2.3.1 Soil

Soils showed no resilience for bulk density and pH but resilience after 44 years for FH layer depth (Table 2-4). While soil bulk density did not show recovery in the GAMM model (Fig. 2-11a), the mixed-effect model found significant differences only for mid-young and old age-criteria groups (Fig. 2-11b). In wells, bulk density ranged from 0.43 to 1.35 g/cm^2 and in references it ranged from 0.15 to 0.93 g/cm^2. Soil pH showed no resilience (difference between GAMM smoother and reference similarity mean was 16.1%) and declining similarity between wells and reference benchmark forests (Fig. 2-12a). The interval plot also showed all wells were significantly different from references, with old wells having the highest pH (Fig. 2-12b). FH depth showed recovery at 44 years post-reclamation (Fig. 2-13a) with the percent difference between the smoother and reference mean at 9.52% (Table 2-4). The interval plot showed all age-criteria groups as significantly different from reference forest plots, with no well age-criteria groups significantly different from each other (Fig. 2-13b).

2.2.3.2 Overstory vegetation

Like PCA and vertical structure visualization, GAMMs for overstory vegetation showed lower tree presence on wells compared to reference benchmark forests (Figs. 14-16). Basal area, canopy cover, and stems per hectare had no resilience recovery curves with the minimum difference in similarity between the smoother and mean reference to reference similarity as 77.5% (basal area), 45.0% (canopy cover), and 32.4% (stems per ha; Table 2-4). Basal area had the largest percent difference between well smoother and reference mean of all the environmental variables. This large dissimilarity matches interval plot and mixed effect model results showing significantly lower basal area between reclaimed wells and references (Fig. 2-14b). In canopy cover and stems per ha interval plots, young and old wells showed greater variance (95% confidence intervals) than mid-aged wells and reference plots (Fig. 2-15b; Fig. 2-16b). The mixed-effect model and post-hoc testing for stems per hectare showed that old wells and references were not significantly different (Fig. 2-16b); however, compared with similarity between wells (little to no evidence of significant difference: P=1.00), there was weak evidence (P=0.081) for the difference between old age-criteria wells and references.

2.2.3.3 Understory vegetation

Richness and both Shannon and inverse Simpson diversity measures showed resilience with recovery (Fig. 2-17-19); however, understory composition showed no resilience (Fig. 20). Shannon and inverse Simpson diversity measures showed resilience around 35-40 years post-certification (with earlier recovery for Inverse Simpson diversity; Fig. 2-17a, Fig. 2-18a), while richness showed resilience after approximately 20 years post-reclamation (Fig. 2-19a). Richness had the smallest percent difference of all environmental variables between GAMM smoother and reference similarity mean at 0.30%, while inverse Simpson and Shannon diversity had percent differences of 3.5% and 6.0% respectively. All three variables interval plots showed significant differences only between Mid-young and Reference age-criteria groups (Figs. 17b, 18b, 19b). Understory composition had the second largest dissimilarity between smoother and reference mean of all the environmental variables at 64.5% difference (Table 2-4).

2.3 Discussion

This study aimed to quantify compositional, structural, and soil recovery on reclaimed wells of various ages in Alberta's boreal forests compared to the reference mature forest benchmark. Due to a 2010 change in reclamation criteria for wells on forested lands, this study also aimed to assess preliminary effects of updated reclamation practices. Our study provided evidence of recovery for certain vegetation and soil variables (i.e., richness, Shannon and inverse Simpson's diversity, and FH layer depth). However, our analysis did not find a strong association between time since reclamation and recovery success, despite multiple previous studies showing a favorable relationship (after a different disturbance - oil sands mining; Audet et al. 2015; Pinno and Hawkes 2015; Chen et al. 2018). While some wells that had been reclaimed 10-20 years prior to data collection had some shrub and/or tree cover (albeit much less than the reference benchmark forests), other wells of the same age-criteria group resembled older wells in arrested succession (exhibiting features more like grasslands). Our study suggests that the trend towards more ecological recovery over time (reflected in criteria change) is partially successful in promoting recovery that leads to succession, but from both our data and previous research, it is apparent that many reclaimed wells remain in slowed or arrested succession (Lupardus et al. 2019; Azeria et al. 2020). This lack of recovery may be due to soil compaction and agronomic species planted in the past. Therefore, well history may be of greater importance for predicting post-disturbance recovery.

Our data showed that wells had higher soil compaction, soil pH, and graminoid cover compared with adjacent forests with greater soil organic layer depth, shrub and tree cover, canopy cover, and basal area. Multivariate ordinations and perMANOVA pairwise comparisons showed a clear difference between reference benchmark sites and wells of all age-criteria groups. We did not observe significant differences between well age-criteria groups in any cases (including interval plots). As with the community composition PCA, the lack of differences between well age-criteria groups suggests time since reclamation may not be the main driver of recovery. Though, GAMMs did show recovery of richness, diversity measures, and FH layer depth after a certain year post-certification.

In contrast with well-developed overstory forest structure in the benchmark forests, vertical structure analysis on wells reflected low plant cover in upper strata and low tree and shrub presence. Instead, wells were dominated by forbs, graminoids, and mosses. Indicator species analysis showed wells commonly had introduced species (e.g., *Taraxacum officinale*, common dandelion) and ruderal species (e.g., *Trifolium hybridum*, alsike clover).

2.3.1 Soil

Soil characteristics determine a large part of what can successfully grow in an ecosystem. Biological (organic layer depth), physical (compaction measured by bulk density), and chemical (pH) soil properties have significant implications for ecological recovery through relationships with vegetation, hydrology, and nutrient cycles. Elevated bulk density values, especially those exceeding 1.4 g cm^{-3}, are known to adversely affect plant growth in boreal forests (Sutton 1991; Binkley and Fisher 2019). Such compaction restricts root penetration and limits water and nutrient availability, which are crucial for tree development and overall plant health (Gale et al. 1991; Arshad and Coen 1992; Zhao et al. 2010; Frerichs et al. 2017). Furthermore, the higher soil pH levels observed on the wells may further complicate recovery, as pH is a reliable indicator of species diversity in boreal ecosystems and plays a critical role in nutrient cycling and community structure (Koptsik et al. 2001). The shift towards more alkaline conditions could favor the persistence of introduced plant species from agronomic seed mixes that were planted, which are often associated with such soil environments (Rose and

Hermanutz 2004), potentially impeding the growth of native species like aspen and white spruce (Zhang et al. 2013; Calvo-Polanco et al. 2017). While soil pH and bulk density did not show evidence of recovery, our FH depth GAMM showed recovery of the soil organic layer after 44 years. Soil water and nutrient status increase with LFH (FH+leaf litter) layer thickness (Lowry 1975; Beckingham and Archibald 1996). Forest clearing and large-scale forest fires decrease the natural LFH layer (Bonan and Shugart 1989; Pennock and van Kessel 1997; Bock and Van Rees 2002; MacKenzie et al. 2004). However, such disturbances rarely remove topsoil as well construction does. Soil organic layer regeneration over time suggests that, despite compaction and altered pH conditions, some aspects of soil structure and function can recover, potentially aiding in the long-term re-establishment of forest vegetation. However, the slow recovery of the FH layer might reflect the lack of leaf litter from trees/shrubs on wells and the more severe and lasting impacts of well disturbances compared to natural disturbances, where compaction and altered soil chemistry are less pronounced. This partial recovery underscores the complexity of post-disturbance ecological processes and highlights the need for targeted reclamation efforts to address specific challenges posed by well reclamation. These findings suggest that the recovery of soil properties on disturbed sites may be delayed or even permanently altered, with lasting consequences for forest regeneration and ecosystem services.

2.3.2 Understory vegetation

Plant richness and diversity showed evidence of recovery on wells over time, with wells having reference-level richness after 20 years and reference-level diversity around 40 years, suggesting some resilience within the plant communities following disturbance. However, the lack of recovery in understory community composition indicates that while the number of species may be increasing, the species composition remains altered. Graminoid, forb, and moss dominance (along with non-native and ruderal species like common dandelion and timothy grass) highlights a significant shift in community structure. This shift is concerning because these species are often associated with disturbed areas and can impede the recovery of native species, particularly woody plants that are essential for the re-establishment of a forest and long-term ecosystem stability and function. The presence of *Trifolium hybridum* (TRIHYB), *Vicia americana* (VICAME), *Taraxacum officinale* (TAROFF) and *Cirsium arvense* (CIRARV), highlights the persistence of introduced and ruderal species on old wells, indicating ongoing challenges in recovering native plant communities. The invasion of introduced and noxious species near anthropogenic disturbances has been well-documented, driven by increased bare ground and light availability (Rose and Hermanutz 2004; Langor et al. 2014). Some of these species may have been seeded rather than invading wells as old reclamation practices often involved unknowingly planting species not suited to the surrounding ecosystem with the aim to avoid soil erosion. Non-native species often have traits (which will be discussed in further detail in Chapter 3) that allow them to quickly colonize and dominate disturbed sites, outcompeting native vegetation and altering ecosystem processes. For instance, species like *Trifolium hybridum* (alsike clover) and *Vicia americana* (American vetch) can establish symbiotic relationships with nitrogen-fixing bacteria, potentially altering soil nutrient dynamics by increasing nitrogen levels (USDA 2024). This shift in soil conditions could further disadvantage native species, which are adapted to the low-nutrient soil conditions typical of boreal forests, therefore sustaining the altered community composition observed on the wells. Indicator species in reference benchmark forests, including *Linnaea borealis*, *Cornus canadensis*, *Mitella nuda*, and *Viburnum edule,* are characteristic of mature boreal forests with well-established native understory communities, which were largely absent from both new and old wells.

The prevalence of graminoids (including native species) on reclaimed wells may be preventing the growth of woody species by creating a physical barrier (preventing seeds of woody species from reaching soil, successfully germinating, and/or growing) and outcompeting woody species for light, water, and nutrients.

2.3.3 Overstory vegetation

Canopy cover, basal area, and stems per hectare have not yet recovered on wells, which is particularly troubling, as it suggests that the structural complexity of these ecosystems has been significantly compromised. The lack of living and dead trees means that key components of the forest structure -such as woody debris and a closed canopy- are missing. This absence has cascading effects on the understory, influencing microclimate conditions on the forest floor, including light availability, soil temperature, air humidity, and nutrient cycling (Andersson and Hytteborn 1991; Mills and Macdonald 2004; Botting and Fredeen 2006; Park and Carpenter 2016). A lack of canopy cover creates open conditions that may further facilitate the establishment and spread of ruderal and non-native species, reinforcing the altered trajectory of ecological recovery on these sites. Additionally, the reduced basal area and tree density observed on wells may be linked to the historical lack of tree planting on sites certified under older reclamation criteria, which did not require active reforestation. This has likely contributed to the persistence of an open, non-forested state on many reclaimed wells, limiting the potential for woody species recovery and the re-establishment of a more typical boreal forest structure. Slow recovery of these key structural elements underscores the challenges of restoring disturbed sites to their pre-disturbance conditions and highlights the need for more targeted reclamation strategies that prioritize the re-establishment of native tree species and the overall structural complexity of the forest.

Visualization of vertical structure occurrence (count and percent) showed that while reference benchmark forest plots have more vegetation in upper strata, Young age class wells have a more similar percent occurrence with more trees than older age class wells. This may be due to newer reclamation practices leading to tree planting. Old and Mid-Old wells were likely planted with agronomic seed mixes popular at the time, so shrubs such as *Salix* species may be present in upper strata rather than *Populus* or *Picea* species that can develop into the overstory canopy over time.

2.3.4 Limitations

Apart from spatial variation (partially accounted for by our paired study design), a principal limitation of this study was the variance in reclamation practices employed by each company to meet government standards. Although regulations aim to set a consistent standard for reclamation, the methods used to achieve these standards can differ widely due to constraints such as time, funding, and logistical considerations like the availability of native plant seeds. This variability introduces uncertainty into our understanding of the patterns observed in vegetation recovery, as differences in seed mix application or the planting of woody species may lead to inconsistent outcomes across sites. The move to digital application processes and a decrease in on-site validation may reduce incentives for companies to reclaim sites to the highest possible standard, potentially leading to further discrepancies in recovery. The assumption of identical starting conditions across reclaimed sites, despite variations in initial biotic and abiotic factors, complicates our ability to draw firm conclusions about the drivers behind the vegetation patterns we observed. Without detailed documentation of the reclamation actions taken at each site, it is challenging to assess whether the presence of introduced species, or the lack of woody species, is due to specific revegetation practices or other factors, such as passive regeneration before reclamation. The observed deviations from expected successional trends may reflect these inconsistencies in reclamation practices, underscoring the need for more standardized and transparent reclamation efforts. In addition, while the paired study design accounts for some spatial variation, it was not possible to create pairs to account for temporal variation. It would be ideal to look at adjacent forests naturally disturbed at the same time as well disturbance (paired start ages).

2.3.5 Management implications and future directions

Our research indicates that reclaimed wells may not develop a community composition or structure like reference sites within 44 years post certification. This lack of recovery may be due to modified soil properties and competition between adjacent forest species and established graminoid/forb species that were seeded during the reclamation process (such as alsike clover and timothy grass). Future research may expand on this study by reevaluating younger wells reclaimed under the updated criteria once they've had more time to recover. Ideally, the same wells could be surveyed to better analyze successional trajectories of sites with existing data. The long-term impact of industrial footprints on resources, biodiversity, ecosystem services, and processes must be considered. Repeated, long-term sampling can reveal a true time-dependent recovery. Alternatively, manipulative studies imitating a range of reclamation practices (e.g., meeting vs. exceeding reclamation criteria) would be helpful in determining the exact mechanisms that lead to some sites progressing and others staying in the pioneer stage (more like grasslands) of forest succession.

A better understanding of the recovery of plant communities from this study will inform the development of more effective reclamation practices and management of wells in the future as well as contribute to the study of cumulative effects of intensive oil and gas exploration in the study area. Additionally, this research will contribute to the broader understanding of succession in anthropogenically-disturbed forests informing ecological theory. The results of this project could improve reclamation and management practices for disturbed forests to increase biodiversity and habitat recovery. As many ungulates use the boreal forest for protection and/or nutrition, returning the fragmented landscape created by wells to pre-disturbance functioning is important for Alberta wildlife. This research also has important implications for policy makers and energy regulators that hold oil and gas companies responsible for the health of the land they lease (e.g. Environmental Protection and Enhancement Act, Conservation and

Reclamation Regulations, Alberta Energy Regulator reclamation requirements). Improved understanding of the oil and gas industry's impact on plant communities is essential for reducing negative consequences caused by anthropogenic disturbance and answering the question: are plant communities recovering from oil and gas disturbance functioning as expected? Learning about the mechanisms behind the plant community's functioning may help management practices move towards more efficient and effective revegetation guidelines. Our study underscores the importance of reclamation decisions on individual sites, as time does not appear to override arrested succession in many cases.

2.4 Tables

Table 2-1. Description of well pad age-criteria categories that were used in analysis. The 25 wells are divided into four age-criteria groups calculated as 2023 (data collection year) - reclamation certification year and pre and post 2007 (when the updated 2010 well site criteria for forested lands applied). Reference age range comes from counting rings on tree cores.

Age-criteria Group	Criteria	n	Age range - certified
Young	New post-2010	5	7-16
Mid-Young	Old pre-2010	9	17-23
Mid-Old	Old pre-2010	7	24-31
Old	Old pre-2010	4	32-44[1]
Ref	n/a	24[2]	(28-141, from tree cores)

[1]44 year-old well was reclaimed in 1990 but abandoned 20 years prior (1970), so it was assigned to old since there was likely passive regeneration.
[2]One reference was adjacent to two wells and was therefore used as the reference benchmark forest for both.

Table 2-2. Indicator species analysis for species in 25 certified reclaimed well pads and 24 reference sites in Alberta's Central Mixedwood (n=15) and Lower Foothills (n=10) Natural Subregions. See species associated with species codes in Appendix table A-1.

Benchmark Reference Group. 39 species.

Species	A	B	R^2	P-Value
CORCAN	0.982	1.000	0.991	0.001

LINBOR	1.000	0.958	0.979	0.001
LONINV	0.947	1.000	0.973	0.001
MITNUD	0.979	0.958	0.969	0.001
PETFRI	0.842	1.000	0.917	0.001
MAICAN	0.910	0.917	0.913	0.001
VIBEDU	1.000	0.833	0.913	0.001
ARANUD	0.994	0.833	0.910	0.001
ROSACI	0.856	0.958	0.906	0.001
RUBPUB	0.853	0.917	0.884	0.001
SPIANN	0.994	0.708	0.839	0.001
GALTRIR	0.834	0.833	0.834	0.001
GYMDRY	1.000	0.667	0.816	0.001
ABIBAL	0.986	0.667	0.811	0.001
RUBIDA	0.711	0.917	0.807	0.002
RIBLAC	0.920	0.708	0.807	0.001
RIBTRI	0.865	0.750	0.805	0.001
BETPAP	0.847	0.750	0.797	0.001
OSMDEP	1.000	0.625	0.791	0.001
PLESCH	0.872	0.667	0.763	0.002
VACMYR	0.982	0.583	0.757	0.001
PTICRI	0.898	0.625	0.749	0.001
ATHFIL	1.000	0.542	0.736	0.001
RHOGRO	1.000	0.500	0.707	0.001
GALBOR	0.783	0.625	0.700	0.010
PYRASA	0.905	0.500	0.673	0.002
HYLSPL	0.891	0.500	0.667	0.004
VIOSPP	0.861	0.500	0.659	0.001

STRAMP	1.000	0.417	0.645	0.001
POPTRE	0.898	0.458	0.641	0.004
VACVIT	1.000	0.375	0.612	0.001
PICMAR	0.969	0.375	0.603	0.002
AMEALN	1.000	0.250	0.500	0.008
VIOCAN	1.000	0.250	0.500	0.014
TOMNIT	0.943	0.250	0.486	0.040
LONDIO	1.000	0.208	0.456	0.023
MAISTEL	1.000	0.208	0.456	0.024
EURCON	0.953	0.208	0.446	0.045
VACOXY	1.000	0.167	0.408	0.049

Wells Group. 27 Species.

Species	**A**	**B**	**R^2**	**P-Value**
SMALLGRAM	0.941	1.000	0.970	0.001
TAROFF	0.943	0.960	0.951	0.001
BRACMOSS	0.891	1.000	0.944	0.001
TRIHYB	0.947	0.880	0.913	0.001
SALSPP	0.875	0.920	0.897	0.001
PLASPP	0.825	0.960	0.890	0.001
VICAME	0.840	0.880	0.860	0.001
EQUARV	0.709	0.960	0.825	0.001
SYMPUN	0.839	0.800	0.819	0.001
ACHALP	0.985	0.640	0.794	0.001
SOLCAN	0.798	0.760	0.779	0.001
ACHMIL	0.883	0.640	0.752	0.002
PHLPRA	1.000	0.560	0.748	0.001
NONBGRAM	0.918	0.600	0.742	0.003

CASMIN	0.975	0.520	0.712	0.002
THUREC	0.886	0.520	0.679	0.002
AULPAL	0.910	0.480	0.661	0.002
SONARV	0.919	0.440	0.636	0.003
CIRARV	1.000	0.400	0.632	0.001
HERMAX	0.857	0.400	0.586	0.042
GEUMAC	0.775	0.440	0.584	0.021
BOTVIR	0.849	0.400	0.583	0.046
CAMSTE	0.771	0.440	0.582	0.019
RANACR	1.000	0.320	0.566	0.006
DREADU	0.938	0.320	0.548	0.01
MELOFF	1.000	0.280	0.529	0.009
NEWGRAMREC	0.953	0.280	0.517	0.023

Table 2-3. Indicator species analysis for species in 25 certified reclaimed well pads grouped into new criteria (n=5) and old criteria (n=20) and 24 reference sites in Alberta's Central Mixedwood (n=15) and Lower Foothills (n=10) Natural Subregions. See species associated with species codes in Appendix table A-1.

Benchmark Reference Group. 28 species.

Species	**A**	**B**	**R^2**	**P-Value**
LINBOR	1.000	0.958	0.979	0.001
CORCAN	0.943	1.000	0.971	0.001
MITNUD	0.952	0.958	0.955	0.001
LONINV	0.856	1.000	0.925	0.001
VIBEDU	1.000	0.833	0.913	0.001
ARANUD	0.992	0.833	0.909	0.001
MAICAN	0.859	0.917	0.887	0.001
ROSACI	0.801	0.958	0.876	0.001
PETFRI	0.737	1.000	0.859	0.001

RUBPUB	0.780	0.917	0.846	0.001
SPIANN	0.972	0.708	0.830	0.003
GYMDRY	1.000	0.667	0.816	0.002
OSMDEP	1.000	0.625	0.791	0.001
ABIBAL	0.933	0.667	0.789	0.003
RIBLAC	0.869	0.708	0.785	0.004
GALTRIR	0.716	0.833	0.772	0.003
RIBTRI	0.770	0.750	0.760	0.01
RUBIDA	0.592	0.917	0.737	0.008
ATHFIL	1.000	0.542	0.736	0.014
VACMYR	0.918	0.583	0.732	0.012
PLESCH	0.780	0.667	0.721	0.027
RHOGRO	1.000	0.500	0.707	0.013
PTICRI	0.760	0.625	0.689	0.044
GALBOR	0.725	0.625	0.673	0.042
STRAMP	1.000	0.417	0.645	0.027
HYLSPL	0.766	0.500	0.619	0.046
VACVIT	1.000	0.375	0.612	0.035
PICMAR	0.912	0.375	0.585	0.05

Newer Criteria Wells. 16 Species.

Species	A	B	R^2	P-Value
SYMPUN	0.639	1.000	0.799	0.005
BRACMOSS	0.565	1.000	0.752	0.004
PLASPP	0.532	1.000	0.730	0.026
SALSPP	0.519	1.000	0.720	0.024
EQUARV	0.480	1.000	0.692	0.022
AULPAL	0.775	0.600	0.682	0.029

DREADU	0.747	0.600	0.669	0.01
CASMIN	0.740	0.600	0.666	0.039
NONBGRAM	0.550	0.800	0.663	0.046
THUREC	0.630	0.600	0.615	0.043
TRIPRA	0.908	0.400	0.603	0.017
MELOFF	0.856	0.400	0.585	0.021
PLAHUR	0.814	0.400	0.570	0.044
HIEUMB	0.800	0.400	0.566	0.046
PINCON	0.779	0.400	0.558	0.032
NEWGRAMREC	0.732	0.400	0.541	0.047

Older Criteria Wells. 7 Species.

Species	A	B	R^2	P-Value
TRIHYB	0.722	0.950	0.828	0.001
VICAME	0.698	0.950	0.814	0.001
ACHMIL	0.755	0.750	0.753	0.007
TAROFF	0.523	1.000	0.723	0.019
SMALLGRAM	0.496	1.000	0.705	0.029
CIRARV	0.946	0.450	0.652	0.022
SONARV	0.751	0.500	0.613	0.04

Table 2-4. Results of Generalized Additive Mixed Models of similarity between wells and references as a function of time since reclamation for soil and vegetation attributes measured at 25 wells and adjacent reference sites (n=24) in Fox Creek. Given is the chosen similarity metric used (Bray-Curtis or Euclidean), *k* value, estimated degrees of freedom (*edf*), *F* value, and significance (*P*) for the smoother. Temporal pattern as determined based on if/when the smoother and reference confidence intervals overlapped (Fig. 2-1). The percent difference between the mean reference similarity and the highest point on the smoother curve is also given. See also Figs. 2-11 to 2-20.

Attribute	*Similarity metric*	*k*	*edf*	*F*	*P-value of smoother*	*Temporal pattern*	*Years to recovery*	*% difference smoother high point and reference mean*
Soil								
Bulk density	Euclidean	4	2.4	10.33	<.0001	No resilience	> 44	6.4%
pH	Euclidean	4	2.6	16.54	<.0001	No resilience	> 44	16.1%
FH depth	Euclidean	4	2.4	20.25	<.0001	Resilience	~ 44	9.5%
Overstory								
Basal area	Bray-Curtis	4	1.0	6.789	0.0094	No resilience	> 44	77.5%
Canopy cover	Bray-Curtis	3	1.9	10.06	<.0001	No resilience	> 44	45.0%
Stems per ha	Bray-Curtis	4	1.0	14.31	<0.002	No resilience	>44	32.4%
Understory								
Richness	Euclidean	4	2.1	17.01	<.0001	Resilience	~ 20	0.30%
Shannon Diversity	Euclidean	4	2.0	12.14	<.0001	Resilience	~ 40	6.0%
Inverse Simpson Diversity	Euclidean	3	1.0	10.54	<0.002	Resilience	~ 37	3.5%
Understory Composition	Bray-Curtis	4	1.6	1.655	0.107	No resilience	> 44	64.5%

2.5 Figures

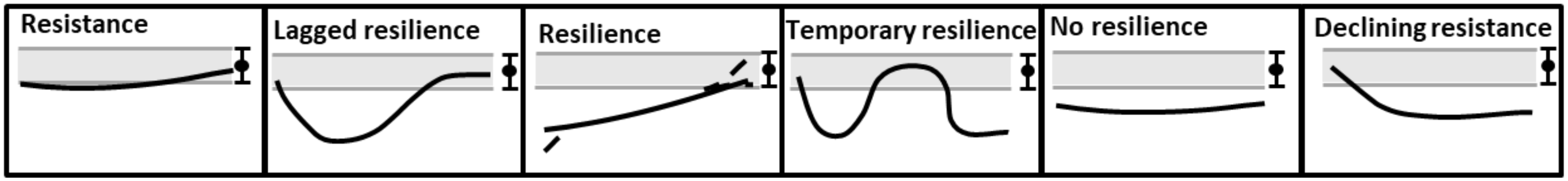

Figure 2-1. Temporal patterns of response curves for the wellsites (y axis: Similarity, x-axis: years post-certification): “Resistance” (well to reference similarity GAMM curve overlaps with the lower confidence interval of the mean reference to reference similarity), “Lagged resilience” (declining similarity between wells and references (i.e., loss of resistance), followed by increasing similarity (resilience) with recovery at the year post-reclamation when the curve intersects the lower confidence interval of the reference to reference similarity), “Resilience” (low well to reference similarity in the recent years post-reclamation indicating a lack of resistance, followed by an increase, indicating resilience and recovery), “Temporary resilience” (low well to reference similarity in recent years post-reclamation followed by the curve intersecting the lower confidence interval of the reference to reference similarity then declining back below the reference to reference similarity range), “No resilience” (well to reference similarity falls below reference to reference similarity), “Declining resistance” (decrease in well to reference similarity over time, indicating a loss of resistance, with no evidence of subsequent resilience).

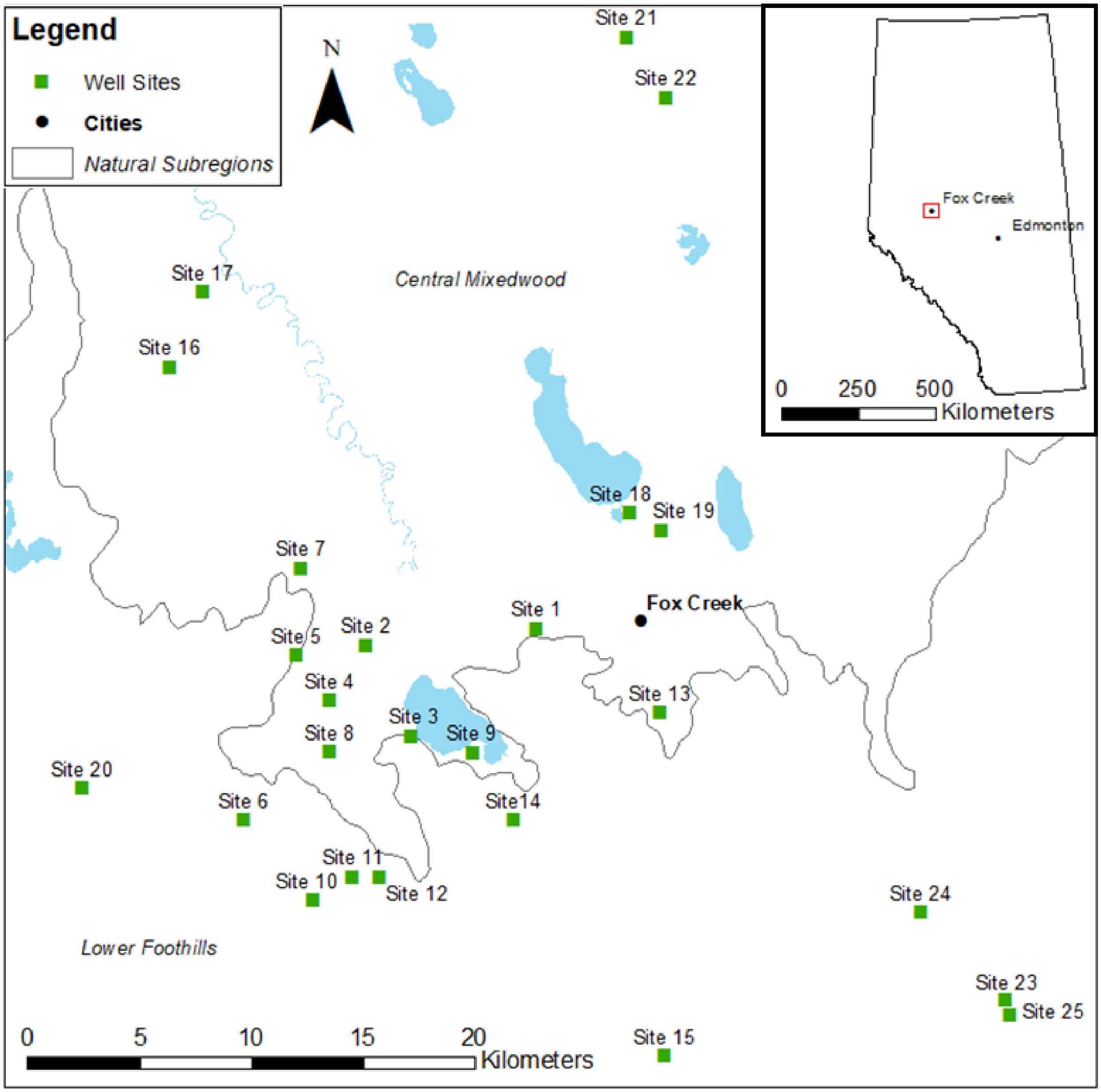

Figure 2-2: Map showing the locations of 25 certified reclaimed oil and natural gas production well pads and adjacent benchmark reference pairs in Alberta's upland forested lands near the city of Fox Creek, Alberta. The sites are distributed across the Central Mixedwood (n=15) and Lower Foothills (n=10) natural subregions. The inset map displays the broader context of Alberta, with Edmonton and Fox Creek highlighted for reference.

