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Ecohydrological functioning of an upland undergoing reclamation on post-mining landscape of the Athabasca oil sands region, Canada

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Abstract

Ecohydrological functioning of natural Boreal forest in Canada's Boreal Plains is a product of interactions between soil hydrophysical characteristics and hydrogeochemical processes. These interactions create a moisture–nutrient gradient within the surface soils, increasing along low-relief transitions from upland to riparian zone, and in turn influence the distribution of vegetation communities. It is not yet known if/when analogous ecohydrological functions can be achieved in constructed uplands following industrial disturbance, such as that following oil sands development. Hence, to assess this, we studied interactions between hydrogeochemical processes and vegetation colonization in a constructed upland relative to hydrophysical properties of 2 reclamation cover substrates during a typical continental climate's growing season. Our results indicated that in 3 years of postconstruction, the establishment of a moisture–nutrient gradient that supports vegetation colonization along slope positions was still limited by heterogeneity of cover substrates. Portions of the upland under peat–mineral mix were characterized by lower nutrient availability, high moisture content, and establishment of planted shrubs and trees. In contrast, forest floor materials plots were characterized by poor soil quality, but higher nutrient availability and greater colonization of invasive grasses and native shrubs. We suggest that the colonization of underdeveloped soils by invasive grasses may facilitate pedogenic processes and thus should be accepted by reclamation managers as a successional milestone in the recovery of ecohydrological functioning of constructed uplands. Poor soil structure under forest floor materials could not support edaphic conditions required by plants to efficiently utilize fertilizer, making this practise futile at the early stage of soil development.

KEYWORDS

Athabasca oil sands region, ecohydrology, forest floor material, hydrogeochemistry, peat–mineral mix, reclamation, upland

1 | INTRODUCTION

Forested uplands are a crucial component of the Western Boreal Forest (WBF), as they serve as conveyors of multiple hydrogeological functions that sustain the ecosystem services (Devito et al., 2005; Ketcheson et al., 2016). Geomorphology and soils of these forested uplands are characterized by a low relief (~7–12%), with deep glaciated substrates varying in hydrologic storage and transmission properties (Devito et al., 2005; Rowland, Prescott, Grayston, Quideau, & Bradfield, 2009). These hydrophysical properties and the subhumid climate of the region controls how moisture and solutes are redistributed within the landscape. Wetlands and open-water ponds interact with the uplands by redistributing moisture during wet and dry conditions, thus ensuring resilience to moisture stress during normal periodic drought (Johnson & Miyanishi, 2008). Differences in topographic positioning and soil substrate generally form a gradient favouring the accumulation of moisture and solutes towards the lower portions of the forested upland, influencing biogeochemical cycling and vegetation establishment (Dimitriu & Grayston, 2010). Despite this gradient, surface hydrological redistribution is rare during a typical WBF growing season because vertical flow dominates over lateral flow, and there is a high water requirement to fill deep unsaturated zone storage and meet vegetation evapotranspiration demands in this subhumid environment (Devito et al., 2005; Johnson & Miyanishi, 2008; M. D. MacKenzie & Quideau, 2010). Thus, an ecohydrologically functional upland is a critical component of the Boreal landscape due to these interactions among landscape units (i.e., forest, peatlands, and open-water wetlands).

Currently, the Athabasca oil sands region (AOSR) within the WBF is being heavily disturbed by industrial development for bituminous oil sand through in-situ recovery and open pit mining (Rooney, Bayley, & Schindler, 2012). The energy companies operating in this region are required by the Alberta government environmental regulations to return all disturbed lands to "equivalent predisturbance land capability" (Government of Alberta, 2017), which implies that post-mining landscapes should be functionally equivalent to what was present under predisturbance conditions. To fulfil this obligation, reclamation of individual landscape units found in predisturbance Boreal ecosystems has been ongoing in AOSR. Indeed, forested uplands and open-water wetlands have been successfully reclaimed in some places (Fung & Macyk, 2000; Pollard et al., 2012). However, these land units only represent a small portion (~23%) of the predisturbed landscape, where fen peatlands dominate (~45%; Audet, Pinno, & Thiffault, 2015; Price, McLaren, & Rudolph, 2010; Rooney et al., 2012). Furthermore, studies that have assessed the functioning of these reclaimed uplands only assessed them as an isolated landscape unit without considering their roles in the provision of hydrogeochemical support to surrounding habitats through hydrologic redistribution and solute transfer.

The process of upland reclamation in the AOSR involves the filling of excavated mine pits with mine overburden materials and tailing sands, which is then capped with salvaged organic amendments such as forest floor material (FFM) or peat–mineral mix (PMM; Naeth et al., 2012; Hemstock, Quideau, & Chanasyk, 2010; M. D. MacKenzie & Quideau, 2010). The capping layer serves as a substrate for vegetation establishment, while also limiting alkaline-sodic mine effluents from penetrating to the rooting zone, discouraging the vertical flow of moisture and nutrients through the formation of a capillary barrier (Carey, 2008; Jung, Duan, House, & Chang, 2014). Although this reconstruction approach works for isolated upland systems, studies have shown that achieving hydrogeochemical connectivity among land units will require the inclusion of atmospherically exposed recharge basins in reclaimed uplands to promote groundwater flow (Kessel, 2016; S. J. Ketcheson & Price, 2016b).

Constructed uplands differ from natural systems not only in terms of hydrogeological storage capacities but also in terms of their surface soils, which vary considerably in terms of vegetation colonization, biogeochemical, and hydrophysical characteristics (Dimitriu, Prescott, Quideau, & Grayston, 2010; Macdonald, Landhäusser, et al., 2015; M. D. MacKenzie & Quideau, 2010). Studies have found that similarities between constructed uplands and natural analogues were dependant on the salvaging and placement of the cover material used in reclamation. For instance, comparison between forested uplands reclaimed with FFM and PMM has shown that sites reclaimed with FFM are structurally similar to pristine Boreal forest soils, whereas PMM supported greater moisture absorption capabilities, which is imperative to sapling survival within the first few years (M. D. Mackenzie, Hofstetter, Hatam, & Lanoil, 2014; B. D. Pinno & Errington, 2015; Schott, Snively, Landhäusser, & Pinno, 2015). Regardless of amendment type, the mixing and degradation of the salvaged material during placement disturbs soil structure, presenting greater heterogeneity within the reclaimed system (S. J. Ketcheson & Price, 2016b; Macdonald, Snively, Fair, & Landhäusser, 2015; D. Mackenzie, 2011). In addition, newly reclaimed soils may possess hydrophobic properties and limited water storage capacities, leading to run-off and the erosion of unconsolidated particles downslope (Keshta, Elshorbagy, & Barbour, 2010; S. J. Ketcheson & Price, 2016b). Because run-off is rare during the growing season within the undisturbed WBF landscape, the infrequent near-surface flushing of soils leads to the accumulation of major nutrients such as soluble reactive phosphorus (SRP) and inorganic forms of nitrogen (i.e., $N\text{-}NO_3^-$, $N\text{-}NH_4^+$; Macrae, Devito, Creed, & Macdonald, 2006; Macrae, Redding, Creed, Bell, & Devito, 2005).

