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# Succession after reclamation: Identifying and assessing ecological indicators of forest recovery on reclaimed oil and natural gas well pads

Randi C. Lupardus[a,*], Anne C.S. McIntosh[a], Arnold Janz[b], Dan Farr[b]

[a] University of Alberta, Augustana Faculty, 4901 46 Ave, Camrose, AB T4V 2R3, Canada
[b] Alberta Environment and Parks, Environmental Monitoring & Science Division, Edmonton, AB T5J 5C6, Canada

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ABSTRACT

Site preparation for oil and gas extraction often requires the complete removal of vegetation and surface soil on the well pad. Although subsequent reclamation then attempts to restore vegetation and soil properties on the well pad, given the magnitude of the extraction disturbance, the potential to shift its successional trajectory is high. The objectives of this study were to: i) assess successional recovery of vegetation and soil on decommissioned and reclaimed well pads and compare it with reference forest of varied successional stages, and ii) identify which above- and below-ground properties were influenced by reclamation and are thus useful ecological indicators for recovery towards forest. We sampled 30 study units in Alberta's boreal forest; each site included a reclaimed well pad and adjacent reference site, with well pads ranging from 7 to 48 years post-reclamation. We conducted multivariate statistical analysis using 62 ecological above- and below-ground properties (e.g., percent cover of plant species, soil bulk density) categorized by: site type (reclaimed vs reference), natural subregion, forest stage, forest type, and time since reclamation. By grouping sites by site type, forest type, forest stage, and time since last disturbance, there was a clear separation of sites, with only two reclaimed well pads (7%) resembling plant community composition of reference areas, and 18 well pads (60%) resembling treeless grasslands, two of which were > 35 years post disturbance, indicating an arrested recovery trajectory. The remaining 33% of well pads are likely on a trajectory towards recovery. We found that reclamation had a significant effect on soil bulk density (E = 0.35), soil pH (E = 0.24), noxious plant species (E = 2.33), canopy cover (E = −0.26), grass cover (E = 0.16), woody cover (E = −0.18), LFH depth (E = −0.15), introduced species richness (E = 0.26), and live tree basal area (E = −0.17) after controlling for forest stage and time since disturbance. Our results indicate well pad impacts can be long lasting and may remain for decades or more post reclamation, potentially arresting their recovery trajectory.

## 1. Introduction

Canada's boreal forest is maintained by a wide variety of natural disturbances (Bergeron et al., 1998; Chen et al. 2016; Dhar et al., 2016; Moroni, 2006; Yeboah et al., 2015), which are increasingly conflated with anthropogenic disturbances including clearcutting, construction of roads, seismic exploration, and oil and natural gas (O&NG) development, resulting in habitat loss and shifts in historical range of boreal forest, outside their long-term natural ranges of variability (Pasher et al., 2013). Collectively there is ~24 million ha of anthropogenic impact to Canada's boreal forest (Pasher et al., 2013 – does not include fire disturbance).

O&NG well pad preparation for drilling and production in Canada's boreal forest generally involves clearing forest vegetation (~100 × 100 m area), salvaging any merchantable timber, and dozing or mulching other woody debris. A drilling rig requires a stable and level foundation for operation, and thus usually requires full removal of surface soil and leveling of subsurface soils. Surface stripping removes the native seedbank and soil, leaving the site in a bare ecological successional state. Allred et al. (2015) estimated that vegetation removal by O&NG development from 2000 to 2012 reduced net primary production by ~4.5 Tg of carbon across central North America. O&NG disturbance has generated unique habitats and landscapes with spatial patterns outside that of the historical analogue (Pickell et al., 2015). The intensity, frequency, and uniqueness of O&NG disturbance and its relative impacts on the canopy, forest floor, and associated below-ground properties and processes may play important roles in determining the structure and function of the disturbed ecosystem

* Corresponding author.
*E-mail address:* lupardus@ualberta.ca (R.C. Lupardus).

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(Rowland et al., 2009).

Although the reclamation process varies by location, it is typically composed of five steps: i) remove structures, ii) remove contamination (remediate), iii) replace salvaged soil, iv) recontour, and v) revegetate (ESRD, 2013). After these steps are completed and the well pad has met legislated requirements an operator can apply for a reclamation certificate. In Alberta, Canada, reclamation certification requires that soil, vegetation, and hydrology are returned to an 'equivalent land capability' (ELC; Bott et al. 2016; Powter et al., 2012), – the ability of the land to support various land uses after conservation, is similar to the ability that existed prior to an activity, but not necessarily identical (ESRD, 2013; Government of Alberta, 1995). Of the ~400,000 wells drilled between 1963 and 2014 in Alberta, ~65,000 reclamation certificates and ~35,000 exemptions were issued (Supplemental Fig. 1; Alberta Energy Regulator, 2018).

Changing O&NG practices and reclamation criteria have likely led to improvements in reclamation outcomes. Early reclamation efforts were focused on revegetation with agronomic species to prevent soil erosion, rather than consideration for use of native plant species and reestablishment of tree cover on forested well pads. The use of local soils and plants was not a reclamation priority and materials varied greatly. These factors have created variable soil qualities and plant communities, with reclaimed sites on different trajectories of recovery, making interpretation of monitoring and management exceedingly difficult (Frerichs et al., 2017; Stuble et al., 2017). Although practices have recently changed to consider use of native plant species and local soils, and reestablishment of trees, older reclamation efforts may have unintentionally promoted the succession of non-native herbaceous vegetation (Powter et al., 2012). These agronomic grasses have potentially inhibited growth of both coniferous and deciduous tree species (Bailey and Gupta, 1973; Eis, 1981; Bedford et al., 2000; Cole et al., 2003; Kokkonen et al., 2018), leaving long-term legacy effects, or lags in ecosystem response (Bürgi et al., 2017). The rate of well pad ecological recovery is not currently monitored and many sites might not be recovering and instead might be in a state of arrested succession (i.e., halted reestablishment of late-successional plant communities, which are similar to older reference sites). Knowledge of this rate of recovery (or lack thereof) is essential for accurate land change estimates, forecasting successional development and ecosystem resilience, and management of human footprint in forested regions.

Overall, the long-term outcomes of high-severity disturbance associated with energy extraction and subsequent reclamation efforts on ecosystem properties and processes remain uncertain. It is not yet known if or when reclaimed well pads will return to plant communities and successional trajectories that are representative of those following other high-severity disturbances in forests, such as fire (Pyne, 2008), insect attack (Dhar et al., 2016) or forestry (Hynes and Germida, 2013; Schmidt et al., 1996). The overall goal of this study was to quantify ecological recovery of certified reclaimed O&NG well pads in forested areas. We estimated that full ecological recovery could be expected when the biological, physical, and chemical properties of the soil, vegetation, and decaying material on reclaimed well pads were similar to (or moving towards) the properties of undisturbed reference areas (Ruiz-Jaen and Aide 2005, Shackelford et al. 2013). The main objectives of this study were to: i) assess successional recovery of vegetation and soils on decommissioned and reclaimed well pads and compare it with reference undisturbed forest of varied successional stages, and ii) identify which above- and below-ground properties were influenced by reclamation and are thus useful ecological indicators of ecological recovery towards forest.

## 2. Materials and methods

### 2.1. Study region

The study area was located in the Central Mixedwood and Lower Foothills Natural Subregions in central Alberta (The Natural Regions Committee, 2006). These forested areas are dominated by mosaics of aspen (*Populus tremuloides*), white spruce (*Picea glauca*), and coniferous/deciduous mixedwood (aspen and white spruce) forest on uplands, with extensive areas of jack pine (*Pinus banksiana*) and lodgepole pine (*Pinus contorta*) stands on coarse soils. At higher elevations, lodgepole pine–jack pine hybrids occur as pure stands or with aspen. Typical soils are moderately fine textured gray luvisols and gleyed subgroups. Grasslands are very rare, occurring only as patches in jack pine or black spruce (*Picea mariana*) forest on dry, coarse, well-drained soils. Stand age in this region is generally younger than 100–120 years old reflecting the regional disturbance regime of relatively frequent stand-initiating wildfire (Corns et al., 2005). Significant aspen and conifer harvesting occur throughout the Natural Subregions for pulp and softwood production. On the 270 sampled plots, there were 27 ecological site types represented based on soil characteristics, soil nutrients, moisture status, and vegetation structural stage. The most common ecological site classification for the reclaimed sites was NT7f (non-treed sites recently disturbed by humans) and the most frequent classification for the reference sites was RG6b (Dogwood/Fern/Feather Moss with poplar overstory; McIntosh et al., 2019).

In 2014, we implemented a reclamation protocol (McIntosh et al., 2019) to sample clustered study sites that included both certified reclaimed well pads and adjacent reference areas (Fig. 1). These study sites contained reclaimed well pads, ranging from 7 to 48 years post-certification, that were selected from a larger pool of well pads that met study requirements (i.e., all certified reclaimed well pads located in upland forest, no additional active well pad directly adjacent, public land, within ~1.5 h of Slave Lake or Fox Creek field bases, accessible on foot (maximum ~1 km from nearest road)).

### 2.2. Sampling design and site selection

A single study site included the ~100 × 100 m (1 ha) reclaimed wellsite footprint (well pad), and adjacent reference quadrants (cumulatively 1 ha) without a human disturbance (reference site). We used systematically-located sampling points (see McIntosh et al., 2019), except when we purposely relocated sampled reference quadrants to avoid areas impacted by anthropogenic disturbance (e.g., road, pipeline). While accounting for similarity of sites within the same study location (clustering effect) we marginally compared (i.e., analyzed as a group) reference sites against reclaimed well pads and assessed their ecological recovery.