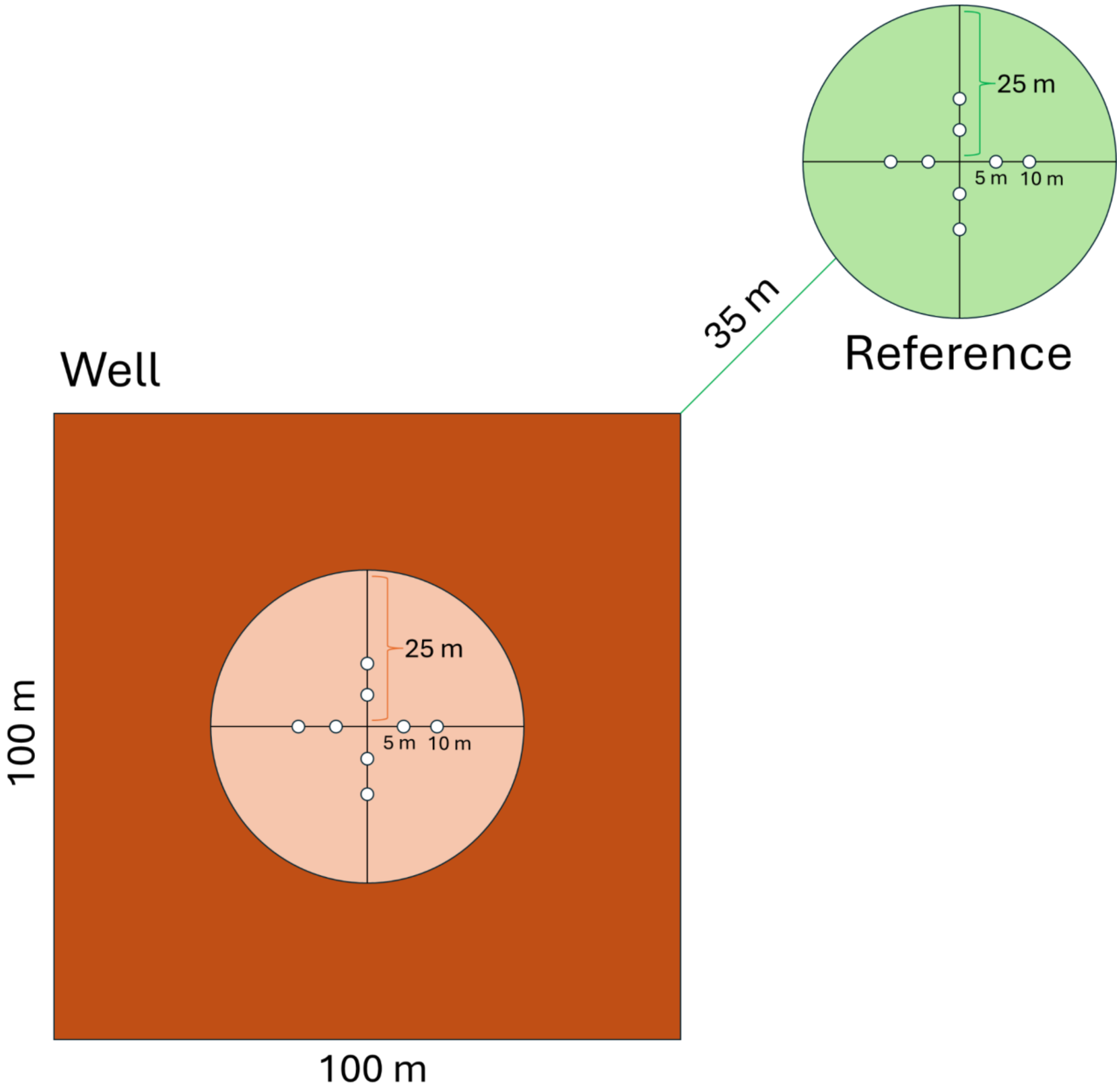

Figure 2-3: Schematic of the site setup for sampling. Wells were sampled within a 25 m radius circle centered on the well bore, with transects extending in each cardinal direction to form four quadrants. Data collection points were established at 5 m and 10 m distances from the center in each direction, where canopy cover measures and soil samples were taken. Within the quadrants, tree data, moss surveys, and tallest tree height measurements/species identification, along with tree cores (if present), were conducted. A reference plot located at least 35 m into the adjacent forest followed the same 25 m radius circle and data collection setup.

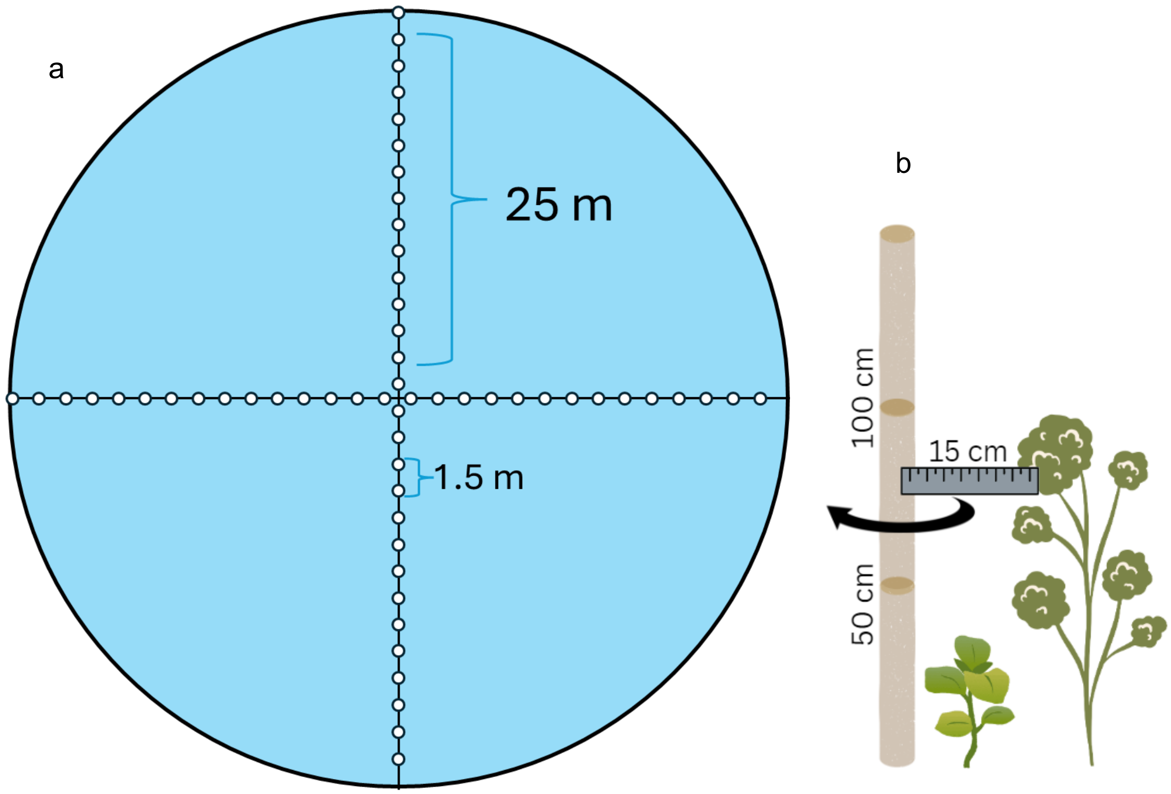

Figure 2-4: Illustration of the point contact method used for vegetation species data collection. **Fig. 2-4a:** Species presence was recorded every 1.5 meters along the transects, with 30 points per 50 m transect, resulting in a total of 60 points per plot. This method was employed to accurately capture species composition across the different height strata within the site. **Fig. 2-4b:** Species presence was defined as photosynthetic material within the 15 cm radius of the center wooden dowel for each 50 cm range height strata (included: 0-50 cm, 50-100 cm, 100-150 cm, 150-200 cm, 200-250 cm, and 250+ cm).

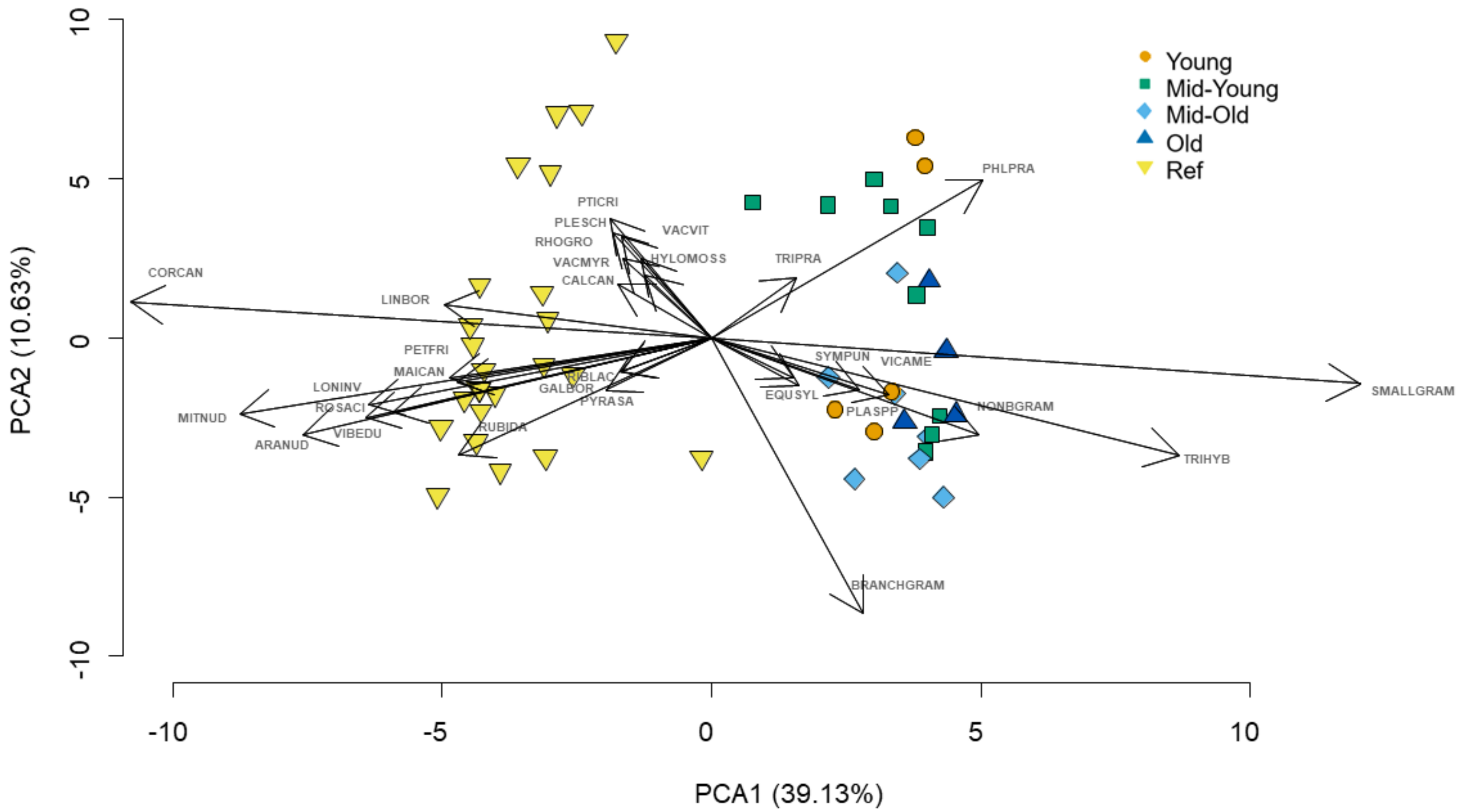

Figure 2-5. Principal component analysis (PCA – scaling 2) of understory composition assessed along a post-reclamation chronosequence and reference sites grouped by age-criteria group. Age- criteria groups: Young (new criteria), Mid-Young (old criteria), Mid-Old (old criteria), Old (old criteria), and Ref (mature benchmark reference forest). Vectors indicate the 30 species with the highest occurrence in the study. See Table 2-1 for age-criteria group descriptions. Refer to appendix table A-1 for species codes.

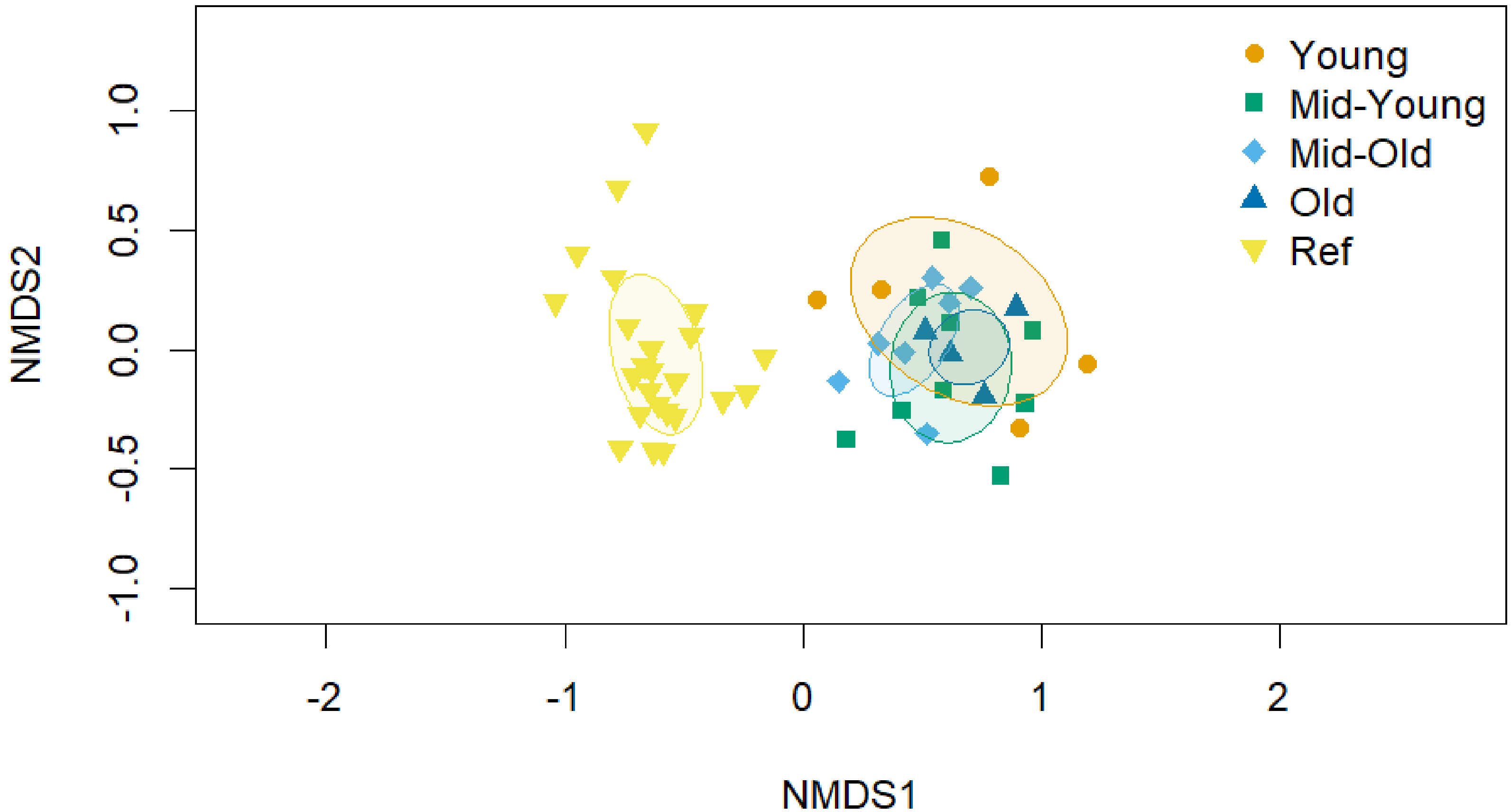

Figure 2-6. Nonmetric multidimensional scaling ordination of vegetation species composition with 95% confidence interval ellipses for age-criteria group showing separation of wells and references but overlap among well age-criteria groups. See Table 2-1 for age-criteria group descriptions

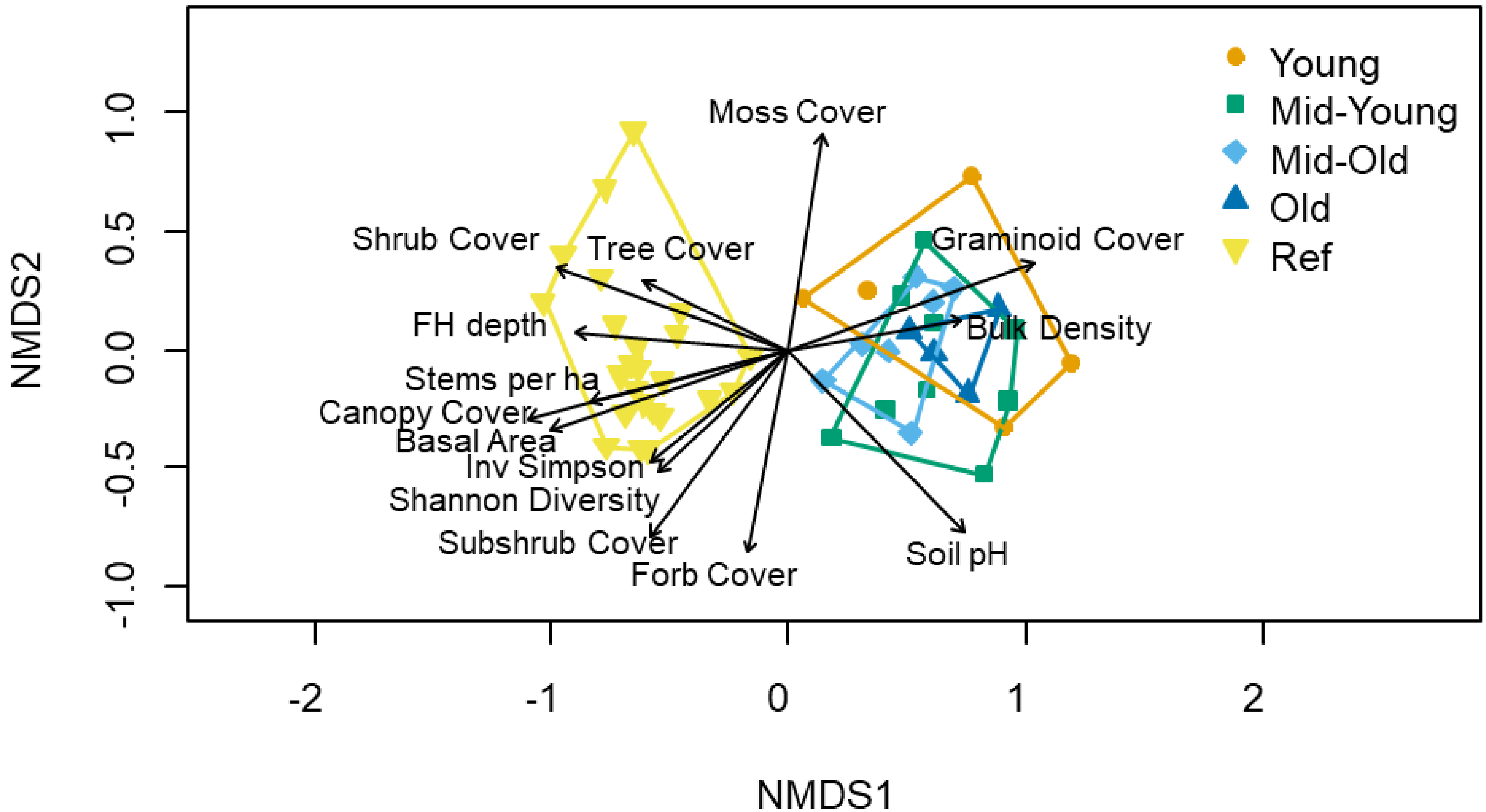

Figure 2-7. Nonmetric multidimensional scaling ordination of vegetation species composition with hulls for age-criteria group: Vectors indicate environmental, vegetation, soil, and diversity variables). Vector direction and length reflect the strength of correlation with the first two axes. See Table 2-1 for age-criteria group descriptions.

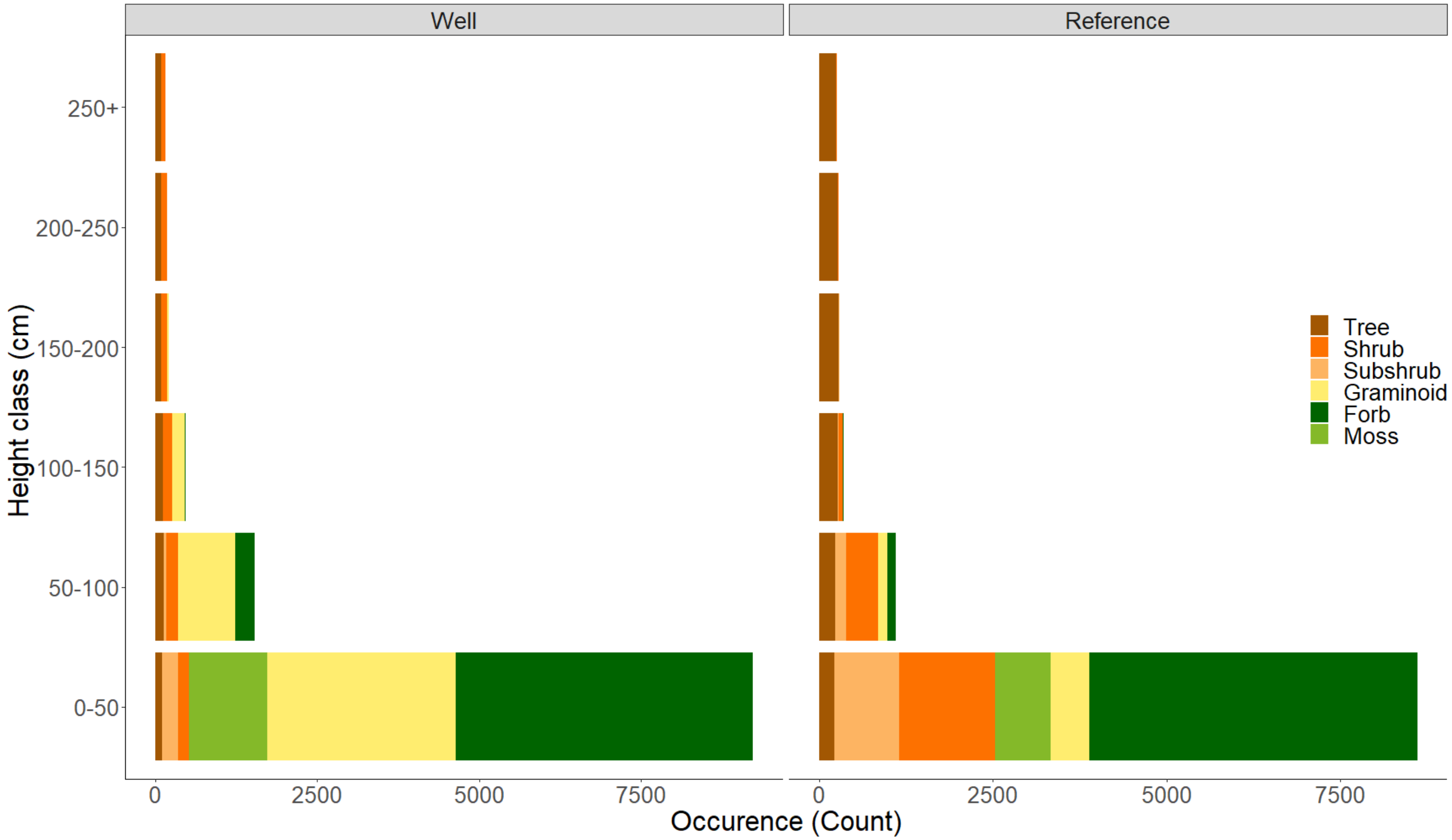

Figure 2-8. Occurrences (count) of understory vegetation grouped into plant life forms in each 50 cm height strata. Data is grouped by well and reference group.

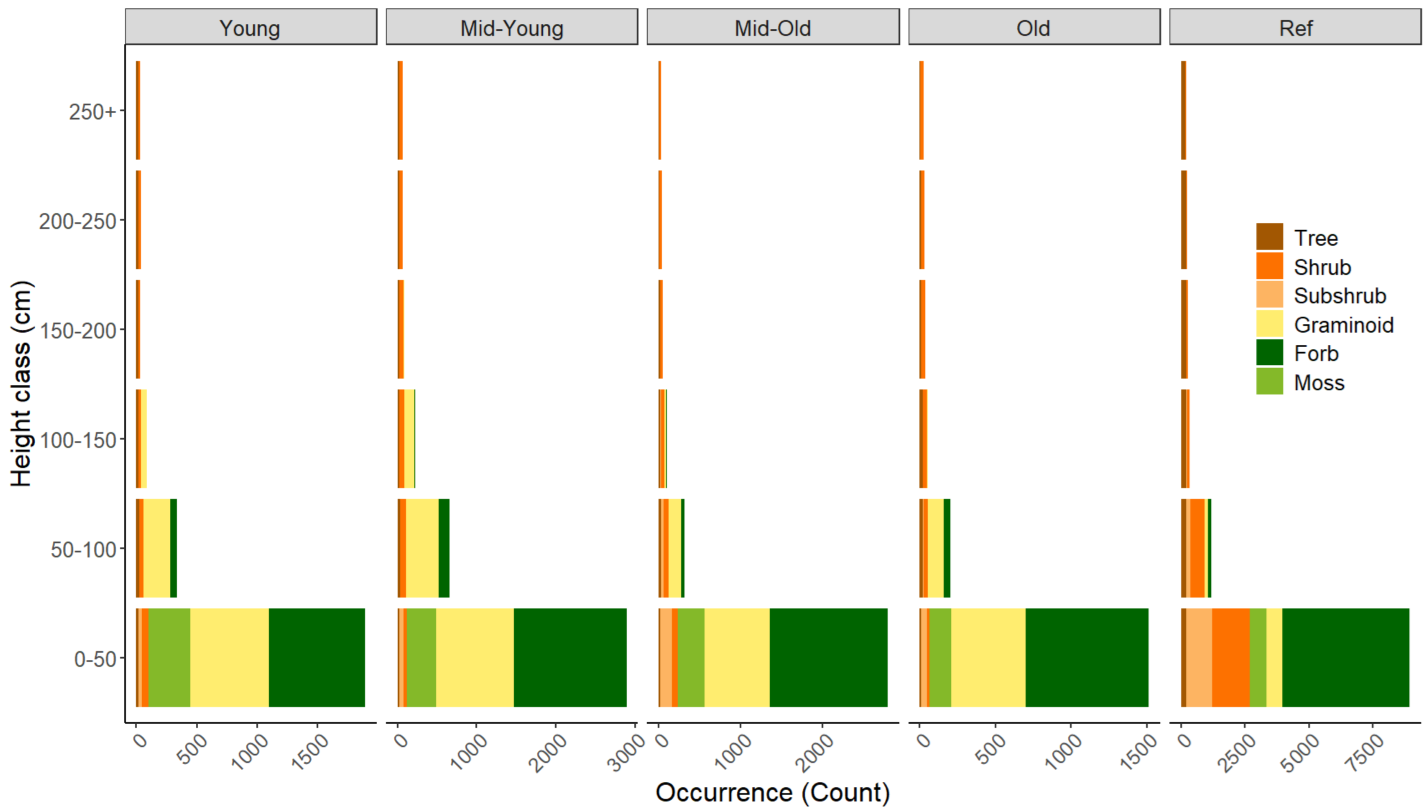

Figure 2-9. Occurrences (count) of understory vegetation grouped into plant life forms in each height strata. Data is grouped by age-criteria group. Note that totals along axes are different (especially reference since there are more plots in that group). See Table 2-1 for age-criteria group descriptions

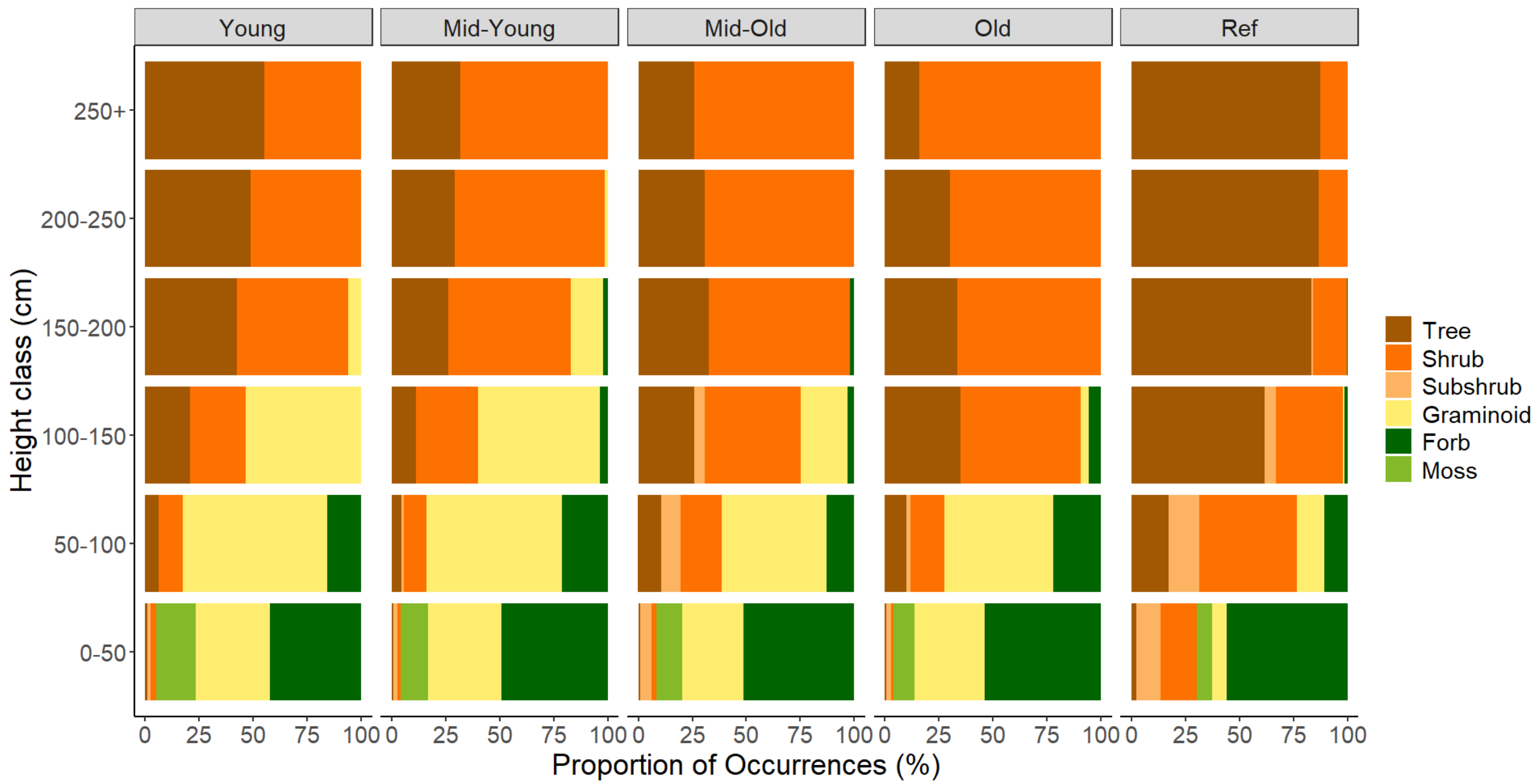

Figure 2-10. Proportion of occurrences (%) of understory vegetation grouped into plant life forms in each height strata. Data is grouped by age-criteria group. see table 2-1 for age-criteria group descriptions