The mobility of major nutrients within the landscape are controlled by hydrophysical and geochemical process. Phosphorus mobility is primarily governed through erosion due to its strong affinity for the mineral-rich forest soils (Kreutzweiser, Hazlett, & Gunn, 2008; Macrae et al., 2005). In contrast, mobility of inorganic nitrogen is strongly influenced by biogeochemical processes (i.e., nitrification), and groundwater flow, given the dominance of vertical flow and unsaturated storage (Macrae et al., 2006). As upland reclamation aims to return equivalent capability to the disturbed land, it is important to understand how the heterogeneity of the soils, erosional processes, and incorporation of structures that promote groundwater recharge will influence the formation of a moisture–nutrient gradient and subsequently the establishment of desired Boreal forest vegetation communities.

In this study, the spatial and temporal characteristics of ecohydrological functions in an upland undergoing reclamation are examined during a growing season typical of the AOSR. The aim is to determine how topographic position and heterogeneity of the cover-soil influences moisture–nutrient gradients and pioneer vegetation colonization shortly (3 years) after reconstruction. The specific objectives are to (a) characterize spatial hydrophysical properties of the reclaimed upland relative to topographic position and reclamation cover substrate; (b) understand the interaction between hydrophysical properties and soil hydrogeochemical processes across the landscape over the growing season; and (c) identify the potential effects of the evolving soil moisture–nutrient gradient on the colonization of pioneer vegetation communities (planted and invasive) across the upland. We hypothesize that hydrophysical properties will not be influenced by a topographic gradient, given the fine-scale spatial heterogeneity in surface properties reported for reclaimed soils at the early stage of soil development (S. Ketcheson & Price, 2016a). We also hypothesize that mobile ions will accumulate in the lower topographic positions because of downhill transport and repeated near surface flushing events.

2 | MATERIALS AND METHODS

2.1 | Study site description

The study was conducted in summer of 2015 (from May to August) on a constructed watershed (56°55.944′N, 111°25.035′W), herein referred to as the Nikanotee Fen Watershed, consisting of an upland-fen system surrounded by three previously reclaimed hillslopes and one natural hillslope. The Nikanotee Fen Watershed was constructed on an overburden dump within the AOSR, ~40 km north of Fort McMurray, Alberta (Figure 1; S. Ketcheson & Price, 2016a). Briefly, overburden was removed with heavy machinery and stockpiled in-situ at the time of mining. The tailing sand used to form the upland aquifer is a by-product of bitumen separation from the oil sands. The upland is underlain by a 3-m thick layer of this tailing sand aquifer, situated above an impermeable engineered geotextile clay liner, designed to support sufficient lateral groundwater flow from upland to fen. The 7.7-ha forested upland consists mostly of FFM, with strategically placed areas of PMM (Figure 1). A 2.2-ha transition zone (low-slope) is incorporated at the toe of the low relief (2–3%) upland, where pockets of peat substrate are located. The remainder of the upland (5.5 ha) is designed as a combination of hummocks and hollows aimed to reduced excessive run-off while promoting infiltration and groundwater recharge (Daly et al., 2012; Pollard et al., 2012; S. Ketcheson & Price, 2016a).

Surface of the FFM was reconfigured through tillage, to increase recharge to the upland aquifer and facilitate the development of plant rooting structure. Planting was initiated on the site in the summer of 2013, and again in the summer of 2015 to actualize the desired stand density. Detailed description of the planting strategy is provided in Daly et al. (2012), whereas information about the plant species and quantity planted during each planting campaign is presented in Table 1. In the 2015 planting campaign, approximately 10,188 tree and shrub saplings (Table 1) were planted on June 21, 2015, with the addition of 10-g Continuum RT^{TM} (18:9:9:9[S]) controlled release fertilizer (CRF) in biodegradable paper packets, applied to each individual sapling at an approximate rate of 1,756 kg ha^{-1}. Vegetation surveys conducted in the upland at the peak of the 2015 growing season showed that forbs and grasses (e.g., *Sonchus arvensis* and *Agropyron tracycaulum*) dominate the vegetation canopy. Sampling plots for this study were located within grids formed by north–south (A–A′; B–B′) and east–west (C–C′; D–D′) transects, running throughout the upland (Figure 1). The gently sloping topography of the upland guided the delineation of the grids into upper, mid, and lower slope positions. Each of the three slope positions had three grid points, which formed the monitoring plots.

2.2 | Hydrophysical characteristics

Soil moisture access tubes (PR2 Delta-T Devices©) were installed in all the monitoring plots to measure volumetric water content twice a week

