Net Nitrogen Mineralization in Boreal Fens: A Potential Performance Indicator for Peatland Reclamation

Journal:	<i>Botany</i>
Manuscript ID	cjb-2015-0263.R2
Manuscript Type:	Article
Date Submitted by the Author:	05-May-2016
Complete List of Authors:	Hartsock, Jeremy; Southern Illinois University, Plant Biology House, Melissa; Southern Illinois University, Plant Biology Vitt, Dale; Southern Illinois University Carbondale, Plant Biology
Keyword:	Nitrogen mineralization, Peatland, Reclamation, Nitrification

Title: Net Nitrogen Mineralization in Boreal Fens: A Potential Performance Indicator for Peatland Reclamation

Authors: Jeremy A Hartsock; Melissa House; Dale H Vitt

Jeremy A Hartsock (corresponding author): Center for Ecology and Department of Plant Biology, Southern Illinois University, Carbondale, IL 62901, USA; Email: hartsock@siu.edu; Phone: 309-721-1323

Melissa House: Center for Ecology and Department of Plant Biology, Southern Illinois University, Carbondale, IL 62901, USA; Email: mhouse@siu.edu

Dale H Vitt: Center for Ecology and Department of Plant Biology, Southern Illinois University, Carbondale, IL 62901, USA; Email: dvitt@siu.edu

Draft

Abstract

The Sandhill Fen reclamation watershed, commissioned by Syncrude Canada, Ltd, is the first attempt to reclaim a self-sustaining peat-forming wetland on a previously mined area. Here, we quantified net nitrogen mineralization rates at Sandhill Fen in the first and second years since initiation (2013-2014).

Our main objective was to determine if nitrogen production potentials at Sandhill Fen were similar to six regional fens sampled across an ombrotrophic-minerotrophic peatland gradient. In the second year, net nitrogen mineralization rates across Sandhill Fen (2014 mean = $20.2 \text{ mg N m}^{-2} \text{ d}^{-1}$; $0.9 \text{ mg N kg}^{-1} \text{ d}^{-1}$) were quite comparable to the benchmark fen sites (2013-2014 pooled means = $20.6 \text{ mg N m}^{-2} \text{ d}^{-1}$; $5.9 \text{ mg N kg}^{-1} \text{ d}^{-1}$). However, in areas exhibiting low gravimetric soil moisture content at Sandhill Fen, net nitrification contributed more than 50% to the net N mineralization total, an uncommon observation in natural fen type wetlands. These findings highlight the importance of managing soil moisture levels during the early stages of reclamation to 1) maintain relatively anaerobic soil conditions, and 2) facilitate microbial-mediated processes to fall within an acceptable range of variation comparable to undisturbed Albertan fens.

Keywords: Nitrogen mineralization, Peatland, Reclamation, Nitrification

Introduction

In the Athabasca oil sands region (AOSR) (northeastern Alberta, Canada), extensive disturbance from open pit mining for 'oil sands' has radically altered the landscape (Woynillowicz 2005). These disturbances have affected surface hydrology (Price et al. 2009), and compromised regional wetlands that are important habitats for Alberta's plant and wildlife species (Foote 2012). Approximately 650 km² of the boreal plain area are currently affected by oil sands exploration, and by year 2030, the projected area disturbed by oil sands activities may triple (Government of Alberta 2011). Oil sands mining companies such as Syncrude Canada, Ltd. and Suncor Energy, Inc. are required to reclaim these disturbed areas, and return previously mined sites into ecological equivalents similar to undisturbed regional wetlands, namely, peat-forming fens, in order to meet mine closure policy requirements (Purdy et al. 2005; Price et al. 2009; Trites and Bayley 2009a; Rooney and Bayley 2011, Foote 2012). Large scale reclamations on previously mined landscapes (i.e., areal extent >50 hectares in size) are quite different from smaller scale (areal extent <5 hectares in size) steam assisted gravity drainage (SAGD) reclamations on mineral soils (Vitt et al. 2011). For example, in-pit (i.e., previously mined areas infilled with tailings sediments and processed water) reclamations generally involve 1) constructing highly engineered watersheds, 2) capping underlying tailings sediments with available reclamation substrates (i.e., overburden, subsoil mineral substrates) to prevent upward movement of compounds such as sodium, chloride, sulfate, and naphthenic acids, and 3) planting desirable wetland species tolerant to high concentrations of dissolved ions (Trites and Bayley 2009a; Price et al. 2009; Rooney and Bayley 2011, Foote 2012; Pouliot et al. 2012; Rezanezhad et al. 2012; Wytrykush et al. 2012). Determining whether these novel wetland systems are representative of undisturbed regional peat-forming fens is challenging, given the many structural, chemical, and functional components that characterize boreal fens at all trophic levels.