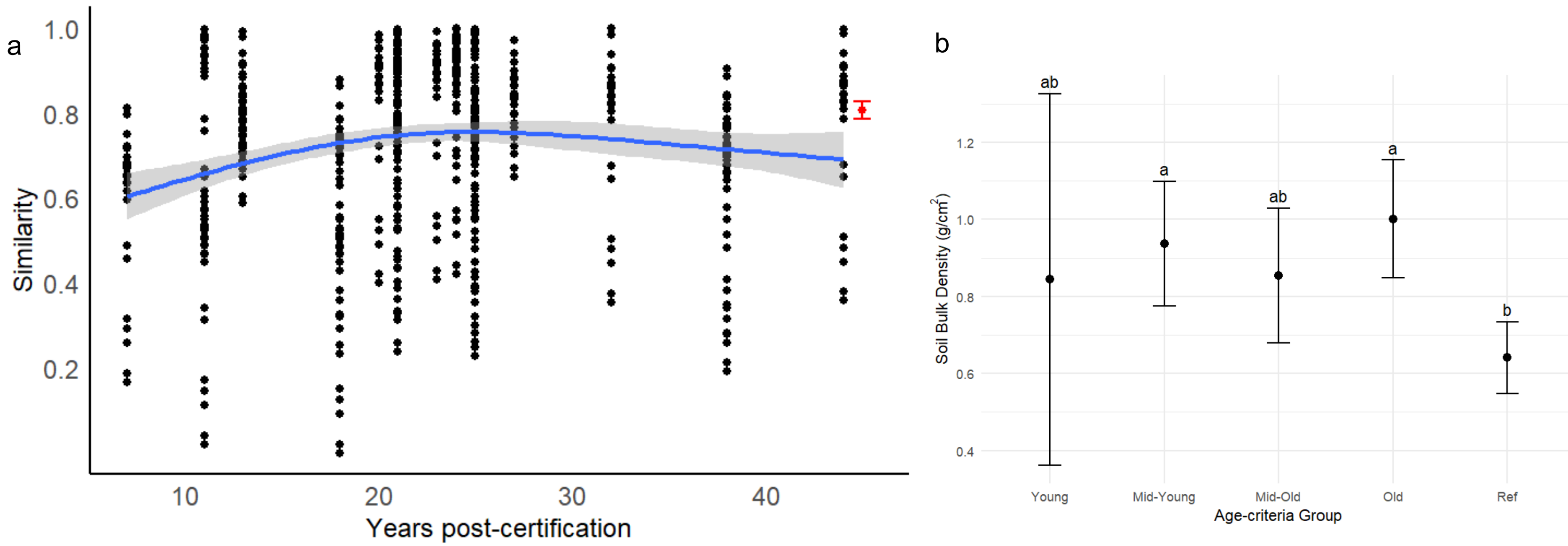

Figure 2-11 - Soil bulk density on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for soil bulk density grouped by age criteria group (see Table 2-1 for age-criteria group descriptions). **Fig. 2-11a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Euclidean distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-11b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

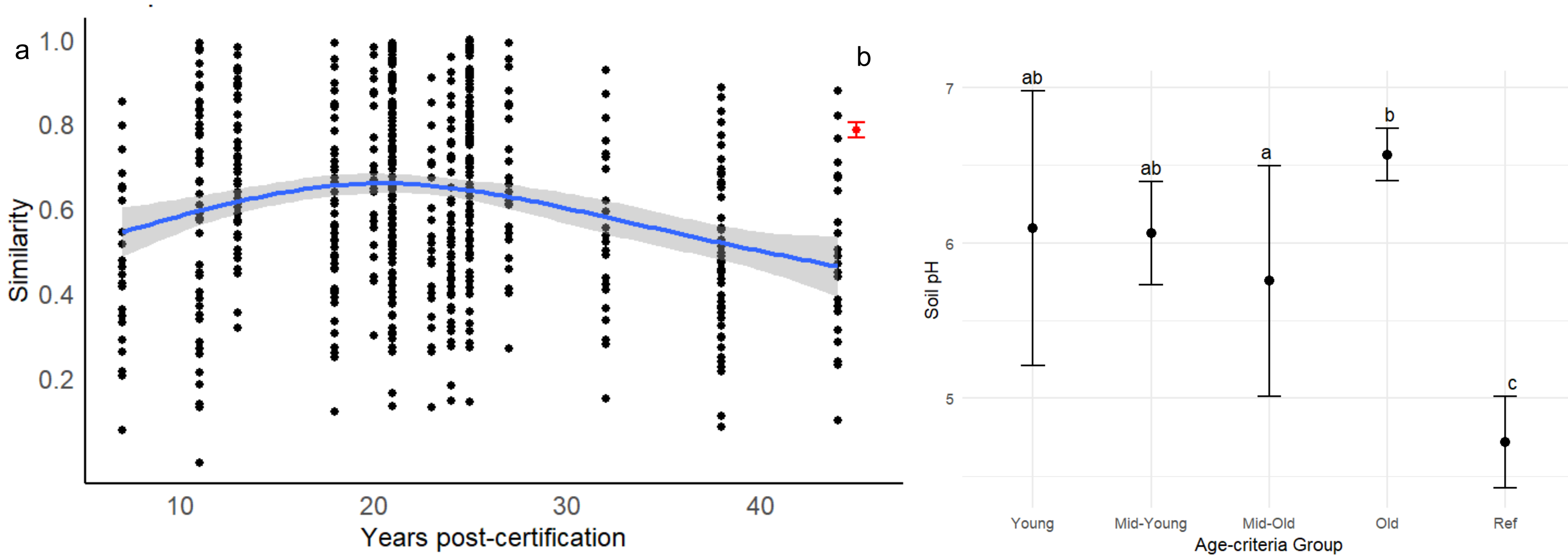

Figure 2-12 - Soil pH on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for soil pH grouped by age criteria group (see table 2-1 for age-criteria group descriptions). **Fig. 2-12a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Euclidean distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-12b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-c) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

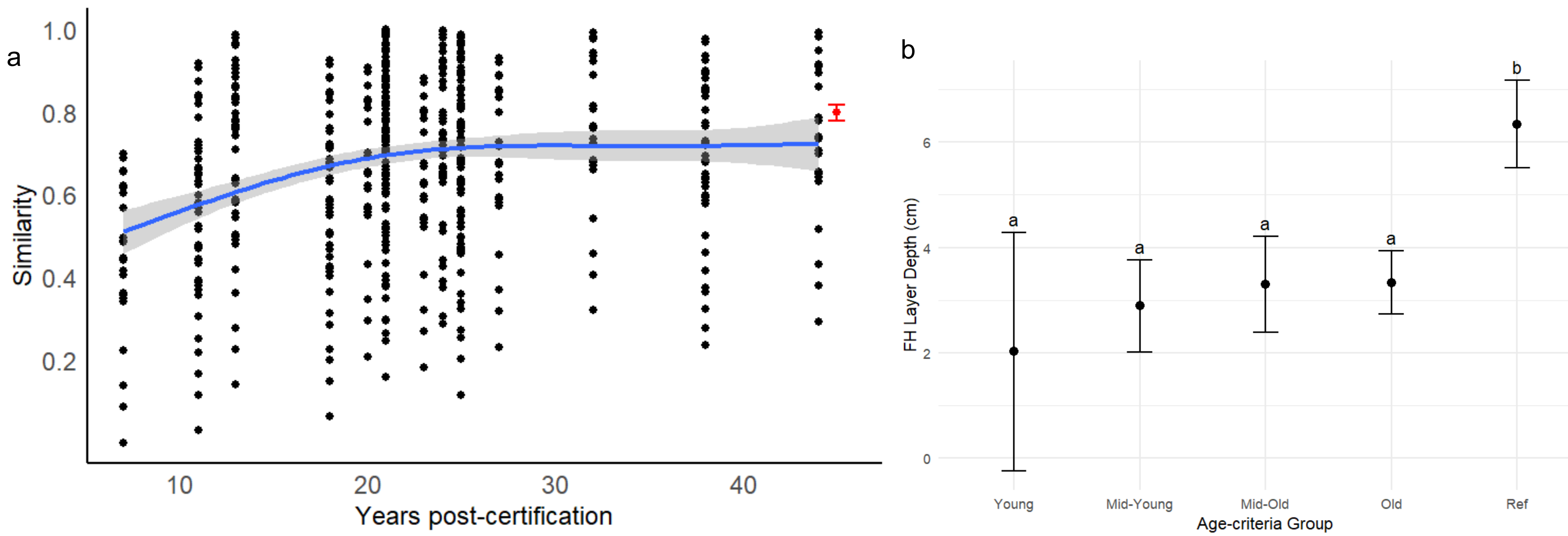

Figure 2-13 - Soil FH depth on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for soil FH depth grouped by age criteria group (see Table 2-1 for age-criteria group descriptions). **Fig. 2-13a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Euclidean distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-13b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-c) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

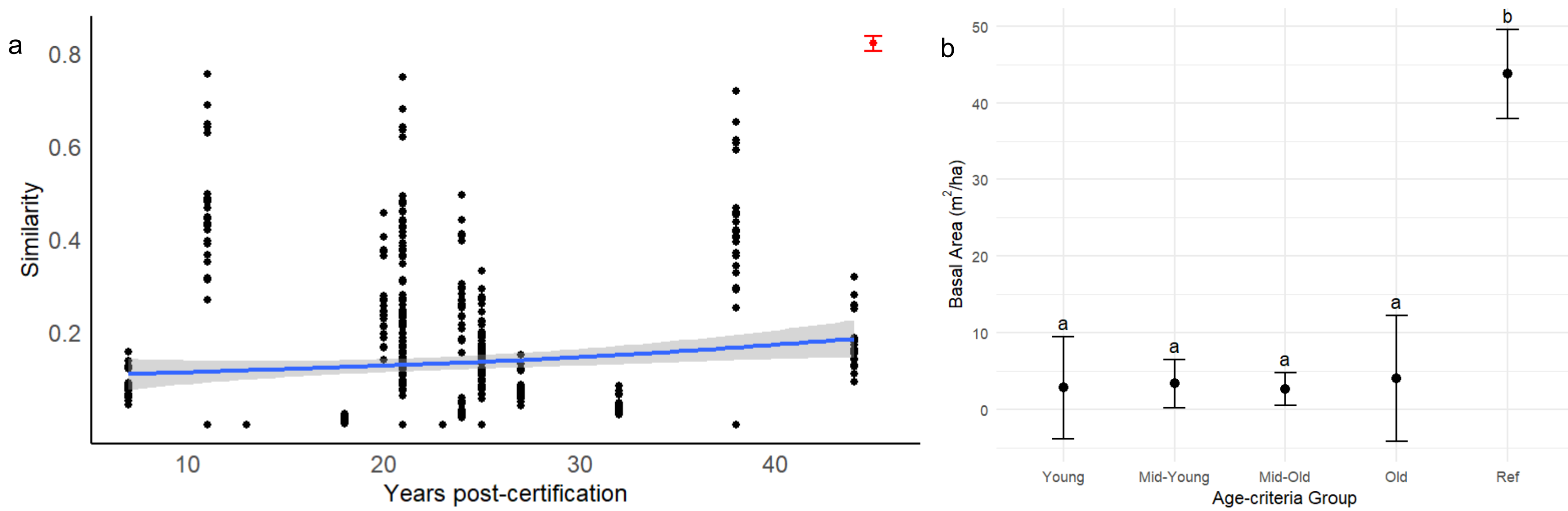

Figure 2-14 - Basal area on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for basal area grouped by age criteria group (see table 2-1 for age-criteria group descriptions). **Fig. 2-14a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Bray-Curtis distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-14b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

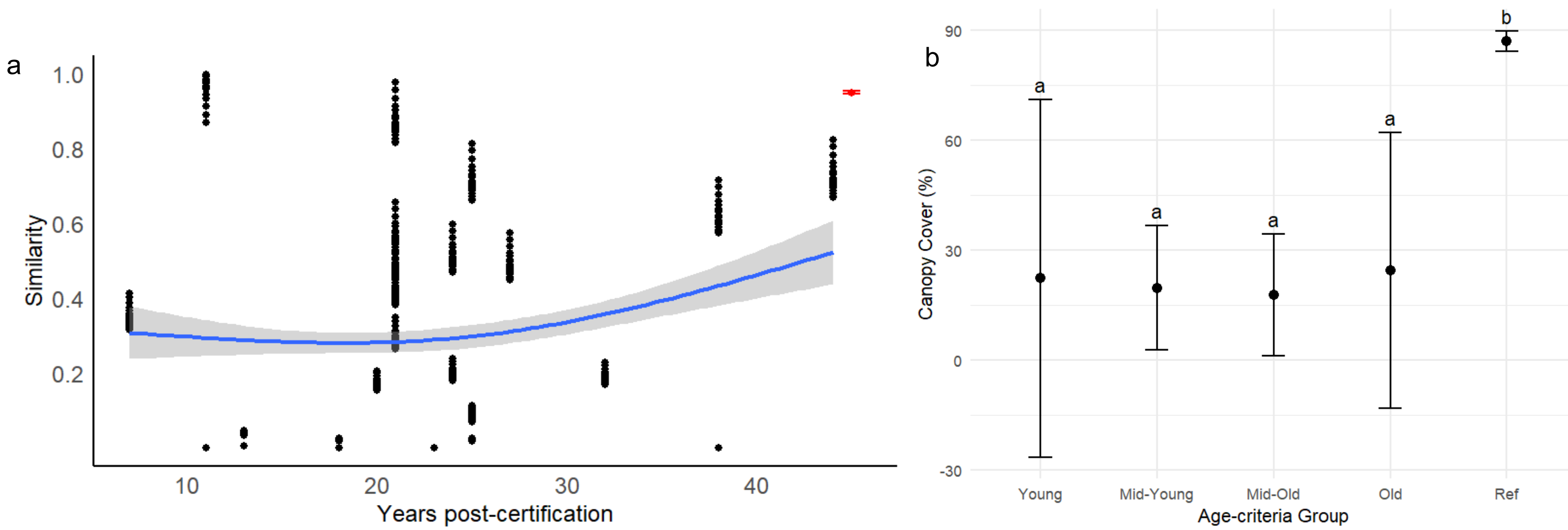

Figure 2-15 - Canopy cover on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for canopy cover grouped by age criteria group (see Table 2-1 for age-criteria group descriptions). **Fig. 2-15a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Bray-Curtis distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-15b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

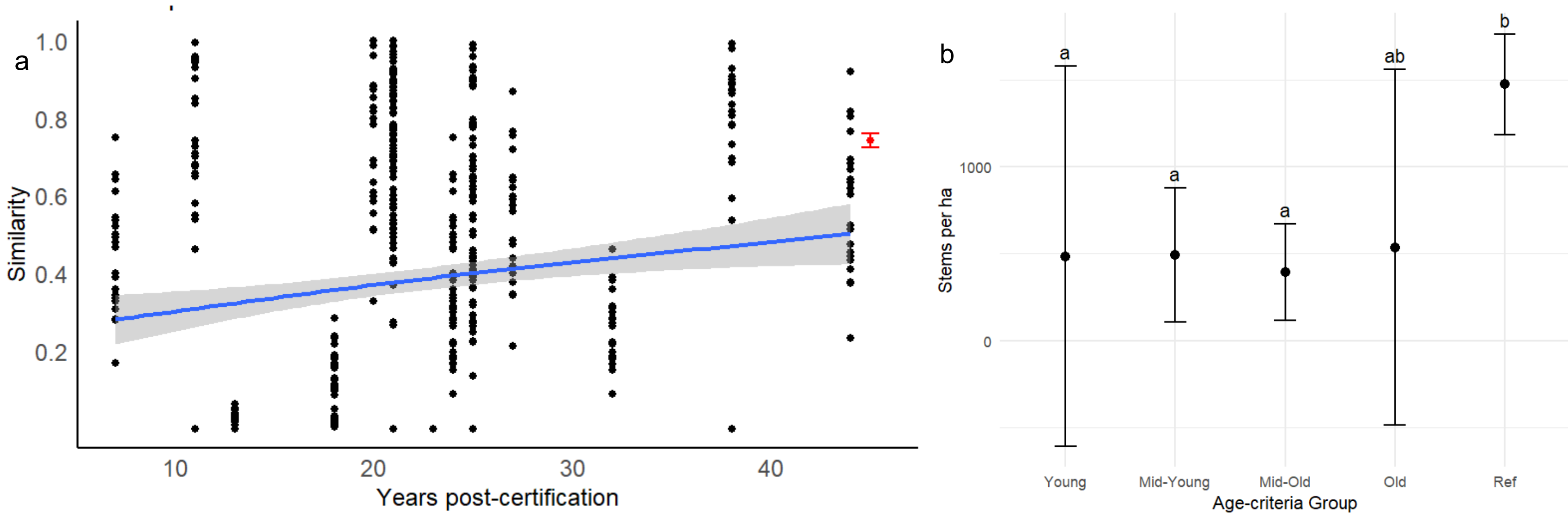

Figure 2-16 - Stems per ha on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for stems per ha grouped by age criteria group (see table 2-1 for age-criteria group descriptions). **Fig. 2-16a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Bray-Curtis distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-16b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

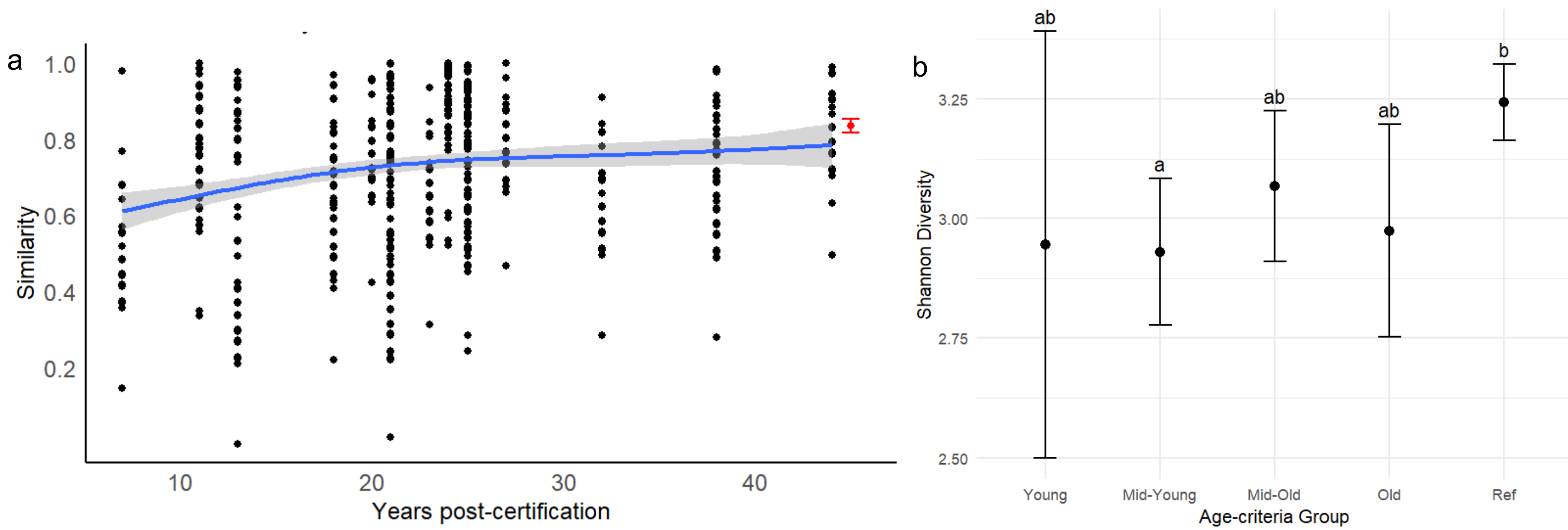

Figure 2-17- Shannon diversity on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for Shannon diversity grouped by age criteria group (see table 2-1 for age-criteria group descriptions). **Fig. 2-17a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Euclidean distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-17b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

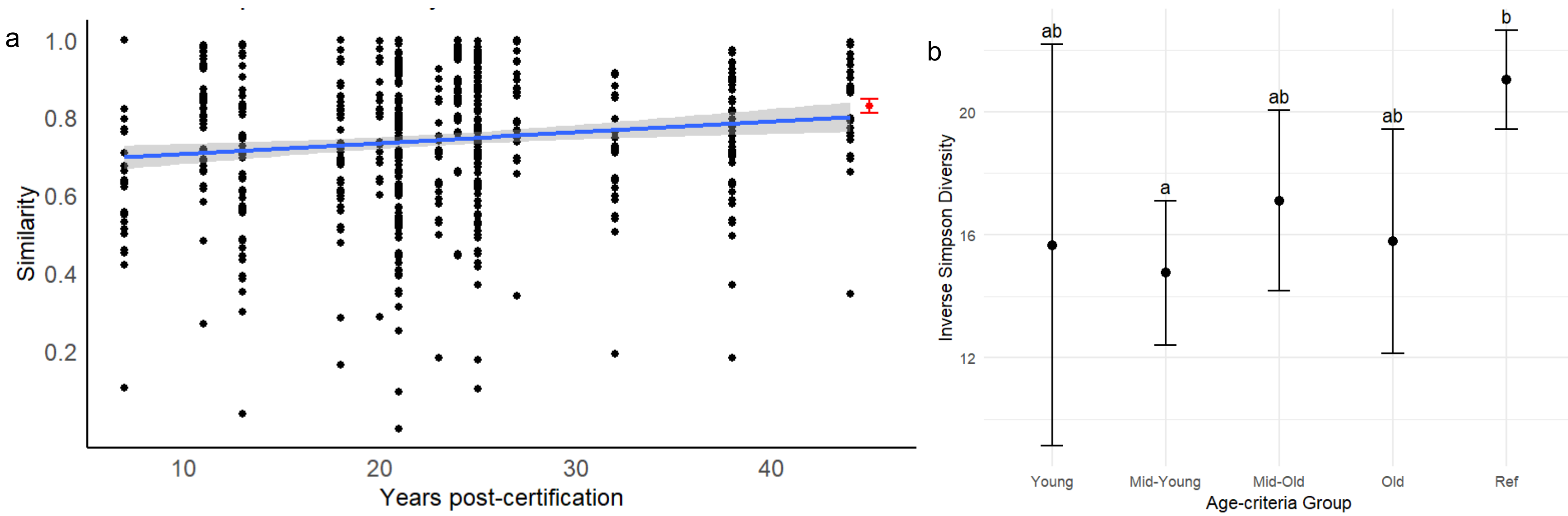

Figure 2-18 - Inverse Simpson's diversity on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for inverse Simpson's diversity grouped by age criteria group (see table 2-1 for age-criteria group descriptions). **Fig. 2-18a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Euclidean distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-18b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

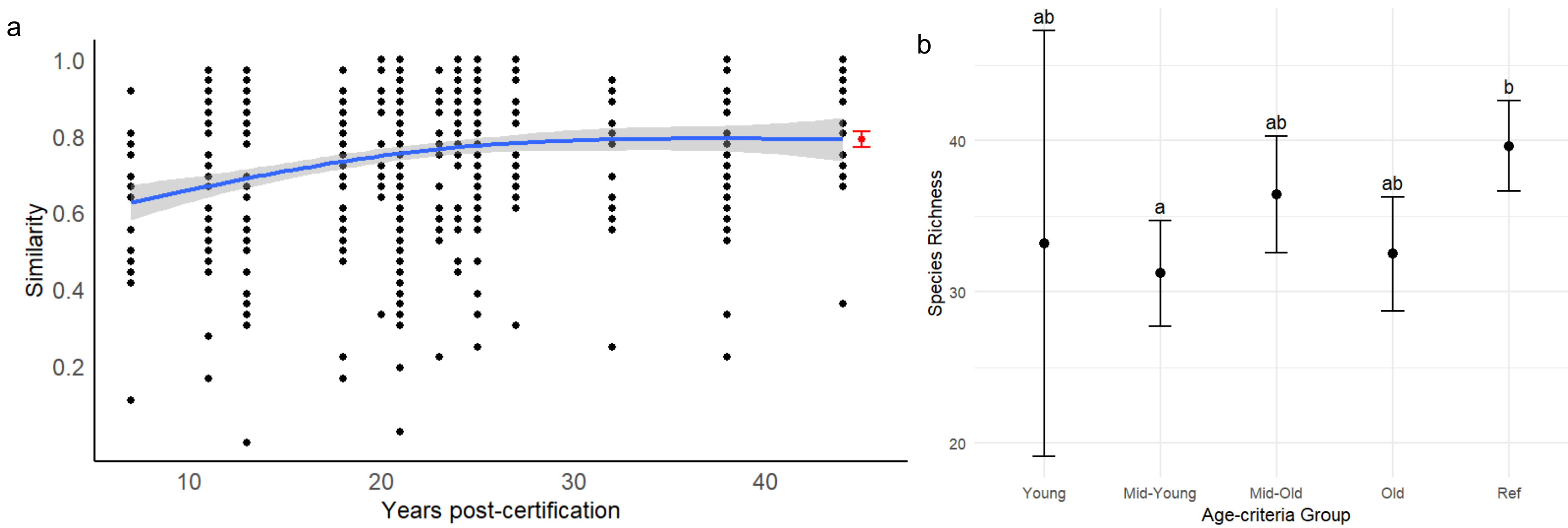

Figure 2-19 - Species richness on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for species richness grouped by age criteria group (see table 2-1 for age-criteria group descriptions). **Fig. 2-19a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Euclidean distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 2-19b:** Points represent the mean raw data for each age-criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

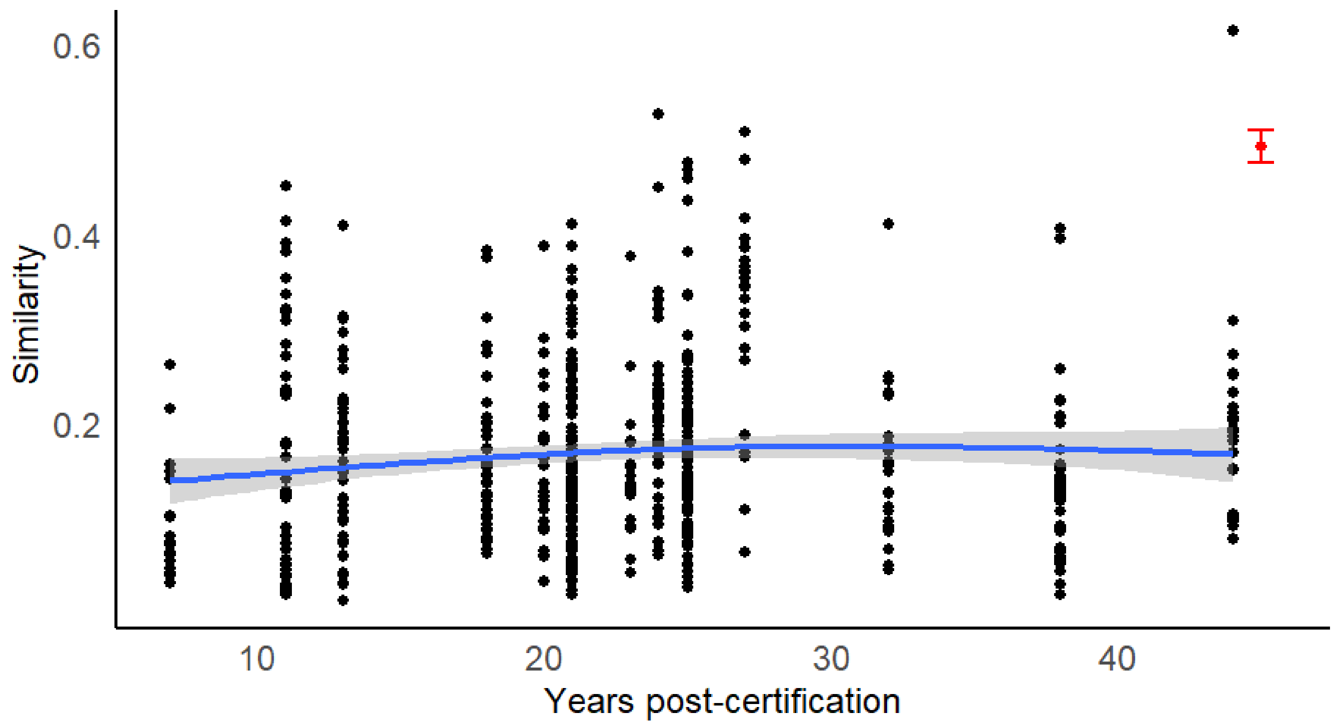

Figure 2-20 - Understory composition on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests**.** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Bray-Curtis distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years.

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Chapter 3. Functional trait recovery on reclaimed well pads of varying age and reclamation criteria towards reference benchmark forests

3.0 Introduction

Boreal forests (also called taiga) are the largest land biome, accounting for 33% of Earth's forested area (Natural Resources Canada 2015). These forests are found in cold, northern latitudes below the Arctic tundra and above temperate deciduous forests. Boreal species are therefore adapted to long, cold winters, spanning large portions of Russia, Canada, and Scandinavia. Boreal forests provide essential ecological, social, and economic services. Ecologically, boreal forests provide carbon sequestration and wildlife habitat (Lemprière et al. 2013; Bradshaw and Warkentin 2015). Humans also benefit through recreation, medicinal herbs, and food sources (Uprety et al. 2012; Kujala et al. 2023). Economically, resource extraction is a major industry in boreal forests. Boreal forests produce approximately 33% of global wood and 25% of paper (Gauthier et al. 2015). In western Canada, the oil and gas industry plays a significant role, with extensive land being cleared annually for petroleum extraction activities (Schneider et al. 2003). However, resource extraction and exploration activities can lead to a long-term decline in forest quality. Habitat fragmentation, hydrologic discontinuity, reduced regeneration and productivity, and increased human access from industrial disturbances could cause the extinction of endemic populations and impaired ecological function (Fahrig 2003; Nishi et al. 2013; Riva and Nielsen 2021). Thus, understanding boreal forest responses to natural and anthropogenic disturbances is essential to preserving the services they provide.

North American boreal forests have evolved along with natural disturbance for 12,000 years (Weber and Stocks 1998; Brandt et al. 2013). Boreal forests experience climatic (e.g., windthrow, frost, and ice damage), biological (e.g., insect outbreaks, disease and herbivory), and pyrogenic (wildfires) disturbances. These natural disturbances play a crucial role in shaping the composition, abundance, and distribution of vegetation by modifying the surrounding environment and the availability of resources over space and time (Pickett and White 1985). Disturbance decreases the overall amount of living plant biomass (Reader 1991), while also providing surviving plants or propagules with more resources such as light, space, soil moisture, and nutrients (Canham and Marks 1985). This decrease in vegetation, and therefore competition, also creates an opportunity for new species to establish in the area (Grime 1973; Collins 1987). Depending on the magnitude, frequency, and intensity of the disturbance, boreal communities are reset to an earlier stage of ecological development, with a gradient of early successional possibilities (De Grandpré et al. 1993). Natural disturbances are vital for supporting species diversity and maintaining ecological processes; many boreal species (e.g., fire-adapted species like jack pine and lodgepole pine) even rely on disturbance for regeneration (Kneeshaw et al. 2011).