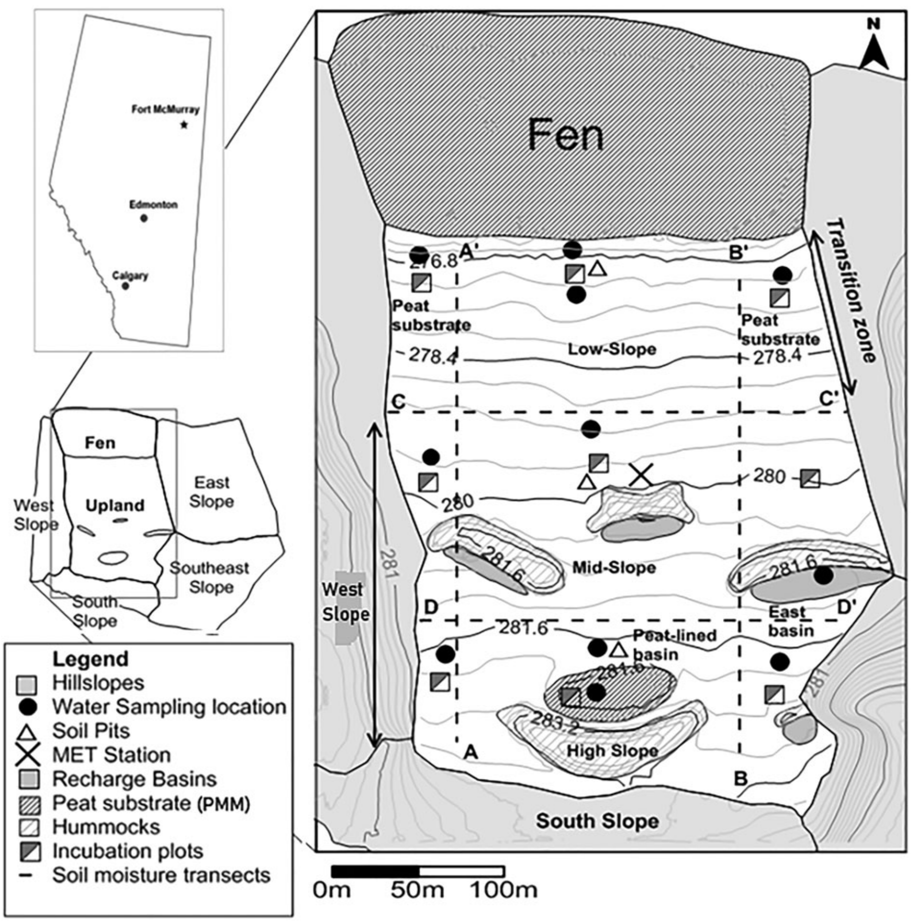

FIGURE 1 Map of the study area showing upland-fen system and indicating the location of monitoring plots in both peat–mineral mix (PMM) and forest floor material substrates. MET, Meteorological Station

TABLE 1 Names of plant species and number of seedlings planted during the two planting campaigns implemented in the constructed upland between 2013 and 2015

	Number of seedlings planted in each planting campaign			
	Transition zone (low slope)		Upland (mid and high slope)	
Names of plants	2013	2015	2013	2015
Trees				
Black spruce (*Picea mariana*)	920	–	5,400	2,160
Jack pine (*Pinus bankstana*)	–	–	3,434	–
Tamarack larch (*Larix laricina*)	–	–	–	2,520
Shrubs				
Labrador tea (*Ledum groenladicum*)	215	–	431	1,434
Willow (*Salix sp.*)	460	720	–	–
Dwarf birch (*Betula pumula*)	460	619	–	–
Bog cranberry (*Vaccinium oxycoccos*)	–	–	260	–
Blueberry (*Vaccinium Cyanococcus*)	–	–	1,155	–
Bunchberry/Dogwood (*Cornus stolonifera*)	–	–	377	935
Prickly wild rose (*Rosa acicularis*)	–	–	116	–
Green alder (*Alnus crispa*)	–	–	–	630
River alder (*Alnus tenuifolia*)	–	–	–	1,170
Sedges/grasses				
Northern reed grass (*Calamagrostis inexpansa*)	193	–	–	–
Water sedge (*Carex aquatilis*)	300	–	–	–

(between May and August, resulting in 32 measurements over the study period) at selected depth intervals (between 10 and 40 cm) within the soil profile. The values obtained were used to determine total soil water (TSW) at 15 and 35 cm. The top 15 cm of soil represents the approximate depth of average root development, and 35-cm depth represented moisture flux within the approximate thickness of the FFM (Burk, Chanasyk, & Mapfumo, 2000; Leatherdale, 2008). At each access tube location, additional measurements of surface soil moisture (i.e., 0–10 cm), surface temperature, and porewater conductivity (ECp) were also recorded twice a week using a Delta-T WET Sensor (Type WET-2). Temperature probes built from Thermocouple Wire (T Type, Duplex Insulated Omega©) were installed within the vicinity of soil moisture profile tubes at various depth intervals (2, 5, 10, 20, and 30 cm). Additionally, measurements of unsaturated surface soil moisture absorption (mm^{-1} hr^{-1}) were conducted once over the study period near each of the soil moisture access tubes using the single ring infiltration tests (Negley & Eshleman, 2006; USDA, 1999).

Three soil pit moisture stations were located at each slope position along the topographic gradient to determine the infiltration capacity of the soil and to identify storm events where precipitation and antecedent moisture conditions led to infiltration into the tailing sand aquifer. Soil samples used in quantification of hydrophysical properties were collected at 10-cm depth, adjacent to moisture profile access tubes within each sampling plot. The physicochemical characteristics analysed include bulk density (g^{-1} cm^3), percentage organic matter content (loss on ignition), and pH. Root to shoot ratio (R/S in g g^{-1}) of established nonplanted pioneer species was also measured using triplicates of rectangular (20 × 50 cm) quadrats, randomly placed around the monitoring plots. Aboveground biomass was removed and placed within a paper bag. Prior to removing belowground biomass, surveys were conducted to visualize where average root depth was located (~20 cm). Two cores (860 cm^{-3} $core^{-1}$) were selected per quadrat. Standard laboratory procedure (Durigan, Melo, & Brewer, 2012; Ravindranath & Ostwald, 2008; Sainju, Allen, Lenssen, & Ghimire, 2017) was used to determine average aboveground and belowground biomass, which were then applied to the entire quadrats volume (20,000 cm^3). The average R/S ratios were correlated with edaphic characteristics to identify soil conditions that facilitated the colonization of native and non-native propagules and seedbanks from the FFM.

2.3 | Hydrogeochemical processes

Water samples for estimating nutrient loss through surface flushing and vertical flow during precipitation events were captured with three V-notched flumes located at the toe of the upland. These run-off samples were collected after major storm events and subsequently processed and analysed for SRP and dissolved inorganic nitrogen (i.e., NO_3-N and NH_4^+-N) using colorimetric techniques (Bran Luebbe AA3, Seal Analytical, Seattle, USA, Methods G-102-93 [NH_4^+], G-109-94 [NO_3^-], G-103-93 [SRP]). Groundwater samples were also collected once every month (i.e., four times over the entire study period) from a network of wells and piezometers installed across the upland to measure groundwater nutrient contribution within the constructed aquifer. Event-based surface water samples were also collected from the recharge basins within the upland to determine if these structures encouraged nutrient leaching. Both ground and surface water samples were analysed with a Dionex ICS-1600 (Method EPA 300.0 with AS-DV auto-sampler) for nutrient ions (PO_4^{2-}, NO_3^--N, and NH_4^+-N) at the Biotron Experimental Climate Change Research Facility, Western University, London, Ontario.