### 2.3. Data collection for ecological indicators of recovery

We characterized soil and vegetation recovery trajectories at reclaimed forested lands (as well as cultivated and grassland – data not presented here) using physical, chemical, and biological indicators. Our protocols attempted to prioritize evidence-based reclamation (Cooke et al., 2018) along with balancing the costs and benefits to monitor ecological recovery (Richardson and Lefroy, 2016). We selected criteria and developed protocols considering method sensitivity, ease of use, cost, sampling season timing, and existing non-destructive methodologies. Supplemental Table 1 describes the suite of properties (potential ecological indicators) we measured. Soils were sampled in four depth increments of 0–15 cm, 15.1–30 cm, 30.1–60 cm, and 60.1–100 cm, although bulk density was only sampled from 0 to 30 cm. Depth zones were used instead of horizons to ensure consistency of sampling and also because horizon boundaries can be difficult to identify in reclaimed soil profiles (Pennock and van Kessel, 1997). We included the organic surface layer in our sample if it was present. See McIntosh et al. (2019) for detailed data collection protocols.

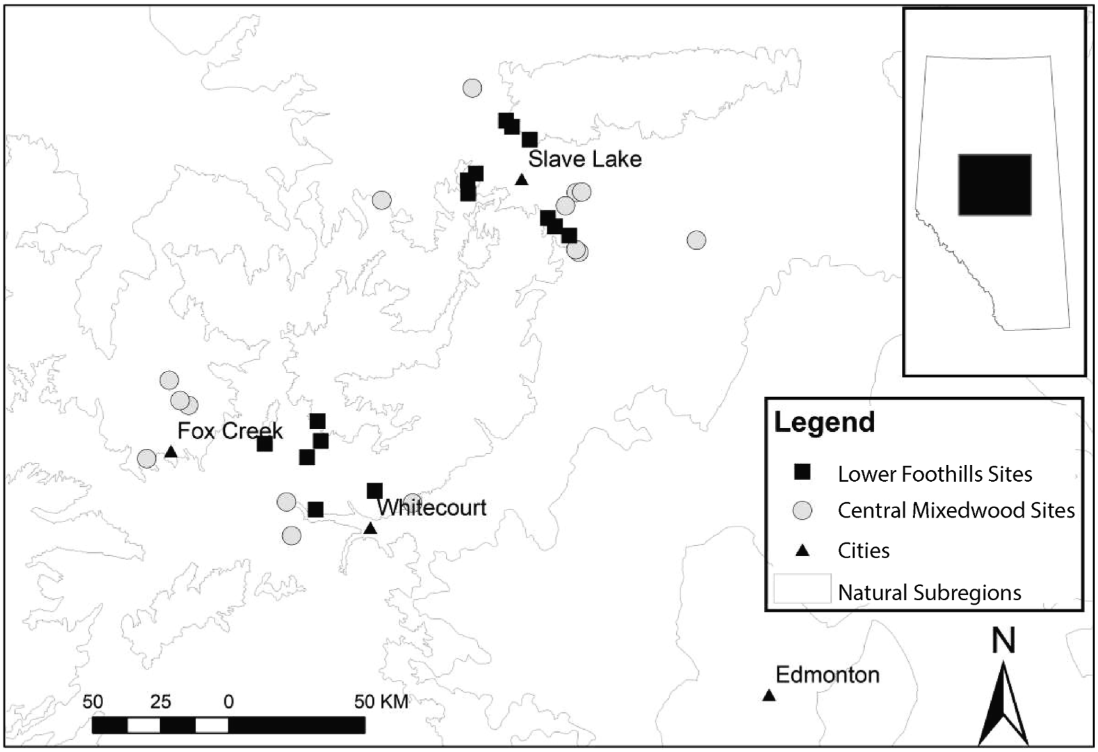


**Fig. 1.** Certified reclaimed oil and natural gas production well pads in Alberta's upland forested lands, including 30 study locations, each with a reclaimed well pad and adjacent reference site, in the Central Mixedwood Natural Subregion (n = 15) and the Lower Foothills Natural Subregion (n = 15). The latitude of sites ranged from 54.1 to 55.6 decimal degrees (N) and longitude ranged from 113.8 to 116.9 (W). Elevations ranged from 609 to 1031 m.

### *2.4. Statistical analysis*

#### *2.4.1. Univariate analysis and summary statistics*

We analyzed 62 variables (Table 1) and grouped them into four data matrices: i) plant diversity properties, ii) soil properties, iii) abiotic properties, and iv) vegetation properties, and had a fifth data matrix of plant species abundances. After preliminary analysis using four soil layer depths (range 0–100 cm), we decided to focus only on the top layer of soil (0–15 cm) for soil multivariate analysis, as the top layer had the greatest interaction with plant communities (although we report summary values for all four depths; Table 1). The plant diversity matrix was composed of introduced cover, native cover, introduced richness, noxious presence, native richness, total richness (S), Shannon diversity (H), Pielou's eveness (J), Simpson diversity (D), and inverse Simpson (D') (Magurran, 2013; Supplemental Table 1). To reduce noise in the vegetation data matrix, we removed all species that only occurred in a single plot.

We created a site matrix composed of seven grouping variables for each site sampled: i) forest type (deciduous, mixedwood, coniferous), ii) forest stage (clearcut within the decade, young forest, mature forest, grassland, burned within the decade), iii) study site (1–30), iv) site type (reclaimed, reference), v) time since last disturbance (short: ST 7–34 years, long: LT 35–48 years), vi) natural subregion (Central Mixedwood, Lower Foothills). Supplemental Table 1 includes a full description of these variables. We extracted data used to categorize reference site quadrants by forest stage, from the Alberta Vegetation Inventory Extended (AVIE) database (Alberta Environment and Parks, 2017). We estimated reference forest stage using fire, clearcut, and wind throw data as well as other historical AVIE records. For young and mature forest we calculated forest stage, describing the general age of a forested stand and the general succession status of the overstory, in each quadrant using age, forest cover, and Natural Subregion. We used the Food and Agriculture Organization of the United Nations definition of a forest, defined as land spanning more than 0.5 ha with trees taller than 5 m with canopy cover of more than 10% (Natural Resources Canada, 2017). Not all certified reclaimed sites were considered forested lands, but were instead classified as grassland. For those sites considered forested, time (since reclamation) was determined using the certification date, unless the difference between certification date and abandonment date or final drill date was greater than 10 years. A number of well pads were naturally recovering for more than 30 years before operators applied for and received reclamation certification. In these instances, we used the abandonment date or final date of industrial activity as the starting point for recovery. We estimated forest stage of reclaimed well pads using AVIE methods and checked all site classifications against quadrant photos and collected data (tree height and diameter to estimate age). Time since most recent disturbance (e.g., crown fire, clearcut, oil and natural gas reclamation or abandonment) was assigned using natural breaks in the dataset (time 1 (7–24 years), time 2 (25–34 years), and time 3 (35–48 years). Vegetation community analysis indicated that groups 1 and 2 were not statistically different and so we combined these two groups for a final analysis with a shorter recovery time, ST (7–34 years) and a longer recovery time, LT (35–48 years). We could not account for variance due to individual operators' drilling practices (e.g., equipment size, winter/summer drill) or reclamation efforts (e.g., reseeding methods, source of seeds, tree planting, mechanical/chemical weed treatment, topsoil removal and replacement methods). These historical efforts were not recorded in accessible log books and could not be accounted for in the model, thus they were lumped into site-level treatment effects.

We avoided the use of analysis of variance (ANOVA) and t-tests, as the dataset included multiple response variables (multiple comparisons) collected in clusters (study site level), which, post transformation, did not all meet test assumptions (i.e., normal distribution, homogeneity of variance, or independence of observations on sites; Anderson, 2001; Hurlbert, 1984; Thiese et al., 2015). Instead we used multivariate, marginal statistical analysis with cluster-specific or population-average inference as an alternative to typical univariate

**Table 1**
Variables used to analyze 30 certified reclaimed well pads and 30 reference sites in Alberta's Central Mixedwood and Foothills Natural Subregions. Means, standard errors of the means, medians, bootstrapped standard errors and 95% confidence intervals of medians are shown for reference and reclaimed sites.