While disturbances are a natural part of boreal forest dynamics, the introduction of anthropogenic disturbances has led to novel conditions (Riva et al. 2020). These anthropogenic disturbances can introduce new challenges due to the absence of a historical equivalence. Larger scale anthropogenic disturbances, including clearcutting for timber and pulp and paper extraction, mining, and oil and gas development (e.g., wells, seismic lines, pipelines), have varying degrees of soil disturbance and impacts on the forest vegetation (Davidson et al. 2020; Filicetti and Nielsen 2022). Many of these disturbances have been concentrated only in the past century and may overlap spatially or temporally with each other and/or natural disturbances. These disturbance dynamics may affect the future successional trajectory of the affected forest (Bergeron and Fenton 2012; Lupardus et al. 2019).

Successional trajectories of forests are determined by the intensity and spatial distribution of disturbance (Turner et al. 1998). In boreal forests, primary succession may occur (e.g., after glacial retreat or strip mining; (Tardif et al. 2019; Anyomi et al. 2022). Secondary succession, the dominant type of succession in boreal forests, occurs as the disturbance agent selectively removes species from the community and modifies the forest structure and composition depending on disturbance frequency, severity, and scale (Kellomäki 2022). It can result from both natural and anthropogenic disturbance, with varying degrees of change in forest structure and function. Localized treefall may create minor forest changes that primarily lead to similar species succeeding, whereas a stand replacing wildfire may burn and consume the overstory canopy, resetting the system and allowing pioneer herb species to restart growth (Payette 1992; Buettel et al. 2017). Disturbance associated with oil and gas extraction is more intense as it not only removes the canopy, but also disturbs the forest floor vegetation community with the leveling of the soil surface prior to drilling and installation of the well (Pickell et al. 2014). Well operation may also occasionally lead to spilling contaminating the soil system (Rowell and Florence 1993). Understanding the variables and processes that contribute to succession patterns is therefore essential, particularly with increasing anthropogenic disturbances such as resource extraction (Oliver and Larson 1996; Messier and Puettmann 2011). Historically, successional research has focused primarily on one or a few components (e.g., only species composition) that impact forest succession, despite the knowledge that multiple components shape the succession process (Meiners et al. 2015). Gaining a deeper understanding of forest succession remains a significant challenge in the field of ecology (Taylor et al. 2020) especially in the boreal given the magnitude of anthropogenic disturbance.

Disturbance type and magnitude will influence the associated impacts on the forest understory vegetation. Understory plants contain most boreal diversity and are essential for maintaining forest ecosystem health (Gilliam 2007). Understory plants (including shrubs, forbs, grasses, and non-vascular plants) provide food and habitat for insects, birds, and mammals. They also affect the environment above and below-ground (Nilsson and Wardle 2005; Hart and Chen 2006). Above ground, they can interact with growing trees (George and Bazzaz 2003); for example, dense understories can alter tree regeneration and succession (Lieffers et al. 1993;

Royo and Carson 2006). Below ground, rapid growth and high turnover of the understory contribute to nutrient cycling and leaf litter. Dynamics of these understory plant communities are shaped by a variety of factors and their interactions (Reich et al. 2012). Understory composition is largely determined by water, nutrients, and light. Anthropogenic disturbances can affect all three parameters. Disturbance impacts on the understory will impact the future development of a forest, making it uncertain how fast natural regeneration of trees and shrubs will occur after disturbance. Factors that impact the plant communities in the understory are still poorly understood. Therefore, an understanding of boreal plant communities' dynamics and functioning is essential for studying their response to anthropogenic disturbance.

Plant functional traits, such as rooting depth and seed weight, are heritable features of a plant species that affect the performance of individuals via survival, growth, and reproduction (Violle et al. 2007; Garnier et al. 2015). These traits play a crucial role in shaping the diversity and productivity of terrestrial ecosystems (Garnier et al. 2015). In recent years, functional traits have become a powerful way to assess ecosystem health and services (Lepš et al. 2011; Garnier et al. 2015). Matching plant characteristics to their function can further our understanding of the specific adaptations species have developed for surviving in different environments. For example, species with a higher specific leaf area (SLA; where SLA is defined as the ratio between leaf area and leaf dry mass) may be more adapted to environments with higher resource availability, while those with a lower SLA may excel in resource-limited conditions (Dahlgren et al. 2006; Grime 2006). Trait-based approaches also allow comparison of ecosystems over spatial and organizational scales (Dawson et al. 2021). Traits provide insights into the adaptive strategies of species and their ecological roles within a community.

The study of plant functional traits has become increasingly popular for determining a community's response to its environment and predicting how ecological systems will respond to disturbance (Lepš et al. 2011). Traits provide a system to quantify and evaluate how species interact with their environment. Trait-based approaches emphasize the functional aspects of biodiversity, allowing researchers to draw conclusions across various ecosystems and scales. For example, plant functional trait data have been used to evaluate plant community successional trajectories (Azeria et al. 2020), responses to climate change (Aubin et al. 2016), persistence of disturbance footprints (Dabros et al. 2022) and invasion resistance (Conti et al. 2018). Studying plant functional traits is particularly important in the context of disturbance ecology, as understanding patterns in characteristics of disturbed plant communities can inform future management strategies (Aubin et al. 2024). Trait-based ecology can be a powerful tool in restoring ecosystem services in forest systems (Aubin et al. 2024).

In Canadian boreal forests, the oil and gas industry constitutes a notable anthropogenic disturbance, with Alberta experiencing a substantial amount of this kind of disturbance. For example, in the Alberta Pacific (ALPAC) forest management area (5.8M ha) in northeastern Alberta, 11,000 ha/yr are cleared for the petroleum industry (Schneider et al. 2003). A variety of operations by oil and gas companies disturb the land, such as seismic lines, pipelines, oil sands, processing plants, tailings ponds, and wells. There has been significant focus on the cumulative consequences of energy extraction in the mineable (surficial) Athabasca Oil Sands, but some argue that in-situ bitumen development may have considerably greater cumulative effects that are anticipated to grow significantly (Nishi et al. 2013). In contrast to linear oil and gas disturbance (i.e. seismic lines, pipelines), drilling a well creates a ~100 × 100 m disturbance. A level surface is required before drilling can begin, so vegetation is cleared, surface soil is temporarily removed, and subsurface soils are leveled. The result is a bare site with removal of native vegetation and seed bank. Approximately 460,000 hectares of Alberta’s land are estimated to be affected by oil and gas wells (ABMI 2018; Janz et al. 2019) of which ~41.5% are in the boreal plains ecozone (AER, 2019 data). Due to growing public awareness of the cumulative effects and industry contributions to habitat fragmentation and biodiversity loss, stakeholders have called for decreasing the total footprint of energy extraction (Powter et al. 2012). Oil and gas exploration and extraction often result in significant environmental disturbance (Pickell et al. 2015), making actions to mitigate these impacts essential.

To reduce this environmental impact, reclamation practices are required after a well is no longer in use and has been abandoned (Flemming et al. 2023). Reclamation practices aim to minimize the damage caused by the wells by restoring the soil, vegetation, and hydrology to a state comparable to the land's original condition. The goal is to ensure that wells in the boreal forest are on a positive trajectory towards returning to a forested state. In 1963, the Alberta government implemented the Surface Reclamation Act to address concerns voiced by landowners over wells. This legislation aimed to regulate the restoration of land that had been disrupted by industrial activity. In the past, reclamation efforts focused on planting agronomic seed mixes to prevent soil erosion, with an emphasis on the belief that "green is good". However, this approach has led to a decline in biodiversity and ecosystem resilience, with certain sites remaining in a stagnant or slow-growing state (Lupardus et al. 2019). This contrast between the well and surrounding forest results in a fragmented landscape leading to decreased connectivity, biodiversity loss for some species, increased edge effects, higher vulnerability to invasive species, and changes to ecosystem processes (Saunders et al. 1991; Fahrig 2003; Riva and Nielsen 2021). Research assessing ecosystem recovery on wells has also shown that wells have low soil quality and land capability (Janz et al. 2019); declines in native plant communities, plant cover, and plant species richness (Lupardus et al. 2019; Azeria et al. 2020); and decreased functional diversity (Lupardus et al. 2020). Approximately 28% of Alberta’s wells are certified reclaimed or exempted (Well Status 2022). Since 2010, reclamation standards have shifted from a simple focus on green is good to the utilization of native plant species and the restoration of woody vegetation on forested lands. Certain methods in well construction have been modified to minimize the extent of disruption, such as the implementation of advanced drilling techniques and more winter drilling. Over 10 years have passed since the well criteria has changed, giving us the opportunity to begin to assess its effectiveness at setting wells on a trajectory towards the target undisturbed ecosystem (i.e., benchmark reference forest). However, there is a lack of post-certification monitoring for recovered areas to determine whether recovery trajectories are developing according to expectations. Therefore, it remains unclear if plant communities, including the traits of the communities, on reclaimed wells will return to a state similar to their pre-disturbance condition, and whether the updated criteria will enhance recovery compared to the older criteria.

In this study, building on plant community taxonomic and structural recovery that I focused on in Chapter 2, I aimed to evaluate the functional recovery of understory plant communities following reclamation of oil and gas wells in Alberta's boreal forests. Six potential pathways for recovery (Fig. 3-1, Macdonald et al. 2024) were considered ranging from "No resilience" (well to reference similarity falls below reference to reference similarity) to "Resistance" (well to reference similarity GAMM curve overlaps with the lower confidence interval of the mean reference to reference similarity consistently over time). Intermediate resilience pathways included: "Lagged resilience" (declining similarity between wells and references (i.e., loss of resistance), followed by increasing similarity (resilience) with recovery at the year post-reclamation when the curve intersects the lower confidence interval of the reference to reference similarity), "Resilience" (low well to reference similarity in the recent years post-reclamation indicating a lack of resistance, followed by an increase, indicating subsequent resilience and recovery), "Temporary resilience" (low well to reference similarity in recent years post-reclamation followed by the curve intersecting the lower confidence interval of the reference to reference similarity then declining back below the reference to reference similarity range), "Declining resistance" (decrease in well to reference similarity over time, indicating a loss of resistance, with no evidence of subsequent resilience). I compared a suite of community weighted mean plant functional traits (above-ground, e.g., light requirement, seed dispersal vector; below-ground, e.g. rooting depth, storage organ) between reclaimed wells and reference forests to determine their plant community's functional recovery over time. Reclaimed wells were grouped by years since reclamation certification and categorized as either pre- or post- 2010 criteria, resulting in a set of age-criteria groups. Primary objectives included: i) quantifying functional syndromes of the understory plant community on wells compared with benchmark reference forest, and ii) identifying how time since reclamation has impacted the similarity of functional traits between wells of different age-criteria groups and adjacent reference forest benchmark stands. I hypothesized that if age post reclamation is important for trait recovery, then older reclaimed wells would have functional trait composition more like the forest benchmark because they have had more time to recover in post-disturbance succession compared with the younger wells. However, I also hypothesized that if reclamation practices are important for recovery, then wells with younger reclamation dates (post-2010 criteria update) would have functional trait composition more like the forest benchmark than the older reclaimed wells that were reclaimed under 'green is good' practices because they were reclaimed under a more ecological framework (e.g., use of native species, minimum requirements of woody species, heterogeneous topography, and downed woody debris).

3.1 Methods

3.1.1 Study Area

This study takes place in west-central Alberta, Canada in the Central Mixedwood Boreal Forest Natural Subregion (Albert et al. 2006), approximately 250 km northwest of Edmonton (near Fox Creek, AB; 54.4009° N, 116.8045° W). The study area supplies resources for timber and oil and gas extraction, with a history of hydrocarbon exploration and development dating back to the 1960s. A high density of wells and associated infrastructure, including pipelines, roads, and access corridors characterize the area. This infrastructure can impact local hydrology, soil conditions, vegetation, and wildlife.

Common terrestrial animal species in this area include deer, black bears, cougars, moose, and coyotes. We sampled upland sites characterized by aspen (*Populus tremuloides* Michaux), mixedwood (mix of conifer and deciduous), and white spruce (*Picea glauca* (Moench) Voss) forests. Other common overstory species in the area include paper birch (*Betula papyrifera* Marshall), balsam poplar (*Populus balsamifera* Linnaeus), and balsam fir (*Abies balsamea* (Linnaeus) Miller). Common understory vegetation includes low bush cranberry (*Viburnum edule* (Michaux) Rafinesque), prickly rose (*Rosa acicularis* Lindley), green alder (*Alnus alnobetula* subsp. *crispa* (Aiton) Raus), bunchberry (*Cornus canadensis* Linnaeus), wild sarsaparilla (*Aralia nudicaulis* Linnaeus), dewberry (*Rubus pubescens* Rafinesque), and bluejoint (*Calamagrostis canadensis* (Michaux) Palisot de Beauvois) growing on Gray Luvisols of various textures. The subregion contains a mix of wetlands and upland forests with short, warm summers and long, cold winters (Albert et al. 2006). The highest and lowest mean monthly temperatures are 15.6°C (July) and -10°C (January) respectively (Government of Canada, 2020) with an average annual temperature of 2.6°C (Fox Creek Junction weather station). Summer precipitation is highest with the greatest rainfall in July at 101 mm, and the average annual precipitation is 595 mm (Smerdon et al. 2019).

3.1.2 Sampling Design and Site selection

We conducted this observational study during the summer of 2023 (early June - early August) using a chronosequence of reclaimed wells that captured a range of ages post-certification. Potential sites were identified using data from AbaData (Abacus Datagraphics Ltd.'s AbaData Oil and Gas Map Software. Accessed May 2023). I selected former wells having homogeneous disturbance with flat or nearly flat topography that was less than 30 minutes of walking from an accessible road. Each well was paired with an upland adjacent reference forest that had not been disturbed by oil and gas and was also uninterrupted by road, pipeline, etc. except for two of the wellsites which shared a reference (due to close proximity). Reference forest stands (n=24) in the study area ranged from 28 to 141 years old with an average age of 74.5 (median age 71) likely due to frequent anthropogenic (e.g., clear cuts) and natural disturbances (e.g., stand-initiating fire). Each reference plot was used as a benchmark with which to compare the recovery of the wellsite. Sites were visited to confirm that there was no additional disturbance that was not captured by AbaData and were excluded when there had been subsequent major disturbance activity in the area. As there had been recent fires in the area and associated road closures, any burned areas were excluded from site exploration.

I sampled 25 sites (well + adjacent reference forest benchmark stand pairs) from a pool of 64 potentially suitable sites, which were all within a ~50 km² area (Fig. 3-2) and their reclamation certification dated from 1985-2016 (38-7 years before data collection). If reclamation certification occurred more than 10 years after site abandonment and within the same criteria time frame, the abandoned year was used for the well age instead to account for passive recovery during that time. Therefore, the oldest well is aged 44 years (post abandonment). Data collection occurred during the peak of the growing season with sampling completed by early August.

Each site (n=25) had a circular well plot and adjacent benchmark forest reference (n=24) plot; each plot had 25 m transects in each cardinal direction from the center point (n=4) and associated quadrants (e.g., NE quadrant). Sample points 5 m and 10 m from the center along each transect were used for soil core and canopy cover data collection. Contact points every 1.5 meters along each transect were used for taxonomic vegetation data collection (Figs. 3-3, 3-4).

3.1.3 Data collection and processing

The first phase of data collection began in mid-June with the first visit to each site (pair of well and forest plot). At each study site, a magnetic locator was used to locate and flag the center of the well (well bore). This marked the center of the study plot (if not found, an estimate based on GPS coordinates, equal distances to the four well edges, or area of heaviest disturbance was used). We then established four 25 m transects oriented to each cardinal direction using 30 and 50 m tapes and marked 5 m and 10 m from the center using pigtails and flagging tape. Each adjacent reference forest benchmark plot was established following the same procedure with its corner that was closest in proximity to the wellsite at least 35 m from the well corner to avoid edge effects and its center approximately 60 m from the well edge (Fig. 3-3). Chosen adjacent forest plots were ideally homogeneous (i.e., in site type, species present, topography, etc.); however, if there was a visible age difference, evidence of other disturbances, or change in topography, transects were adjusted to attempt a mostly homogeneous representation of the reference forest. If the reference forest on one side of the well appeared older than the other, the older forest was most often chosen to keep the average age of the reference forests in a similar range (avoiding large variation in benchmark conditions).

Ecological variables (e.g., diversity indices, soil variables, etc.) were measured as described in Chapter 2. Leaves for functional trait measures (SLA, leaf C, leaf N) were collected on site, while all other trait data were taken from the Traits of Plants in Canada (TOPIC) + GROOT database (Aubin et al. 2012**,** Table 3-1).

3.1.3.1 Leaf collection and analysis methods

We used plant occurrence data from the contact-point method (Aubin et al. 2008) to identify which species occurred at least 25% within each plot (i.e., at least 15 of 60 contact points contained the species) to select species to collect leaves from (Fig. 3-4). We also filtered those species to those which were present in $\geq$ 25% of all plots (e.g., at least 6 of the 25 well plots) for both well and reference plant species. We then visited sites (in late July and early August) which had multiple selected species and collected leaves from multiple individual plants. The number of leaves collected depended on the size of the leaves (e.g., we collected

30 leaves of small-leaf species like *Linnaea borealis* but for medium-leaf species like *Aralia nudicaulis,* we collected 10 leaves). We collected specimens from three different sites for each species and had separate bags filled with leaves for nutrient analysis that would later be sent to the Great Lakes Forestry Centre. Once leaves (with petioles) were removed from a branch, they were stored in containers with wet paper towels and placed in a cooler until they could be refrigerated overnight to prevent desiccation before scanning.

We scanned all leaves and petioles from each species within each plot prior to storage in separate envelopes (labeled with plot, date, number of leaves, and species) using Image J (Schindelin et al. 2012) to calculate leaf and petiole areas from the scanned images. We then dried the leaves and petioles in a 70°C oven for 48 hours and weighed them separately. SLA was calculated using leaf area (m^2) / leaf mass(kg) (Pérez-Harguindeguy et al. 2013).

Leaves were shipped to the Great Lakes Forestry Centre Analytical Lab for carbon and nitrogen content measurements. To determine the carbon and nitrogen content of leaves, a small sample (10-20 mg) of homogenized leaf material was weighed and placed into a tin capsule. This capsule was then introduced into an oxidation column within a furnace set at 900°C, which contained catalysts to facilitate complete combustion in an oxygen-rich atmosphere. The resulting combustion gasses (including NO_x, CO_2, H_2O, N_2, etc.) were transported by a helium carrier gas through a series of purification steps. First, the gasses passed through a reduction tube at 680°C filled with copper, which converted NO_x to N_2 and removed excess O_2. Next, the gasses moved through a magnesium perchlorate tube to remove any H_2O. The purified gas stream was directed through a gas chromatography column where N_2 and CO_2 were separated. These gasses then reached a thermal conductivity detector, which generated signals corresponding to the presence of N_2 and CO_2. Carbon and nitrogen content of the leaf samples were quantified based on these signals. Calibration of the instrument prior to these measurements was achieved by analyzing a pure compound with known carbon and nitrogen content, typically aspartic acid (personal communication with Jamie Dearnley, 2024).

3.1.4 Statistical analyses

First, data were grouped by age to categorize changes over time and criteria. To group the sites by criteria and age, first references were grouped into one category (majority of references were mature and of a similar age). Next, wells were grouped into four age-criteria categories: Young (ages 7-13, 5 wells), Mid-young (ages 18-23, 9 wells), Mid-old (ages 24-27, 7 wells), and Old (ages 32-44, 4 wells). Young wells were reclaimed under the updated 2010 criteria, and all other categories were reclaimed using the criteria from before the 2010 criteria change (Table 3-2).

All statistical analysis was done using R version 4.2.2 (2022-10-31). Significance testing was always done using alpha=0.05.

Initial exploration of functional trait data involved quantifying mean and confidence interval values for each trait separated by age-criteria group (Appendix table B-1) as well as checking distributions for normality, outliers, etc. Any rare species (e.g., only present on one plot) were removed prior to analysis to avoid potential effects of uncommon species on the overall relationship between traits and environmental variables.

Community-weighted trait means (CWM) for each site were calculated by assigning weights to species attributes based on the relative occurrence of species at each site (Garnier et al. 2004). CWM therefore quantifies the dominant trait values in a plant community. A Principal Component Analysis (PCA) was conducted on the CWM trait matrix to investigate variations between site types (wells and benchmark reference) in multiple trait space and illustrate the overall association among different trait types. To assess variations in the general composition of the trait matrix among different age-criteria groups, I conducted a permutational multivariate analysis of variance (perMANOVA; Anderson 2005) using the *adonis2* function from the *vegan* R package. Post-hoc pairwise comparisons were run with Bonferroni correction using the *pairwise.perm.manova* function from the *RVAideMemoire* package (number of permutations=9999). The multi-trait functional dissimilarity was computed for each pair of communities (sites) using the Gower dissimilarity index (Gower 1971). This index is suitable for handling variables (e.g., the community-weighted mean of each trait) that are measured in different units (Pavoine et al. 2009).

Because Gower dissimilarity index is ideal for trait data (especially when they are of different types, e.g., a mix of continuous, ordinal, and categorical variables; Pavoine et al. 2009), a distance-based redundancy analysis (dbRDA) was used for carrying out constrained ordination as standard RDA uses Euclidean distance. I used the function “capscale” in R to carry out a dbRDA constraining functional trait data by soil, canopy, and diversity variables, which allowed me to examine the relationship between environmental variables (e.g., soil properties, diversity measures, cover data- calculated using 0-50 cm strata occurrence data from contact-point method) and CWM trait values of communities. Redundancy analysis requires explanatory variables not to be highly correlated with one another. Therefore, multicollinearity was tested, and the following variables were retained as explanatory variables (all others were omitted): Shannon diversity, basal area, canopy cover, tree cover, moss cover, graminoid cover, soil pH, and soil bulk density. I tested the statistical significance of the environmental constraints using permutation tests. These tests evaluated whether the observed relationships between soil (e.g., bulk density), understory (e.g., life form cover), and overstory (e.g., canopy cover) variables and trait composition were stronger than those expected by chance.

I used Nonmetric Multidimensional Scaling (NMDS) to detect functional traits associated with age-criteria groups and to investigate the relationships between functional trait community and soil characteristics, diversity variables, and other ecological variables (Thessler et al. 2005; Austin 2013). NMDS is a statistical method that does not rely on assumptions of normality, dimensionality, linearity, or the shape of species-response curves to gradients. NMDS is used for analyzing biological communities and is a commonly used ordination method (McCune et al.

2002; Oksanen 2011; Legendre and Legendre 2012; Borcard et al. 2018). NMDS space-sample relationships are determined by the ranking of dissimilarity in compositional space, as described by Legendre and Legendre (2012). Final scores are measurements that indicate the level of dissimilarity in the composition of *n*-dimensional data. I employed the *metaMDS* function from the *vegan* package in R to do numerous runs and identify stable configurations (Oksanen et al., 2018). MetaMDS utilizes square root transformation and scaling techniques such as centering, PC rotation, and half change scaling. It also employs extended scores that are derived from Wisconsin double standardization. The final model was chosen based on the solution that had the least dissimilarity between ordination and Gower distances. My ordination included convex hulls (polygons enclosing all sample points in a group) and 95% confidence intervals to place all sites into age-criteria categories. A high degree of overlap indicated similarity in functional structure between overlapping groups, whereas a low degree of overlap indicated dissimilarity.

In addition to assigning wells to the age-criteria groups for analysis, I also investigated the recovery of functional traits over time using Generalized Additive Mixed Models (GAMMs). GAMMs, rather than regression, are ideal for this study because they can represent non-linear responses over time. Because there are confounding influences of time and the associated change in reclamation criteria, I didn’t expect linear relationships in the data.

GAMMs were performed using the gamm function from the *gamm4* package (Wood et al. 2017) version 0.2-6. I used GAMMs to analyze the relationship between the age post-certification for each wellsite and the reference similarity compared to all other reference sites. The predictor variable was years-post-reclamation, and the random variable wellsite number was included to account for paired sampling. I selected the “*k*” value (either 3, 4, or 5) based on AIC outputs for models run under each scenario. The GAMM model output, including the 95% confidence interval around the smoother, was generated using the "predict" function.

I calculated similarity (1-Gower’s distance) between reclaimed wells and reference forests over years post-reclamation, which was visualized using the *ggplot2* package (Wickham et al. 2016). In these visualizations, each point on the GAMM plot represents the similarity of a well at a given age to each reference site. Additionally, the GAMM model was plotted with a smoother and gray shading showing 95% confidence interval of the correlation. Mean similarity among benchmark reference forests was computed by comparing each benchmark reference to all other references using a custom function in R. On the right side of the plot, the mean similarity of the reference sites is displayed with its 95% confidence interval for comparison. Graphs were analyzed to determine the "full recovery" point, defined as when the GAMM curve crossed the lower confidence interval of the reference vs. reference similarity. To help the reader see the mean values associated when the data were grouped into age-criteria classes, I also plotted interval plots with the mean trait values and included those next to the GAMM plot. A mixed effect model was run for each functional trait grouped by age-criteria group with site ID (well and reference shared location) as a random variable. The model identified significant differences among age-criteria groups (which includes the benchmark forests) and accounted for spatial pairing to get a more reliable signal. Each model was then tested for normal distribution using the Shapiro-Wilk test. Post-hoc tests including either Tukey test (if data were normal) or pairwise comparisons with Bonferroni adjustment (if data didn't meet normality assumption) were used to identify which groups were different, with differences shown using significance letters applied to interval plots.

3.2 Results

3.2.1 Differences in plant trait composition

Principal component analysis of the community weighted mean trait matrix explained 49.2% and 10.3% of trait co-variation in the first two axes (Fig. 3-5). Dimension one had high positive loading for numeric/ordinal traits: relative growth rate, light requirement, non-native status, rooting depth, and leaf nitrogen. Categorical traits with positive loading in the first dimension included: taproot, corm, bulb, and absent storage organs; geophyte, hemicryptophyte, and therophyte Raunkiaer life forms; and unassisted, wind, water, human, and animal (exo-zoochorous) seed dispersal vectors. Dimension one had negative loading for numeric traits: height, seed weight, water preference, lateral spread, specific leaf area, and leaf carbon. Categorical traits with negative first dimension loading included: rhizome and caudex storage organs; chamaephyte, mega and meso phanerophyte, and micro and nano phanerophyte Raunkiaer life forms; and insect, bird, explosive, and animal (endo-zoochorous) seed dispersal vectors. Most of the functional trait locations had high alignment with the first dimension of the PCA.

Along the first axis, there was a clear trend from negative to positive of reference forest plots (left) to reclaimed well plots (right). However, lack of separation of well plots of different age-criteria groups matched perMANOVA ($F_{4, 48}$=4.58, P=0.003) and post-hoc pairwise comparison results (Ref vs Mid-old, P=0.018 and Ref vs Mid-young P=0.019, all age groups vs other age groups p $\geq$ 0.10) that don't provide strong evidence for significant differences between wells based on Age-criteria grouping. P-values for the pairwise perMANOVAs between reference and Young and Old wells were greater than our alpha value, meaning we could not conclude that they were significantly different in their trait properties.

Dimension one differentiated plant communities with traits often associated with fast-resource acquisition (high relative growth rate), lack of canopy (high light requirement), and competitive colonization (low seed weight, far seed dispersal) from those with conservative resource acquisition (high leaf carbon, low growth rate), greater vertical structure (high height, phanerophyte Raunkiaer life forms), and lower competitive colonization capacity (high seed weight, short seed dispersal). There was a significant difference in dispersion among site types (PERMDISP, $F_{4,44}$ =3.75, P=0.01) with the highest dispersion (0.172 distance from median) for young wells indicating that the samples within this group are more dispersed. Old wells had the lowest variability (0.095 distance from median) suggesting that the samples are more clustered around the median.

3.2.2 Functional trait recovery over time

Out of 12 recovery curves, 83.3% showed no resilience within the maximum years post certification of this study (44 years) with variation in decline in similarity between references and wells (Table 3-3). Criteria did not appear to have a strong effect on any of the recovery curves. Resistance and resilience were equally common (8.3%; Table 3-3). Height was resistant to disturbance (immediate recovery), while leaf carbon recovered ~26 years after reclamation. Water preference was nearly resilient but fell just short of reference values. For traits without resistance or resilience, the difference in similarity between wells and reference forest (as a percentage of mean similarity among references) ranged from 12.0% to 53.7% (Table 3-3), with a mean of 28.4% and median of 30.1%. While many curves showed no resilience, some recovery curves showed trends towards recovery on a longer timescale (e.g., SLA).

3.2.2.1 Reproduction traits

Seed weight and seed dispersal vector values did not show signs of recovery, exhibiting a no resilience recovery pattern (Table 3-3). Seed dispersal vector (translated to relative distance, see Table 3-1) values in wells showed no similarity to the reference benchmark forests in the GAMM and interval plot (Fig. 3-6) with the smallest difference between GAMM smoother and reference similarity mean at 30.08% (Table 3-3). Seed dispersal distance was higher in reference benchmark plots than in well plots of all age-criteria groups. Similarly, seed weight did not show signs of recovery (Table 3-3; Fig. 3-7), with all well age-criteria groups determined as having significantly lower seed weight compared to the reference benchmark plots (Fig. 3-7b). Seed weight was highest in reference forest plots, with mid-old wells being closest to reaching the benchmark range (Fig. 3-7b).

3.2.2.2 Above-ground growth and survival traits

Values of five of the seven functional traits in the above-ground growth and survival traits category did not show signs of recovery (Table 3-3, Figs. 3-8 through 3-14). Height was the only trait with a resistant recovery pattern (Table 3-3, Fig. 3-8a). Height was variable for wells; young and old age-criteria groups were not significantly different from reference range, while mid-young and mid-old were significantly shorter than reference benchmark forests based on post-hoc testing (Fig. 3-8b). Both the interval plot and recovery curve for leaf carbon show a pattern of increasing leaf carbon with well age post-certification (Fig. 3-9). There was a resilient recovery pattern for this trait, with recovery after ~26 years post-reclamation (Table 3-3, Fig 3-9a). Leaf carbon content in reference benchmark forests was generally higher than well leaf carbon content. Wells exhibited increasing leaf nitrogen content with age which created decreasing similarity between older wells and the reference forest range (Fig. 3-10a). Young and mid-young age-criteria groups were not statistically different from the reference condition, and the mid-old and old leaf nitrogen content extended above the benchmark leaf nitrogen content (Fig. 3-10b). The light requirement recovery curve had the greatest percent difference between smoother and reference range at 53.68% (Table 3-3, Fig. 3-11a). Light requirement (relative shade intolerance) was significantly greater on wells than in reference forest (Fig. 3-11b). Non-native status was lowest for reference forests and variable among wells (Fig. 3-12b). Young and mid-young wells had more similar non-native status to the reference benchmark on the recovery curve but were still well below (43.76% difference) the lower reference range confidence interval (Table 3-3, Fig. 3-12a). Relative growth rate was greater on wells than in reference forest (Fig. 3-13b). The recovery curve for relative growth rate had a mild lagged or temporary recovery pattern, though it never reached the reference benchmark (20.03% difference; Table 3-3, Fig. 3-13a). Reference forest plant communities had significantly higher SLA than all age-criteria well plant communities (Fig. 3-14b); however, the recovery curve showed a trajectory towards resilience with age, so SLA might recover on a longer time scale (Fig. 3-14a).