Fen type wetlands occupy approximately 63% of the peatland area of the province of Alberta (Vitt et al. 2000), and are important regional sinks of carbon and nitrogen (N) (Gorham 1991; Loisel et al. 2014). Boreal fens are separated into four primary types, poor fens, moderate rich fens, extreme rich fens, and regionally rare saline fens based on specific floristic and chemical attributes (saline fens were not monitored in this study, and are not further described). Poor fens exhibit low surface-water pH values ranging from 4.0-5.5, low reduced electrical conductivity (EC) values below $50 \mu\text{S cm}^{-1}$, and are dominated by a ground layer of *Sphagnum* mosses. Moderate rich fen surface-waters exhibit pH ranges from 5.6-7.0, and EC values typically less than $150 \mu\text{S cm}^{-1}$ (Vitt et al. 1995). For extreme rich fens, surface-water pH ranges from 7.0-8.5, and EC values can exceed $500 \mu\text{S cm}^{-1}$. Both moderate and extreme rich fens are dominated by a ground layer of true mosses, however, moderate rich fens may harbor some mesotrophic *Sphagnum* species (Vitt and Andrus 1977). The overstory of both poor and rich fens can be shrub or tree-dominated, or in many cases, no woody vegetation is present and the system is dominated by mosses and sedges (see Vitt and Chee 1990 for list of species that define specific fen site types).

Under ideal climatic conditions, the vegetation in boreal fens exhibit high rates of net primary production (NPP) over the summer growing season (approximately $300 \text{ g m}^{-2} \text{ yr}^{-1}$; Bartsch and Moore 1985; Campbell et al. 2000) that is sustained in part by microbial-mediated mineralization of organic matter, which supplies both organic and inorganic forms of N to peatland plants and microbes (Bayley et al. 2005; Basiliko et al. 2005; Wiedermann et al. 2009). Nitrogen mineralization is a bacterial and fungal mediated process that converts organically bound N into ammonium (Schimel and Bennett 2004). In inorganic soils and peat, bacteria and fungi excrete enzymes able to break C-N covalent bonds, competing to immobilize inorganic N liberated in these reactions. The total sum of ammonium produced in an ecosystem (per gram, volume, area, or time) is termed gross N mineralization. Net N mineralization refers to the total amount of available inorganic N remaining following bacterial/fungal immobilization

and denitrification, and is an index of plant available N (Hart et al. 1994; Schimel and Bennett 2004; Booth et al. 2005). Net N mineralization rates vary considerably among ecosystems depending upon whether carbon or N is the primary limiting nutrient to soil heterotrophs (McGill and Cole 1981; Damman 1988). The most common index of carbon and/or N limitation is based on soil C:N ratios. Soil C:N ratios exceeding 30:1 are indicative of N limited systems that typically have slow net N mineralization rates due to rapid N immobilization by soil microbes (Schimel and Bennett 2004). When substrate C:N ratios are < 30:1, however, net N mineralization potentials increase (Damman 1988). Net N mineralization rates in boreal fens are controlled by several edaphic and abiotic factors (e.g., soil C:N ratios, the degree of soil humification, aeration, soil moisture levels, pH, temperature), and rates are highly variable across differing peatland site types, ranging from net N immobilization to fast net N mineralization rates exceeding $40 \text{ mg N m}^{-2} \text{ day}^{-1}$ (Humphry and Pluth 1996; Bayley et al. 2005; Wray and Bayley 2008; Wood et al. 2015). In general, net N mineralization rates in rich fens are faster than *Sphagnum*-dominated peatlands due to higher soil N contents in rich fens, and net nitrification rates are slow for both fen types (Regina 1996; Bayley et al. 2005) due to the relatively anaerobic conditions characteristic of these systems. Nitrogen mineralization differences between fen types additionally relate to differences in overall decomposition/production values, with poor fens having less decomposition and low NPP compared to rich fens (Bayley et al. 2005; Vitt et al. 2009).

Few studies have quantified nutrient cycling potentials of soil substrates used for oil sands wetland reclamations (McMillan et al. 2007; Hemstock et al. 2010; Mackenzie and Quideau 2012). In a controlled laboratory experiment, Mackenzie and Quideau (2012) found that net N mineralization potentials for stockpiled peat-mineral soils were slower (about $0.5 \text{ mg N kg}^{-1} \text{ day}^{-1}$) than net N mineralization rates observed in undisturbed Albertan fens (Bayley et al. 2005). However, the effects of environmental factors (e.g., temperature, total precipitation, and soil moisture gradients) may cause unexpected nutrient turnover patterns in some reclaimed wetland systems that may not manifest in a

controlled laboratory setting. Designing field net N mineralization experiments on reclaimed sites therefore are important to 1) assess abiotic effects on N mineralization, and 2) determine if inorganic N production potentials are sufficient to meet plant and microbial N demands; seeing that Yu et al. (2010) showed fens exhibit the highest productivity rates subsequent to initiation. Moreover, N deficiencies at reclamation sites may reduce fitness of desirable fen species (i.e., planted species) during early stages of wetland reclamation, potentially enabling colonization by undesirable transient plants whose presence has long-term effects on community structure (Trowbridge 2007). On the other hand, nutrient supply rates that exceed the typical range of variation observed in boreal fens may create niche environments preferred by monotypic stands of either *Calamagrostis canadensis* or *Typha latifolia* (Wild et al. 2001; Drexler and Bedford 2002; Zedler and Kercher 2004), highlighting the importance for nutrient availability to fall within an acceptable range at recently reclaimed sites.