| | | Reclaimed | | | | | | Reference | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Variable | Mean | SE | Median | SE | 2.50% | 97.50% | Mean | SE | Median | SE | 2.50% | 97.50% |
| Soil | LFH Depth (cm)* | 2.7 | 0.1 | 2.7 | 0.1 | 2.4 | 2.9 | 7.8 | 0.6 | 6.3 | 0.6 | 5.1 | 7.6 |
| | BD1 (0–15 cm; g/cm$^3$)* | 0.86 | 0.02 | 0.90 | 0.05 | 0.80 | 0.99 | 0.49 | 0.02 | 0.49 | 0.03 | 0.43 | 0.56 |
| | BD2 (15.1–30 cm; g/cm$^3$)* | 1.20 | 0.02 | 1.21 | 0.04 | 1.13 | 1.30 | 1.01 | 0.03 | 1.08 | 0.05 | 0.97 | 1.19 |
| | pH1 (0–15 cm)* | 7.01 | 0.10 | 7.23 | 0.33 | 6.54 | 7.91 | 5.90 | 0.10 | 5.77 | 0.38 | 5.00 | 6.55 |
| | pH2 (15.1–30 cm)* | 7.30 | 0.10 | 7.54 | 0.33 | 6.86 | 8.21 | 6.04 | 0.09 | 6.04 | 0.15 | 5.74 | 6.34 |
| | pH3 (30.1–60 cm)* | 7.30 | 0.11 | 7.68 | 0.34 | 6.99 | 8.38 | 6.39 | 0.10 | 6.25 | 0.18 | 5.89 | 6.62 |
| | pH4 (60.1–100 cm)* | 7.26 | 0.11 | 7.40 | 0.21 | 6.97 | 7.83 | 6.69 | 0.11 | 6.38 | 0.38 | 5.61 | 7.15 |
| | EC1 (0–15 cm; μS/cm$^3$)* | 561 | 29 | 553 | 74 | 401 | 705 | 490 | 23 | 443 | 46 | 349 | 537 |
| | EC2 (15.1–30 cm; μS/cm$^3$)* | 340 | 20 | 342 | 46 | 249 | 435 | 245 | 17 | 234 | 36 | 160 | 307 |
| | EC3 (30.1–60 cm; μS/cm$^3$)* | 271 | 34 | 251 | 40 | 169 | 332 | 163 | 12 | 136 | 16 | 103 | 169 |
| | EC4 (60.1–100 cm; μS/cm$^3$) | 302 | 38 | 191 | 35 | 118 | 263 | 174 | 16 | 123 | 17 | 89 | 157 |
| | TN1 (0–15 cm; %)* | 0.18 | 0.01 | 0.14 | 0.02 | 0.10 | 0.17 | 0.45 | 0.04 | 0.34 | 0.04 | 0.26 | 0.43 |
| | TN2 (15.1–30 cm; %) | 0.10 | 0.01 | 0.08 | 0.00 | 0.07 | 0.09 | 0.25 | 0.03 | 0.11 | 0.04 | 0.03 | 0.19 |
| | TN3 (30.1–60 cm; %) | 0.08 | 0.01 | 0.07 | 0.01 | 0.05 | 0.09 | 0.15 | 0.03 | 0.06 | 0.01 | 0.04 | 0.08 |
| | TN4 (60.1–100 cm; %) | 0.09 | 0.01 | 0.06 | 0.01 | 0.04 | 0.08 | 0.13 | 0.03 | 0.05 | 0.01 | 0.03 | 0.07 |
| | TOC1 (0–15 cm; %)* | 2.96 | 0.32 | 2.00 | 0.27 | 1.45 | 2.56 | 9.78 | 1.09 | 5.76 | 0.82 | 4.08 | 7.45 |
| | TOC2 (15.1–30 cm; %) | 1.54 | 0.17 | 1.03 | 0.11 | 0.81 | 1.25 | 5.28 | 0.97 | 1.80 | 0.59 | 0.59 | 3.00 |
| | TOC3 (30.1–60 cm; %) | 1.53 | 0.47 | 0.85 | 0.17 | 0.50 | 1.20 | 3.07 | 0.79 | 0.82 | 0.08 | 0.66 | 0.99 |
| | TOC4 (60.1–100 cm; %) | 1.70 | 0.41 | 0.74 | 0.14 | 0.45 | 1.04 | 2.52 | 0.75 | 0.61 | 0.14 | 0.33 | 0.90 |
| | C:N1 (0–15 cm)* | 15.9 | 0.3 | 16.0 | 0.3 | 15.5 | 16.6 | 19.9 | 0.5 | 19.1 | 0.8 | 17.6 | 20.7 |
| | C:N2 (15.1–30 cm) | 14.7 | 0.3 | 14.5 | 0.4 | 13.7 | 15.3 | 16.1 | 0.5 | 16.0 | 0.9 | 14.1 | 17.8 |
| | C:N3 (30.1–60 cm) | 14.0 | 0.5 | 13.3 | 0.6 | 13.2 | 15.8 | 13.8 | 0.5 | 14.5 | 0.7 | 11.9 | 14.7 |
| | C:N4 (60.1–100 cm) | 14.0 | 0.6 | 13.4 | 0.6 | 13.1 | 15.5 | 13.4 | 0.5 | 14.3 | 0.4 | 12.5 | 14.3 |
| Vegetation | BA Live (m$^2$/ha)* | 5.47 | 0.98 | 2.62 | 1.69 | −0.84 | 6.07 | 23.28 | 1.52 | 22.16 | 3.12 | 15.78 | 28.53 |
| | BA Dead (m$^2$/ha)* | 0.32 | 0.08 | 0 | 0.04 | −0.07 | 0.07 | 4.89 | 0.74 | 3.20 | 0.92 | 1.31 | 5.09 |
| | BA Live Coniferous (m$^2$/ha) | 1.4 | 0.6 | 0.2 | 0.2 | −0.2 | 0.5 | 8.8 | 1.8 | 5.1 | 2.6 | −0.1 | 10.4 |
| | BA Live Deciduous (m$^2$/ha)* | 4.2 | 1.1 | 0.3 | 1.2 | −2.1 | 2.7 | 14.8 | 2.0 | 13.7 | 3.3 | 6.8 | 20.5 |
| | BA Dead Snags (m$^2$/ha)* | 0.4 | 0.1 | 0 | 0 | −0.1 | 0.1 | 3.0 | 0.5 | 2.1 | 0.7 | 0.6 | 3.5 |
| | TPH Live (< 7 cm dbh) | 1970 | 429 | 784 | 377 | 13 | 1554 | 3230 | 553 | 1250 | 667 | −114 | 2614 |
| | TPH Dead (< 7 cm dbh)* | 347 | 147 | 0 | 66 | −135 | 135 | 477 | 91 | 300 | 102 | 91 | 509 |
| | TPH Live (7–25 cm dbh)* | 361 | 73 | 125 | 158 | 502 | 1148 | 973 | 89 | 825 | 158 | 502 | 1148 |
| | TPH Dead (7–25 cm dbh)* | 30 | 16 | 0 | 1 | −1 | 1 | 144 | 24 | 88 | 39 | 7 | 168 |
| | TPH Live (> 25 cm dbh)* | 8 | 2 | 0 | 1 | −1 | 1 | 68 | 8 | 58 | 19 | 20 | 96 |
| | TPH Dead (> 25 cm dbh) | 0 | 0 | 0 | 0 | 0 | 0 | 19 | 3 | 8 | 4 | 0 | 16 |
| | Total TPH Live* | 2339 | 463 | 1367 | 605 | 129 | 2604 | 4270 | 561 | 2831 | 588 | 1628 | 4034 |
| | Total TPH Dead* | 377 | 149 | 0 | 74 | −151 | 151 | 640 | 103 | 433 | 129 | 168 | 698 |
| | Total DWD (Mg/ha)* | 1.5 | 0.6 | 0.1 | 0.1 | −0.1 | 0.4 | 22.0 | 2.9 | 21.7 | 3.0 | 15.5 | 27.8 |
| | CWD (Mg/ha)* | 1.1 | 0.5 | 0 | 0 | −0.1 | 0.1 | 21.1 | 2.6 | 18.6 | 3.3 | 11.8 | 25.3 |
| | CWDs (Mg/ha)* | 1.0 | 0.5 | 0 | 0 | 0 | 0 | 19.4 | 2.6 | 15.9 | 3.5 | 8.7 | 23.1 |
| | SWD (Mg/ha)* | 0.4 | 0.1 | 0.1 | 0.1 | 0.0 | 0.2 | 2.2 | 0.3 | 2.0 | 0.1 | 1.7 | 2.3 |
| | Shrubs < 0.5 m tall (%)* | 4.2 | 0.8 | 2.8 | 1.1 | 0.6 | 5.0 | 18.8 | 1.3 | 18.8 | 2.6 | 13.5 | 24.0 |
| | Shrubs 0.5–2 m tall (%)* | 7.7 | 1.2 | 6.3 | 1.9 | 2.5 | 10.1 | 18.0 | 1.8 | 18.8 | 2.9 | 12.8 | 24.8 |
| | Shrubs > 2 m tall (%) | 6.5 | 1.4 | 3.5 | 1.5 | 0.5 | 6.5 | 5.8 | 1.4 | 1.3 | 1.2 | −1.2 | 3.7 |
| | Grass Cover (%)* | 47.1 | 2.7 | 45.8 | 8.2 | 29.1 | 62.4 | 11.9 | 1.8 | 5.7 | 2.9 | −0.2 | 11.5 |
| | Moss Cover (%) | 1.4 | 0.3 | 0.6 | 0.5 | −0.3 | 1.6 | 16.6 | 2.4 | 7.1 | 3.2 | 0.6 | 13.6 |
| | Lichen Cover (%)* | 0.1 | 0 | 0 | 0 | 0 | 0 | 0.4 | 0.1 | 0.1 | 0 | 0.1 | 0.1 |
| | Canopy Cover (%)* | 29.4 | 2.9 | 23.5 | 8.4 | 6.2 | 40.8 | 72.1 | 2.4 | 77.5 | 3.8 | 69.8 | 85.3 |
| | Forb Cover (%)* | 31.7 | 2.2 | 28.8 | 3.2 | 22.2 | 35.3 | 24.2 | 1.6 | 20.6 | 4.0 | 12.5 | 28.7 |
| Diversity | Introduced Cover (%)* | 5.6 | 0.6 | 23.8 | 3.6 | 16.5 | 31.0 | 0.6 | 0.3 | 0.0 | 0.1 | −0.2 | 0.2 |
| | Native Cover (%)* | 74.6 | 2.5 | 76.3 | 3.5 | 69.1 | 83.4 | 99.1 | 0.3 | 100.0 | 0.1 | 99.8 | 100.2 |
| | Introduced Richness* | 11.3 | 1.0 | 11.5 | 1.9 | 7.6 | 15.4 | 0.8 | 0.2 | 0.0 | 0.2 | −0.3 | 0.3 |
| | Noxious Presence* | 0.6 | 0.1 | 1.0 | 0.3 | 0.4 | 1.6 | 0.2 | 0.1 | 0 | 0 | 0 | 0 |
| | Native Richness* | 20.6 | 2.0 | 22.0 | 1.2 | 19.5 | 24.5 | 34.3 | 1.1 | 33.5 | 1.4 | 30.7 | 36.3 |
| | Richness (S) | 31.9 | 1.1 | 34.0 | 1.3 | 31.4 | 36.6 | 35.0 | 1.1 | 33.5 | 1.5 | 30.5 | 36.5 |
| | Shannon Diversity (H)* | 2.3 | 0.1 | 2.3 | 0.1 | 2.2 | 2.5 | 2.5 | 0 | 2.5 | 0.1 | 2.4 | 2.7 |
| | Simpson Diversity (D) | 0.9 | 0 | 0.9 | 0 | 0.9 | 0.9 | 0.9 | 0 | 0.9 | 0 | 0.9 | 0.9 |
| | Inverse Simpson (D') | 7.7 | 0.4 | 7.6 | 0.6 | 6.3 | 8.9 | 8.7 | 0.5 | 8.6 | 0.8 | 6.9 | 10.3 |
| | Pielou Eveness (J) | 0.8 | 0 | 0.8 | 0 | 0.8 | 0.8 | 0.8 | 0 | 0.8 | 0 | 0.8 | 0.9 |
| Abiotic Cover | Wood Cover (%)* | 1.8 | 0.5 | 0.1 | 0.3 | −0.6 | 0.8 | 12.0 | 0.9 | 12.5 | 1.4 | 9.6 | 15.4 |
| | Litter Cover (%) | 57.4 | 4.0 | 45.0 | 19.0 | 6.1 | 83.9 | 56.9 | 3.1 | 59.4 | 9.1 | 40.7 | 78.1 |
| | Water Cover (%) | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Bare Ground Cover (%)* | 0.6 | 0.2 | 0.1 | 0.1 | −0.2 | 0.3 | 0.3 | 0.1 | 0 | 0 | 0 | 0 |