3.2.2.3 Below-ground growth and survival traits

All three below-ground growth and survival traits showed no resilience recovery curves; however, water preference only had 6.5% difference between the smoother and reference benchmark mean (Table 3-3). The lateral extension recovery curve showed a mild upward trend (Fig. 3-15a), but the interval plot revealed significantly lower lateral spread in wells of all age-criteria groups compared to reference benchmark plots (Fig. 3-15b). While reference forest species had high relative lateral extension, they had low relative rooting depth (Fig. 3-16b). Root depth was lowest in reference forests and not statistically different among wells of different age-criteria groups. There was no evidence of recovery over time for this trait (Fig. 3-16a). Water preference did not show resilience but appeared to be on a trajectory towards recovery (Fig. 3-17a), with only the mid-old age-criteria group as significantly lower than the reference (Fig. 3-17b).

3.2.3 Community-level functional traits and environmental variables

Distance-based redundancy analysis (dbRDA) of the traits matrix constrained by environmental variables demonstrated that 69.62% (63.54% adjusted R^2, corrected for the number of explanatory variables) of the variance in community traits across sites could be explained by the included environmental factors, of which 54.74% was explained in the first axis, and 5.59% explained in the second axis. As with the PCA, the dbDRA had clear groupings for reference plots and well plots, with wells of different criteria/age classes overlapping (Fig. 3-18). Soil bulk density, soil pH, graminoid cover, and moss cover had positive loadings on Axis 1, as did wells. Other environmental variables such as richness, Shannon diversity, tree cover, canopy cover, and basal area had negative loadings on Axis 1, as did reference sites.

A permutation test of the dbRDA model showed strong evidence that the following environmental variables are associated with trait variability: canopy cover ($F_{1,40}$ =62.9211, P=0.001), graminoid cover ($F_{1,40}$ =7.0463, P=0.002) , soil pH ($F_{1,40}$ =5.9117, P=0.003), moss cover ($F_{1,40}$ =4.7527, P=0.008), soil bulk density ($F_{1,40}$ =4.0548, P=0.019). There was moderate evidence that tree cover and basal area significantly explained the variance in functional traits ($F_{1,40}$ =2.3069, P=0.066 and $F_{1,40}$ =2.5945, P=0.061 respectively), and there was weak evidence ($F_{1,40}$ =2.0780, P=0.112) that Shannon diversity had influenced trait variation.

While approximately two thirds of the variation in the traits matrix could be explained by constrained analysis (dbRDA), I also used NMDS to assess unconstrained patterns within the data. The best solution was repeated once in 63 tries (max 100) from try 10 (random start). The final NMDS 2-dimensional solution converged on the 9th attempt and had a final stress of 0.097. I used Gower dissimilarity, which is ideal for variables measured in different units, and because it had the highest value (0.772) when testing rank correlations between dissimilarity indices (Euclidean, Manhattan, Bray-Curtis, Kulczynski) and gradient separation. As with the PCA and dbRDA, NMDS showed clear separation between wells and reference plots, with overlap among well age-criteria groups (Fig. 3-19). Vector analysis showed that the following ecological (e.g., soil, diversity, and cover) and functional variables showed weak evidence of affecting the variation seen in the unconstrained analysis: storage organ-annuals, storage organ-caudex, leaf nitrogen, Raunkiaer life form-Therophyte, dispersal vector–animal other than bird (ingestion, endo-zoochorous), dispersal vector-insect (mostly ants), dispersal vector-explosive discharge; forb cover, and moss cover.

Ecological vectors also aligned with previous PCA and dbRDA ordinations (same positive loading for soil bulk density, graminoid cover, and soil pH, with negative loadings for diversity measures, soil FH depth, and indicators of woody species occurrence; Fig. 3-21, Appendix figure B-1 for all vectors). Similarly, functional trait vectors in the NMDS overall aligned with PCA and dbRDA results. Vectors associated with wells (higher loading on first axis) included light requirement, non-native status, relative growth rate, relative rooting depth, dispersal types: ground, wind, water, human, and exo-zoochorous, storage organs: annuals, bulb, corm, and tuber, and Raunkiear life form: hemicryptophyte (Fig. 3-20, Appendix figure B-2 for all vectors). However, some trait vectors in the unconstrained analysis had different associations than previously represented. For example, leaf nitrogen had more horizontal directionality compared with its position in the PCA where it had higher negative loading in the second dimension. NMDS ordination also had some variables that were previously associated with dimension 1 shift to stronger associations with dimension 2 (negative loading), such as water preference, geophyte Raunkiaer life form, and ground seed dispersal vector (Appendix figure B-2).

3.3 Discussion

This study evaluated the long-term patterns of recovery of functional composition on reclaimed wells. My findings showed that many functional trait values were not trending towards the reference range, although there were exceptions in some trajectories. My observations indicate that oil and gas disturbance and reclamation practices have long-term impacts on the functional composition of plant communities. All analyses showed differences between the reference forest and reclaimed wells of one or more age-criteria groups. These differences indicate that there are significant functional differences between the two groups, even though some functional traits (e.g., leaf Carbon) have shown trait values closer to reference sites. It also reflects patterns observed in Chapter 2 (taxonomic and structural data) that many reclaimed wells may be in slowed or arrested succession. Some functional trait values may be similar to the adjacent reference (e.g., height, leaf carbon, and possibly water preference). However, most (ten out of 12 recovery curves) did not successfully recover at any age post-certification. Multiple analyses showed reference forests were characterized by conservative resource acquisition trait values associated with late succession (higher seed weight, water preference, lateral spread, specific leaf area, and leaf carbon), while reclaimed wells were characterized by trait values associated with rapid colonization associated with early succession (faster relative growth rate and higher light requirement, non-native status, rooting depth, and leaf nitrogen).

My findings suggest that well plant communities have trait values associated with early succession such as wind seed dispersal, higher light preference, and faster relative growth rate. Wells can have unique soil properties (e.g., anthropogenic compaction, lack of organic layer) compared with naturally disturbed areas, which could prevent late-successional species from colonizing and lead to the persistence of early-successional and invasive species. High light requirement (shade intolerance) in wells matches low observed canopy cover and basal area, so species in wells are better adapted to absent canopy conditions. Low canopy cover, basal area, and phanerophyte presence on wells suggests that trees are not easily growing on wells and may not be present there unless planted. Complementary to findings in Chapter 2, high light requirement, non-native status, and fast relative growth rate associated with wells suggest previous use of agronomic seed mixes have had long lasting effects. Many of the old class reclaimed wells still exhibit these functional traits, which creates uncertainty concerning if or how long it will take for trees to grow on them. Seed dispersal vectors on wells were typically long-distance strategies ideal for colonization such as wind, water, and human dispersal. In contrast to Azeria et al., (2020), the sampled wells in this study had less seed dispersal by animals than reference plots, suggesting that animals may not be frequenting these disturbed sites. When employed in reclamation, herbaceous annuals (or biennials like clover) can either directly compete with other plants (Zipper et al. 2011), have priority effects (Fukami et al. 2005; Kardol et al. 2013), or compete with other plants by drawing in browsers (Holt and Lawton 1994). Evidence suggesting minimal animal presence on wells is likely due to limited cover and protection from predators or plant communities unsuitable for browsing compared with the adjacent forests.

3.3.1 Traits exhibiting recovery or marginal recovery

Higher height, canopy cover, basal area, and phanerophyte Raunkiaer life form occurrence vectors were all associated with reference plots in ordinations, but the height recovery curve (GAMM) showed resistance to disturbance (overlap between reference and well plant heights). Typically, tall plants are associated with late-successional species that grow taller to compete for light in a crowded canopy. I suspect that the GAMM for the height functional trait is not reflecting the current state of the wells. Looking at the interval plot for each age-criteria group, species with higher height potential in Old and Young age-criteria groups could be causing this pattern. Height functional trait data are derived from the average or upper average height of a species. It does not represent the current height of that species in this study or take into consideration tree diameter. Alternatively, this finding may be due to community weighted mean values coming from understory sampling, and therefore not capturing the abundance of species with greater heights present in the reference forest overstory.

Recovery of leaf carbon over time observed in this study is a promising indicator of the gradual re-establishment of functional ecosystem processes in reclaimed wells. Leaf carbon content is linked to plant strategies for resource acquisition and allocation (Zhao et al. 2018). Woody species have higher carbon content than herbaceous plants, and conifers have higher carbon content than broad-leaved woody species (Ma et al. 2018). High leaf carbon concentration indicates a shift towards more conservative resource use (i.e., prioritizing long-term survival and efficiency over rapid growth), which is typical of late-successional species that invest in long-term tissue durability and stress resistance (Navas et al. 2010; Chai et al. 2015). On younger wells, lower leaf carbon may indicate that plants are investing in below-ground biomass, especially early on. This allocation to root systems helps the plant obtain water and nutrients, stabilize soil, and compete in a recovering ecosystem during establishment. Early-successional species often have higher growth rates and rapid nutrient intake, which may be linked to decreased leaf carbon. The gradual rise in leaf carbon content indicates the maturation of above-ground biomass and a probable shift in vegetation community strategy as it proceeds from early to late successional phases.

Water preference reflects a plant's soil moisture preference, and shows how the plant adapts to dry (xeric), moist (mesic), and wet (hydric) settings. In early successional stages after disturbances like fire or harvest, species with low water preference may dominate because they may better colonize and establish in dry, open conditions. Lower water preference (higher water use efficiency) is usually linked with better competitive ability. Higher water preference may reflect the shade conditions of reference forests which reduces evapotranspiration compared with low shade wells. It may also reflect higher FH layer depth (organic matter) which retains soil moisture. Water preference of plants on wells of three age-criteria groups was not significantly different from the reference benchmark range (Fig. 3-17b). This overlap may be due to many species having mesic or mesic adjacent (mesic-xeric, mesic-hydric) water preferences. This recovery curve did not show resilience; however, a trend towards resilience is evident in the GAMM, so older wells might show recovery after more time.

3.3.2 Traits exhibiting slowed/arrested recovery

The following numeric functional trait values - seed dispersal vector, seed weight, light requirement, relative growth rate, leaf nitrogen, SLA, lateral spread, rooting depth, and non-native status - did not show evidence of recovery. However, interval plots occasionally showed overlap between young age-criteria wells and reference (i.e., leaf nitrogen, rooting depth, non-native status). Confidence intervals of young plots in these interval plots often had a large range compared with other groups (e.g., rooting depth mean (95% CI): Young 0.306 (0.226-0.39) vs Mid-old 0.32 (0.304-0.337)). This large variability is supported by dispersion analysis results showing high variability within the young age-criteria category, which may reflect the initial colonization phase of early stage succession, which is characterized, even in natural succession, by rapid species turnover and heterogeneity as colonizing plants compete to establish.

Neither reproduction trait recovery curve showed recovery. Reference forest plots had further relative seed dispersal distance (converted from categorical as indicated in Table 3-1) compared with all well age-criteria groups. Even though human dispersal had the highest assigned value (1.0) and had positive loadings in first axes of ordinations (associating human seed dispersal with wells), ground (unassisted), wind, and water dispersal vectors were associated with wells in ordinations and all had low assigned values for dispersal distance. References had more bird and animal dispersal (assigned value 0.75) which could account for higher seed dispersal values for benchmark forests. References had high seed weight compared with all age-criteria well groups, which is associated with late successional species, as heavier seeds use more resources for seedling establishment and are better adapted for seed survival and establishment in deep shade (Pérez-Harguindeguy et al. 2013).

The above-ground growth and survival traits seen in wells plant communities reflect the early successional dynamics and challenges of ecological recovery on reclaimed well pads. The observed significant increase in leaf nitrogen in mid-old and old age-criteria wells compared to reference benchmark forests suggests the dominance of early successional and acquisitive species, which thrive in disturbed environments due to their high resource acquisition strategies and rapid growth rates (Rawat et al. 2021). Similarly, higher light requirements and relative growth rates were seen on wells, both of which are characteristic of early successional species that inhabit open, disturbed regions. There was low non-native status seen in reference forests compared to wells, with old wells having the highest mean value for non-native status. This pattern mirrors changing reclamation practices which discouraged use of non-native plants. Young and mid-young wells showing grater similarity to the reference benchmark in terms of non-native status could also reflect these changes in practice. Higher SLA is typically linked with higher natural resource efficiency under stressful conditions (Yu et al. 2022), which contrasts with the slower recovery trajectory of SLA on wells; however, species may also increase their SLA in shade conditions (Liu et al. 2016). Wells may still be enduring elevated levels of stress or resource scarcity, which may be impeding the full recovery of these functional traits.

Root trait (lateral extension and relative rooting depth) values showed a no resilience recovery pattern. Reference benchmark species had greater lateral spread and shallower relative rooting depths. Greater lateral spread is associated with late succession when species need to outcompete neighbors or increase reproductive sites. Boreal forest species like *Populus tremuloides* have shallow and extensive lateral roots (Howard 1996). Deep rooting depth in wells could reflect ruderal species which have taproots (e.g., *Cirsium arvense* and *Taraxacum officinale*). While well soil compaction may be higher than reference benchmark soil compaction, bulk density values are still below 1.4 g cm^{-3} where bulk density can adversely affect plant growth in boreal forests (Sutton 1991; Binkley and Fisher 2019). Higher moisture and organic matter in reference forests might also explain the shallower rooting depth compared to wells (as plants would not need to root as deeply to access resources).

3.3.3 Age group recovery and environmental variables

Similarity among reference and young and/or old wells for height (both), leaf carbon (old), leaf nitrogen (young), water preference (both) suggest that these two age-criteria groups might be more like reference condition and therefore functionally more like the benchmark reference than the mid-age groups. These similarities may be explained by older wells having sufficient time to begin to functionally recover and young wells under new criteria experiencing more efficient recovery than older wells due to ecological reclamation practices.

In the dbRDA, environmental variables explained 63.5% of the variation in functional composition. Ecologically, environmental variables may constrain which traits can persist in a plant community. For example, soil compaction and canopy cover likely have a large influence on certain functional traits (e.g., root traits and shade tolerance). I did not often see a significant effect of age or criteria for the functional composition of the wells. Large variation in reclamation practices employed by individual companies and in response to site-specific conditions could explain the lack of influence from age and criteria. However, some evidence, such as the recovery of leaf carbon and other trends in recovery curves, suggests a minor influence of age/criteria on certain traits (e.g., non-native status, seed weight). Interestingly, species richness and diversity measures (Shannon, inverse Simpson) were not observed as important environmental factors in the dbRDA. The data showed higher diversity on reference plots compared to well plots; however, within this study we did not detect a relationship between diversity and functional composition. Abiotic factors (e.g., soil type, moisture, pH) may strongly determine which species can survive, causing the functional traits of surviving species to be similar. Certain habitats may favor species with specific traits, leading to high functional similarity despite species diversity. For example, (Akram et al. 2020) found converging leaf traits in an arid environment (not due to phylogeny), suggesting that environmental factors filtered species with similar traits (e.g., thick leaves in desert plants).

3.3.4 Limitations

I acknowledge that there are limitations to my conclusions based on the effect of time due to the unrealistic assumption of identical biotic and abiotic starting conditions on reclaimed sites. I do not know what specific reclamation actions were taken on each site (outside of what criteria required at the time), making it difficult to draw conclusions about the drivers of the patterns I observed. Even within the criteria, there could be variation in execution (e.g., the way Alberta Energy and Natural Resources recommended seed mixes be applied). We might assume the presence of introduced species is due to revegetation planting using non-native seed mixes, but without documentation that cannot be confirmed. Other characteristics are more difficult to estimate, such as if woody species were planted, or what (if anything) was done for soil compaction. Notably, the current certification process does not consider potential passive regeneration that may occur before reclamation, nor does it account for time that a wellsite may have been reclaimed before applying for certification. My study observed that certain reclaimed sites exhibited variations in the expected successional trend over time. These deviations could be attributed to factors unique to each site, such as local conditions or landscape characteristics, as well as historical factors related to reclamation/restoration practices (Young et al. 2005). Finally, comparing recently disturbed wells to mature forests limits the conclusions of this study due to the mismatch in time scales (e.g., up to 44 years given to recover to average 74.5-year-old forests). Nevertheless, despite these limitations, our findings offer evidence of the directional temporal change in the functional composition of highly disturbed forests after reclamation.

3.3.5 Management implications and future directions

This study highlights the importance of utilizing a trait-based approach to uncover overarching patterns that go beyond variations in taxonomy and structure (e.g., Diaz et al. 2004; McGill et al. 2006). Our research supports previous studies indicating that reclamation practices have the potential to impact the recovery of plant communities, but frequently do not fully restore the original conditions (. Our findings contrast with those in some previous studies which have found some lagged recovery of reclaimed wells on slightly longer timescales (Lupardus et al. 2019; Azeria et al. 2020), which could be attributed to site heterogeneity or differences in chosen recovery indicators.

A key management recommendation emerging from our study is to adopt a trait-based approach for restoring disturbed boreal ecosystems. This approach is useful because it focuses on the functional characteristics of plant species, which directly influence ecosystem processes and resilience. Reclamation efforts can then be tailored to promote species that contribute to desired ecosystem functions, such as nutrient cycling, water retention, and resistance to invasive species. Trait-based approaches can identify and prioritize species that possess traits promoting ecosystem stability and resilience, thereby enhancing the long-term success of restoration projects. In addition, these approaches facilitate the comparison and transfer of knowledge across different ecosystems and scales, making them widely applicable and adaptable to various restoration contexts. Incorporating trait-based insights into reclamation practices ensures that the selection of species is based on their functional roles and contributions to ecosystem recovery, rather than solely on their presence in reference sites. The following framework can be useful to implement this strategy: selection of services to be restored, trait selection, data acquisition, analytical planning, and empirical testing and monitoring (Aubin et al. 2024). Other suggested actions involve integrating a variety of native species to encourage functional diversity, attending to soil properties (e.g., tilling or aerating soil to reduce bulk density, adding soil amendments like compost to improve organic layer depth) to improve site suitability, and observing long-term successional dynamics to evaluate the efficacy of reclamation endeavors. These strategies are in line with overarching principles of ecological restoration and can enhance the results of reclamation projects.

Future research should prioritize monitoring wells under new criteria, since our study could only assess a maximum of 16 years of recovery for new criteria sites. Over time, new patterns in the Young age-criteria group may arise in the absence of agronomic seeding. In addition, performing manipulative field experiments instead of relying solely on observational studies would yield more reliable insights into the precise mechanisms that influence successional trajectories and the recovery of functional composition. These studies could investigate the impacts of various reclamation practices, combinations of species, and environmental conditions on the dynamics of plant communities, providing valuable guidance for improving reclamation efforts.

3.4 Tables

Table 3-1. Plant functional traits compiled from field data and Traits of Plants in Canada (TOPIC) database.

Trait (abbreviation)	**Variable type**	**Trait classes / Units**	**Code (value)**	**Description**	**Species (n)**
			Above-Ground Traits		
Height (HT)	Continuous	cm	N/A	Competitive ability for light	132
Leaf Carbon/Nitrogen (Leaf_C, Leaf_N)	Continuous	% Nitrogen % Carbon	N/A	Amount of carbon and nitrogen present per unit of leaf tissue, Nutrient use efficiency in the plant, Indicator of soil nutrient content	n=20 (field collected species)
Light requirement (LightReq)	Ordinal	Shade tolerant Mid tolerant Shade intolerant	i (1) m (0.5) s (0)	Shade tolerance + succession (which species dominate early vs later stages of succession)	40 45 47
Non-native status (ST)	Binary	Native Exotic	AB_i (0) AB_t (1)	Presence of non-native species (may outcompete native plants)	116 16
Raunkiaer life form (RA)	Categorical	Chamaephyte Geophyte Hemicryptophyte Micro & nano phanerophyte Mega & meso phanerophyte Therophyte	ch g h mc mg t	Adaptation to adverse (e.g., cold/dry) environmental conditions	7 23 71 20 8 3

Relative growth rate (RelGrow)	Ordinal	Slow, Moderate, Rapid	Slow (0) Moderate (0.5) Rapid (1.0)	Rate at which a plant grows relative to its initial size Adaptation to resource availability, response to environmental stress, biomass production	26 58 48
Seed dispersal vector (DI)	Categorical and numeric	unassisted (autochorous, barochorous) insect (mostly ants, myrmecochorous) explosive discharge (ballistichorous) water (hydrochorous) wind (anemochorous) bird (ingestion, endo-zoochorous) animal other than bird (ingestion, endo-zoochorous) animal (carried externally, exo-zoochorous) human dispersal (anthropochorous)	g (0) an (0.25) ex (0.25) e (0.5) w (0.5) bi (0.75) ez (0.75) zz (0.75) hd (1.0)	Also called dispersal syndrome (e.g., wind, animal, water) Colonization of new habitats, reproductive success	13 6 8 17 57 20 54 14 11
Seed weight (SDWT)	Continuous	g/1,000 seeds	N/A	Reproductive success, seedling establishment, resources invested in seed production	132

Specific leaf area (SLA)	Continuous	m^2 / kg		Ratio of leaf area to leaf dry mass, How much carbon plant is investing in photosynthetic capacity, leaf longevity, competition for light, growth strategy	N=20 (field collected species)
Below-Ground Traits					
Lateral extension (LE)	Ordinal	a absent (does not reproduce clonally) cl Clonal compact (non phanerophyte) P_cl Clonal limited (phanerophyte) cm Clonal intermediate (non phanerophyte) P_cm Clonal moderate (phanerophyte) ce Clonal extensive (non phanerophyte) P_ce Clonal extensive (phanerophyte)	a (0) cl (0.2) P_cl (0.4) cm (0.6) P_cm (0.8) ce (1.0) P_ce (1.2)	Horizontal spread of vegetative structures (stems, branches, leaves) Competition for light, water, and nutrients especially in crowded environments	5 13 7 43 17 37 10
Rooting soil depth (RSD)	Ordinal	hs other superficial Raunkier life forms (includes shallow roots spreading through soil) as superficial phanerophyte (includes shallow	"hs" ~ 0, "as" ~0.2,	Depth at which a plant’s roots occur relative to the depth of the soil profile (belowground competition)	72 19

		roots spreading through soil) hi other intermediary Raunkier life forms ai intermediary phanerophyte hp other deep Raunkier life forms (includes tap roots) ap deep phanerophyte (includes tap roots)	"hi" ~ 0.4, "ai" ~ 0.6, "hp" ~0.8, "ap" ~ 1.0		5 8 23 5
Storage organ (SO)	Categorical	annuals bulb or pseudobulb corm caudex rhizome tuber/tuberous roots	a b c cd r t	Organs used for storing carbohydrates or water, also perenniating organs + vegetative propagation. competitive vigor and the ability to exploit patches rich in key resources	33 5 2 15 76 27
Water preference (WP)	Ordinal	Xeric Mesic - xeric Mesic Mesic - humid Humid	x (0) mx (0.25) m (0.5) mh (0.75) h (1)	Drought/moisture tolerance, indicate moisture conditions post-disturbance	3 26 45 43 15

Table 3-2. Description of the categories that were used in analysis. The 25 wells are divided into four age-criteria groups calculated as 2023 (the year sampled) - reclamation certification year and pre and post 2007 (when the updated 2010 well site criteria for forested lands applied). Reference age range comes from counting rings on tree cores.

Age-criteria Group	**Criteria**	**n**	**Age range - certified**
Young	New post-2010	5	7-16
Mid-Young	Old pre-2010	9	17-23
Mid-Old	Old pre-2010	7	24-31
Old	Old pre-2010	4	32-44[1]
Ref	n/a	24[2]	(28-141, from tree cores)

[1]44 year-old well was reclaimed in 1990 but abandoned 20 years prior (1970), so it was assigned to old since there was likely passive regeneration.
[2]One reference was adjacent to two wells and was therefore used as the reference benchmark forest for both.

Table 3-3. Results of Generalized Additive Mixed Models of similarity between wells and references as a function of time since reclamation for the various soil and vegetation attributes measured at 25 wells and adjacent reference sites (n=24) in Fox Creek. Gower similarity metric was used. Given is the *k* value, estimated degrees of freedom (*edf*), *F* value, and significance (*P*) for the smoother. Temporal pattern as determined based on if/when the smoother and reference confidence intervals overlapped (Fig. 3-1). The percent difference between the mean reference similarity and the highest point on the smoother curve is also given. See also Figs. 3-6 to 3-17.

Trait	***k***	***edf***	***F***	***P-value for smoother***	***Temporal pattern***	***Years to recovery***	***% difference smoother high point and reference mean***
Reproduction							

Seed dispersal vector (numeric)	3	1.8	1.749	0.242	No resilience	>44	30.08%
Seed weight	4	2.5	10.74	<0.0001	No resilience	>44	14.94%
Growth and Survival (Above ground)							
Height	4	1.0	2.398	0.122	Resistance	~0	1.57%
Leaf Carbon	4	2.5	47.05	<0.0001	Resilience	~ 26 years	2.11%
Leaf Nitrogen	4	2.4	11.12	<0.0001	No resilience	>44	12.04%
Light requirement	4	2.4	15.78	<0.0001	No resilience	>44	53.68%
Native status	4	2.9	18.02	<0.0001	No resilience	>44	43.76%
Relative growth rate	4	2.7	2.082	0.059	No resilience	>44	20.03%
Specific leaf area	3	1.9	45.32	<0.0001	No resilience	>44	20.65%
Growth and Survival (Below ground)							
Lateral spread	4	2.6	20.08	<0.0001	No resilience	>44	30.21%
Root depth	3	1.9	4.576	0.012	No resilience	>44	30.39%
Water preference	4	2.0	7.957	<0.0004	No resilience *(yet)	>44	6.5%

3.5 Figures

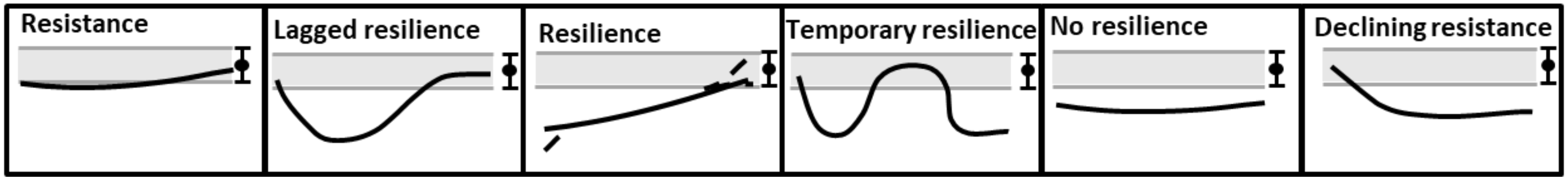

Figure 3-1. Temporal patterns of response curves for the wells (y axis: Similarity, x-axis: years post-certification): “Resistance” (well to reference similarity GAMM curve overlaps with the lower confidence interval of the mean reference to reference similarity), “Lagged resilience” (declining similarity between wells and references (i.e., loss of resistance), followed by increasing similarity (resilience) with recovery at the year post-reclamation when the curve intersects the lower confidence interval of the reference to reference similarity), “Resilience” (low well to reference similarity in the recent years post-reclamation indicating a lack of resistance, followed by an increase, indicating resilience and recovery), “Temporary resilience” (low well to reference similarity in recent years post-reclamation followed by the curve intersecting the lower confidence interval of the reference to reference similarity then declining back below the reference to reference similarity range), “No resilience” (well to reference similarity falls below reference to reference similarity), “Declining resistance” (decrease in well to reference similarity over time, indicating a loss of resistance, with no evidence of subsequent resilience).

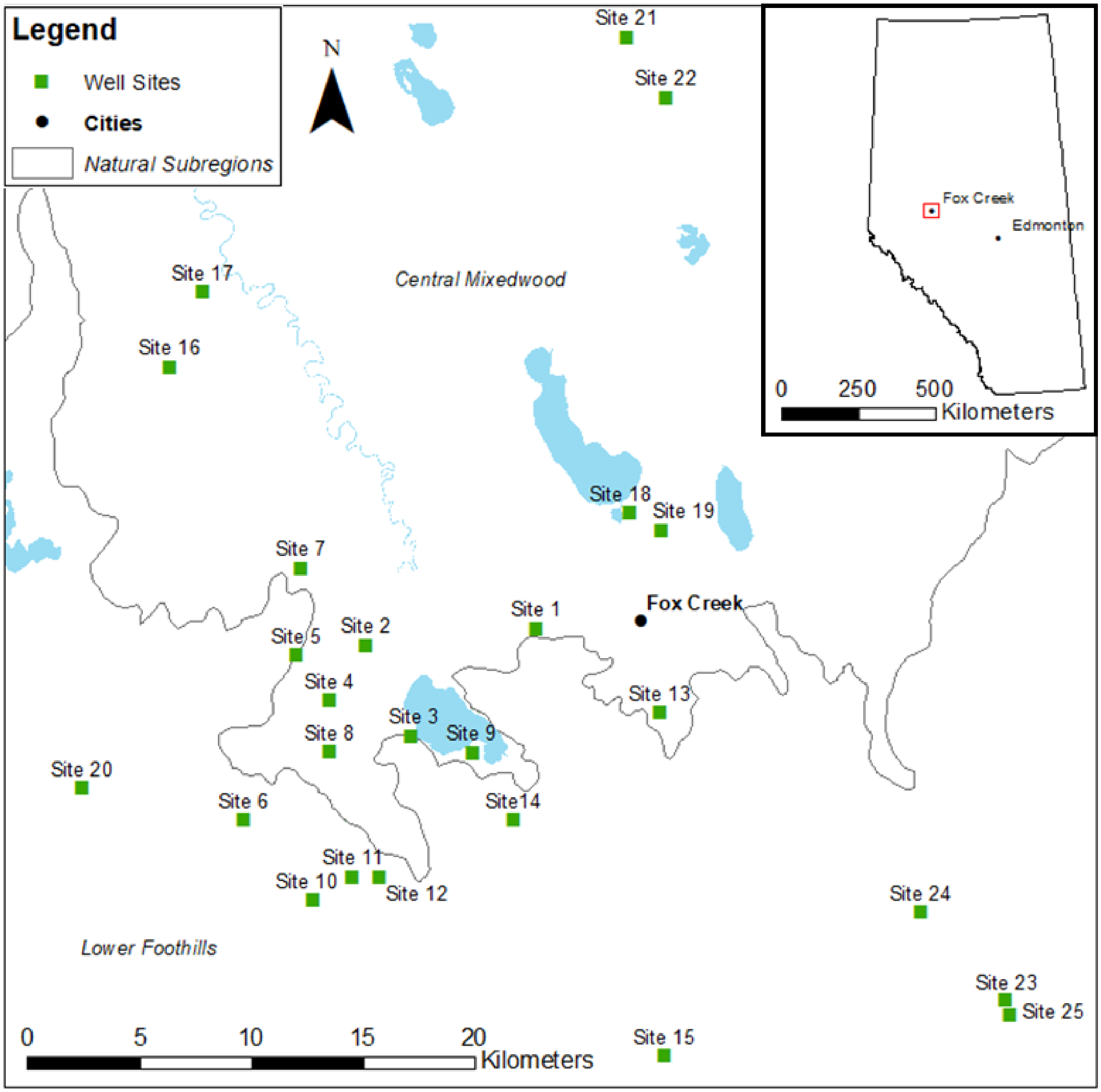

Figure 3-2. Map showing the locations of 25 certified reclaimed oil and natural gas production well pads and adjacent benchmark reference pairs in Alberta's upland forested lands near the city of Fox Creek, Alberta. The sites are distributed across the Central Mixedwood (n=15) and Lower Foothills (n=10) natural subregions. The inset map displays the broader context of Alberta, with Edmonton and Fox Creek highlighted for reference.