The rate of porewater nutrient supply was measured using Plant Root Simulator probes (Western Ag Innovations, Saskatoon, Saskatchewan, Canada). The Plant Root Simulator probes were installed in triplicates within each of the nine sampling plots over three incubation cycles (i.e., early, middle, and late growing season) to monitor seasonal variability in supply rates of bio-available nutrients and anions (nitrate [NO_3^-], phosphate [$H_2PO_4^-$ and HPO_4^{2-}], and sulphate [SO_4^{2-}]) and cations (ammonium [NH_4^+], potassium [K^+], calcium [Ca^{2+}], magnesium [Mg^{2+}], aluminium [Al^{3+}], iron [Fe^{3+}], manganese [Mn^{2+}], copper [Cu^{2+}], zinc [Zn^{2+}], and boron [B^+]) to plants (μg 10 cm^{-2} incubation $periods^{-1}$; Nwaishi et al., 2016).

2.4 | Statistical methods

All statistical analyses were performed with R© (R Core Team, 2015). Prior to analyses, data were assessed for normality using the Shapiro–Wilks test ($p > .05$; "Shapiro.test" stats package R.3.2.3), and datasets that did not conform to the criteria of normality were transformed using general relativization followed by arcsine square root transformation (Rooney & Bayley, 2011; Rowland et al., 2009; Turcotte, Quideau, & Oh, 2009). Transformation was not effective on all variables; hence, non-parametric tests were performed for data analyses.

To assess spatial (along topographic gradients) and temporal (over the growing season) variability in hydrophysical properties, a Scheirer–Ray Hare extension of the Kruskal–Wallis test was applied as the non-parametric equivalent to a two-way analysis of variance (Dytham, 2011), followed by a post hoc analysis (function "kruskal," package "agricolae"). A Mann–Whitney *U* test was used to measure the degree of variability in hydrophysical properties between both cover substrates (i.e., FFM and PMM). Permutation multivariate analysis of variance (function "Adonis," package "Vegan") was used to determine significant spatial and temporal variability in ion availability. This was followed by a multi-response permutation procedure (MRPP; function "mrpp," package "Vegan"), which indicates when and where these significant variations occurred. Indicator species analysis (function "indicators," package "indicspecies") was used to complete the analysis by identifying the monitoring plots that were significantly different. The accepted significance level for all statistical tests was $p \le 0.05$. Ordination (Non-metric multidimensional scaling -NMDS) was used to display correlations between hydrophysical properties and hydrogeochemical dynamics of ions, as well as the relationship between moisture–nutrient gradients and vegetation community establishment.

3 | RESULTS

3.1 | Spatial variation in hydrophysical properties of the constructed upland soils

The results of our field study indicate that the topographic gradient did not significantly ($p = .5$) influence surface soil moisture conditions (TSW_{15}) but influenced ($p = .02$) subsurface soil moisture redistribution (TSW_{35}), with the highest soil moisture content observed in the mid-slope position. Volumetric water content was influenced by the type of reclamation cover substrate, with plots under PMM supporting significantly higher ($p = .03$) moisture content than plots under FFM. Unsaturated surface soil moisture absorption was found to be heterogeneous throughout the upland and significantly greater in PMM ($p < .001$; Table 2). Likewise, infiltration was variable throughout the upland. Major infiltration and groundwater recharge rarely occurred for both the low and mid-slope sections of the upland all through the study period; however, infiltration occurred frequently within the upper slope area of the upland (Figure 2). Surface FFM layers (5 and 15 cm) often responded to precipitation events; however, the base (25 cm) and tailing sands top (~30 cm) solely responded to precipitation events when antecedent moisture conditions and the magnitude of the events were optimal.

The heterogeneity of reclamation cover substrate was reflected in the spatial variability in soil physical properties of the constructed upland (Table 1). Bulk density (ρb) was significantly higher in the mid slope ($p = .02$) than the other slope positions and higher within the FFM than the PMM ($p < .001$). Conversely, the R/S ratio was significantly lower ($p = .008$) at the mid-slope position and in plots under FFM ($p < .001$), with higher bulk densities. Organic matter content was not significantly different among slope positions but was higher ($p < .001$) in plots under PMM than FFM. More neutral pH was observed at the lower slope position ($p = .025$), especially within PMM plots ($p = .03$), whereas more acidic pH was observed in the mid-slope and in plots under FFM cover.

3.2 | Spatiotemporal variability in hydrogeochemical processes

Seasonal dynamics in soil hydrogeochemical processes varied spatially and temporally (Table 3). As expected, topographic position ($p = .003$) and cover substrate ($p < .001$) significantly contributed to the observed spatial variability of hydrogeochemical processes, but with

TABLE 2 Soil functional characteristics (mean ± standard deviation) relative to topographic position and substrate; showing root to shoot ratio (R/S), bulk density (ρb), unsaturated surface soil moisture absorption (SSMA), organic matter (%), and pH

	Slope position			Substrate	
Parameter	High slope	Mid slope	Low slope	FFM	Peat
R/S (g g^{-1})	1.95 (0.29)^a	0.32 (0.01)^b	0.95 (0.02)^a	0.86 (0.01)^b	1.18 (0.17)^a
ρb (g cm^{-3})	1.26 (0.03)^b	1.51 (0.01)^a	1.26 (0.4)^b	1.42 (0.02)^a	1.18 (0.04)^b
SMMA (mm^{-1} hr^{-1})	48.00 (19.17)^a	37.65 (9.81)^a	39.19 (14.72)^a	37.33 (9.81)^b	49.49 (13.75)^a
Organic matter (%)	18.83 (0.87)^a	19.5 (0.59)^a	18.33 (0.97)^a	17.94 (0.54)^b	22.04 (0.68)^a
pH	6.43 (0.18)^{ab}	6.13 (0.04)^b	6.5 (0.06)^a	6.22 (0.65)^b	6.66 (0.13)^a

Note. Characters a, ab, and b are used to indicate significant differences (Kruskal test) among treatments (comparing slope position and substrate separately). FFM = forest floor materials.