In this field study, we assess net N mineralization, net nitrification, water chemistry patterns, and soil characteristics at a recently constructed fen reclamation site, and compare observations to six benchmark fens in the AOSR. From these data, we attempt to capture the entire range of net N mineralization variance across the diverse gradient of regional fen types, and determine if inorganic N production rates at a recently constructed fen reclamation site are on trajectories to exhibiting rates equivalent to undisturbed fens. We also assess the major controls on N mineralization and nitrification, in order to determine if the factors controlling nutrient turnover at a reclamation site are similar to benchmark fens.

Materials and Methods

Climate

Weather records for precipitation and mean daily temperatures over the 2013 and 2014 growing seasons (May – September) were obtained from the Fort McMurray airport weather station database

(<http://www.theweathernetwork.com>) located about 50 km south of the reclamation site. The twenty-year total precipitation and mean annual air temperature averages (1994-2014) for Fort McMurray, AB were 406 ± 87 cm and $1.4 \pm 1.1^{\circ}\text{C}$, respectively.

Site Descriptions

Benchmark sites

The six regional benchmark fens monitored in this study are located within a 100 km driving radius of Fort McMurray, AB. Benchmark sites were selected based on indicators that captured the full range of chemical and floristic variability characteristic of fens in the AOSR (with the exception of saline fens).

High-Test Rich Fen (abbreviation: ERF) (56.950 N, -111.531 W; 314 m elevation) is an open patterned extreme rich fen with a ground layer dominated by the moss *Scorpidium scorpioides*, and by the *Carex* species *C. chordorrhiza*, and *C. lasiocarpa*. The site contains sporadic populations of *Triglochin maritima*, and *Campyllum stellatum*. Hummocks are absent from the fen interior.

Anzac Rich Fen (abbreviation: MRF1) (56.382 N, -111.026 W; 466 m elevation) is an open moderate rich fen closely flanked by an ombrotrophic bog to the west, and by Highway 881 to the east (repaved in 2006). The site recently has developed over the last 8 years following road construction disturbance, and accumulated peat deposits are quite shallow (< 20 cm). The site is dominated by an especially dense cover of the early successional sedge *Carex aquatilis*. Within the fen main body are sporadic occurrences of mosses within the genera *Calliergon*, and *Drepanocladus*. Along the western edge nearest to the bog, *Sphagnum* becomes increasingly abundant. Hummocks are absent from the fen main body.

Stoney Mountain Rich Fen (abbreviation: MRF2) (56.365 N, -111.276 W; 726 m elevation) is a shrubby moderate rich fen dominated by several species of *Salix*, *Carex aquatilis*, *Carex utriculata*, and *Calamagrostis canadensis*. Hummocks are colonized by *Aulacomnium palustre* and *Sphagnum teres*, hollows are colonized by mosses within the genera *Calliergon* and *Drepanocladus*.

Beaver Pond Rich Fen (abbreviation: MRF3) (56.378 N, -111.253 W; 763 m elevation) is an open moderate rich fen with a floating mat. The site is dominated by *Carex* species, namely *C. aquatilis* and *C. utriculata*, and a ground layer composed largely of *Sphagnum squarrosum*. The site receives water inputs from a spillover culvert that drains a beaver pond. Over the 2014 growing season, a trench developed below the outflow culvert, and water was diverted away from the site. Consequently, the water table in 2014 was markedly lower compared to year 2013.

Stoney Mountain Poor Fen (abbreviation: PF1) (56.364 N, -111.276 W; 725 m elevation) is dominated by a ground layer of *Sphagnum* species (*S. angustifolium*, *S. magellanicum*, *S. fuscum*, and *S. majus*). *Carex aquatilis* is the dominant sedge present at the site. Other vascular plants include *Andromeda polifolia*, *Kalmia polifolia*, and *Oxycoccus microcarpus*.

Mariana Lake Poor Fen (abbreviation: PF2) (55.898 N, -112.095 W; 695 m elevation) is an oligotrophic site dominated by a ground layer of *Sphagnum* species, namely, *S. angustifolium*, *S. fuscum*, and *S. magellanicum*. The dominant vascular species include *Carex aquatilis*, *Kalmia polifolia*, *Andromeda polifolia*, *Scheuchzeria palustris*, *Eriophorum angustifolium* and *Oxycoccus microcarpus*.