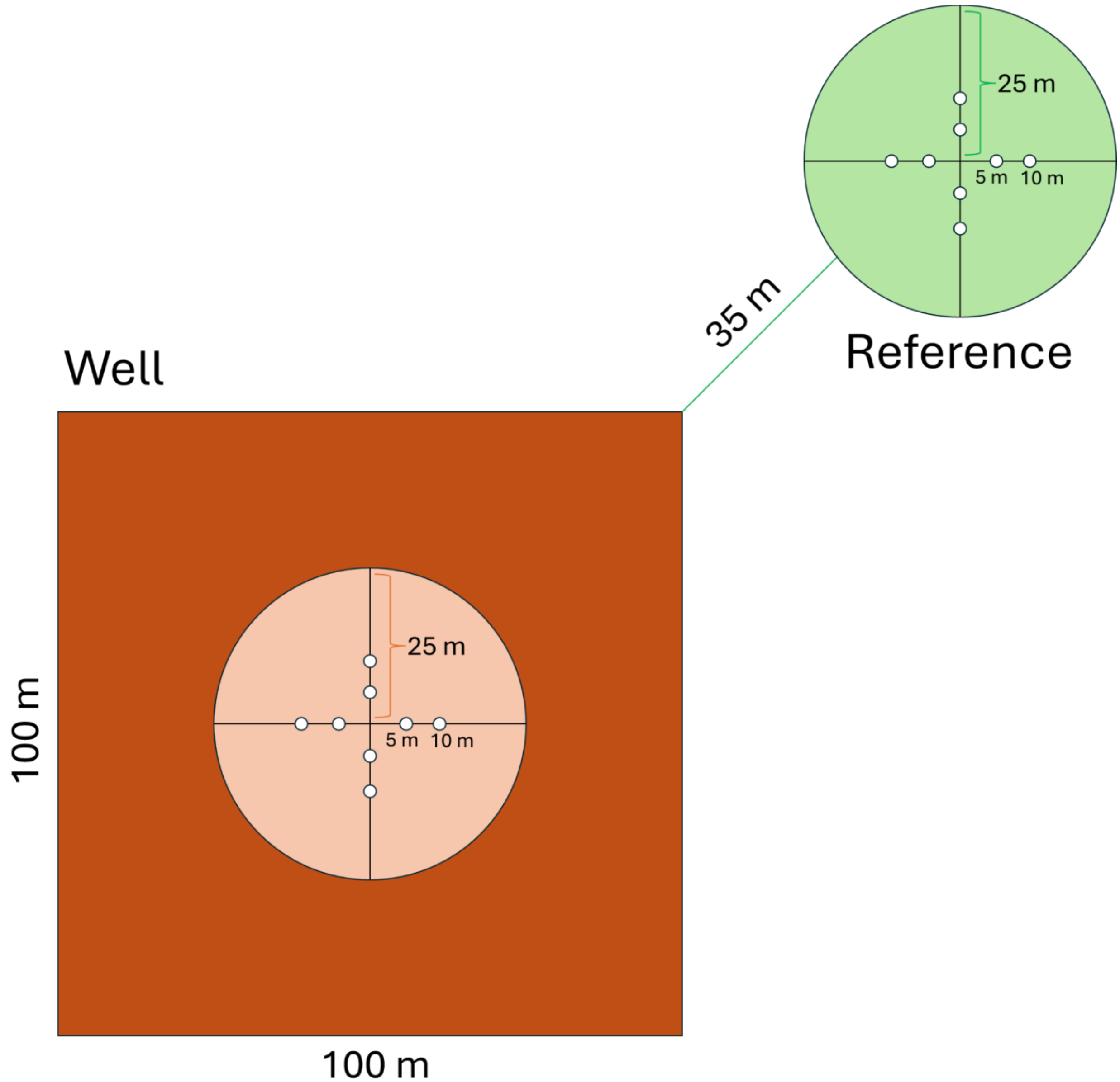

Figure 3-3. Schematic of the site setup for sampling. Wells were sampled within a 25 m radius circle centered on the well bore, with transects extending in each cardinal direction to form quadrants. Data collection points were established at 5 m and 10 m distances from the center in each direction, where canopy cover measures and soil samples were taken. Within the quadrants, basal area data, moss surveys, and tallest tree height measurements/species identification, along with tree cores (if present), were conducted. A reference plot located at least 35 meters into the adjacent forest followed the same 25 m radius circle and data collection setup.

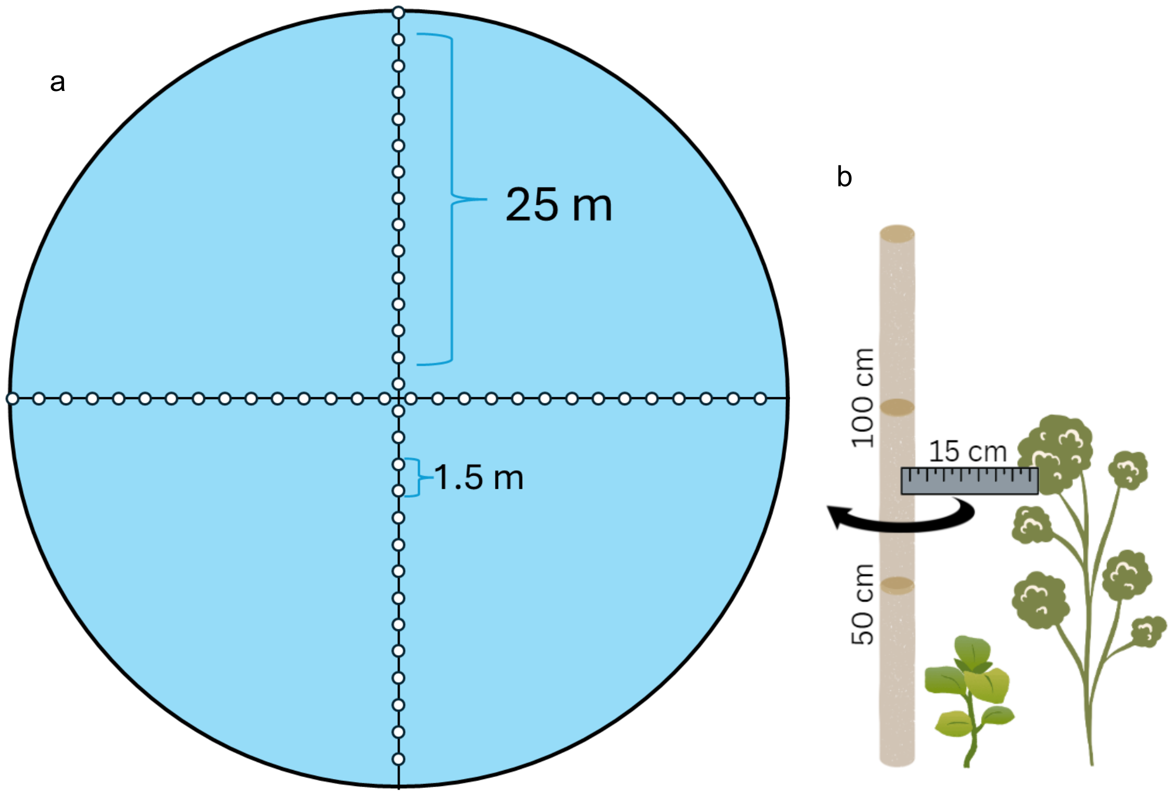

Figure 3-4. Illustration of the point contact method used for species data collection. **Fig. 3-4a:** Species presence was recorded every 1.5 meters along the transects, with 30 points per 50 m transect, resulting in a total of 60 points per plot. This method was employed to accurately capture species composition across the different height strata within the site. **Fig. 3-4b:** Species presence was defined as photosynthetic material within the 15 cm radius of the center wooden dowel for each 50 cm range height strata (included: 0-50 cm, 50-100 cm, 100-150 cm, 150-200 cm, 200-250 cm, and 250+ cm).

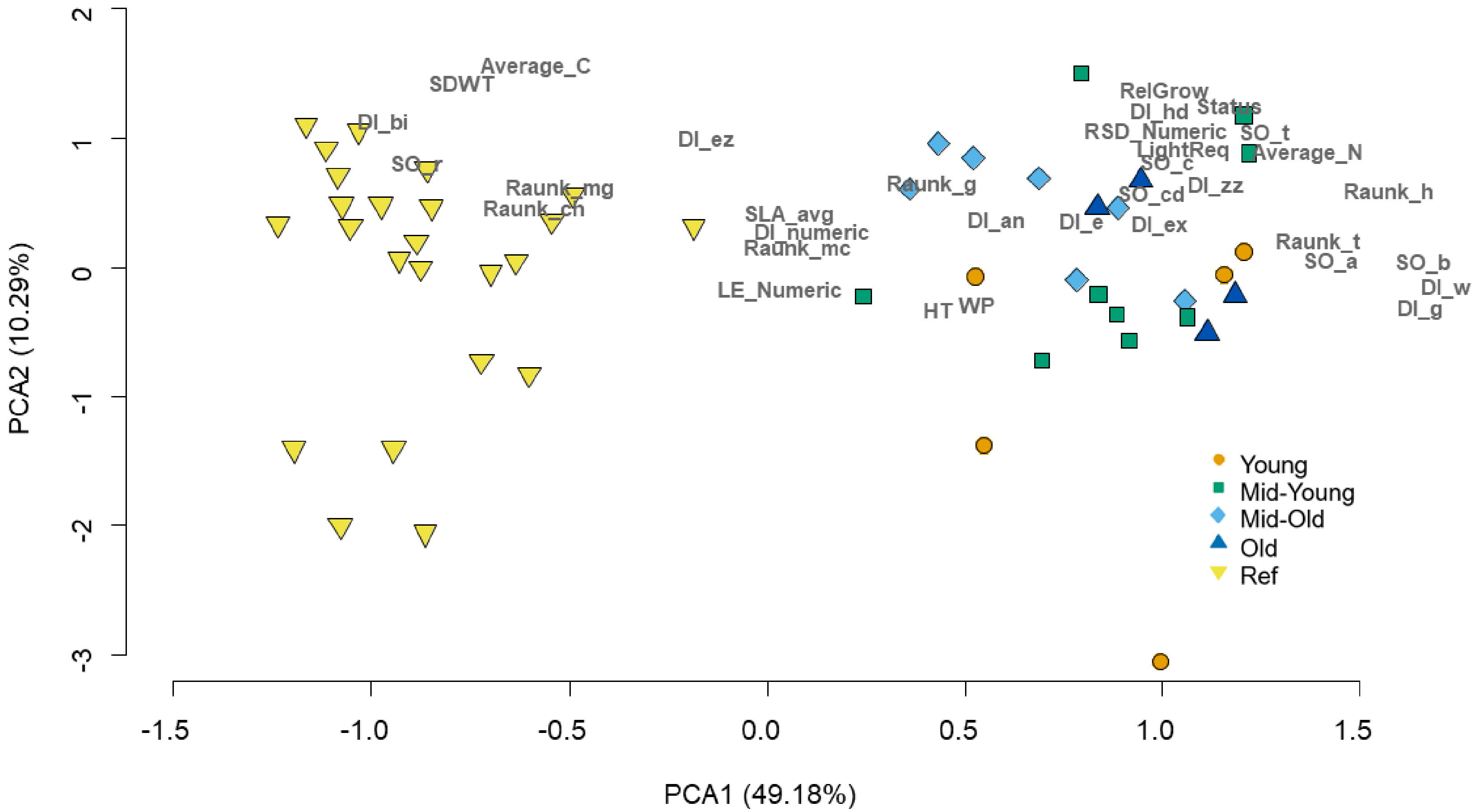

Figure 3-5. Principal component analysis (PCA – scaling 1) of functional trait composition assessed along a post-reclamation chronosequence and reference sites grouped by age-criteria group. Age- criteria groups: Young (new criteria), Mid-Young (old criteria), Mid-Old (old criteria), Old (old criteria), and Ref (mature benchmark reference forest). Gray text represent the functional characteristics associated to the sites which they are close to. See Table 3-1 for trait explanations. See Table 3-2 for age-criteria group descriptions.

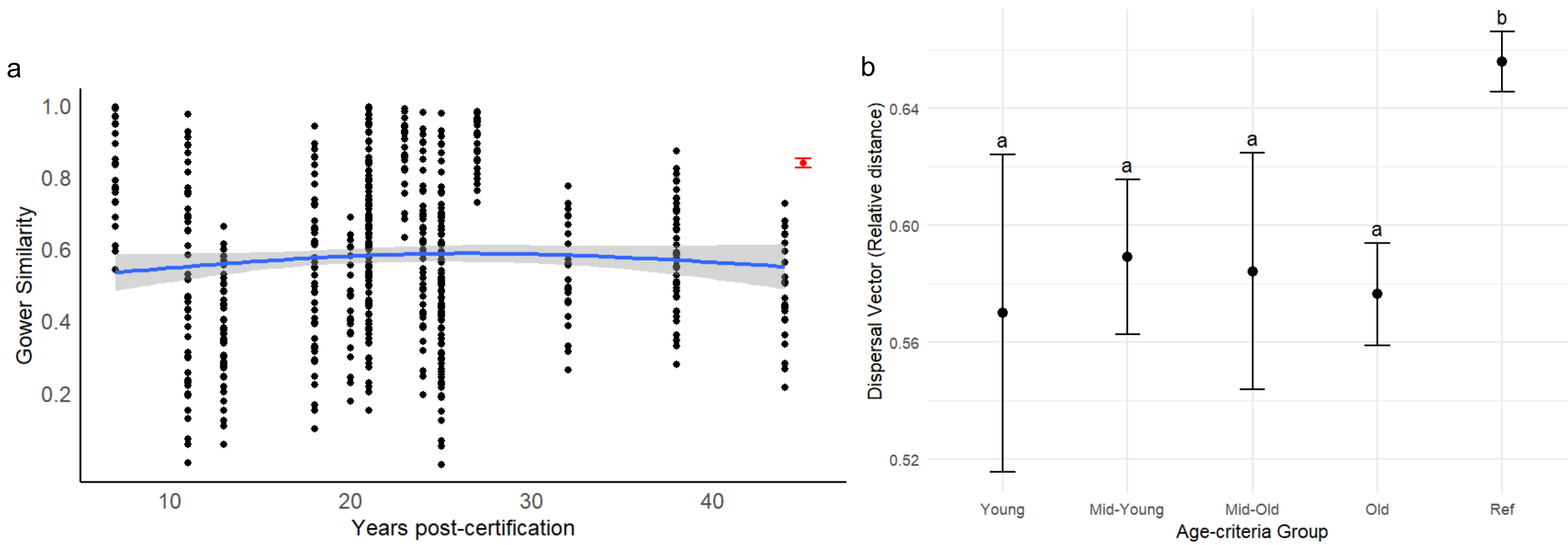

Figure 3-6. Seed dispersal vector (converted to relative distance, see Table 3-1) on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for seed dispersal vector grouped by age criteria group. **Fig. 3-6a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-6b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

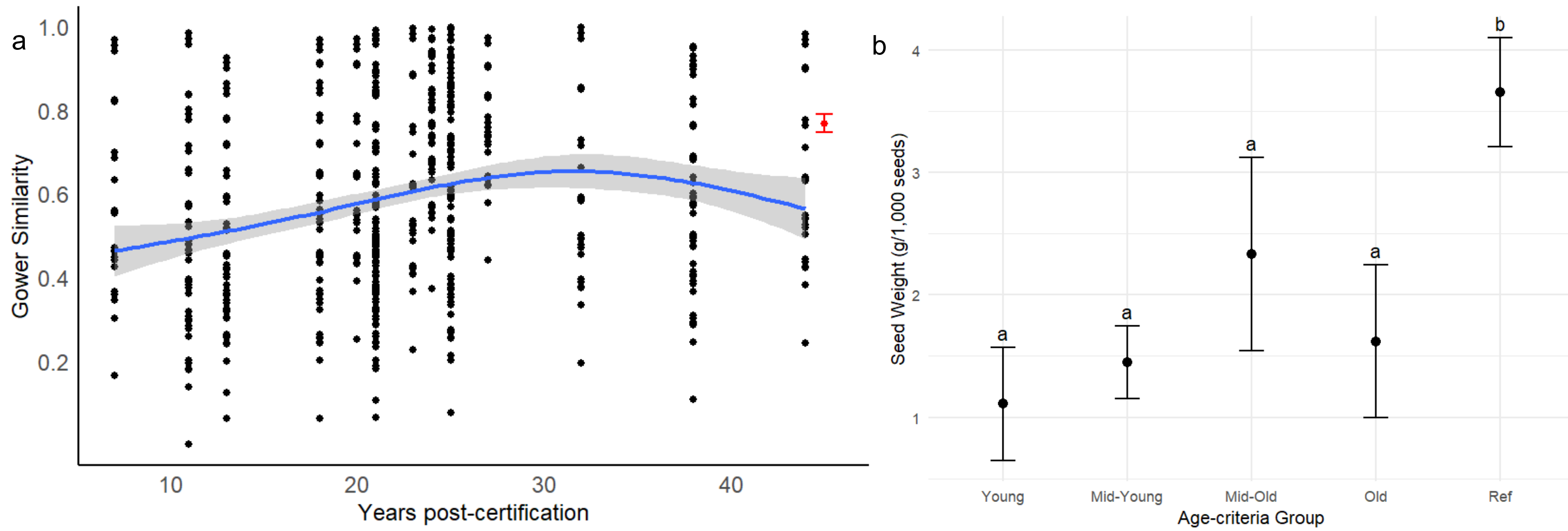

Figure 3-7. Seed weight on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for seed weight grouped by age criteria group. **Fig. 3-7a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-7b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

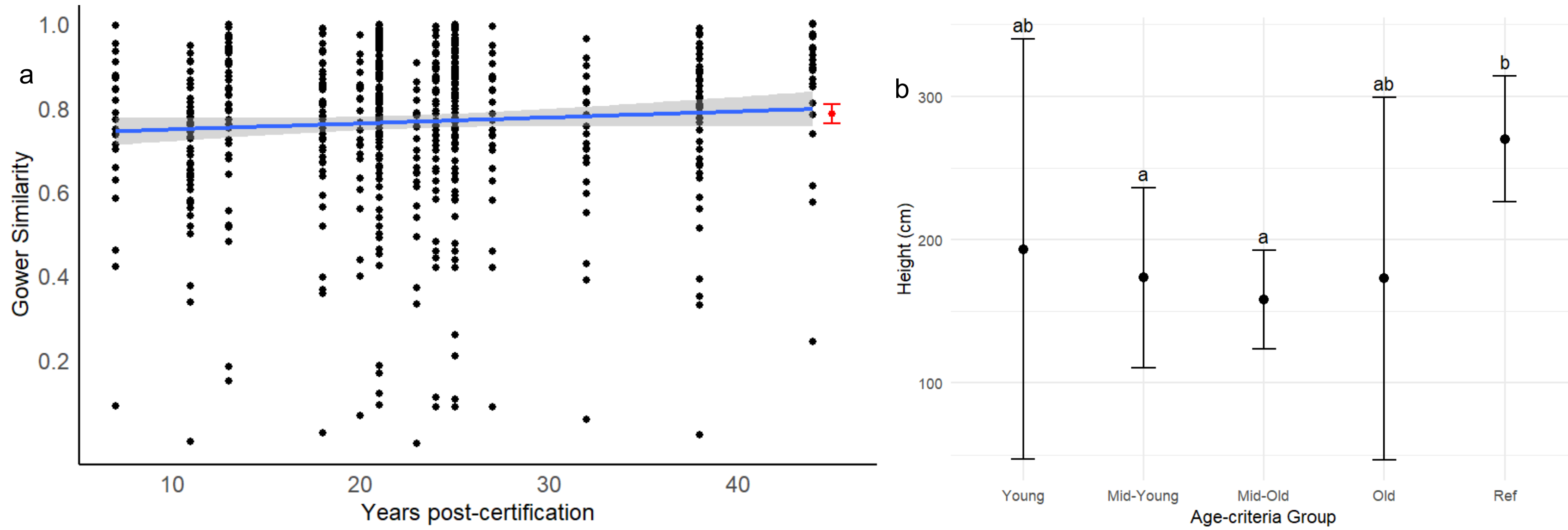

Figure 3-8. Height on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for height grouped by age criteria group. **Fig. 3-8a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-8b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

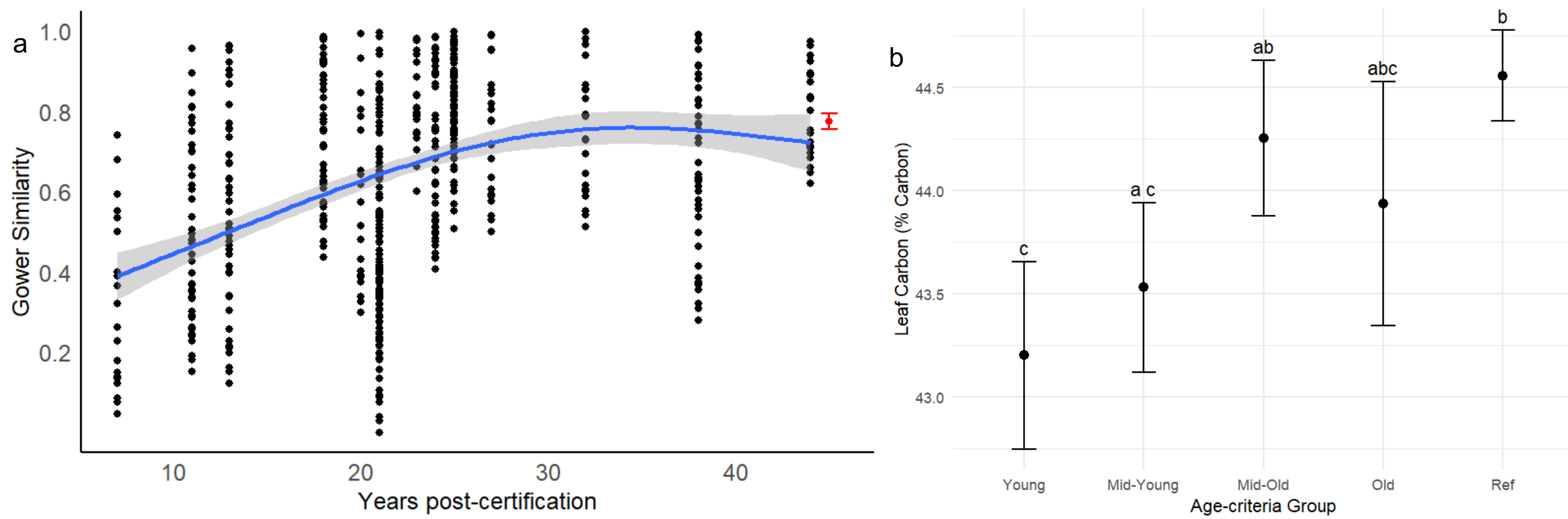

Figure 3-9. Leaf Carbon content (%) on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for leaf Carbon content grouped by age criteria group. **Fig. 3-9a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 9b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-c) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

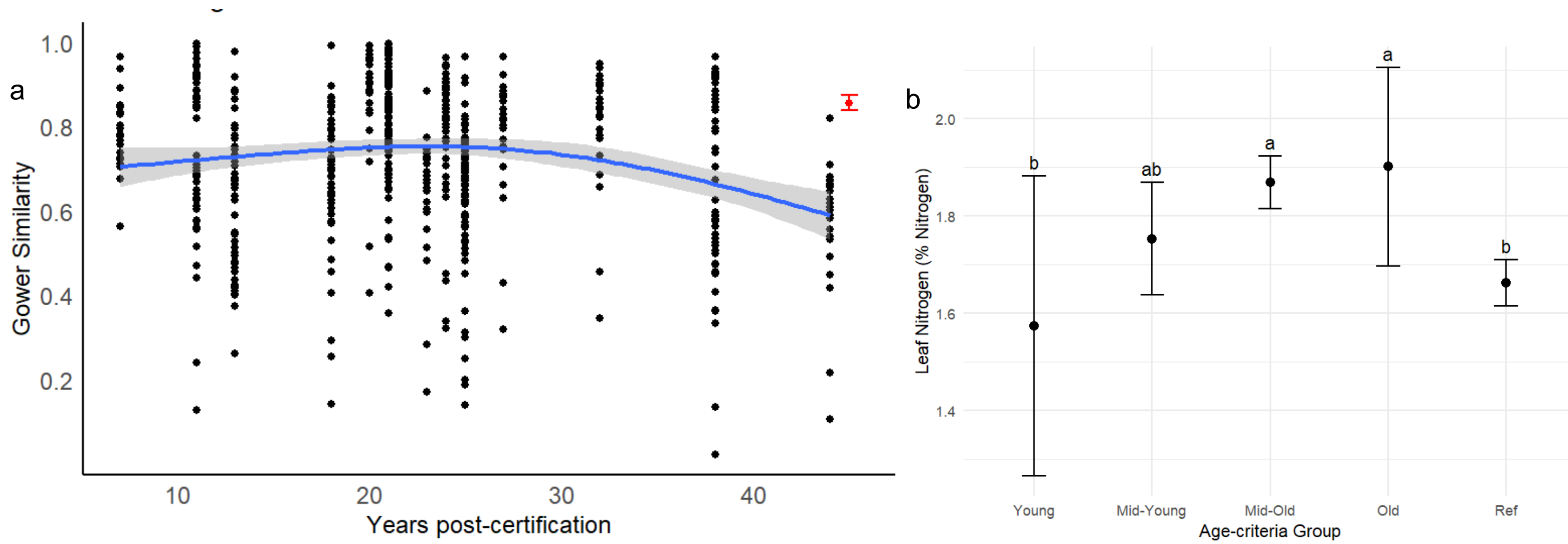

Figure 3-10. Leaf Nitrogen content (%) on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for leaf Nitrogen content grouped by age criteria group. **Fig. 3-10a**: The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-10b**: Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

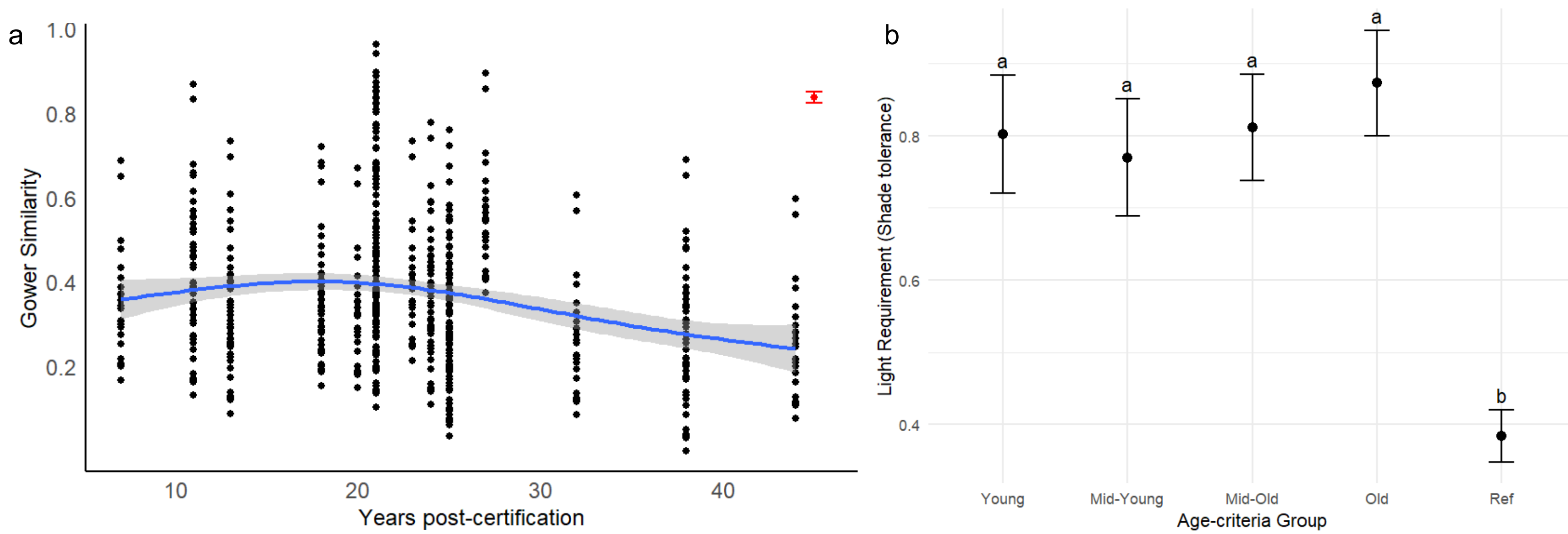

Figure 3-11. Light requirement (shade tolerance, see Table 3-1; higher values=greater light requirement/low shade tolerance) on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for light requirement grouped by age criteria group. **Fig. 3-11a**: The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-11b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

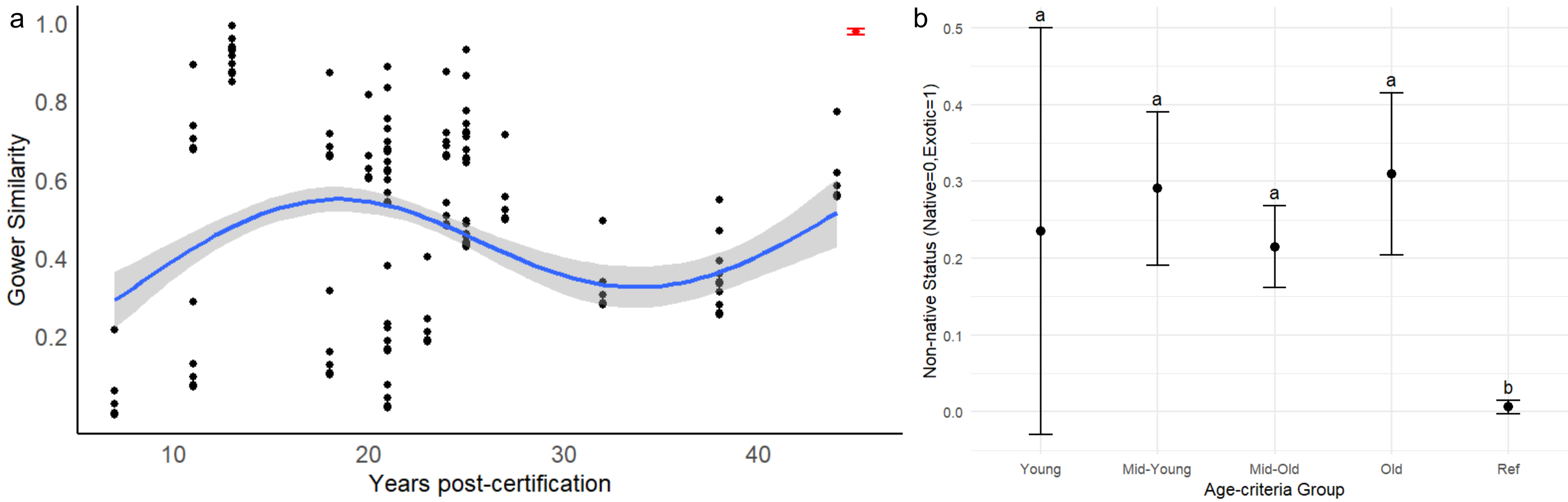

Figure 3-12. Non-native status (0=native, 1=introduced) on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for non-native status grouped by age criteria group. **Fig. 3-12a**: The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-12b**: Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