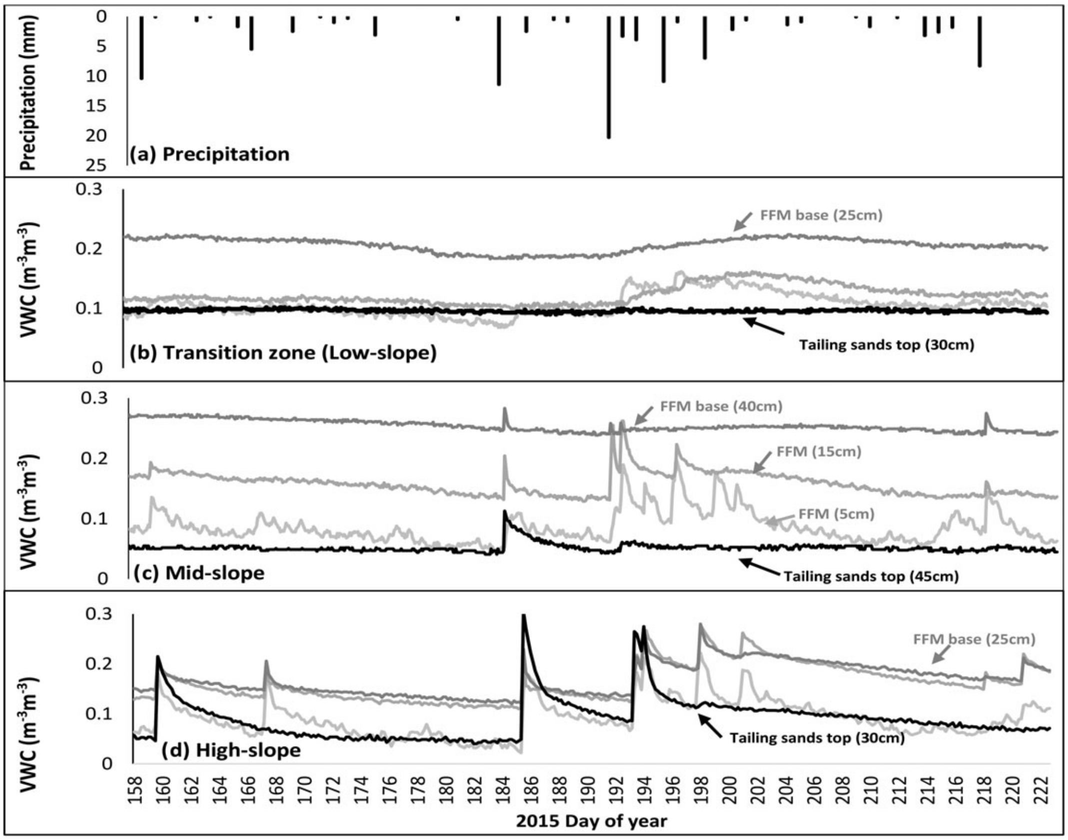

FIGURE 2 Response of soil volumetric water content (VWC) in the forest floor materials (FFM) cover substrate (grey) and tailing sands top layer (black) to growing season's precipitation (a). The water content within the soil profile was recorded at three different soil moisture stations along the topographic gradient, covering the transition zone (b), mid-slope (c), and high-slope (d) areas. The base FFM layer for both the transition and high slope was located at~25 cm, whereas the base of the FFM layer at the mid slope was located at ~40 cm

TABLE 3 Outputs of multi-response permutation procedure analysis on ion availability relative to slope position and time of season. Results demonstrate the separation (T) between slope positions and separation (A) among same-slope replicates

	Time of season							
	Throughout season		Early		Middle		Late	
Position	T	A	T	A	T	A	T	A
High versus mid versus low	**−3.75**	0.083	**−4.06**	0.098	**−4.73**	0.091	**−2.47**	0.065
High versus mid	**−3.72**	0.036	**−3.47**	0.079	−3.06	0.043	−0.87	0.039
High versus low	−0.47	0.014	−0.74	0.007	**−2.92**	0.077	−0.82	0.016
Mid versus low	**−12.77**	0.111	**−5.35**	0.164	**−4.63**	0.096	**−4.27**	0.11

Note. Bold numbers represent significant differences ($p > .05$).

a very low coefficient of determination (R^2 = .11 and .25 for slope position and cover substrate, respectively). Results from the MRPP (Table 3), which tested the null hypothesis of no difference in ion availability among slope positions, show that over the course of the study period, the greatest degree of separation in availability of ions occurred between the mid- and low-slope positions (T = −12.77). No significant difference was observed between monitoring plots in the high- and low-slope positions (p = .301), where within group replicates had very minimal group homogeneity (A = 0.014). Subsequent cluster and indicator species analysis divided the monitoring plots into three groups on the basis of nutrient and ion availability, where Ca^{2+}, Mg^{2+}, B^+, and SO_4^{-2} corresponded strongly with Group 1, whereas Mn, K^+, NO_3^-, NH_4^+, and SRP corresponded strongly to Group 2. A third group was significantly different from Groups 1 and 2 but did not possess specific indicators. Group 1 corresponded to all monitoring plots under peat, whereas other groups corresponded with plots under FFM cover.

Temporal variability in plant nutrient ion supply rates was not observed over the growing season (Table 3). Among slope positions, the greatest degree of separation occurred during the middle of the growing season (T = −4.73, A = 0.098), between the mid- and low-slope positions (T = −4.63, A = 0.096). The least degree of spatial variability in nutrient ion supply occurred during the late season (T = −2.47, A = 0.065), when supply rates in both mid (T = −0.87, A = 0.039, p = .15) and low slope (T = −0.82, A = 0.016) were not significantly different (p = .23) from the supply rates in the high slope.

Concentration of SRP was consistently high in run-off water over the growing season and did not reflect concentrations in rhizosphere porewater (TSW_{15}). In contrast, the concentration of dissolved inorganic N in run-off water strongly reflected total inorganic N (TIN) availability in porewater. The application of fertilizer (on day of the year 172) increased the concentration of SRP and TIN in rhizosphere porewater pools considerably (~114% and 82% increase, respectively), which was strongly reflected in the concentration in run-off water following a storm event (Figure 3a). However, later in the growing season, the availability of TIN per area of the rhizosphere decreased, returning to baseline concentrations (10.99 and 7.49 μg/10 cm^2), which was detected in the run-off water (Figure 3b).