Reclamation site

Sandhill Fen reclamation watershed (abbreviation: SHF) (57.040 N, -111.596 W; approximately 50 hectares in size; 310 m elevation) overlays a previously mined in-pit (East In-pit) located near Syncrude Canada, Ltd.'s main operations center. Construction of the watershed began in 2008, and was fully commissioned in 2012. The wetland portion of the watershed (approximately 17 hectares) is flanked on both sides by seven upland hills constructed to provide habitat for upland species. In fall 2010, peat and small amounts of underlying clay mineral were harvested from a nearby peatland complex, stored approximately 4 months, and spread throughout the wetland in January 2011. Peat was leveled by a bulldozer, and prior to flooding, the final peat layer thickness was about 0.5 meters. Seeding of the wetland main body with several wetland vascular plant species was completed in fall 2011. Active

pumping of lake water into the watershed facilitated initial site flooding in August 2012. The wetland water table in 2013, however, was highly variable due to active draining and subsequent re-wetting of the fen to allow testing and/or proper calibration of water balance monitoring instrumentation. In general, in early June 2013, the majority of the wetland area was quite saturated with water. Nearly all standing-water was actively drained from the site in mid-June, and active re-wetting did not occur until late June because of mechanical difficulties with the pumping system. Consequently, for an approximate two-week period in June, the hydration status of SHF soil was dissimilar to benchmark fen sites (data not shown). In 2014, active pumping of water was suspended, and water levels were not further managed. The water table at SHF is highest following spring thaw periods, and decreases as the growing season progresses due to evapotranspiration and hydrologic outflow. At the end of growing season 2014, the dominant vascular plant species within the wetland main body was *Carex aquatilis*. In areas with ponded water, the marsh species *Typha latifolia* was common, and in areas prone to late season drying, the grass, *Calamagrostis canadensis*, was dominant. Vegetation patterns and species assemblages across SHF are described in Vitt et al. (2016). Detailed information on the construction and engineering of SHF are found in Wytrykush et al. (2012) and Vitt and House (2015).

Experimental Design

Across the six benchmark fen sites, in years 2013 and 2014, research plots (n=8 for each site) were established randomly along transects bisecting the interior of each peatland. At SHF, in fall 2011, research plots (n=20; 10m x 10m) were established in locations where it was predicted water would saturate the soil surface subsequent to site flooding. Several unforeseen watershed design factors at SHF, however, constricted the wetland area from the proposed wetland boundaries. Consequently, some research plots were localized in areas that remained moist over the growing season, and others occupied areas where water tables were below the placed peat surface. To account for within-site

moisture gradient differences at SHF, the twenty original research plots were separated into 'wet' and 'dry' groupings (n=10 for each group), and treated as two independent sites (SHF wet and SHF dry), thereby capturing potential consequences of low soil moisture levels on ecosystem processes during the early stages of wetland reclamation. The presence or absence of standing water across SHF research plots was assessed in June, July and September of each year. Wet plots were those that remained submerged for at least one month each growing season, and for dry plots the soil surface was exposed above the water table throughout the entirety of both growing seasons.

Net N mineralization rates were quantified using the *in situ* buried polyethylene bag technique (Robertson 1999; Bayley et al. 2005). In mid-June (2013 and 2014), soil cores (0-10 cm) were collected with a cylindric aluminum corer (6.5 x 10 cm), halved longitudinally with a serrated bread knife, and placed into separate polyethylene Whirl Pak® bags (hereafter referred to as initial extractable N core, and final extractable N core) at field moisture content. At SHF, soil cores were collected two days after the fen drainage event in 2013, and soil cores were still quite moist across the wet plots. Live plants were removed from each soil core prior to placement into Whirl Pak® bags. Immediately after collection from the field, initial extractable N cores were taken to a laboratory for processing and extraction (60 min per extraction) with 1.0 molar KCl to displace initial extractable ammonium and nitrate contents (modified from Verhoeven et al. 1990). For the final extractable N cores, after being placed into polyethylene Whirl Pak® bags, samples were reburied in the same peat profile location from which they were removed. Incubation cores remained in the peat matrix for approximately 30 days. Following the field incubation period, final extractable N cores were collected and immediately taken to a laboratory for processing and extraction. Net N mineralization was calculated from the difference between final extractable ammonium + nitrate contents (obtained from the field incubation cores) and initial extractable ammonium + nitrate contents (obtained from the initial cores) (Robertson 1999). Net nitrification was calculated from the difference between final extractable nitrate contents and initial

extractable nitrate contents (Robertson 1999). Extractable ammonium and nitrate concentrations were measured on a N autoanalyzer using the alkaline phenol and cadmium reduction methods, respectively.

In addition to measuring N mineralization, additional volumetric peat cores (6.5 x 10 cm) and surface-water samples were collected several times over the 2013 and 2014 growing seasons (June–September) to assess differences both across sites and between years for soil characteristics (e.g., gravimetric soil water content, bulk density, % organic matter (as loss on combustion), total N, total carbon, and soil temperature) and water chemistry patterns (e.g., pH, EC, and dissolved ammonium and nitrate). For sites exhibiting surface-water pH < 5.5, reported EC values were reduced for H⁺ ions (Sjörs 1950). Seasonal surface-water samples were collected in acid-washed 60 mL HDPE bottles. Water samples were filtered with whatman 541 filter paper and frozen on the same day of collection. Reduced EC and pH were measured in the lab at 20°C. Ammonium and nitrate concentrations were determined colorimetrically using the aforementioned methods.