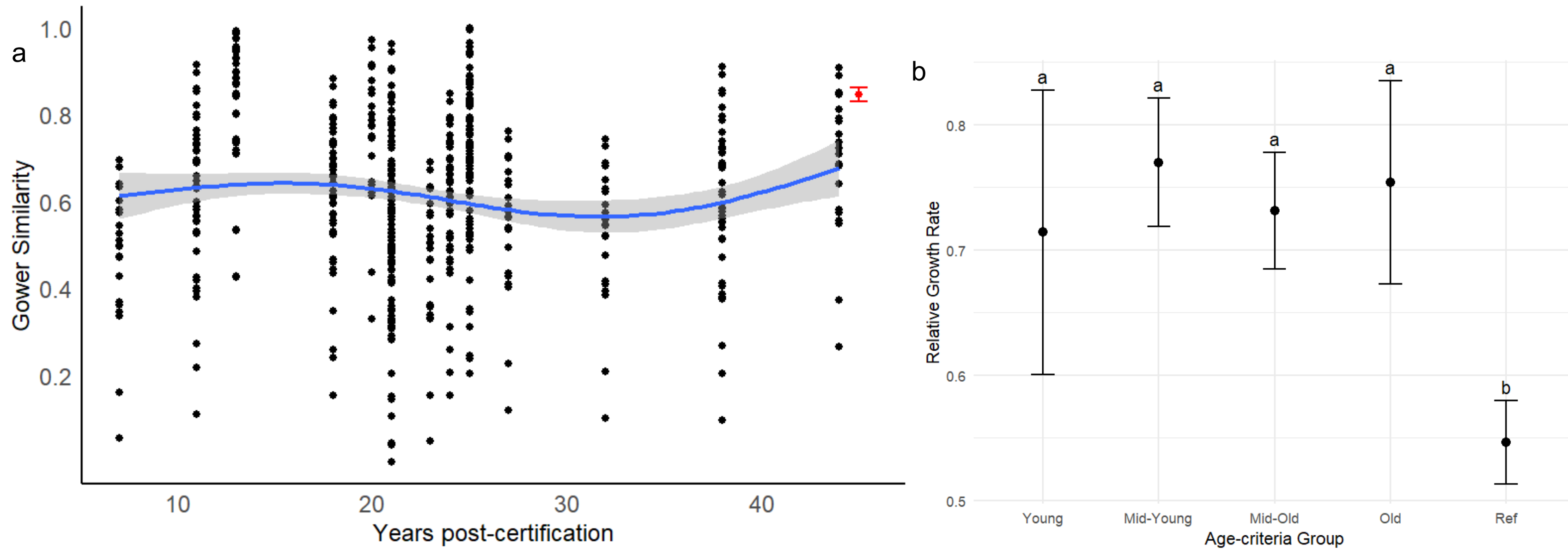

Figure 3-13. Relative growth rate (see Table 3-1; higher values=faster growth) on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for relative growth rate grouped by age criteria group. **Fig. 3-12a**: The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-12b**: Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

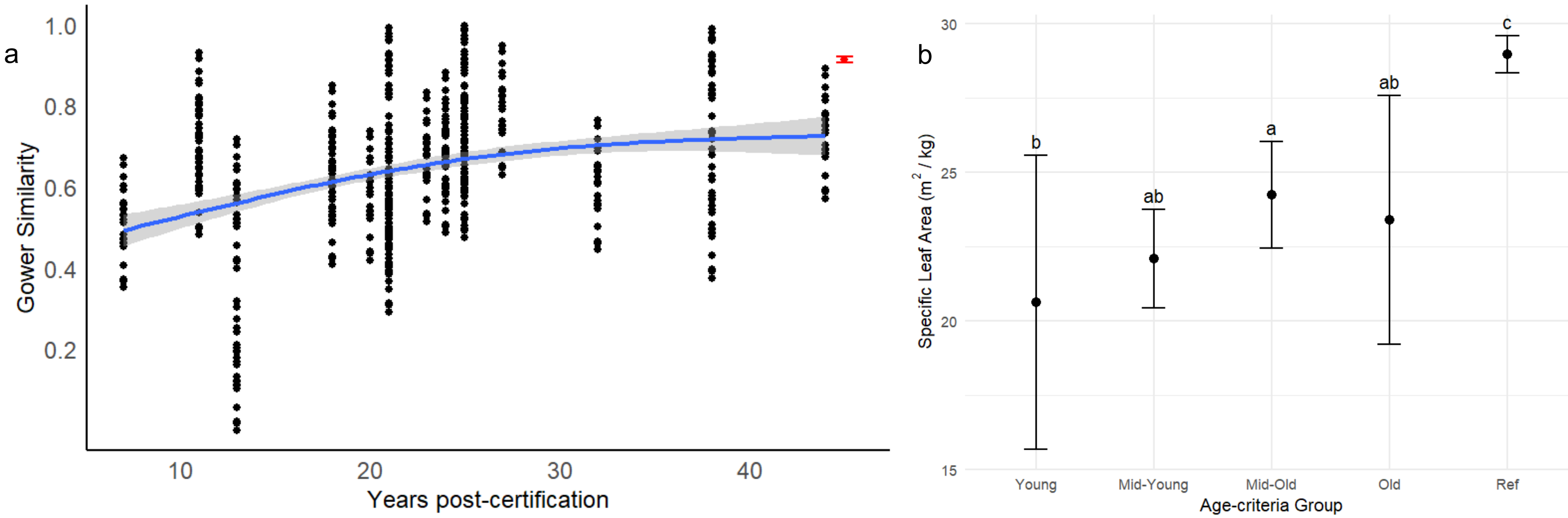

Figure 3-14. Specific leaf area on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for specific leaf area grouped by age criteria group. **Fig. 3-14a**: The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-14b**: Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-c) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

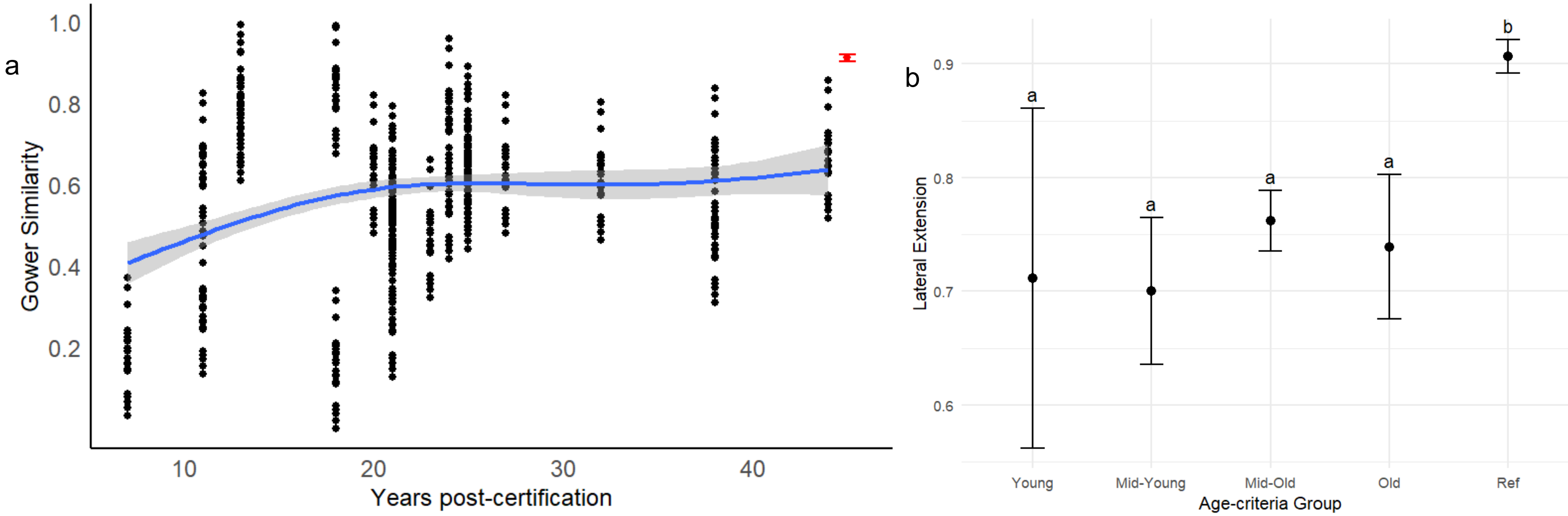

Figure 3-15. Lateral extension (see Table 3-1; higher values=greater lateral spread) on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for lateral extension grouped by age criteria group. **Fig. 3-15a**: The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-15b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using pairwise comparisons with Bonferroni adjustment).

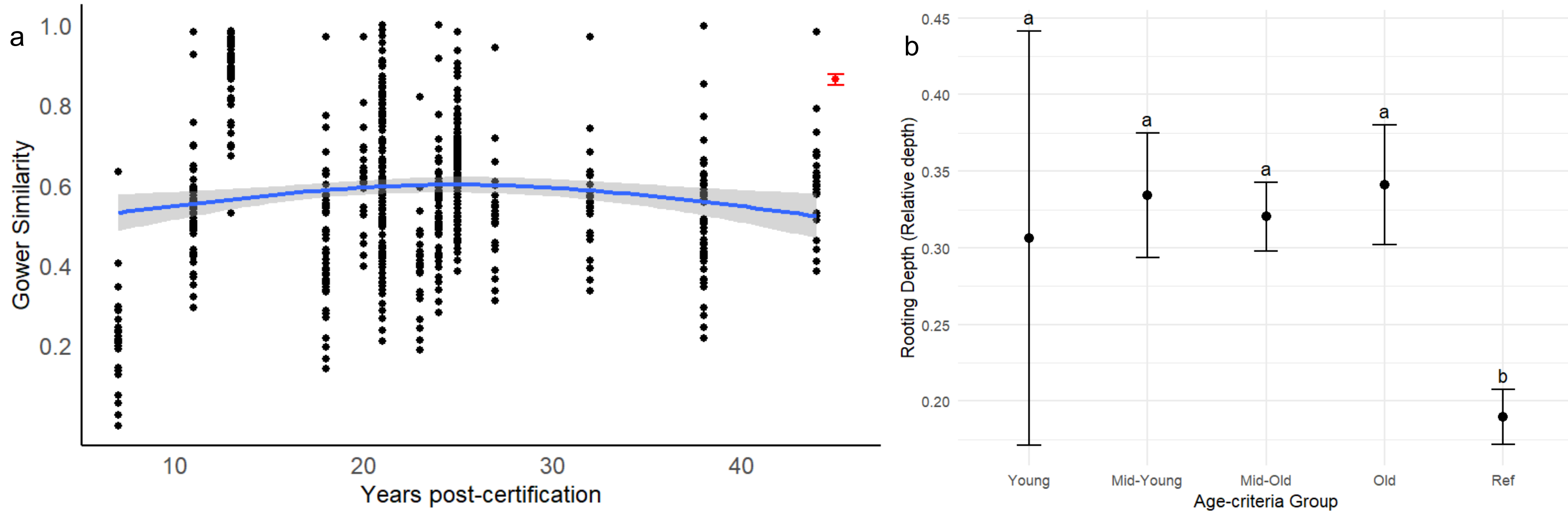

Figure 3-16. Relative rooting depth on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for relative rooting depth grouped by age criteria group. **Fig. 3-16a**: The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-16b**: Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

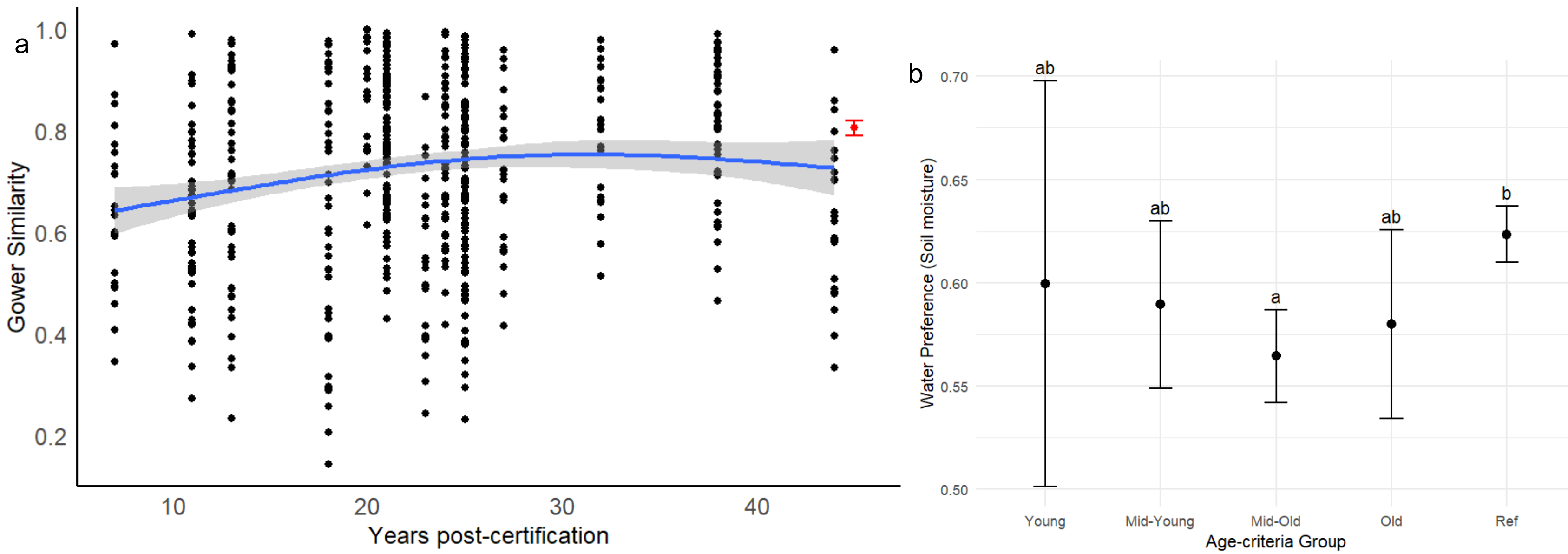

Figure 3-17. Water preference on reclaimed well pads and benchmark reference forests a) across years post-reclamation compared to reference forests, and b) interval plot for water preference grouped by age criteria group. **Fig. 3-17a:** The black line represents the modeled (Generalized Additive Mixed Model) community similarity (using Gower distance) between reclaimed well pads and reference benchmark forests. The gray shading shows the 95% confidence interval around the smoother. Points represent the raw data (similarity between each well pad and each reference forest). The red point with error bars on the right represents the overall mean similarity between reference benchmark forests, with a 95% confidence interval. Reclaimed well pads to the left of 16 years post-certification were reclaimed under newer criteria (2010 update) than those older than 16 years **Fig. 3-17b:** Points represent the mean raw data for each age criteria group, with 95% confidence interval error bars. Letters (a-b) above each interval indicate significant differences based on post-hoc testing (using Tukey test).

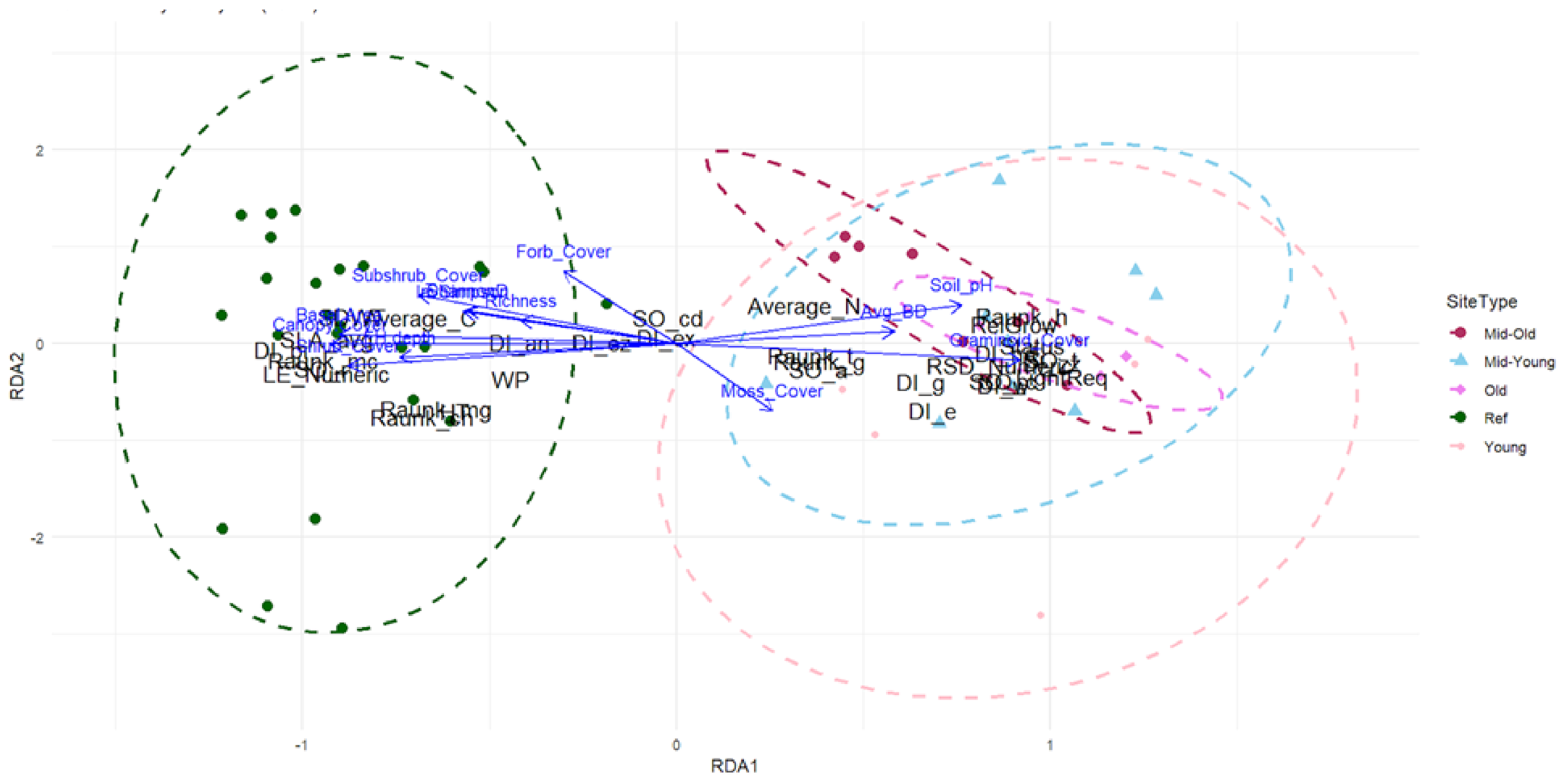

Figure 3-18. Distance-based redundancy analysis (dbRDA) model of functional trait Community Weighted Means. The first two axes explained 63.54% of the total CWM trait variance, of which 54.74% and 5.59% were explained in axis 1 and axis 2, respectively. Points are study plots and ecological variables are blue arrows, black arrows are short names for traits, refer to Table 3-1. Ecological variables are soil bulk density (Avg_BD), pH (Soil_pH), soil FH layer depth (FH_depth), Shannon and inverse Simpson diversity, richness, and cover of plant life forms (format: Lifeform_Cover).

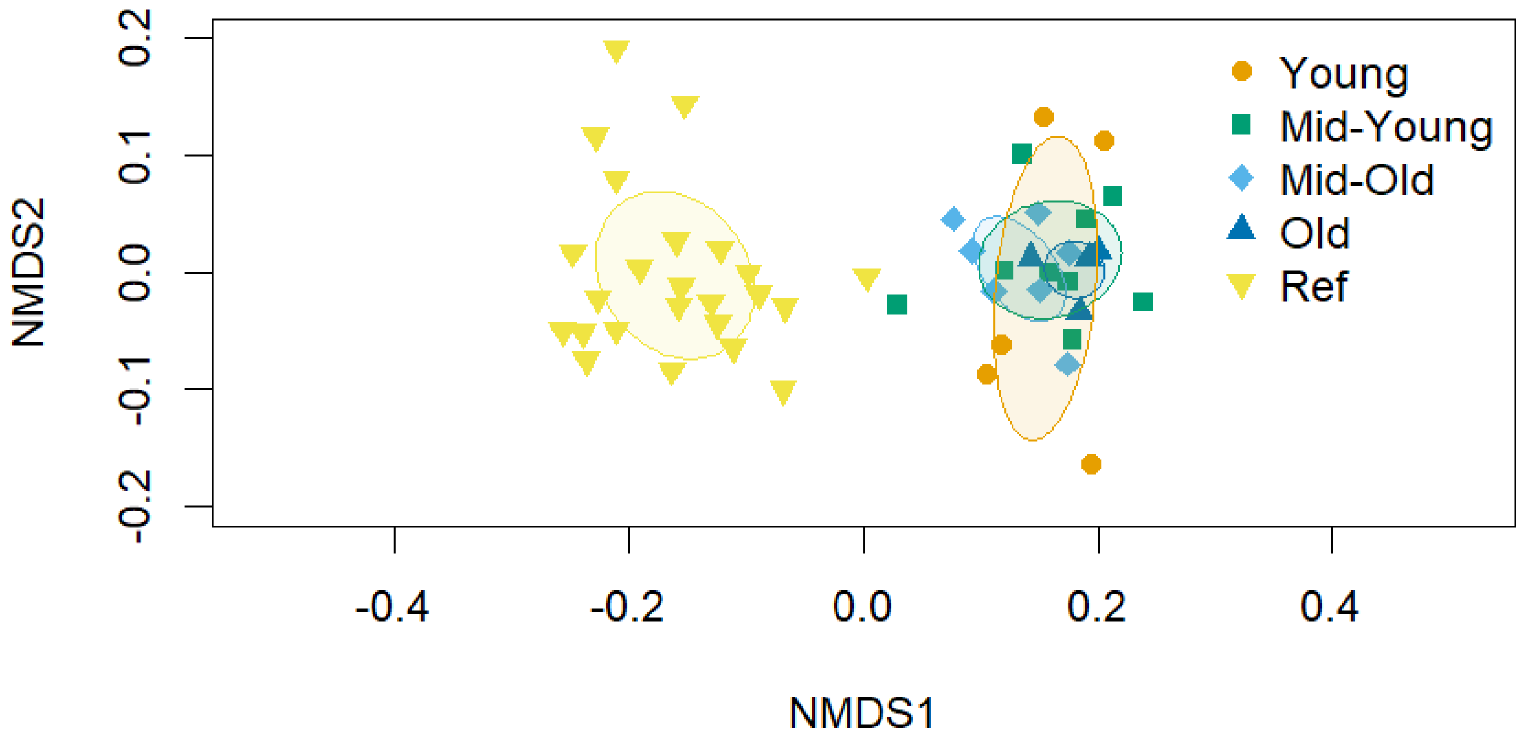

Figure 3-19. Nonmetric multidimensional scaling (NMDS) ordination of plant functional trait composition with 95% confidence interval ellipses for age-criteria group showing separation of wells and references but overlap among well age-criteria groups. See Table 3-2 for age-criteria group descriptions.

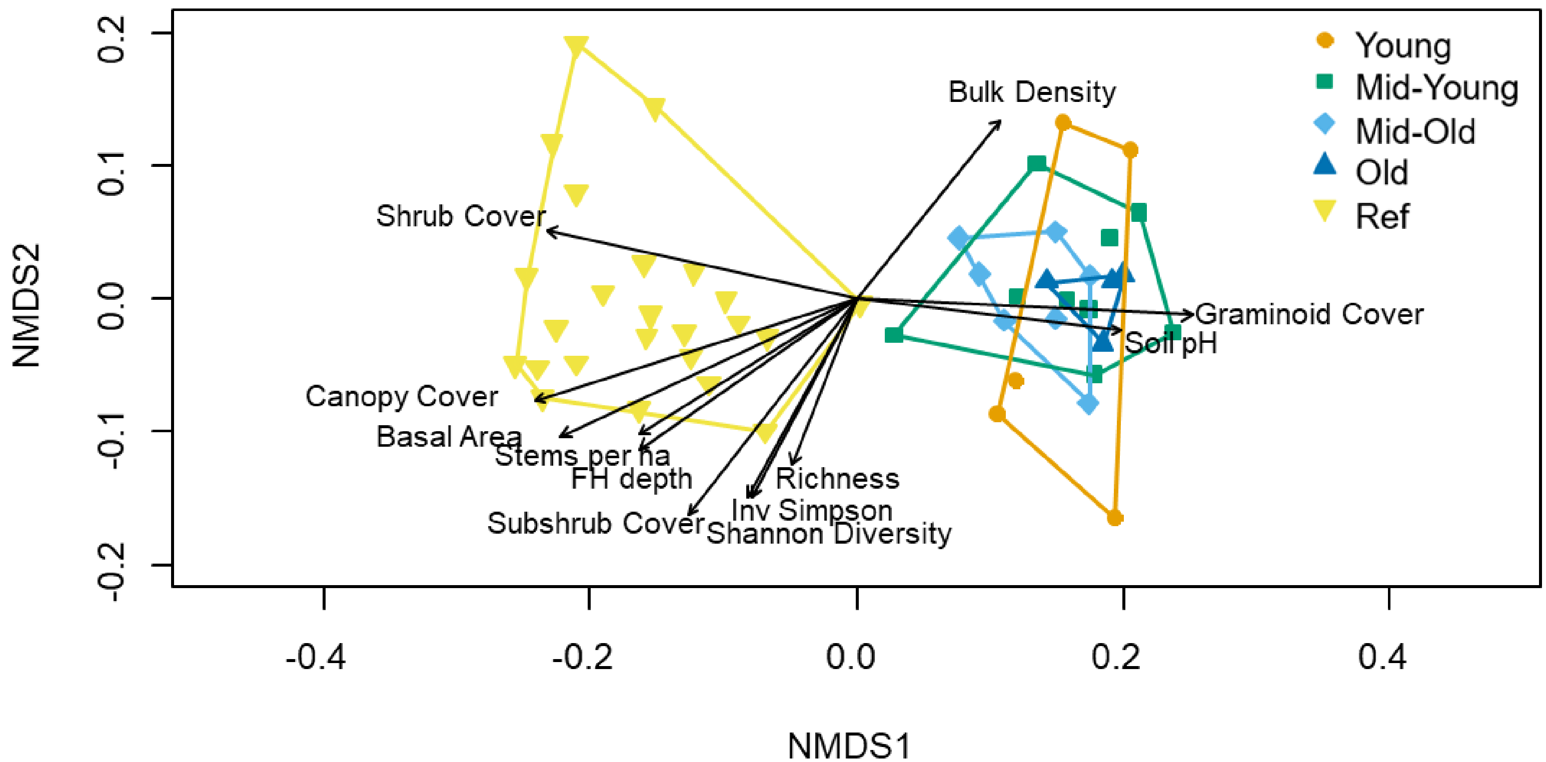

Figure 3-20. Nonmetric multidimensional scaling ordination of vegetation species composition with hulls for age-criteria group: Vectors indicate environmental, vegetation, soil, and diversity variables). Vector direction and length reflect the strength of correlation with the first two axes. See Table 3-2 for age-criteria group descriptions. To avoid overcrowding, only vectors with P<0.001 from vector analysis were plotted. See Appendix Fig. B-1 for all vectors.

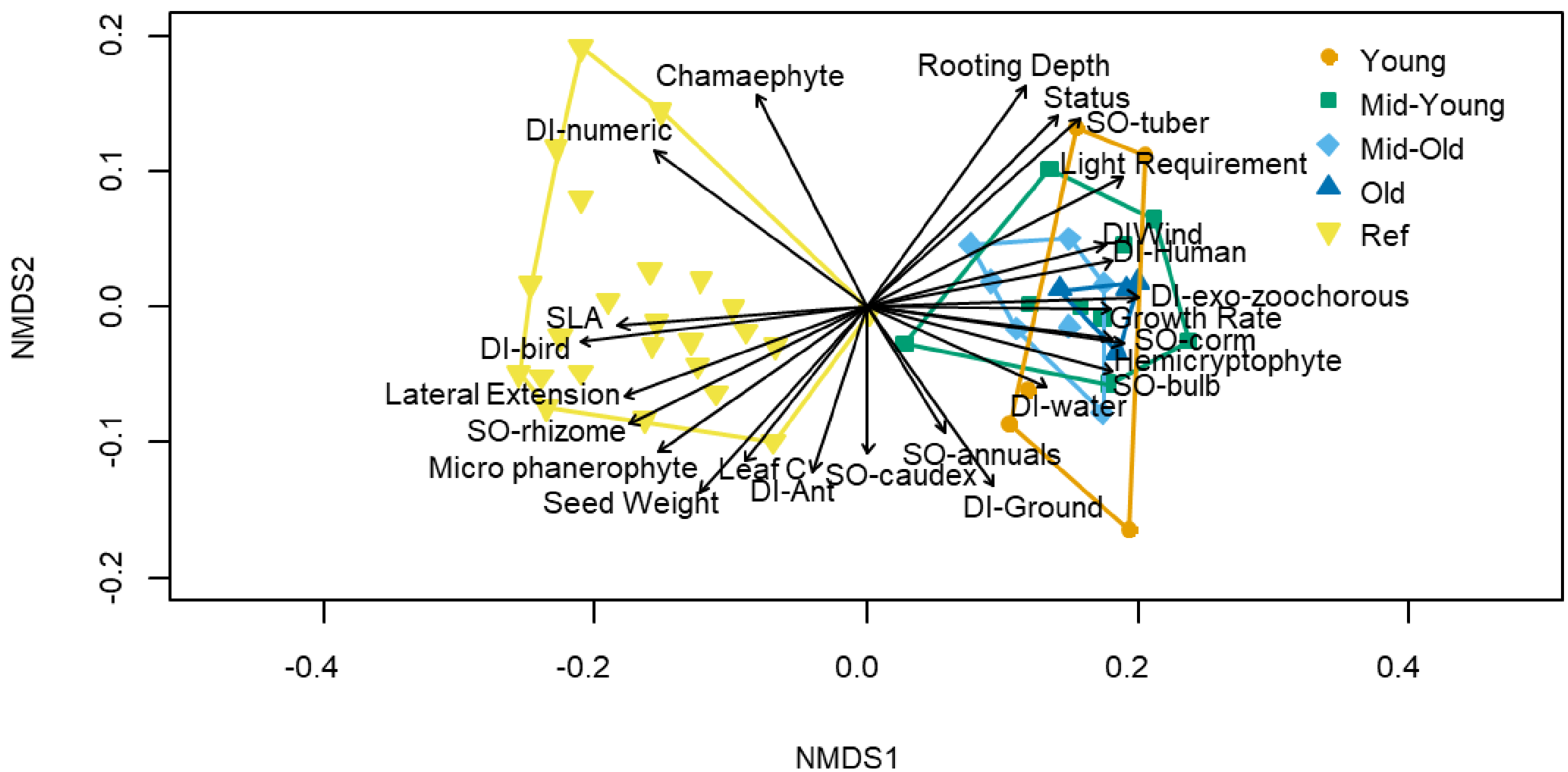

Figure 3-21. Nonmetric multidimensional scaling ordination of vegetation species composition with hulls for age-criteria group. Vectors indicate functional traits. Vector direction and length reflect the strength of correlation with the first two axes. See Table 3-2 for age-criteria group descriptions. To avoid overcrowding, only vectors with P<0.001 from vector analysis were plotted. See Appendix Fig. B-2 for all vectors.