*Notes:* BD is Bulk Density, TN is Total Nitrogen, TOC is Total Organic Carbon, C:N is Carbon to Nitrogen Ratio, BA is basal area, TPH is Trees per hectare, DWD is Downed Woody Debris, CWD is Coarse Woody Debris (CWDs is soft), SWD is Small Woody Debris. Median, SE and CI values calculated using a nonparametric bootstrap method. Asterisk indicates reclamation values are outside the 95% CI for the reference sites. Fungi, rock, and animal cover were 0% and were removed from the table.

analyses (Heagerty and Zeger, 2000; Inan, 2015; Nielsen et al., 2007), which can only predict a single response variable and cannot account for correlation between response variables. We calculated summary statistics for sites, including calculations of standard errors (SEs), means, medians, and 95% confidence intervals (CIs) using a nonparametric bootstrap method. For all statistical analyses, we used the software environment R (version 3.4.3; R Core Team, 2018).

#### 2.4.2. Multi-response permutation procedures and permutational analysis of variance

We derived a compositional distance matrix among sites using the Bray-Curtis dissimilarity index (Legendre and Legendre, 2012). We used multi-response permutation procedures (MRPP) with 1000 permutations to determine if the community structure of different sites could be explained by categorical grouping variables including: *forest type* (coniferous, mixedwood, deciduous), *forest stage* (clearcut, young forest, mature forest, grassland, burned), *site type* (reclaimed, reference), *natural subregion* (Central Mixedwood, Lower Foothills), *time since last disturbance* (ST, LT), and all combinations of these groups (McCune et al., 2002; Berry et al., 2016). We also ran a permutational analysis of variance (PERMANOVA) using the R package 'Adonis' (R 'vegan' package) for comparison against MRPP to validate our findings (Anderson, 2001). The *p*-values from multiple pairwise comparisons were adjusted (Padj) using the sequential Holm correction.

#### 2.4.3. Non-metric multidimensional scaling (NMDS) ordination

We used NMDS to identify vegetation clusters related to the significant grouping categories and to examine correlations between plant community and soil variables, diversity variables, and other abiotic variables (Austin and Pillar, 2013; Thessler et al., 2005). NMDS is free from assumptions of normality, dimensionality, linearity, and the shape of species-response curves to gradients (Kruskal, 1964) and is the most widely used ordination method for biological community analysis (Borcard et al. 2018; Legendre & Legendre, 2012; McCune et al., 2002; Oksanen 2011). With NMDS ordination space-sample relationships are based on ranked dissimilarity in compositional space (Legendre and Legendre, 2012). Final scores are relative indicators of n-dimensional compositional dissimilarity (where $n + 1$ represents the number of total species in a species-by site data matrix of relative abundances) in a k-dimensional ordination space (where $k < n-1$). We used 'metaMDS' (R 'vegan' package) to execute multiple runs and looked for stable configurations (Oksanen et al., 2018). MetaMDS applies a square root transformation, scaling (i.e., centering, PC rotation, halfchange scaling), and uses expanded scores based on Wisconsin double standardization. The solution with the lowest dissimilarity between ordination and Bray-Curtis distances was selected as the final model (converged after 69 tries, stress = 0.2).

#### 2.4.4. Indicator species analysis

We performed an Indicator Species Analysis (ISA; Iqbal et al., 2018; Khan et al., 2016; Jiang et al., 2018; Xiao et al., 2017) using the 'multipatt' function in R package 'indicspecies' (De Cáceres and Legendre, 2009), using 'strassoc' and function "A.g" (accounting for unequal sample size) to identify species affiliations and function 'combinespecies' to identify pairs of species affiliations to a particular site type group (reclaimed vs reference; De Cáceres et al., 2012). Indices included aspecificity (A) and sensitivity (B) of species as indicators of group. Quantity A is the probability of site association with the site-group combination when the species was present at that site (Murtaugh, 1996). Quantity B indicates how frequently the species was found at sites of the site-group combination. ISA calculates indicator values for each species based on species abundance scores and the proportional frequency of all species in a particular group. The coefficient of determination ($R^2$) assesses the positive or negative association of species for environmental conditions fundamental to sites belonging to the site-group combination, compared to all other sites. Species occurring in only one site were omitted. Only species with $p < 0.001$ and $R^2 > 0.7$ were considered strong indicator species for a group and all species forming joint pairs had to occur in the site to use the combination as an indicator.

#### 2.4.5. Multivariate joint generalized estimating equation (JGEE)

We were not interested in targeting individual site reclamation success or failure, but rather the overall (marginal) effects of reclamation. A practical and commonly employed marginal modeling approach in a correlated data analysis framework is to utilize generalized estimating equations (GEEs; Zeger and Liang, 1986). Unlike conditional models, which may include high-dimensional random effect terms, GEEs offer a computationally non-intensive parameter estimation algorithm, which provides marginal, population averaged inference for clustered (non-independent) data (Colford et al., 2009). Our data required joint modeling of multiple response variables using joint GEEs (JGEEs; Lipsitz et al., 2009), an extension of traditional GEEs to multiple responses, implemented in R package 'JGEE' (Inan, 2015). We could not separate effects of reclamation activities from other O&NG activities such as drilling, cleaning, and production; instead we refer to them cumulatively as reclamation effects, since this is the last phase of the well lifecycle. We analyzed the effect of reclamation on multiple gaussian and binary marginal mixed response variables. We selected this model due to its ability to handle a clustered sampling design. To account for the association among the outcomes, we used the autoregressive correlation structure and computed robust standard errors on the coefficient estimates (Lipsitz et al., 2009; Inan and Yucel, 2017). We also used variance inflation factors (VIFs) and a correlation matrix to determine multi-collinearity. The fit of a JGEE model cannot be assessed without the use of a second, comparable dataset, to which we did not have access; however, estimates have been assessed using residuals and summary measures. We considered each study site a cluster, with effects interpreted directly on continuous variables and as the rate ratio of the binary response variable with reverse log (i.e., exponentiated) effect. Instead of generating all possible effects of the reclamation treatment, we determined effects using those variables (vectors) correlated with axes ($R^2 > 0.4$ and $p < 0.001$) in the previously mentioned NMDS ordination, as well as a binary variable representing noxious species presence or absence. Marginal (site-averaged) effects are an absolute change in the probability of an outcome while holding all other variables constant. If negative, this indicated a decrease in probability. Estimate standard errors (SEs) were determined using the JGEE package sandwich estimator, which resamples clusters instead of individual observations in order to preserve the dependence within each cluster (Inan, 2015)

## 3. Results

### 3.1. Univariate statistics

We used the range of values measured at reference sites to define "healthy" conditions, for comparison with the range of values measured at well pads (Table 1, Supplemental Fig. 2). For many variables, the range of values measured on reclaimed well pads were outside the bootstrapped CIs of reference medians, indicating they are outside the range for healthy forest (Table 1). The range of values for several environmental variables including LFH depth, grass cover, lichen cover, woody debris biomass and cover, and canopy cover were below the CIs for reference sites, whilst bare ground was above the reference CI. Of the soil variables, bulk density, electrical conductivity, and pH were above the reference CIs in most soil layers of reclaimed sites, whereas total nitrogen, total organic carbon, and the carbon to nitrogen ratio were all below the healthy range for reference site soil (0–15 cm). Vegetation parameters were also very different on reclaimed sites with tree basal area, trees per ha (live and dead), all downed woody materials (large, small, coarse, and soft), and shrub cover (< 2 m height) below the reference sites range. The distribution of introduced, noxious, and native plant species were also profoundly different when comparing reference to reclaimed, although diversity indices generally were not (i.e., S, D, D', E; Table 1). We identified 10 bryophyte species, 3 lichens, and 103 vascular species, including 6 trees, 2 clubmosses, 2 ferns, 52 forbs, 16 graminoids, and 25 shrubs, on our study sites. Of these identified species, 19 were introduced, four of which were also noxious species (Supplemental Table 2). Introduced and noxious species were more prevalent on reclaimed sites with corresponding, reduced native species cover. Sixty percent of reclaimed well pads had at least one noxious species present, compared to only 20% on the reference sites. There were only 5 (17%) reference sites containing a single noxious species and 18 (60%) reclaimed sites containing one or two noxious species. Although reference and reclaimed sites had similar vegetation species richness (S), introduced species median cover on reference sites was lower (0%) than on the reclaimed sites (24%; Table 1).

### 3.2. Community structure

The MRPP analysis determined *site type* ($A = 0.09$, $p < 0.001$), *forest stage* ($A = 0.12$, $p < 0.001$), and *forest type* ($A = 0.07$, $p < 0.001$) were significant grouping variables; however, the effect sizes were relatively small. *Natural region* ($A = 0.005$, $p = 0.07$) and *time since disturbance* ($A = 0.005$, $p = 0.06$), were not significant grouping variables, nor did they have large effect sizes. PERMANOVAs indicated there were no significant interaction effects between the grouping variables and that *site type* ($F = 14.8$, $R^2 = 0.18$, $p = 0.001$), *time* ($F = 2.1$, $R^2 = 0.02$, $p = 0.021$), *forest type* ($F = 2.6$, $R^2 = 0.09$, $p = 0.001$), and *forest stage* ($F = 1.8$, $R^2 = 0.09$, $p = 0.001$) were significant (alpha = 0.05), mostly confirming MRPP results. Together, these results indicate that differences in *site type*, *forest type*, *forest stage*, and *time since last disturbance*, are significant independently, yet do not cluster tightly based on vegetation community. Multivariate analysis using NMDS was required to further examine differences in plant community composition between combinations of groups.