Statistical Analyses

Differences across study sites for each parameter were assessed using a two-way analysis of variance (ANOVA) design with site and year as fixed factors. Because surface-water was absent from dry plot localities at the reclamation site, reported surface-water observations at SHF were from wet plots only. Non-normal datasets were square root or log transformed to improve homoscedasticity. Significance was assigned at $\alpha = 0.05$. Post hoc comparisons were made using Tukey's test. All two-way ANOVA tests were conducted using R version 3.1.1. Single-factor regression analyses were used to coarsely assess surface-water and soil properties that controlled net N mineralization and nitrification within each peatland site type (observations from peatlands of the same type were pooled for regression analyses). We subsequently examined whether the same properties controlling N mineralization and nitrification in SHF wet and dry plots were similar to benchmark sites. All datasets used for regression analyses were

log transformed (except for pH) to improve homoscedasticity (Booth et al. 2005). Because surface-water was absent from dry plot localities at the reclamation site, determining relationships with surface-water characteristics was precluded.

Results

Climate Patterns

Total growing season (May – September) precipitation was nearly equivalent between 2013 and 2014 (301 and 306 mm, respectively - Fig. 1). However, heavy rainfall prior to buried bag placement in 2013 noticeably raised water tables across the benchmark fen sites. Consequently, differences between years for some soil characteristics likely were due to the inundated conditions prior to sampling in 2013. For growing season temperatures, year 2013 experienced an additional 303 cumulative degree-days compared to year 2014 (2366 and 2063 total degree-days, respectively - Fig. 1). Total annual precipitation and mean annual air temperatures for years 2013 (390 cm and 0.8°C) and 2014 (407 cm and 0.1°C) were similar to historic (1994-2014) Albertan climate patterns.

Surface-water Chemistry

There was a significant interaction between the factors site and year for surface-water pH ($F_{6, 324}=8.9$; $p<0.001$), however, MRF2 was the only site that experienced a significant change in pH between the two years (Table 1). Surface-water pH ranged from 3.8-9.9, with SHF wet plots and ERF exhibiting the most basic conditions, while the poor fens were the most acidic. The moderate rich fens MRF1 and MRF2 were slightly basic, whereas MRF3 was slightly acidic. There was a significant interaction between the factors site and year for surface-water EC ($F_{6, 323}=37.3$; $p<0.001$), and SHF wet plots were the only site to experience a change in EC between years (Table 1). Surface-water EC values were quite low at the moderate rich and poor fen sites and ranged from 7 - 160 $\mu\text{S cm}^{-1}$ over the two growing seasons. In

2013, EC across wet plots at SHF (range: 190 - 992 $\mu\text{S cm}^{-1}$) were most similar to the ERF site (range: 192 - 448 $\mu\text{S cm}^{-1}$), however, in 2014, surface-water EC across SHF wet plots increased markedly (range: 875 - 2550 $\mu\text{S cm}^{-1}$) beyond what is typical for Albertan fens (with the exception of saline fens). For surface-water ammonium and nitrate concentrations, our observations across benchmark sites were up to ten times higher than concentrations reported by Vitt and Chee (1990), Bayley et al. (2005), and Vitt et al. (2009), and we are uncertain why these discrepancies are present. There was a significant effect of site for surface-water ammonium ($F_{6,329}=19.0$; $p<0.001$), and concentrations generally were highest across the *Sphagnum*-dominated systems (Table 1). For dissolved nitrate in surface-water, a weak interaction was observed ($F_{6,323}=2.1$; $p=0.049$; no post hoc differences) with no differences across sites ($F_{6,323}=1.7$; $p=0.122$) or between years ($F_{1,323}=1.7$; $p=0.195$) (Table 1). Summary statistics for surface-water properties are presented in supplementary table 1 (S1).

Soil Analysis

For gravimetric soil water content, differences were apparent across sites only ($F_{7,373}=194.0$; $p<0.001$) (Table 2). At the benchmark sites, gravimetric soil water content was greater than 90%, however, for SHF wet and dry plots, mean gravimetric soil water content was markedly lower (73% and 61%, respectively). Differences for soil bulk density (0 – 10 cm cores) were apparent across sites only ($F_{7,378}=421.2$; $p<0.001$) (Table 2). Bulk density was exceptionally high across SHF wet and dry plots, and linked across the benchmark fens by peatland site type, with SHF wet and dry plots > ERF, MRF1-3 > PF1-2. For soil organic matter content, sampling was carried out at SHF in 2013 and 2014, and in 2013 only for benchmark sites; thereby precluding statistical analysis. In general, soil organic matter trends were inversely related to bulk density, with SHF wet and dry plots < ERF, MRF1-3 < PF1-2 (Table 2). For N contents in soil, differences were observed across sites only ($F_{7,378}=100.8$; $p<0.001$) (Table 2). Soil N contents generally were highest across rich fens, intermediate across wet and dry plots at SHF, and