In groundwater, NO_3^- concentration was greatest following large precipitation events, and this was only witnessed within the recharge basins, particularly the east basin. Conversely, NH_4^+ concentrations decreased following precipitation events within these recharge basins and increased during the drier sampling periods. NO_3^- concentrations were consistently minimal within the groundwater samples, where either PMM or FFM cover substrate was located. Dissolved inorganic N concentration in groundwater was relatively consistent in all the four samples collected during the study period (5.5, 5.8, 6.4, and 5.9 mg ml^{-1}). The concentration of SRP in groundwater was below detection limit, but it is worthy to note that the availability of SRP in porewater pools was negatively correlated with ions such as Ca^{2+} (p = .044, R^2 = −.36), Fe^{3+} (p = .007, R^2 = −.4), and SO_4^{2-} (p < .001, R^2 = −.47), which are found in high concentrations in post-mining overburden materials.

3.3 | Colonization of vegetation functional groups in the constructed upland

Analysis of vegetation survey data indicates that the β diversity of the constructed system was considerably low (βw = 1.54). Vegetation colonization patterns were influenced by the distinctive soil properties such as nutrient conditions, moisture, and temperature (Figure 4). For

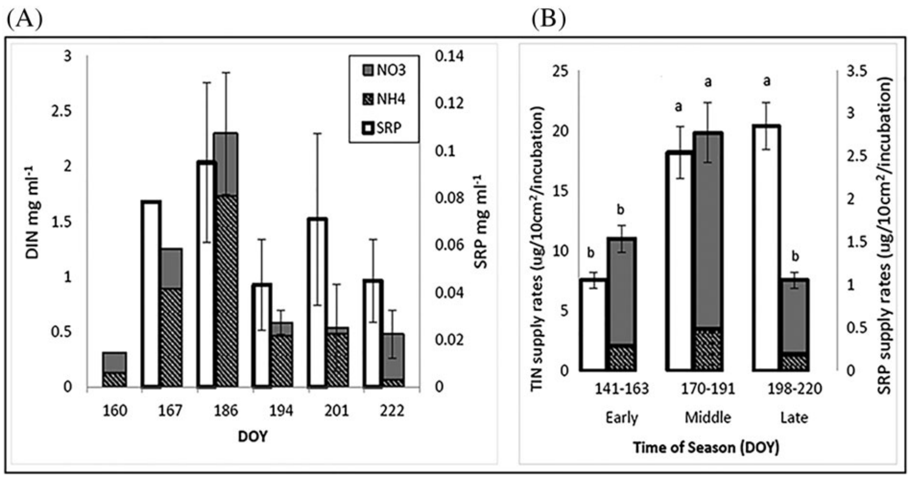

FIGURE 3 (a) Nutrients (dissolved inorganic nitrogen [DIN] and soluble reactive phosphorus [SRP]) concentrations (mean ± standard error) in run-off samples following major storm events (>5 mm) and (b) seasonal (mean ± standard error) supply rates of total inorganic nitrogen (TIN) and SRP within the reclaimed upland soils. Characters A and B are used to indicate significant differences over temporal scale (Kruskal) throughout the season. Note that fertilizer was applied to the site on June 21, 2015 (day of the year [DOY] 172)

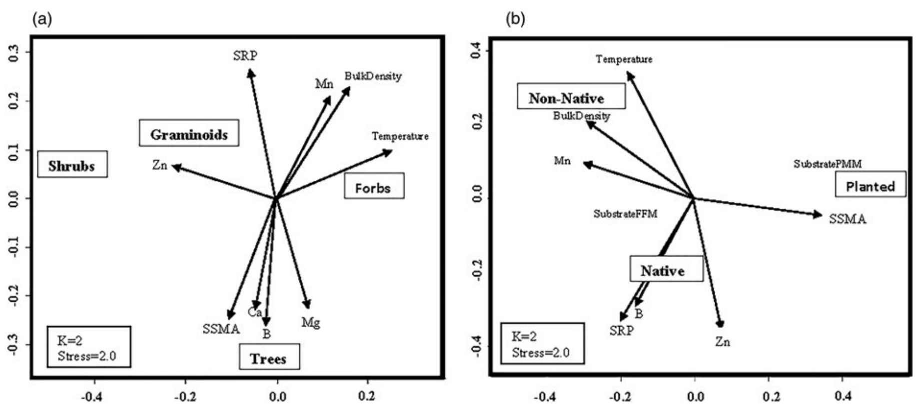

FIGURE 4 NMDS ordination plots demonstrating relationships between environmental variables and functional vegetation groups. Vectors represent strong correlations (p < .05). (a) Relationships between environmental variables and forbs, gramminoids, shrubs, and trees.
(b) Relationship between environmental factors and native, non-native, and planted species. K represents the number of dimensions for best fit, and the stress value describes the fit of the ordination model. SRP = soluble reactive phosphorus; FFM = forest floor materials; PMM = peat–mineral mix; SSMA = surface soil moisture absorption

instance, forb colonization was correlated with areas of higher soil temperatures (p = .029, R^2 = .70), whereas grasses with SRP (p = .023, R^2 = .73) and tree saplings correlated (p = .017, R^2 = .73) with areas of elevated unsaturated surface soil moisture absorption (Figure 4a). The soil properties (e.g., high moisture content and availability of cations) that are associated with the abundance of tree saplings and other planted species are predominant in PMM substrate (Figure 4b). The presence of species that are native to the region is associated with areas of elevated SRP (p < .001, R^2 = .76) and K^+ (p = .05, R^2 = .58) availability, whereas the presence of non-native species was strongly correlated with areas of higher bulk density (p = .05, R^2 = .64) and warmer soil conditions (p = .007, R^2 = .78).

4 | DISCUSSION

4.1 | Characteristics of the constructed upland relative to topographic position and cover substrate

Our results show that the constructed upland is characterized by a high degree of heterogeneity in physical properties of the cover substrates, which may have led to the lack of a consistent topographic influence on soil hydrophysical characteristics, such as moisture content and nutrient availability. Hence, we did not observe the formation of moisture-nutrient regimes along the topographic gradient. These findings are consistent with those of previous upland reclamation studies in AOSR (Leatherdale, 2008; M. D. MacKenzie & Quideau, 2010), which attributed the poor structure of the reclaimed soil to the fragmentation of donor LFH layers by large operational equipment used in transfer and placement of the cover substrates in the constructed upland. Consistent with our hypothesis, these results suggest that 3 years of postconstruction is not sufficient time for the recovery of homogeneous and optimum soil conditions in reclaimed uplands. It is worthy to note that a bigger sample size will be required to elucidate the magnitude of heterogeneity that exists in reconstructed upland soils.