Three combinations of groups, including *forest stage* × *forest type*, *forest stage* × *forest type* × *time since last disturbance*, and *site type* × *forest stage* × *forest type* × *time since last disturbance* were used as grouping categories for NMDS. Although time was only a significant grouping variable for PERMANOVA and not MRPP, the combination of *site type* × *forest stage* × *forest type* × *time since last disturbance* explained community structure better than any other categorical variable combination, based on results of both MRPP and NMDS. Moreover, it was important to control for *time since last disturbance* to identify older sites that were not on a positive trajectory of recovery. Post-hoc MRPP contrasts indicated differences between 19 out of 49 comparisons (Table 2). None of the grouping variables explained a large amount of variance, likely due to similarities in reclamation effects across forest types and forest stages. Site-specific reclamation practices, such as type of seed used in re-vegetation, quantity and quality of top soil used, active vs. passive re-vegetation, as well as site specific conditions during reclamation, such as climate, precipitation, and traffic density, likely accounted for the greatest variance between sites, however these variables could not be determined nor controlled.

The final NMDS 2-dimensional solution had a final stress of 0.2, final instability < 0.001, and final $R^2$ of 0.685 (axis 1 = 0.503, axis 2 = 0.182). The ordination with convex hulls (polygons enclosing all sample points in a group) placed all sites into categories (Fig. 2A). A high degree of overlap indicated similarity in community structure between overlapping groups. The only groups without overlap were reference mature coniferous forest and reclaimed young deciduous forest at *time LT*. All other groups overlapped to varying degrees and fit within the broader category, *site type* (reclaimed or reference). Reclaimed sites were contained within grassland, young coniferous, and young deciduous forest, with the exception of two mature mixed forest at *time LT*, which were more similar in community structure to the bunched reference sites. The reference sites were all contained within burned, clearcut, mature deciduous, and mature mixedwood groups. Successional vectors indicated reference and reclaimed sites in the same location (study unit) were often less similar to each other than they were to other locations (Fig. 2B), indicating that location similarity (cluster effect) is potentially weak.

Our ordination revealed that only two of the eight oldest well pads were grouped with the reference sites and recovering to forest (including being high in live basal area, woody debris, shrub cover, TPH, LFH, native richness, and live deciduous trees; Supplemental Table 3). With the exception of the aforementioned sites, reclaimed well pads clustered together, strongly correlated with higher levels of bulk density, grass cover, and introduced spp. richness and cover, with the grassland sites likely in a state of arrested trajectory of recovery. In

**Table 2**
Multi-response permutation procedure (MRPP) pairwise contrasts on similarity in vegetation species composition in Alberta's Central Mixedwood (n = 15) and Lower Foothills (n = 15) Natural Subregions, comparing among reference sites (Ref): burned, clearcut, young (Y), mature (M) coniferous, deciduous, mixedwood, and reclaimed well pads (Rec): Grassland, Y, M coniferous, deciduous, mixedwood, at times ST (7–34 years) and LT (35–48 years) since disturbance.

| Group1 | | Group 2 | *A* | *p* |
|---|---|---|---|---|
| Rec Grassland ST | vs. | Rec Grassland LT | 0.02 | 0.170 |
| Rec Grassland ST | vs. | Rec M Mixedwood LT | 0.04 | 0.016 |
| Rec Grassland ST | vs. | Rec Y Coniferous ST | 0.03 | 0.056 |
| Rec Grassland ST | vs. | Rec Y Deciduous ST | 0.00 | 0.590 |
| Rec Grassland ST | vs. | Ref Burned | 0.06 | 0.001 |
| Rec Grassland ST | vs. | Ref M Deciduous LT | 0.13 | < 0.001 |
| Rec Grassland ST | vs. | Ref M Mixedwood LT | 0.08 | < 0.001 |
| Rec Y Coniferous ST | vs. | Rec Grassland LT | −0.02 | NaN |
| Rec Y Coniferous ST | vs. | Rec M Mixedwood LT | 0.18 | NaN |
| Rec Y Coniferous ST | vs. | Ref M Mixedwood LT | 0.22 | 0.022 |
| Rec Y Deciduous ST | vs. | Rec Grassland LT | 0.00 | 0.399 |
| Rec Y Deciduous ST | vs. | Rec M Mixedwood LT | 0.07 | 0.077 |
| Rec Y Deciduous ST | vs. | Rec Y Coniferous ST | 0.00 | 0.441 |
| Rec Y Deciduous ST | vs. | Ref M Mixedwood LT | 0.13 | 0.009 |
| Rec Y Deciduous LT | vs. | Rec Grassland ST | 0.05 | 0.003 |
| Rec Y Deciduous LT | vs. | Rec Grassland LT | 0.05 | 0.127 |
| Rec Y Deciduous LT | vs. | Rec M Mixedwood LT | 0.06 | 0.069 |
| Rec Y Deciduous LT | vs. | Rec Y Coniferous ST | 0.06 | 0.074 |
| Rec Y Deciduous LT | vs. | Rec Y Deciduous ST | 0.02 | 0.285 |
| Rec Y Deciduous LT | vs. | Ref Burned | 0.10 | 0.011 |
| Rec Y Deciduous LT | vs. | Ref Clearcut | 0.08 | 0.003 |
| Rec Y Deciduous LT | vs. | Ref M Deciduous LT | 0.09 | < 0.001 |
| Rec Y Deciduous LT | vs. | Ref M Mixedwood LT | 0.15 | 0.009 |
| Ref Burned | vs. | Rec Grassland LT | 0.13 | NaN |
| Ref Burned | vs. | Rec M Mixedwood LT | 0.06 | NaN |
| Ref Burned | vs. | Rec Y Coniferous ST | 0.17 | 0.024 |
| Ref Burned | vs. | Rec Y Deciduous ST | 0.08 | 0.030 |
| Ref Clearcut | vs. | Rec Grassland ST | 0.10 | < 0.001 |
| Ref Clearcut | vs. | Rec Grassland LT | 0.12 | 0.006 |
| Ref Clearcut | vs. | Rec M Mixedwood LT | 0.00 | 0.487 |
| Ref Clearcut | vs. | Rec Y Coniferous ST | 0.16 | < 0.001 |
| Ref Clearcut | vs. | Rec Y Deciduous ST | 0.10 | < 0.001 |
| Ref M Coniferous LT | vs. | Rec Grassland ST | 0.14 | < 0.001 |
| Ref M Coniferous LT | vs. | Rec Grassland LT | 0.21 | 0.025 |
| Ref M Coniferous LT | vs. | Rec M Mixedwood LT | 0.17 | 0.031 |
| Ref M Coniferous LT | vs. | Rec Y Coniferous ST | 0.25 | 0.009 |
| Ref M Coniferous LT | vs. | Rec Y Deciduous ST | 0.20 | < 0.001 |
| Ref M Coniferous LT | vs. | Rec Y Deciduous ST | 0.23 | < 0.001 |
| Ref M Deciduous LT | vs. | Rec Grassland LT | 0.07 | 0.001 |
| Ref M Deciduous LT | vs. | Rec M Mixedwood LT | 0.00 | 0.309 |
| Ref M Deciduous LT | vs. | Rec Y Coniferous ST | 0.12 | < 0.001 |
| Ref M Deciduous LT | vs. | Rec Y Deciduous ST | 0.08 | < 0.001 |
| Ref M Mixedwood ST | vs. | Rec Grassland LT | 0.18 | < 0.001 |
| Ref M Mixedwood ST | vs. | Rec Grassland ST | 0.14 | NaN |

*Notes:* NaN = comparisons could not be made as a group contains a single site. Ref vs Ref comparisons were removed for clarity of interpretation. The Holm method was used for family wise adjusted p-values. The distance matrix was Bray-Curtis.