lowest across the poor fens. For total soil carbon contents, differences were observed both across sites ($F_{7,377}=70.3$; $p<0.001$) and between years ($F_{1,377}=7.8$; $p=0.006$) (Table 2). Soil carbon contents were slightly higher in 2013 compared to 2014 (40% vs. 38%, respectively). Across sites, the poor fens exhibited the highest soil carbon concentrations, values were intermediate across rich fens, and lowest across wet and dry plots at SHF. For soil C:N ratios, differences were observed across sites only ($F_{7,378}=594.8$; $p<0.001$). Poor fen C:N ratios ranged from 39:1 - 83:1, and from 13:1 - 39:1 across the rich fens and SHF wet and dry plots. For soil temperatures (0-10 cm below green moss segments) there was a significant interaction between the factors site and year ($F_{7,370}=5.4$; $p<0.001$) (Table 2). Between years, peat temperatures were higher in 2013 (mean=15.8°C) compared to 2014 observations (mean=12.9°C), especially for the moderate rich fens. The only site exhibiting an increase in mean soil temperature between years 2013 and 2014 was PF2. Summary statistics for soil characteristics are presented in supplementary table 2 (S2).

Soil Extractable Nitrogen Contents

There was a significant interaction between the factors site and year for soil extractable ammonium contents (0-10 cm cores) on both areal and gravimetric normalization metrics ($F_{7,119}=17.5$; $p<0.001$ and $F_{7,118}=15.2$; $p<0.001$, respectively) (Fig. 2a-b). Between years, extractable ammonium contents were markedly higher across all sites in 2014 (areal mean= 830.3 mg N m⁻²; gravimetric mean=181.4 mg N kg⁻¹) compared to 2013 (areal mean=182.8 mg N m⁻²; gravimetric mean=21.4 mg N kg⁻¹) (Fig. 2a-b). For soil extractable nitrate, patterns were similar to extractable ammonium observations, and interactions were observed on both areal and gravimetric normalization metrics ($F_{7,116}=13.0$; $p<0.001$ and $F_{7,114}=12.0$; $p<0.001$, respectively) (Fig. 3a-b). Between years, extractable nitrate contents were markedly higher across all sites in 2014 (areal mean= 214.0 mg N m⁻²; gravimetric mean=87.9 mg N kg⁻¹) compared to 2013 (areal mean=9.9 mg N m⁻²; gravimetric mean=1.0 mg N kg⁻¹) (Fig. 3a-b). Overall, we suspect that

heavy rainfall prior to sampling in 2013 mobilized extractable N contents from soil exchange sites in the peat profile, given the wide discrepancies between years across all sites. Summary statistics for extractable nutrients are presented in supplementary table 3 (S3).

Nitrogen Mineralization and Nitrification

There was a significant interaction between the factors site and year for net N mineralization rates on an areal basis ($F_{7,118}=3.2$; $p=0.004$) (Fig. 4a). On a gravimetric basis, differences were observed across sites only ($F_{7,124}=11.2$; $p<0.001$) (Fig. 4b). At benchmark sites, areal and gravimetric rates ranged from $-1.9 - 82.2 \text{ mg N m}^{-2} \text{ day}^{-1}$ and $-12.9 - 26.6 \text{ mg N kg}^{-1} \text{ day}^{-1}$, respectively (Fig. 4a-b). At the SHF reclamation site, N mineralization rates exhibited considerable overlap with the benchmark fens. Across wet plots, areal and gravimetric rates ranged from $-1.9 - 53.9 \text{ mg N m}^{-2} \text{ day}^{-1}$ and $-0.3 - 3.5 \text{ mg N kg}^{-1} \text{ day}^{-1}$, respectively (Fig. 4a-b). Across dry plots, net N mineralization ranged from $0.6 - 64.2 \text{ mg N m}^{-2} \text{ day}^{-1}$ and $-0.6 - 3.1 \text{ mg N kg}^{-1} \text{ day}^{-1}$ (Fig. 2a-b). Net N mineralization rates (pooled across the 2013 and 2014 growing seasons) generally were fastest across the moderate rich fens (MRF1-3 = $29.9 \text{ mg N m}^{-2} \text{ day}^{-1}$; $8.1 \text{ mg N kg}^{-1} \text{ day}^{-1}$), intermediate at the extreme rich fen (ERF = $18.5 \text{ mg N m}^{-2} \text{ day}^{-1}$; $4.3 \text{ mg N kg}^{-1} \text{ day}^{-1}$), and slowest across the poor fens (PF1-2 = $7.9 \text{ mg N m}^{-2} \text{ day}^{-1}$; $3.2 \text{ mg N kg}^{-1} \text{ day}^{-1}$). Across SHF wet plots, areal and gravimetric means (2013-2014) were $17.0 \text{ mg N m}^{-2} \text{ day}^{-1}$ and $0.8 \text{ mg N kg}^{-1} \text{ day}^{-1}$, respectively. For SHF dry plots, areal and gravimetric N mineralization means ($14.0 \text{ mg N m}^{-2} \text{ day}^{-1}$; $0.6 \text{ mg N kg}^{-1} \text{ day}^{-1}$) were nearly equivalent to wet plot observations.