In novel environments, the establishment of pedogenic processes such as freeze–thaw cycles, organic matter accumulation, and plant root development is needed to facilitate the development of edaphic characteristics (e.g., preferential flow paths and rhizosphere effect) that can sustain the recovery of a functional soil with consistent structural profile (Guebert & Gardner, 2001; S. J. Ketcheson, Price, et al., 2016). Notwithstanding the poor surface structure of the constructed upland, surface reconfiguration achieved through tillage that was performed perpendicular to the topographic gradient created microtopographic features, which functioned as areas of water accumulation during storm events. In addition, areas of microforms (hummocks and hollows) incorporated within the engineering design of the constructed watershed also created areas of preferential flow, directing percolating water towards hollows and into groundwater recharge basins (Kessel, 2016; S. J. Ketcheson, Price, et al., 2016). The role of these water-conveying features is evident in the disparity between topographic influence on surface (TSW_{15}) and subsurface (TSW_{35}) soil moisture redistribution. Furthermore, textural interfaces formed by placing the fine-grained FFM over a coarser grained material (tailing sands) led to infrequent water percolation into the tailing sands aquifer, creating a capillary barrier that limits moisture losses below FFM. Although these textural discontinuities were purposely implemented to limit any vertical seepage of soil moisture and nutrients below the cover soil, while impeding highly sodic water from rising above the tailing sands (Huang, Barbour, Elshorbagy, Zettl, & Si, 2013; Jung et al., 2014; Naeth, Chanasyk, & Burgers, 2011), they can also inhibit soil development by limiting percolation through the soil profile, thereby complicating the re-establishment of hydrogeochemical connectivity. Although the rapid infiltration response of FFM following precipitation events suggests that it will be a suitable surface cover material to facilitate the re-establishment of hydrogeochemical connectivity in the constructed upland, PMM with lower bulk density and higher organic matter and moisture content likely presents a more suitable substrate for rhizosphere development with the higher R/S ratio.

The availability of nutrient ions in the constructed upland also varied between cover substrates, with PMM plots supporting higher ion contents (Ca^{2+}, Mg^{2+}, B^+, and SO_4^{2-}) than FFM. However, the availability of macronutrients (K^+, NO_3^-, NH_4^+, and SRP) was highest in FFM. Previous studies have shown that higher concentration of sulphate found in the peat substrates is artefacts of the oxidation of donor peat during stockpiling (Nwaishi et al., 2016), whereas the higher concentration of K^+ and SRP in plots under FFM was attributed to the presence of mycorrhizae inoculum, which produce extracellular enzymes catalysing phosphate and K^+ mineralization (Brown & Naeth, 2014). Furthermore, the elevated concentration of Ca^{2+} in the peat substrate could be immobilizing phosphates by forming insoluble precipitates (Gurevitch, Scheiner, Fox, 2006).

The presence of NO_3^- as the dominant form of inorganic N confirms that nitrification is a major pathway for N mineralization in the uplands, especially in plots under FFM cover substrates. The lower availability of NO_3^- under PMM substrate cover is typical of peat substrate due to a higher C:N ratio and the metabolic constraints of peatland microbial communities in the novel forest floor environment (M. D. Mackenzie et al., 2014; McMillan, Quideau, MacKenzie, & Biryukova, 2007). Peatland microbial communities are adapted to low temperature, acidic pH, and anaerobic conditions. Hence, when exposed to contrasting forest floor conditions, the microbial community structure takes longer to adapt and reactivate mineralization of organic matter into available nutrients, thereby limiting nitrification (which is not a dominant process in peatlands) in PMM (Jamro, Chang, & Naeth, 2014; M. D. MacKenzie & Quideau, 2012).

4.2 | Interactions between hydrophysical characteristics and spatiotemporal variability in soil hydrogeochemical processes

Spatiotemporal variability in the hydrogeochemical process of moisture and nutrient redistribution was primarily controlled by the interaction between hydrophysical characteristics of cover substrates and slope position. This interaction was more evident in the hydrogeochemical response of the constructed upland to the application of

CRFs in the middle of the growing season. Although CRFs are designed to gradually release nutrient ions to the soil matrix over time, the poor structure and underdeveloped state of the reconstructed soil resulted in a rapid loss of major nutrient constituents in the CRFs, through run-off and leaching. A similar outcome was reported by Sloan, Uscola, and Jacobs (2016), who found that the majority of nutrient constituent of CRFs was unaccounted for within the saplings and competing vegetation, in an upland undergoing reclamation.

With lower organic matter content and higher infiltration capacity, FFM plots were more susceptible to NO_3^- loss through leaching following precipitation events. The high concentration of SRP in run-off water suggests that the underdeveloped state of LFH in the constructed upland increased the susceptibility of available SRP to near-surface leaching. Studies have shown that the presence of litter in a forested upland system significantly reduces soil nutrient loss through run-off (Li, Niu, & Xie, 2014; Sayer, 2006). Furthermore, near-surface tilling perpendicular to the topographic gradient on the upland could have also affected spatial nutrient availability through changes in soil hydraulic, aeration, and diffusive properties, increasing the lability of the organic matter and thereby mobilizing many essential plant nutrients (Lipiec & Stepniewski, 1995). The effect of surface reconfiguration on hydrogeochemical processes likely varied between cover substrate, but this was not directly tested in this study. However, the high degree of intragroup variability in nutrient-ion availability between FFM and PMM, along slope positions, suggests that the heterogeneity of the cover substrate could potentially mask any topographical influence on nutrient-ion availability (Leatherdale, 2008).

Vertical fluxes of nutrients within the soil profile were also affected by the physical structure of the reconstructed soil layers. A capillary barrier formed at the interface of the FFM and tailing sand created a perched water table, inducing localized anoxic conditions within the FFM layer. Hence, NO_3^- being the dominant and most mobile form of TIN in the surface soil layer was easily leached into anoxic subsurface layers, where it was reduced to NH_4^+. Although NH_4^+ is not very mobile, the hydrophobic properties of the tailing sands suggest that once moisture penetrates below the cover soil, water and ions gradually seep into groundwater aquifers (Huang, Lee Barbour, Elshorbagy, Zettl, & Cheng Si, 2011; Jung et al., 2014). Consequently, NH_4^+ was observed as the dominant form of TIN in shallow groundwater samples. The effect of this capillary barrier on N dynamics was confirmed by the increased NO_3^- concentrations observed in groundwater samples collected after precipitation events, from wells where such a capillary barrier did not exist (e.g., recharge basins). As with natural forested uplands, SRP did not seem susceptible to groundwater leaching, likely due to the formation of insoluble precipitates within the cation-rich subsoils layers (~20 to 50 cm).