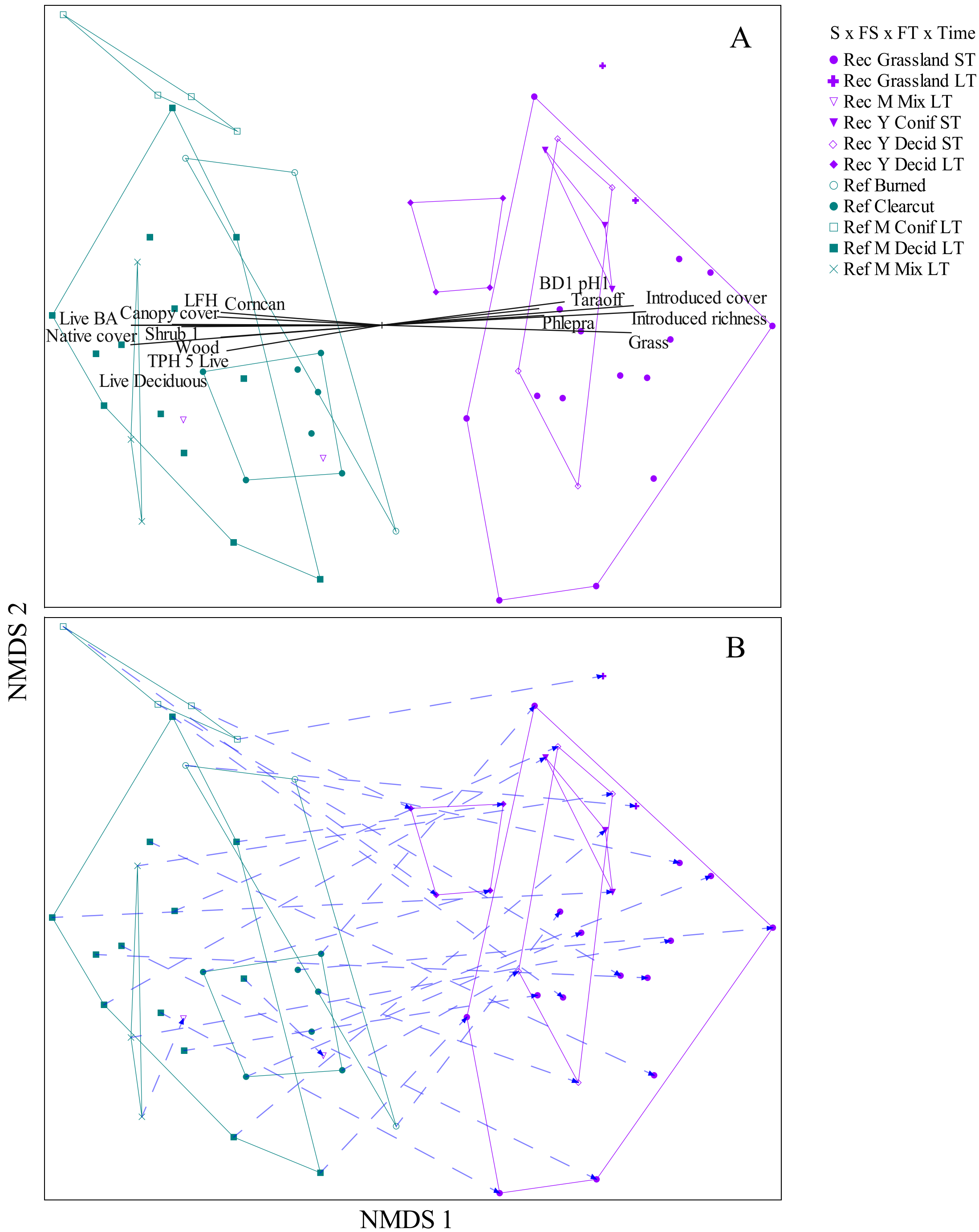


**Fig. 2.** Nonmetric multidimensional scaling ordination of vegetation species composition for S: *site type* (reclaimed vs reference), FS: *forest stage* (mature (M) vs young (Y) forest), FT: *forest type* (grassland, mix: mixedwood, conif: coniferous, decid: deciduous, burned, clearcut), and Time: *time since disturbance* (ST = 7–34 years, LT 35–48 years). A) vectors indicate significant (alpha 0.05) environmental, vegetation, soil, and diversity variables (see Supplemental Table 1 for detailed description of variables). Vector direction and length reflect the strength of correlation with the first two axes. B) Successional vectors join reference to reclaimed sites (arrows indicate direction) from the same location.

contrast to the older sites, some recently reclaimed sites are seemingly in transition, moving towards full recovery, specifically the Rec Y Decid LT and some of the sites within the Rec Grassland ST groups; although the latter grassland sites should be followed closely to ensure a transition back to forested land with time. To provide the reader with a visual illustration of the lack of recovery over time (~10, 20, and 35 years post-disturbance) we have provided several examples in Fig. 3.

### 3.3. Indicator species analysis

Our analyses of the understory vascular plant species data matrix on single species or two species-combinations for reclaimed and reference sites allowed us to find valid indicators for the two site types (Table 3; see Supplemental Table 4 for the complete list of indicators). For the single species ISA, there were 110 total species in the analysis and 32 species associated to a single group, with 19 species strongly correlated with the Reference group and 13 species correlated with the reclaimed group (Supplemental Table 4). For the joint species analysis, there were


**Fig. 3.** Examples of three different certified reclaimed well pads in Alberta's Central Mixedwood and Foothills Natural Subregions covered with non-forested vegetation: A) ~30 years post reclamation B) ~20 years post reclamation, and C) ~10 years post reclamation. Adjacent, undisturbed, forested lands are in the background at each location.

6105 pairs of species and 350 pairs associated with a single group. The final set of indicators for groups ranged in aspecificity (A) from 0.72 to 1 and sensitivity (B) from 0.51 to 0.97 (Table 3; Supplemental Fig. 3). The strongest indicator species for the reference group was for *Cornus canadensis* (A = 0.99, B = 0.97, $R^2$ = 0.98, $p$ = 0.001; native shrub) and for the reclaimed well pads it was *Taraxacum officinale* (A = 0.99, B = 0.89, $R^2$ = 0.943, $p$ = 0.001; introduced forb). *C. canadensis* was strongly correlated with other native forbs and shrubs including *Mitella nuda* (native forb), *Lonicera involucrata* (native shrub), *Viburnum edule* (native shrub), *Rosa acicularis* (native shrub) *Rubus pubescens* (native

**Table 3**
Indicator species analysis for single or two-species combinations for 30 certified reclaimed well pads and 30 reference sites in Alberta's Central Mixedwood (n = 15) and Lower Foothills (n = 15) Natural Subregions.

| Reference Joint Species[a] | $A^b$ | $B^c$ | $R^{2,d}$ | Growth Form |
|---|---|---|---|---|
| *Cornus canadensis* | 0.99 | 0.97 | 0.98 | shrub |
| *Cornus canadensis + Rosa acicularis* | 0.98 | 0.79 | 0.88 | shrubs |
| *Cornus canadensis + Rubus pubescens* | 0.98 | 0.76 | 0.86 | shrubs |
| *Mitella nuda* | 1 | 0.72 | 0.85 | forb |
| *Cornus canadensis + Mitella nuda* | 1 | 0.69 | 0.83 | shrub + forb |
| *Cornus canadensis + Petasites palmatus* | 0.97 | 0.66 | 0.80 | shrub + forb |
| *Rosa acicularis* | 0.76 | 0.83 | 0.79 | shrub |
| *Mitella nuda + Rosa acicularis* | 1 | 0.62 | 0.79 | forb + shrub |
| *Lonicera involucrata* | 1 | 0.59 | 0.77 | shrub |
| *Viburnum edule* | 0.99 | 0.59 | 0.76 | shrub |
| *Rubus pubescens* | 0.73 | 0.79 | 0.76 | shrub |
| *Aralia nudicaulis + Cornus canadensis* | 0.98 | 0.59 | 0.76 | shrubs |
| *Cornus canadensis + Lonicera involucrata* | 1 | 0.55 | 0.74 | shrubs |
| *Cornus canadensis + Viburnum edule* | 1 | 0.55 | 0.74 | shrubs |
| *Mitella nuda + Rubus pubescens* | 1 | 0.55 | 0.74 | forb + shrub |
| *Rosa acicularis + Viburnum edule* | 0.99 | 0.55 | 0.74 | shrubs |
| *Lonicera involucrata + Rosa acicularis* | 1 | 0.52 | 0.72 | shrubs |
| *Lonicera involucrata + Viburnum edule* | 1 | 0.52 | 0.72 | shrubs |
| *Rubus pubescens + Viburnum edule* | 0.99 | 0.52 | 0.72 | shrubs |
| *Aster ciliolatus + Cornus canadensis* | 0.98 | 0.52 | 0.71 | forb + shrub |
| *Rosa acicularis + Rubus pubescens* | 0.76 | 0.66 | 0.71 | shrubs |
| *Galium boreale + Rosa acicularis* | 0.95 | 0.52 | 0.70 | shrubs |
| **Reclaimed Joint Species[a]** | $A^a$ | $B^b$ | $R^{2\ c}$ | **Growth Form** |
| *Taraxacum officinale* [1] | 0.99 | 0.90 | 0.94 | forb |
| *Phleum pratense*[1] | 1 | 0.76 | 0.87 | graminoid |
| *Trifolium hybridum*[1] | 0.96 | 0.76 | 0.85 | forb |
| *Taraxacum officinale + Trifolium hybridum* | 1 | 0.72 | 0.85 | forbs |
| *Phleum pratense + Taraxacum officinale* | 1 | 0.69 | 0.83 | graminoid + forb |
| *Phleum pratense + Trifolium hybridum* | 1 | 0.66 | 0.81 | graminoid + forb |
| *Vicia americana* | 0.94 | 0.69 | 0.80 | forb |
| *Cirsium arvense*[1,2] | 0.95 | 0.62 | 0.77 | forb |
| *Taraxacum officinale + Vicia americana* | 1 | 0.59 | 0.77 | forbs |
| *Cirsium arvense + Taraxacum officinale* | 1 | 0.55 | 0.74 | forbs |
| *Agropyron scabra* | 0.97 | 0.55 | 0.73 | graminoid |
| *Agropyron scabra + Taraxacum officinale* | 1 | 0.52 | 0.72 | graminoid + forb |
| *Cirsium arvense + Phleum pratense* | 1 | 0.52 | 0.72 | forb + graminoid |
| *Fragaria virginiana + Taraxacum officinale* | 1 | 0.52 | 0.72 | forbs |
| *Phleum pratense + Vicia americana* | 1 | 0.52 | 0.72 | forbs |
| *Trifolium hybridum + Vicia americana* | 0.96 | 0.52 | 0.70 | forbs |

*Notes:*[1]Introduced species, [2]noxious species, [a]only species with R2 ≥ 0.7 and p ≤ 0.001 were reported, [b]aspecificity, [c]sensitivity, [d]Pearson correlation between species and site type group (reclaimed or reference).

shrub), *Petasites palmatus* (native forb), and *Aster ciliolatus* (native forb; Table 3). *T. officinale* was associated with species including *Phleum pratense* (introduced grass), *Trifolium hybridum* (introduced forb), *Vicia americana* (native forb), *Agropyron scabra* (native grass), and *Cirsium arvense* (introduced, noxious forb). *C. arvense* was present on 63% of reclaimed quadrants ranging from 10 to 55% cover, yet was absent from all but 3/120 reference quadrants (2.5%).