For net nitrification, there were significant effects for the factors site ($F_{7,126}=13.1$; $p<0.001$) and year ($F_{1,126}=14.4$; $p<0.001$) on an areal basis only, and no differences were observed on a gravimetric basis (Fig. 5a-b). Across benchmark sites and SHF wet plots, net nitrification rates were slow and contributed little to the net N mineralization total for both areal and gravimetric normalization metrics (Fig. 5a-b). For SHF dry plots, however, nitrification rates were quite fast (mean = $10.2 \text{ mg N m}^{-2} \text{ day}^{-1}$

and $0.3 \text{ mg N kg}^{-1} \text{ day}^{-1}$), and exceeded rates reported in previous Albertan studies (Bayley et al. 2005, Wood et al. 2015). Although differences were slight between years, areal nitrification rates in 2013 were slower than 2014 observations (mean = 1.1 and $3.7 \text{ mg N m}^{-2} \text{ day}^{-1}$, respectively). Summary statistics for net N mineralization and nitrification are presented in supplementary table 4 (S4).

For single-factor regression analyses, we only reported relationships with slopes significantly different from zero (Tables 3-4). Plotting net N mineralization rates against surface-water and soil characteristics across the benchmark sites yielded several weak correlations (R^2 fit values <0.30), suggesting that N mineralization in these systems were controlled by many interacting properties (Table 3). At the extreme rich fen (ERF), no surface-water or soil properties correlated with net N mineralization observations, however, the likelihood of distinguishing causal effects at ERF was low due to low statistical power from small sample size ($n=16$) (Table 3). Net N mineralization across the MRF sites were positively correlated with surface-water pH on both areal and gravimetric metrics (areal $R^2=0.15$; gravimetric $R^2=0.21$) (Table 3). Soil extractable nitrate contents yielded a negative correlation with net N mineralization across MRF sites (gravimetric $R^2 = -0.18$), whereas increasing soil temperatures correlated with fast N mineralization rates (areal $R^2=0.09$). Surface-water nitrate concentrations weakly correlated with net N mineralization on a gravimetric basis only ($R^2=-0.11$). Overall, all properties potentially influencing N mineralization patterns across the MRF sites yielded weak regression fit lines, whereby R^2 values ranged from $0.09 - 0.21$ (Table 3). Across the poor fen benchmark sites (PF1-2), net N mineralization was positively correlated with increasing gravimetric soil water content only (areal $R^2 = 0.29$) (Table 3), suggesting that N mineralization potentials may decrease under moisture deficient conditions in poor fens. At the reclamation site, SHF wet plots were positively correlated with dissolved nitrate concentrations (areal $R^2=0.50$; gravimetric $R^2=0.48$), surface-water EC (gravimetric $R^2=0.57$) and soil extractable ammonium (gravimetric $R^2=0.40$) (Table 3). For SHF dry plots, net N mineralization was positively correlated with soil extractable ammonium on an areal basis only ($R^2=0.25$) (Table 3).

Plotting net nitrification rates against surface-water and soil properties at benchmark sites produced several strong correlations, suggesting that nitrification is a tightly controlled process in fens (Table 4). At the extreme rich fen, nitrification was positively correlated with a myriad of factors such as pH (areal $R^2=0.39$; gravimetric $R^2=0.63$), surface-water ammonium concentrations (areal $R^2=0.50$; gravimetric $R^2=0.56$), soil extractable ammonium (areal $R^2=0.81$; gravimetric $R^2=0.82$) and extractable nitrate contents (areal $R^2=0.58$; gravimetric $R^2=0.63$) (Table 4). Across MRF sites, in contrast to N mineralization observations, nitrification potentials decreased as a function of increasing surface-water pH (areal $R^2= -0.30$; gravimetric $R^2= -0.28$) (Table 4). Other processes across MRF sites that positively influenced nitrification rates were surface-water nitrate concentrations (areal $R^2=0.09$), and soil extractable ammonium (gravimetric $R^2=0.12$) and nitrate contents (gravimetric $R^2=0.21$) (Table 4). Surface-water nitrate concentrations were the only factor that correlated with nitrification across the poor fens, and the relationship was quite weak (areal $R^2=-0.13$) (Table 4). At the reclamation site, similar to the extreme rich fen and moderate rich fen sites, several factors influenced nitrification rates. Across wet plots, nitrification correlated positively with soil extractable ammonium contents (areal $R^2=0.48$; gravimetric $R^2=0.52$), surface-water EC (gravimetric $R^2=0.47$), soil moisture (areal $R^2=0.34$), and soil C:N ratios (areal $R^2=0.29$) (Table 3). Nitrification was negatively correlated with soil bulk density (areal $R^2= -0.27$) and surface-water pH (gravimetric $R^2= -0.49$). For dry plots, bulk density was the only factor that correlated with nitrification ($R^2= -0.21$), and produced a negative relationship similar wet plot observations.