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Chapter 4. Conclusions

My thesis offers important insights into the recovery of taxonomic, structural, and functional trait measures on reclaimed well pads over time, using a chronosequence of wells at different reclamation ages and reference benchmark forests, while simultaneously examining the impact of the 2010 criteria change on reclaimed well recovery.

In Chapter 2, I investigated the recovery of taxonomic, structural, and soil properties on reclaimed wells in Alberta's boreal forests compared to the reference mature forest benchmark. I found evidence of certain vegetation and soil variables recovering towards the reference condition, but did not find a strong association between time since reclamation and recovery success. This lack of recovery may be due to soil compaction and agronomic species planted in the past. Wells had higher soil compaction, soil pH, and graminoid cover compared to adjacent forests, which had greater soil organic layer depth, shrub and tree cover, canopy cover, and basal area. Understory indicator species analysis showed wells commonly had introduced species and ruderal species. Soil characteristics, such as high bulk density, high pH, and low FH layer depth showed evidence of significantly altered soils that likely support the survival of planted ruderal species. The results of this chapter suggested that the recovery of soil properties on disturbed sites may be delayed or even permanently altered, with lasting consequences for forest regeneration and ecosystem health. My research revealed that graminoids, including native species, on reclaimed wells may be preventing the growth of woody species by creating a physical barrier and outcompeting them for light, water, and nutrients. Overstory structure, such as canopy cover and basal area, failed to recover on wells, indicating the compromised structural complexity of these ecosystems. This lack of canopy cover likely created open conditions that further facilitated the establishment and spread of ruderal and non-native species (which were planted on older wells), reinforcing the altered trajectory of ecological recovery on these sites. My findings highlighted the need for more targeted reclamation strategies that prioritize the re-establishment of native tree species and the overall structural complexity of the forest. My results also showed the importance of reclamation decisions on individual sites, as time does not appear to override arrested succession in many cases.

In Chapter 3, I examined the long-term recovery of functional trait composition on reclaimed well pads, revealing significant differences between reference forests and reclaimed wells. My results showed that reclamation practices have long-term impacts on plant communities' functional traits, as a majority did not recover compared to mature benchmark forests in the timescale of this study. Many functional traits were not trending towards the mature reference benchmark condition, although some showed signs of improvement. Many reclaimed wells remained in slowed or arrested succession, exhibiting traits associated with rapid colonization and early succession, such as wind seed dispersal, higher light preference, and faster relative growth rate. Some wells showed unique soil properties compared to references, like compaction and lack of organic layer, which may have prevented late-successional species from colonizing and led to the persistence of early-successional and invasive species. I also found that while some functional traits were similar to the adjacent reference, most did not show similarity to the reference benchmark forest at any age post-certification. My results revealed that ecological variables play a significant role in the functional composition of disturbed forests after reclamation. However, age and criteria did not significantly affect the functional composition of well pads. My findings suggested the benefits of a trait-based approach to restore disturbed boreal ecosystems, focusing on the functional characteristics of plant species that directly influence ecosystem processes and resilience. This approach can identify and prioritize species that contribute to desired ecosystem functions, enhancing the long-term success of reclamation projects. Future research should prioritize monitoring wells under the newer criteria and perform manipulative field experiments (e.g., with treatment plots under various reclamation scenarios) to better understand the mechanisms influencing successional trajectories and plant functional trait recovery.

The findings of this study have important implications for forest management practices. It may take more than 44 years for some wells to show signs of recovery. While there is some hope for younger wells reclaimed under the new criteria, the outcomes are still unclear due to the relatively short time since their implementation. Additionally, the limited number of younger wells in my study area makes it difficult to draw definitive conclusions. Nonetheless, some younger wells showed promising signs of recovery, including fewer introduced species and a higher proportion of trees in the upper strata of the understory (200-250+ cm) despite limited basal area and stem density. These findings suggest that ongoing monitoring and adaptive management will be crucial in ensuring the successful reclamation and recovery of well pads, particularly as wells are now being reclaimed under the updated criteria.

While this study provides valuable insights into the recovery of reclaimed well pads, it is important to recognize that the long-term success of reclamation efforts remains uncertain. It remains to be seen if the 2010 criteria change will set wells on an effective recovery trajectory. Nevertheless, my results can be applied to forest management and reclamation practices. As evidenced by many wells having limited vertical and horizontal structure (i.e. lack of species in upper strata of understory, low basal area and canopy cover), reclamation strategies should consider prioritizing planting native shrubs and trees since they're unlikely to naturally propagate on reclaimed wells. This will promote structural complexity that is lacking on many wells. Reclamation specialists should also consider strategies to mitigate soil compaction such as deep-ripping and to stimulate organic horizon genesis such as use of soil amendments (e.g., mulches, biosolids). Trait-based management of reclaimed well pads in boreal forests should prioritize plant functional traits associated with reference benchmark forests (resource conservation strategy) rather than those observed on disturbed sites (resource acquisition strategy). For example, species with high seed weight, specific leaf area, and lateral extension will have greater functional similarity to mature benchmark forests compared to fast-growing deep-rooting plants. While early successional species will inherently have a more acquistive strategy than mature forests, selecting for more similar functional traits may better set reclaimed well pads on a trajectory towards functioning similarly to pre-disturbance conditions. Forest management could also involve identifying functional traits that reflect beneficial services such as carbon content reflecting carbon sequestration or water uptake (by roots or leaves) reflecting water filtration.

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Appendices

Appendix A

Appendix Table A-1. List of species and taxa for vascular and non-vascular plants sampled in the study.

Code	Genus	Species	Scientific authority	Growth form
ABIBAL	*Abies*	*balsamea*	(Linnaeus) Miller	Tree
ACHALP	*Achillea*	*alpina*	Linnaeus	Forb
ACHMIL	*Achillea*	*millefolium*	Linnaeus	Forb
ACTRUB	*Actaea*	*rubra*	(Aiton) Willdenow	Forb
AEGPOD	*Aegopodium*	*podagraria*	Linnaeus	Forb
ALNALN	*Alnus*	*alnobetula*	(Ehrhart) K. Koch	Tree
AMEALN	*Amelanchier*	*alnifolia*	Nuttall) Nuttall ex M. Roemer	Shrub
ARANUD	*Aralia*	*nudicaulis*	Linnaeus	Forb
ASTAME	*Astragalus*	*americanus*	(Hooker) M.E. Jones	Forb
ASTCAN	*Astragalus*	*canadensis*	Linnaeus	Forb
ASTCIC	*Astragalus*	*cicer*	Linnaeus	Forb
ATHFIL	*Athyrium*	*filix-femina*	(Linnaeus) Roth	Forb
AULPAL	*Aulacomnium*	*palustre*	(Hedwig) Schwagrichen	Moss
BETPAP	*Betula*	*papyrifera*	Marshall	Tree
BOTVIR	*Botrypus*	*virginianus*	(Linnaeus) Michaux	Forb
BRACMOSS	*Brachythecium*	*spp.*		Moss
BRANCHGRAM	*Phleum*	*pratense*	Linnaeus	Graminoid
	Calamagrostis	*canadensis*	(Michaux) Palisot de Beauvois	
	Festuca	*Rubra*	Linnaeus	
BROINE	*Bromus*	*inermis*	Leysser	Graminoid
BULRUSH	*Scirpus*	*microcarpus*	J. Presl & C. Presl	Graminoid
	Scirpus	*atrovirens*	Willdenow	
	Scirpus	*atrocinctus*	Fernald	
CALCAN	*Calamagrostis*	*canadensis*	(Michaux) Palisot de Beauvois	Graminoid
CALLEP	*Caltha*	*leptosepala*	de Candolle	Forb
CALPAL	*Caltha*	*palustris*	Linnaeus	Forb
CASMIN	*Castilleja*	*miniata*	Douglas ex Hooker	Forb
CERFON	*Cerastium*	*fontanum*	Baumgarten	Forb
CHAANG	*Chamaenerion*	*angustifolium*	(Linnaeus) Scopoli	Forb

Code	Genus	Species	Scientific authority	Growth form
CICDOU	*Cicuta*	*douglasii*	(de Candolle) J.M. Coulter & Rose	Forb
CICSPP	*Cicuta*	*douglasii*	(de Candolle) J.M. Coulter & Rose	Forb
	Cicuta	*maculata*	Linnaeus	
CIRARV	*Cirsium*	*arvense*	(Linnaeus) Scopoli	Forb
CLIDEN	*Climacium*	*dendroides*	(Hedwig) F. Weber & D. Mohr	Moss
COPTRI	*Coptis*	*trifolia*	(Linnaeus) Salisbury	Forb
CORCAN	*Cornus*	*canadensis*	Linnaeus	Forb
CORSER	*Cornus*	*sericea*	Linnaeus	Shrub
CORSTR	*Corallorhiza*	*striata*	Lindley	Forb
DELGLA	*Delphinium*	*glaucum*	S. Watson	Forb
DELMOSS	*Hypnum*	*spp.*		Moss
	Campylium	*spp.*		
	Bryum	*spp.*		
DENDEN	*Dendrolycopodium*	*dendroideum*	(Michaux) A. Haines	Forb
DREADU	*Drepanocladus*	aduncus	(Hedwig) Warnstorf	Moss
EQUARV	*Equisetum*	*arvense*	Linnaeus	Forb
EQUSYL	*Equisetum*	*sylvaticum*	Linnaeus	Forb
EURCON	*Eurybia*	*conspicua*	(Lindley) G.L. Nesom	Forb
FRAVIR	*Fragaria*	*virginiana*	Miller	Forb
GALBOR	*Galium*	*boreale*	Linnaeus	Forb
GALTRI_D	*Galium*	*trifidum*	Linnaeus	Forb
GALTRI_R	*Galium*	*triflorum*	Michaux	Forb
GENAMA	*Gentianella*	*amarella*	(Linnaeus) Börner	Forb
GEUMAC	*Geum*	*macrophyllum*	Willdenow	Forb
GEURIV	*Geum*	*rivale*	Linnaeus	Forb
GEUTRI	*Geum*	*triflorum*	Pursh	Forb
GOODREP	*Goodyera*	*repens*	(Linnaeus) R. Brown	Forb
GYMDRY	*Gymnocarpium*	*dryopteris*	(Linnaeus) Newman	Forb
HALDEF	*Halenia*	*deflexa*	(Smith) Grisebach	Forb
HERMAX	*Heracleum*	*maximum*	W. Bartram	Forb
HIEUMB	*Hieracium*	*umbellatum*	Linnaeus	Forb
HYLOMOSS	*Hylocomium*	*splendens*	(Hedwig) Schimper	Moss
KNIGHTMOSS	*Ptilium*	*crista-castrensis*	(Hedwig) De Notaris	Moss
LAROCC	*Larix*	*occidentalis*	Nuttall	Tree
LATOCH	*Lathyrus*	*ochroleucus*	Hooker	Forb

Code	Genus	Species	Scientific authority	Growth form
LATVEN	*Lathyrus*	*venosus*	Muhlenberg ex Willdenow	Forb
LEUVUL	*Leucanthemum*	*vulgare*	Lamarck	Forb
LILPHI	*Lilium*	*philadelphicum*	Linnaeus	Forb
LILY	*Lilium*	*philadelphicum*	Linnaeus	
	Lilium	*columbianum*	Leichtlin	
	Prosartes	*trachycarpa*	S. Watson	
LINBOR	*Linnaea*	*borealis*	Linnaeus	Forb
LONDIO	*Lonicera*	*dioica*	Linnaeus	Vine
LONINV	*Lonicera*	*involucrata*	(Richardson) Banks ex Sprengel	Shrub
LONVIL	*Lonicera*	*villosa*	Linnaeus	Shrub
MAICAN	*Maianthemum*	*canadense*	Desfontaines	Forb
MAIDIL	*Maianthemum*	*dilatatum*	(Alph. Wood) A. Nelson & J.F. Macbride	Forb
MAIRAC	*Maianthemum*	*racemosum*	(Linnaeus) Link	Forb
MAISTEL	*Maianthemum*	*stellatum*	(Linnaeus) Link	Forb
MEDSAT	*Medicago*	*sativa*	Linnaeus	Forb
MELOFF	*Melilotus*	*officinalis*	(Linnaeus) Lamarck	Forb
MENARV	*Mentha*	*arvensis*	Linnaeus	Forb
MERPAN	*Mertensia*	*paniculata*	(Aiton) G. Don	Forb
MITNUD	*Mitella*	*nuda*	Linnaeus	Forb
MOELAT	*Moehringia*	*lateriflora*	(Linnaeus) Fenzl	Forb
MONUNI	*Monotropa*	*uniflora*	Linnaeus	Forb
NEWGRAM42	*Leymus*	*innovatus*	(Beal) Pilger	Graminoid
	Elymus	*trachycaulus*	*(Link) Gould ex Shinners*	
NEWGRAM REC	*Phalaris*	*arundinacea*	Linnaeus	Graminoid
	Poa	*palustris*	Linnaeus	
	Poa	*interior*	Rydberg	
	Poa	*nemoralis*	Linnaeus	
NONBGRAM	*Acorus*	*calamus*	Linnaeus	Graminoid
	Carex	*aquatilis*	Wahlenberg	
OPLHOR	*Oplopanax*	*horridus*	(Smith) Miquel	Shrub
OSMDEP	*Osmorhiza*	*depauperata*	Philippi	Forb
PARPAL	*Parnassia*	*palustris*	Linnaeus	Forb
PEDGRO	*Pedicularis*	*groenlandica*	Retzius	Forb
PETFRI	*Petasites*	*frigidus*	(Linnaeus) Fries	Forb

Code	Genus	Species	Scientific authority	Growth form
PHLPRA	*Phleum*	*pratense*	Linnaeus	Graminoid
PICGLA	*Picea*	*glauca*	(Moench) Voss	Tree
PICMAR	*Picea*	*mariana*	(Miller) Britton, Sterns & Poggenburgh	Tree
PILAUR	*Pilosella*	*aurantiaca*	(Linnaeus) F.W. Schultz & Schultz Bipontinus	Forb
PINCON	*Pinus*	*contorta*	Douglas ex Loudon	Tree
PLASPP	*Plagiomnium*	*spp.*		Moss
PLAHUR	*Platanthera*	*huronensis*	(Nuttall) Lindley	Forb
PLAMAJ	*Plantago*	*major*	Linnaeus	Forb
PLAOBT	*Platanthera*	*obtusata*	(Banks ex Pursh) Lindley	Forb
PLESCH	*Pleurozium*	*schreberi*	(Willdenow ex Bridel) Mitten	Moss
POPBAL	*Populus*	*balsamifera*	Linnaeus	Tree
POPTRE	*Populus*	*tremuloides*	Michaux	Tree
PROTRA	*Prosartes*	*trachycarpa*	S. Watson	Forb
PYRASA	*Pyrola*	*asarifolia*	Michaux	Subshrub
RANACR	*Ranunculus*	*acris*	Linnaeus	Forb
RHIMIN	*Rhinanthus*	*minor*	Linnaeus	Forb
RHOGRO	*Rhododendron*	*groenlandicum*	(Oeder) Kron & Judd	Shrub
RIBGLA	*Ribes*	*glandulosum*	Grauer	Shrub
RIBHIR	*Ribes*	*Hirtellum*	Michaux	Shrub
RIBLAC	*Ribes*	*lacustre*	(Persoon) Poiret	Shrub
RIBTRI	*Ribes*	*triste*	Pallas	Shrub
ROSACI	*Rosa*	*acicularis*	Lindley	Subshrub
RUBIDA	*Rubus*	*idaeus*	Linnaeus	Subshrub
RUBPAR	*Rubus*	*parviflorus*	Nuttall nom. cons.	Subshrub
RUBPED	*Rubus*	*pedatus*	Smith	Forb
RUBPUB	*Rubus*	*pubescens*	Rafinesque	Forb
SALSPP	*Salix*	*bebbiana*	Sargent	Shrub
	Salix	*pedicellaris*	Pursh	
	Salix	*planifolia*	Pursh	
	Salix	*pyrifolia*	Andersson	
	Salix	*scouleriana*	Barratt ex Hooker	
SAMRAC	*Sambucus*	*racemosa*	Linnaeus	Shrub
SCUGAL	*Scutellaria*	*galericulata*	Linnaeus	Forb
SHECAN	*Shepherdia*	*canadensis*	(Linnaeus) Nuttall	Shrub

Code	Genus	Species	Scientific authority	Growth form
SMALLGRAM	*Deschampsia*	*caespitosa*	(Linnaeus) Palisot de Beauvois	Graminoid
	Carex	*deweyana*	Schweinitz	
	Carex	*aurea*	Nuttall	
SOLCAN	*Solidago*	*canadensis*	Linnaeus	Forb
SONARV	*Sonchus*	*arvensis*	Linnaeus	Forb
SPHAG	*Sphagnum*	*spp.*		Moss
SPIANN	*Spinulum*	*annotinum*	(Linnaeus) A. Haines	Forb
SPILUC	*Spiraea*	*lucida*	Douglas ex Greene	Shrub
STRAMP	*Streptopus*	*amplexifolius*	(Linnaeus) de Candolle	Forb
SYMCIL	*Symphyotrichum*	*ciliolatum*	(Lindley) Á. Löve & D. Löve	Forb
SYMLAE	*Symphyotrichum*	*laeve*	(Linnaeus) Á. Löve & D. Löve	Forb
SYMLAN	*Symphyotrichum*	*lanceolatum*	(Willdenow) G.L. Nesom	Forb
SYMPUN	*Symphyotrichum*	*puniceum*	(Linnaeus) Á. Löve & D. Löve	Forb
TAROFF	*Taraxacum*	*officinale*	F.H. Wiggers	Forb
THAVEN	*Thalictrum*	*venulosum*	Trelease	Forb
THINGRAM	*Puccinellia*	*distans*	(Jacquin) Parlatore	Graminoid
	Agrostis	*gigantea*	Roth	
	Agrostis	*scabra*	Willdenow	
THUREC	*Thuidium*	recognitum	(Hedwig) Lindberg	Moss
TIATRI	*Tiarella*	*trifoliata*	Linnaeus	Forb
TOMNIT	*Tomentypnum*	*nitens*	(Hedwig) Loeske	Moss
TREEMOSS	*Climacium*	*dendroides*	(Hedwig) F. Weber & D. Mohr	Moss
TRIHYB	*Trifolium*	*hybridum*	Linnaeus	Forb
TRIPRA	*Trifolium*	*pratense*	Linnaeus	Forb
TRIREP	*Trifolium*	*repens*	Linnaeus	Moss
VACMEM	*Vaccinium*	*membranaceum*	Douglas ex Torrey	Shrub
VACMYR	*Vaccinium*	*myrtilloides*	Michaux	Shrub
VACOXY	*Vaccinium*	*oxycoccos*	Linnaeus	Shrub
VACVIT	*Vaccinium*	*vitis-idaea*	Linnaeus	Shrub
VIBEDU	*Viburnum*	*edule*	(Michaux) Rafinesque	Shrub
VICAME	*Vicia*	*americana*	Muhlenberg ex Willdenow	Vine
VICCRA	*Vicia*	*cracca*	Linnaeus	Forb
VIOCAN	*Viola*	*canadensis*	Linnaeus	Forb

Code	Genus	Species	Scientific authority	Growth form
VIOREN	*Viola*	*renifolia*	A. Gray	Forb
VIOSPP	*Viola*	*palustris*	Linnaeus	Forb
	Viola	*canadensis*	Linnaeus	
	Viola	*renifolia*	A. Gray	

Appendix Table A-2. Mean values and 95% confidence intervals visualized in interval plots of Figs. 2-11 through 2-20 for each variable. Age-criteria group descriptions are given in Table 2-1.

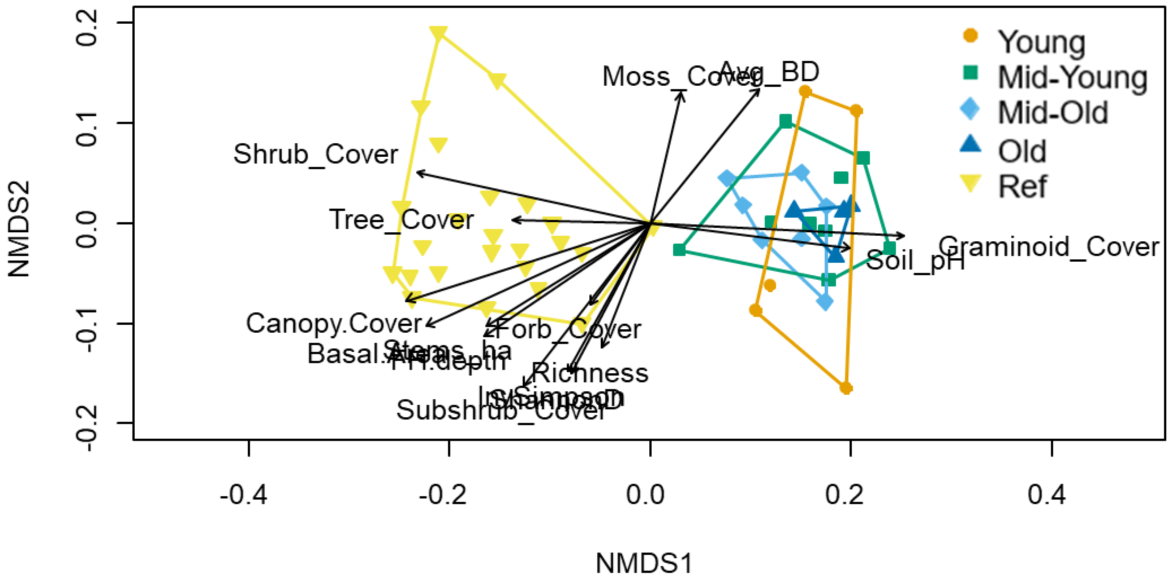
	Age Criteria group Mean value (95% confidence interval)				
Variable	**Young**	**Mid-Young**	**Mid-Old**	**Old**	**Ref**
Bulk density (g/cm^2)	0.84 (0.54-1.15)	0.94 (0.81-1.1)	0.85 (0.72-0.99)	1.0 (0.92-1.1)	0.64 (0.55-0.73)
Soil pH	6.10 (5.53-6.66)	6.062 (5.76-6.33)	5.76 (5.13-6.27)	6.57 (6.47-6.64)	4.73 (4.45-4.99)
FH depth (cm)	2.0 (0.71-3.53)	2.89 (2.15-3.59)	3.3 (2.61-3.95)	3.34 (2.99-3.64)	6.34 (5.52-7.22)
Shannon diversity	2.95 (2.66-3.23)	2.93 (2.81-3.05)	3.07 (2.94-3.17)	2.97 (2.85-3.10)	3.24 (3.16-3.31)
Inverse Simpson diversity	15.7 (11.75-19.7)	14.76 (12.95-16.70)	17.11 (14.81-19.13)	15.79 (13.76-17.77)	20.95 (19.46-22.50)
Richness	33.2 (24.4-42)	31.2 (28.34-34.11)	36.43 (33.71-39.28)	32.5 (30.5-34.5)	39.54 (36.71-42.63)
Basal Area (m^2/ha)	2.81 (0-7.73)	3.35 (1.06-6.05)	2.64 (1.14-4.33)	4.02 (0.44-8.74)	43.73 (38.39-49.46)
Canopy Cover (%)	22.3 (0.40-58.38)	19.71 (7.74-34.56)	17.83 (6.23-30.59)	24.40 (4.50-44.30)	87.05 (84.35-89.87)
Forb cover (%)	41.3 (27.92-52.95)	49.11 (45.26-53.55)	50.65 (45.71-55.1)	53.41 (48.31-62.38)	55.22 (50.25-59.25)
Graminoid cover (%)	34.67 (23.66-45.72)	34.06 (29.00-39.33)	29.51 (23.70-36.82)	32.72 (23.76-41.69)	6.55 (4.91-8.38)
Moss cover (%)	18.52 (13.93-23.54)	12.44 (8.53-16.34)	11.57 (7.78-15.84)	9.73 (5.74-15.37)	8.07 (4.47-12.36)
Shrub cover (%)	3.02 (0.61-5.30)	1.79 (0.62-3.16)	2.25 (0.66-4.71)	1.07 (0.66-1.66)	16.90 (14.63-19.78)
Subshrub cover (%)	1.42 (0.05-3.52)	1.61 (0.81-2.60)	5.08 (1.21-9.73)	2.33 (0.51-3.66)	11.01 (8.61-13.2)

Tree cover (%)	1.04 (0.34-1.97)	1.0 (0-2.50)	0.95 (0.54-1.39)	0.74 (0-1.48)	2.26 (1.74-2.77)

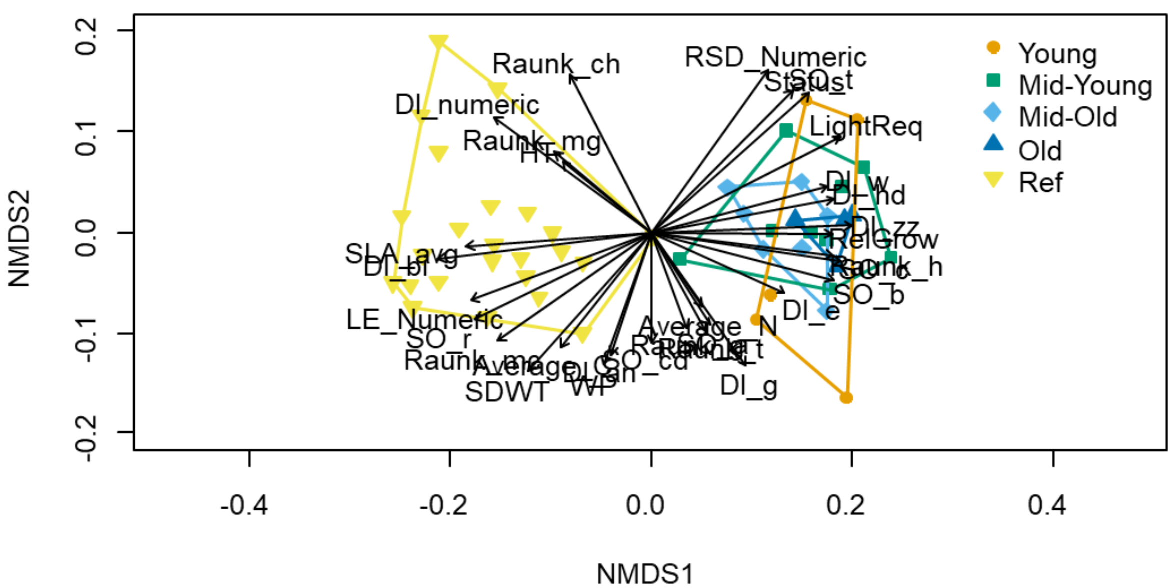
Appendix B

Appendix Table B-1. Means and 95% confidence intervals for each plant functional trait separated by age-criteria group.

	Age-criteria age Group				
Functional Trait	***Young***	***Mid-Young***	***Mid-Old***	***Old***	***Ref***
Light Requirement	*0.80 (0.75-0.85)*	*0.77 (0.70-0.82)*	*0.81 (0.75-0.86)*	*0.87 (0.83-0.91)*	*0.39 (0.36-0.42)*
Water Preference	*0.6 (0.54-0.67)*	*0.59 (0.56-0.62)*	*0.56 (0.55-0.58)*	*0.58 (0.55-0.61)*	*0.62 (0.61-0.64)*
Relative Growth Rate	*0.71 (0.64-0.79)*	*0.77 (0.73-0.81)*	*0.73 (0.69-0.76)*	*0.75 (0.71-0.80)*	*0.54 (0.51-0.57)*
Seed Weight	*1.11 (0.81-1.41)*	*1.45 (1.20-1.68)*	*2.34 (1.72-2.92)*	*1.62 (1.22-1.91)*	*3.64 (3.17-4.09)*
Height	*193.26 (122.01-300.23)*	*173.33 (127.66-226.82)*	*158.16 (135.75-186.14)*	*172.81 (106.39-239.21)*	*270.76 (233.91-314.05)*
Leaf Nitrogen	*1.57 (1.40-1.78)*	*1.75 (1.66-1.84)*	*1.87 (1.83-1.91)*	*1.9 (1.79-2.01)*	*1.66 (1.61-1.70)*
Leaf Carbon	*43.20 (42.94-43.53)*	*43.53 (43.23-43.86)*	*44.25 (43.98-44.54)*	*43.94 (43.57-44.21)*	*44.53 (44.32-44.73)*
Specific Leaf Area	*20.63 (17.10-23.21)*	*22.08 (20.84-23.46)*	*24.23 (23.08-25.71)*	*23.39 (21.34-25.51)*	*28.88 (28.28-29.46)*
Lateral Spread	*0.71 (0.62-0.80)*	*0.7 (0.65-0.75)*	*0.76 (0.74-0.78)*	*0.74 (0.70-0.77)*	*0.91 (0.89-0.92)*
Rooting Depth	*0.31 (0.23-0.39)*	*0.33 (0.30-0.37)*	*0.32 (0.30-0.34)*	*0.34 (0.33-0.36)*	*0.19 (0.18-0.21)*
Native Status	*0.24 (0.07-0.40)*	*0.29 (0.21-0.37)*	*0.22 (0.18-0.26)*	*0.31 (0.25-0.35)*	*0.01 (0-0.02)*
Seed dispersal vector (numeric)	*0.57 (0.54-0.60)*	*0.59 (0.57-0.61)*	*0.58 (0.56-0.62)*	*0.58 (0.57-0.59)*	*0.66 (0.65-0.67)*



Appendix Figure B-1. Nonmetric multidimensional scaling ordination of vegetation species composition with hulls for age-criteria group: Vectors indicate environmental, vegetation, soil, and diversity variables). Vector direction and length reflect the strength of correlation with the first two axes. See Table 3-2 for age-criteria group descriptions.



Appendix Figure B-2. Nonmetric multidimensional scaling ordination of vegetation species composition with hulls for age-criteria group. Vectors indicate functional traits. Vector direction and length reflect the strength of correlation with the first two axes. See Table 3-2 for age-criteria group descriptions.