### 3.4. Multivariate JGEE estimates

A model with nine response variables was best at determining effects of reclamation after controlling for forest stage and time since last disturbance (Table 4). We found that reclamation had a significant ($p < 0.001$) and positive effect on bulky density, pH, noxious presence, grass cover, and introduced richness, and had a negative effect on wood cover, LFH, live basal area, and canopy cover. We did not have a separate dataset available to test the model's ability to predict responses. The results of our binary response variable "noxious" had to be exponentiated before interpretation, whereas continuous variables we interpreted directly. Our analyses showed that reclaimed sites were

**Table 4**
Estimates of coefficients for 30 certified reclaimed well pads in Alberta's Central Mixedwood (n = 15) and Lower Foothills (n = 15) Natural Subregions model fitted using a Joint Generalized Estimating Equation (JGEE).

| Response Variable | Covariate | Estimate | Robust SE | Robust *Z* | *p* |
|---|---|---|---|---|---|
| Bulk density | Intercept | 0.43 | 0.03 | 12.91 | < 0.001 |
| | Reclaimed | 0.35 | 0.02 | 14.16 | < 0.001 |
| | Grassland | −0.10 | 0.04 | −1.28 | 0.20 |
| | Mature Forest | −0.13 | 0.04 | −2.98 | 0.00 |
| | Young Forest | −0.13 | 0.04 | −2.09 | 0.04 |
| | ≥35 years post disturbance | 0.05 | 0.04 | 1.14 | 0.26 |
| pH | Intercept | 0.56 | 0.06 | 8.94 | < 0.001 |
| | Reclaimed | 0.24 | 0.08 | 3.43 | < 0.001 |
| | Grassland | −0.15 | 0.10 | −1.61 | 0.11 |
| | Mature Forest | −0.17 | 0.10 | −1.71 | 0.09 |
| | Young Forest | −0.03 | 0.10 | −0.82 | 0.41 |
| | ≥35 years post disturbance | 0.03 | 0.08 | 0.46 | 0.64 |
| Noxious presence | Intercept | −0.94 | 0.54 | −1.87 | 0.06 |
| | Reclaimed | 2.33 | 0.70 | 2.94 | 0.00 |
| | Grassland | −0.14 | 0.82 | 0.21 | 0.84 |
| | Mature Forest | −1.08 | 0.73 | −1.66 | 0.10 |
| | Young Forest | −0.04 | 0.87 | 0.06 | 0.96 |
| | ≥35 years post disturbance | −1.15 | 0.81 | −0.97 | 0.33 |
| Grass cover | Intercept | 0.21 | 0.04 | 4.79 | < 0.001 |
| | Reclaimed | 0.16 | 0.08 | 1.77 | 0.08 |
| | Grassland | 0.18 | 0.09 | 2.58 | 0.01 |
| | Mature Forest | −0.03 | 0.11 | −0.57 | 0.57 |
| | Young Forest | 0.08 | 0.11 | 1.02 | 0.31 |
| | ≥35 years post disturbance | −0.10 | 0.10 | −0.30 | 0.76 |
| Wood cover | Intercept | 0.27 | 0.04 | 6.32 | < 0.001 |
| | Reclaimed | −0.18 | 0.05 | −3.73 | 0.00 |
| | Grassland | −0.07 | 0.06 | −0.88 | 0.38 |
| | Mature Forest | −0.11 | 0.06 | −1.75 | 0.08 |
| | Young Forest | −0.05 | 0.06 | −0.87 | 0.39 |
| | ≥35 years post disturbance | 0.06 | 0.05 | 1.37 | 0.17 |
| LFH depth | Intercept | 0.16 | 0.01 | 11.37 | < 0.001 |
| | Reclaimed | −0.15 | 0.03 | −4.43 | < 0.001 |
| | Grassland | 0.06 | 0.03 | 1.91 | 0.06 |
| | Mature Forest | 0.10 | 0.02 | 3.73 | 0.00 |
| | Young Forest | 0.07 | 0.03 | 2.15 | 0.03 |
| | ≥35 years post disturbance | 0.01 | 0.02 | 0.55 | 0.58 |
| Introduced richness | Intercept | 0.10 | 0.03 | 2.71 | 0.01 |
| | Reclaimed | 0.26 | 0.09 | 2.89 | 0.00 |
| | Grassland | 0.00 | 0.10 | 0.08 | 0.93 |
| | Mature Forest | −0.02 | 0.11 | −0.11 | 0.91 |
| | Young Forest | 0.09 | 0.11 | 0.77 | 0.44 |
| | ≥35 years post disturbance | −0.08 | 0.10 | −0.55 | 0.58 |
| Live tree BA | Intercept | 0.14 | 0.03 | 4.48 | < 0.001 |
| | Reclaimed | −0.17 | 0.04 | −4.00 | < 0.001 |
| | Grassland | 0.03 | 0.04 | 0.09 | 0.93 |
| | Mature Forest | 0.25 | 0.05 | 5.12 | < 0.001 |
| | Young Forest | 0.14 | 0.04 | 2.19 | 0.03 |
| | ≥35 years post disturbance | 0.04 | 0.03 | 1.52 | 0.13 |
| Canopy cover | Intercept | 0.55 | 0.07 | 8.06 | < 0.001 |
| | Reclaimed | −0.26 | 0.07 | −3.65 | 0.00 |
| | Grassland | −0.15 | 0.10 | −1.51 | 0.13 |
| | Mature Forest | 0.25 | 0.10 | 2.61 | 0.01 |
| | Young Forest | 0.14 | 0.08 | 1.59 | 0.11 |
| | ≥35 years post disturbance | 0.02 | 0.08 | 0.26 | 0.80 |

*Note:* Coefficients with $p < 0.05$ are statistically significant at the 95% confidence level.

$\exp^{(2.33)} = 10.3\%$ more likely to have noxious species present than on reference sites. Bulk density increased by a multiple of 0.35 after reclamation, controlling for all other factors in the model. This means we expect a bulk density of 1.3 g cm$^{-3}$ on reclaimed sites when that of the reference site is only 0.9 g cm$^{-3}$. Although bulk density was higher on reclaimed sites in general, mature forest had significantly lower bulk density ($E = -0.13$, $p = 0.002$) when compared to grassland sites and young forest. Time since reclamation did not significantly affect bulk density. pH increased by a multiple of 0.24 after reclamation, controlling for all other factors in the model. The model also predicted high grass cover on reclaimed grassland sites ($E = -0.18$, $p = 0.009$), and lower grass cover on older sites ($E = -0.03$, $p = 0.07$) and mature forest ($E = -0.1$, $p = 0.05$). Wood cover decreased significantly by 0.18 on reclaimed well pads. Introduced richness on the other hand significantly increased by 0.26 on reclaimed well pads. LFH was 0.15 times lower on reclaimed well pads and for each cm increase in LFH depth, mature forest floors were 0.1 times thicker. By no surprise, there was also a decrease in live tree basal area ($E = -0.35$, $p < 0.001$) and canopy cover ($E = -0.26$, $p < 0.001$) on reclaimed well pads with mature forest having significantly higher rates of each ($E = 0.25$, $p < 0.001$; $E = 0.25$, $p = 0.009$). The model residuals ranged from $-0.75$ to 0.94, supporting model fit. We did not include grouping variable *forest type* due to its small effect size. Additionally, adding another categorical predictor variable would cause rank deficiency.

## 4. Discussion

The first objective of this study was to assess vegetation succession on decommissioned and reclaimed well pads and compare properties with reference undisturbed forest of varied successional stages. We found evidence that some reclaimed well pads were in a state of arrested succession, with vegetation dissimilar to reference sites up to 48 years post reclamation. Successional vectors indicated that plant species composition differed on reference and reclaimed sites in the same location (only 30 m apart) and that there are enduring reclamation effects on community structure. Although reclamation practices likely varied among well pads with time and different regulatory and policy regimes, we have clearly shown that a majority of our certified reclaimed well pads, of various age and forest stage, were treeless grassland with introduced species. In the current study, reference sites recovering from harvest and fire have understory vegetation similar to young forest; whereas many sites disturbed by O&NG have been converted from forest to well pad to grassland. The current rate of land-use change from forested to other land types is likely underestimated if certified reclaimed (and exempt) well pads, ~31,000 (~12,000 on crown land) in Alberta's Central Mixedwood and Lower Foothills, are still considered boreal forest (Alberta Energy Regulator, 2018). This conversion inevitably influences forest ecology, ecosystem functioning, and the plant community composition (Aerts and Honnay, 2011; Chazdon, 2008).

Time since restoration or reclamation has been shown in numerous studies to be positively correlated with restoration success (Audet et al., 2015; Chen et al., 2018; Pinno and Hawkes, 2015; Rowland et al., 2009), yet there was not a strong effect of time in our analysis. If time since disturbance was not driving the succession of plant community structure on well pads, then what was? Why were reclaimed well pads so different from adjacent reference sites at the same location after accounting for time since disturbance, forest stage, and forest type?

The second objective of our study was to determine which properties were significantly influenced by post well pad reclamation and are thus useful ecological indicators of recovery. We found that reclamation had strong effects on nine indicators within three categories: soil (bulk density, pH), vegetation (introduced richness, grass cover, live tree BA, noxious presence), and environmental (canopy cover, downed wood cover, LFH depth).