4.3 | Effects of soil moisture–nutrient gradient on vegetation colonization across the upland

A moisture–nutrient gradient did not exist along the topographic gradient but did exist between cover substrates. Hence, the hydrogeochemical characteristics of the cover substrate strongly influenced the pattern of vegetation colonization within the constructed upland. The combination of low moisture content with higher nutrient concentration in plots under FFM cover substrate led to the colonization of these plots by invasive forbs (e.g., *S. arvensis*) and grasses (e.g., *A. tracycaulum*) that are typical of drier climates. With higher moisture retaining abilities, nutrient adsorption capacities, and decreased competition from a less viable seed bank, PMM had higher planted-sapling cover than FFM. Because FFM had a higher availability of essential macronutrients (N–P–K), this result implies that planted-sapling survival is influenced by a moisture deficit rather than nutrient supply. Similar results were reported by previous aspen saplings establishment studies, which demonstrated that the survival rate of planted saplings is greater under PMM than FFM, especially during the first few years following soil placement (B. D. Pinno & Errington, 2015; B. D. Pinno, Landhäusser, MacKenzie, Quideau, & Chow, 2012; Schott et al., 2015).

Under the FFM soil cover, planted trees and shrubs were the least abundant plant species, whereas nonplanted native species were more abundant than what was observed in PMM plots. Consistent with previous studies, our results confirm that salvaged FFM is advantageous to the recovery of native species in reclaimed uplands because it contains greater densities of viable propagules and seedbank, typical to that of natural forested upland ecosystems (Archibald, 2014; Errington & Pinno, 2015; D. D. Mackenzie & Naeth, 2010). The availability of nutrient-ions in FFM influenced both the regeneration of native species seedbank and colonization by invasive species, which are known to out-compete planted saplings for nutrients within reclaimed soils (Errington & Pinno, 2015; Hangs, Knight, & Van Rees, 2003; Landhäusser & Lieffers, 1994). Given that a greater proportion of the upland is under FFM cover substrate, these invasive forbs and grasses were the dominant vegetation cover throughout the upland. Similar results were observed in an upland reclamation study, which confirmed that non-native forbs and grasses dominate reclaimed uplands for the first 5 years post soil placement but gradually decline with development of a canopy cover by planted saplings (B. Pinno & Hawkes, 2015).

The ability of these pioneer non-native species to colonize portions of the reconstructed soil that are currently uninhabitable to planted shrub and tree saplings highlights their crucial role in facilitating the recovery of optimum soil conditions that can later support the establishment of native planted tree saplings (Gómez-Aparicio, 2009; Padilla & Pugnaire, 2006). This ecological phenomenon known as "facilitation" has been adopted in the restoration of soil functions in novel ecosystems, where restoration fails because of harsh environmental conditions (Gómez-Aparicio et al., 2004). Through facilitation, "Nurse Grass Effect" creates a better soil condition because of the ability of the invasive plants to efficiently capture resources that are limiting under more harsh environmental conditions and use them in above and belowground biomass accumulation, which in turn adds labile organic matter to the soil (Jordan, Larson, & Huerd, 2008; Maestre, Bautista, Cortina, & Bellot, 2001; Wolkovich, Bolger, & Cottingham, 2009). The extensive architecture of their fibrous roots also facilitates a rhizosphere effect and improves the physical structure of the soil. Indeed, our results suggest that this ecological concept of using nurse grass to facilitate the establishment of planted saplings holds promising potentials to support the recovery of native species in plots under FFM cover. However, it is yet to be adopted in AOSR as a reclamation practice for constructed uplands.

5 | CONCLUSIONS AND RECOMMENDATIONS FOR UPLAND RECLAMATION PROJECTS

This study tested the effect of topographic gradient under different cover substrates, on the ecohydrological functioning of a constructed upland, 3 years postconstruction. Topographic gradient had no significant effect on the ecohydrological functioning of the constructed upland likely due to heterogeneity of the cover substrates and the lack of a developed soil structure, typical of newly reclaimed soils, which limits gravitational influences on moisture redistribution. Hence, in the early years following reclamation, the trajectory of ecohydrological functions in a constructed upland is largely dependent on the functional characteristics of the cover substrates. Although the sample size may not have provided the best representation of site heterogeneity, it provided a reasonable baseline information for future studies to build on. By comparing key attributes of FFM and PMM, our findings suggest that the hydrophysical characteristics that will support moisture redistribution are more efficient under FFM, whereas those that support moisture retention and pedogenic process of root development are more efficient in PMM. On the other hand, the macronutrients required for vegetation establishment are readily available in FFM, whereas the PMM contains high concentrations of ions that support the exchange of nutrients between plants and soil solution. Hence, to facilitate the recovery of ecohydrological functioning in constructed uplands, the combination of FFM and PMM across and within the constructed soil layers will likely yield the best results. Across the upland, PMM would be a suitable cover substrate for the creation of wet ecosites such as transitional riparian zones, whereas if placed within the constructed soil layers as an interface horizon between FFM and tailing sand, PMM could facilitate the retention of infiltrating moisture and limit nutrient losses through run-off and leaching.

Colonization of the constructed upland by vegetation functional groups was mainly influenced by the hydrophysical and physicochemical properties of the cover substrate. Although FFM appears to have facilitated the colonization of both invasive and native species, peat substrates have often demonstrated favourable conditions for the survival of planted saplings, therefore reducing the need for early fertilization of reclaimed soils. Our findings also suggest that fertilizer application at the early stage of soil development is not economically viable, because the underdeveloped-constructed soil cannot support edaphic conditions required by planted saplings to efficiently access the applied fertilizer with minimal loss to the environment. Thus, we recommend that future upland reclamation strategies should monitor the stability in physicochemical soil properties prior to the application of inorganic soil amendments such as fertilizer, in order to reduce potential loss through run-off and leaching. The proliferation of non-native invasive species in heavily disturbed constructed soils should be considered a successional milestone because the autogenic processes initiated by these invasive-pioneer species can help accelerate pedogenic development and facilitate the establishment of planted saplings in reclaimed upland ecosystems.

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