The biological, chemical, and physical attributes of forests are driven largely by the ecological properties and processes in the soil. Forest soils accept, hold, and supply water; promote root growth; hold, supply, and cycle mineral nutrients; promote biological activity and optimum gas exchange; and accept, hold, and release carbon (Burger and Kelting, 1999). Two important physical properties of soils, bulk density and pH, can be elevated post O&NG reclamation (Calvo-Polanco et al., 2017; Frerichs et al., 2017), inhibiting the aforementioned properties and processes. Soil bulk density and pH values outside the natural range of variability may have negative effects on tree growth and nutrient availability (Arshad and Coen, 1992; Frerichs et al., 2017; Gale et al., 1991; Zhao et al., 2008; Zhao et al., 2010). Soil pH is one of the best predictors of species diversity parameters in boreal forest (Koptsik et al. 2001) and can drive nutrient availability and thus, community structure. If negatively altered by O&NG post reclamation, associated properties and processes may recover at slower rates and some may be permanently inhibited. Introduced plants are often associated with high pH conditions (Rose and Hermanutz, 2004), which have been shown to decrease net assimilation and transpiration rates in species such as black spruce, and can inhibit root growth in species such as aspen and white spruce (Calvo-Polanco et al., 2017; Zhang et al., 2013). Twenty three percent of our reclaimed well pads had a mean pH > 8, a level at which forest health might decline, aiding in the permanent inhibition of tree growth (Amacher et al., 2007; Gale et al., 1991; Schoenholtz et al., 2001) and providing favorable soil conditions for introduced plants. In our study, neither forest type nor time since disturbance significantly affected pH levels, suggesting that pH is relatively stable over time without the aid of natural disturbance such as fire. Although bulk density was statistically higher on reclaimed well pads than reference sites, none of the site mean bulk densities were greater than 1.4 Mg m$^{-3}$ (1.4e$^{6}$ g cm$^{-3}$), a level that adversely influences plant growth in boreal forest (Binkley and Fischer, 2013; Sutton, 1991). In our study, time since reclamation did not significantly affect bulk density, which is not surprising, as natural ameliorative processes do not rapidly loosen compacted soil (Froehlich et al., 1985). Moreover, the rate of recovery may vary among environmental parameters; for example, plant cover and biomass production may recover quickly after reclamation while pH and bulk density remain outside the range of natural variability (Hansen and Gibson, 2014). By altering soil conditions, rapid recovery of plant growth may alter soil conditions and produce alternative states due to positive feedback (e.g., carbon, nitrogen, and phosphorous exchange, microbial symbiont activity) by dominant species (Del Moral et al., 2007).

The species richness of introduced plants, presence of noxious species, and grass cover, three of our nine indicators, were important components of plant community composition and indicators of progress toward ecological recovery. Invasion of introduced plants near anthropogenic disturbance is becoming more common due to an increase in bare ground and light availability facilitating invasion (Langor et al., 2014; Rose and Hermanutz, 2004). As human footprint and the extent of bare ground in the boreal forest increases in association with O&NG activity there is an increasing likelihood or risk that introduced species will invade adjacent, undisturbed landscapes. Additionally, the number (richness) of invasive plant species may be a poor predictor of negative impacts on species diversity and ecosystem processes. The degree of adverse impact may depend largely on the traits of individual invasive species rather than overall richness, thus a few successful, noxious, introduced species can have a large impact on the system (Clavero et al., 2009; Dillemuth et al., 2009).

Species that showed environmental preference for reclaimed well pads included a common weed, *T. officinale* and an intentionally introduced (seeded) agronomic species, *P. pratense*. These were also the top two indicator species for reclaimed well pads. Both were found on most reclaimed plots, but were absent from reference plots. The ISA joint species list for reclaimed sites was composed of a combination of grasses and forbs including introduced and noxious species, which are often found in disturbed areas (Halpern, 1989; Knapp, 1991). They can also have biologically relevant impacts on system function. For example, *T. hybridum* and *V. americana* have symbiotic relationships with nitrogen-fixing bacterium *Rhizobium*, and can alter soil conditions by fixing nitrogen (USDA, 2002).

In contrast, our ISA indicated the strongest indicator species for reference sites was *C. canadensis*. Often considered a dominant species in mature, coniferous stands, *C. canadensis* may also be present, yet less abundant, in young, deciduous stands with species such as alder, willow, birch, or aspen. Soils in *C. canadensis* habitats are often associated with decaying wood and a thick organic surface horizon, as it dominates in the understory of cool, low-light boreal forest with saturated, slightly acidic to neutral, relatively rich soils. *C. canadensis* was strongly correlated with other native forbs and shrubs. These communities are common in coniferous and mixedwood forest of this region (e.g., McIntosh et al., 2016), yet they were absent from the reclaimed well pads. The absence of a species characteristic of local plant communities (*C. canadensis*), and the presence of persistent introduced species such as dandelion and timothy, on reclaimed well pads, likely has important implications for ecological function.

Of the noxious plant species found on sites, only one species, *C. arvense* (Canada thistle), had notable abundance. Canada thistle is a successful invader with traits including vegetative growth, long seed viability, dense root structures, and production of phytotoxins, which inhibit the growth of other plants (Kazinczi et al., 2004; Pilipavicius, 2008; Stachon and Zimdahl, 1980). The environmental and economic impacts of Canada thistle are of high concern for land managers due to negative impacts on native plant and animal diversity (Carter and Lym, 2017; Lym and Duncan, 2005) and fire frequency (Hogenbirk and Wein, 1991). Canada thistle was established on more than half the reclaimed well pads and removal is likely necessary to prevent spread (Schuster et al., 2018).

Biodiversity and ecosystem-level processes of the boreal forest have been studied extensively with respect to climate change and forest management (Dominic et al., 2009, Haeussler et al., 2004; Malmström and Raffa, 2000; Volney et al., 2000; Yan et al., 2018). By the year 2100, the Canadian boreal biome could increase 4–5 °C in mean temperature (Price et al., 2013). It is likely that introduced and noxious species will be among those species already moving northward at a pace of 16.9 km per decade in response to climate change (Chen et al., 2011). Introduced and noxious species can drastically alter the biological functions and processes of an ecological system. Once they become established, altering the state of the system, eradication of species, and reseeding with desirable species is required (Desserud and Naeth, 2014; Espeland and Perkins, 2017). Persistent noxious species on reclaimed sites interfere with the structure and functioning of vegetative communities by preventing native species recolonization, competing with target species, and reducing the integrity of adjacent landscapes by expanding into intact areas (Espeland and Perkins, 2017). Heavy grass cover can have competitive effects on conifer and deciduous tree species establishment (Bailey and Gupta, 1973; Eis, 1981; Bedford et al., 2000; Cole et al., 2003; Kokkonen et al., 2018). Forest stands may naturally establish on reclamation sites via ingress of tree species that produce air borne seeds such as *Populus* spp. and *Betula* spp. (Frouz and Kalčík, 2006); However, dense, persistent grass communities are often a filter in determining future successional trajectories of forests by inhibiting or delaying tree growth and regeneration (Royo and Carson, 2006). Graminoid cover shades the ground, decreasing soil temperatures, which in turn can reduce photosynthetic rate, nutrient uptake, and growth in boreal tree species, including aspen (Landhäusser and Lieffers, 1998). Shading creates cooler, moister micro-climates, and micro-habitats (Chávez and Macdonald, 2012), supporting various organisms including fungi, invertebrates, birds, and mammals; however, the success of shade-intolerant pioneer species, such as aspen, is very much dependent on the absence of shade from graminoid cover in the early stages of succession (Landhäusser and Lieffers, 1998).

In this study, grass and shrub roots were able to penetrate the soil and become established on recovering sites, yet fewer trees (both dead and alive) were observed on reclaimed well pads, likely because sites certified under older criteria were not required to plant trees. Reduced LFH on reclaimed sites could decrease water retention, thereby reducing the establishment success of germinated trees (Qi and Scarratt, 1998). The lack of trees in turn contributes to a long-term lack of woody debris and canopy cover, which affect understory abundance (Hart and Chen, 2008) and influence microclimate conditions on the forest floor by regulating light, soil temperature, air humidity, and nutrients (Andersson and Hytteborn, 1991; Botting and Fredeen, 2006; Mills and Macdonald, 2004; Park and Carpenter, 2016). These indicators are major contributors to plant community structure, productivity, plant and fungi diversity, water retention, erosion control, soil formation, energy exchange, and nutrient and mineral cycling (Brown and Naeth, 2014; Harmon et al., 1986; Hély et al., 2000; Ódor et al., 2006; Ódor and Standovár, 2001; Rambo, 2001; Rambo and Muir, 1998; Shorohova and Shorohov, 2001).

We have shown that most certified, reclaimed well pads have failed to become similar in community structure to reference sites with time (up to ∼50 years). The combination of soil physical modifications, changes in soil chemistry caused by soil layer mixing, the very low colonizing potential of some forest vegetation species, and their inability to compete with grasses, contribute to soil and vegetation impacts that differentiate reclaimed site characteristics from surrounding forest (Bockstette et al., 2017). This lingering industrial footprint effect post-reclamation is important to acknowledge as it can affect the long-term availability of resources, biodiversity, ecosystem services and processes.

We can only know more about the true time-dependent recovery with repeated, long-term sampling. Future research should focus on determining whether changes in relevant ecological indicators on reclaimed sites affect the abiotic (e.g., soil stability, hydrology, nutrient cycling) and biotic (e.g., plant functional traits, species turnover and regeneration, wildlife dynamics) components of the boreal forest's ecological function. By accurately accounting for the impacts of O&NG, we can better evaluate reclamation success.

## 5. Conclusions

Our results, based on a rigorous sampling design and statistical analysis, demonstrated that a limited number of certified reclaimed well pads were on a positive successional trajectory for recovery. We identified soil bulk density and pH, introduced species richness, presence of noxious species, grass cover, wood cover, LFH thickness, live tree basal area, and canopy cover as important ecological indicators of recovery for sites of all ages and forest stages. Of the eight oldest reclaimed well pads (35–48 years post-disturbance) two had similar plant and soil structure to reference sites, two showed an arrested successional recovery state, and four were potentially on a positive successional trajectory, with successional plant communities in-between those of established forest and grassland. Fifteen younger sites sampled 7–34 years post-disturbance, were grassland, and could potentially face a similar fate of arrested succession (treeless for decades). Additional research is also needed to understand the post-reclamation recovery of a wider range of forest regions; long-term monitoring is needed to characterize recovery status beyond the period considered in this study. We envision that other scientists and governmental bodies can apply our protocols and statistical methods to quantify recovery trajectories at reclaimed forested lands in other parts of the world where O&NG development is a dominant anthropogenic disturbance agent.

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## Appendix A. Supplementary data

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