Discussion

The Sandhill Fen (SHF) experimental watershed is Syncrude Canada, Ltd's first attempt to construct wetlands on a previously mined in-pit. Planning for the reclamation project was highly collaborative and spanned many scientific disciplines (e.g., vertebrate ecology, invertebrate ecology, plant ecology,

microbial ecology, soil biogeochemistry, and hydrology). Here, our discussion is centered on surface-water properties, soil characteristics and net N mineralization rates at SHF after two years of reclamation. In order to assess the current status and/or potential succession trajectories of SHF, we compared observations to six regional benchmark fens that encompass three unique fen site types (e.g., extreme rich, moderate rich, and poor fens).

Surface-water Properties

For most surface-water properties measured, observations across benchmark sites coincide with other Albertan fen studies (Vitt and Chee 1990; Bayley et al. 2005; Locky and Bayley 2010b). For the reclamation site, active draining and subsequent refilling of the wetland over the 2013 growing season surely influenced surface-water properties in the first year. However, surface-water properties at SHF in 2013 were quite comparable to the ERF site, namely for pH, EC, dissolved ammonium, and dissolved nitrate concentrations. In the second growing season (2014), active water table management practices were suspended; leading to changes in hydrologic outflow, and higher residence time of surface-waters. Consequently, mean surface-water pH levels decreased slightly across SHF wet plots in the second growing season, potentially from the accumulation of dissolved organic acids that previously were flushed from the site during draining events in 2013. Surface-water EC levels increased sharply in the second year beyond what is typical for Albertan fens (with the exception of saline fens), and may exceed threshold levels ($>2000 \mu\text{S cm}^{-1}$) detrimental to some ion-sensitive Ephemeroptera (mayfly) species in the near future (Echols et al. 2010).

Soil Properties

Across benchmark sites, soil parameter observations were consistent with other Canadian fen studies (Bayley et al. 2005; Vitt et al. 2009; Locky and Bayley 2010b; Macrae et al. 2013; Andersen et al. 2013;

Wood et al. 2015). Overall, the MRF1 site was an outlier, and exhibited many soil characteristics that differed from MRF2-3. Soil N and carbon contents at MRF1 generally were lower than MRF2-3, and may be attributed to the relatively young age of MRF1 (approximately 8 years since initiation). Species richness at MRF1 was low (data not shown) and the early succession sedge *Carex aquatilis* was quite abundant. High *C. aquatilis* abundance coupled with low species richness potentially conspired at MRF1 to yield peat deposits of poorer quality.

For SHF soil properties, observations were similar to peat-mineral mix observations assessed in McMillan et al. (2007), Hemstock et al. (2010) and Mackenzie and Quideau (2012). The stockpiled peat placed across the wetland interior of SHF in 2011 was quite heterogeneous, and contained material from *Sphagnum*-dominated systems, rich fens, and significant fractions of mineral mix soil substrates. Overall, total soil N contents, C:N ratios, and soil temperatures at SHF were comparable to the benchmark sites, whereas gravimetric soil moisture, soil bulk density, total soil carbon contents, and organic matter contents were markedly different. The intrinsic properties of mineral soils are quite dissimilar from organic soil types, and were responsible for causing the across site differences for bulk density and organic matter contents between SHF and benchmark sites. High bulk density at SHF additionally affected peat moisture levels, whereby highly compacted soil cores at SHF contained fewer water filled pore-water spaces in comparison to the benchmark sites. As the reclamation site continues to mature, we predict that inputs of organic material from root production and litter deposition may transform SHF wet plot soils into an environment most similar to the MRF1 site. However, the future trajectories of SHF dry plot soils are unclear, given that carbon accumulation potentials may be affected by the aerobic status of soils exposed above the water table.

Soil Extractable Nitrogen

We reported soil extractable nutrient contents using both areal and gravimetric normalization metrics to highlight the effects of soil bulk density, namely for wet and dry plots at SHF. On a gravimetric basis, observations for extractable nutrient contents across benchmark sites were similar to Bayley et al. (2005) over both growing seasons. With the exception of PF2 in 2014, ammonium was the dominant nutrient extracted from soils, coinciding with Macrae et al. (2013), Andersen et al. (2013), and Wood et al. (2015). Across site differences for extractable nutrient contents were slight, with the exception of extractable nitrate contents observed at PF2 in 2014, which were markedly higher than the other sites. Unexpected spikes for soil extractable nitrate were also reported by Wood et al. (2015), suggesting that unique site conditions at the time of sampling may affect soil extractable nutrient contents a great deal.

At SHF, similar to benchmark sites, extractable ammonium and nitrate contents were higher in 2014 compared to 2013, and ammonium was the dominant nutrient extracted from soils. On an areal basis, soil extractable ammonium and nitrate contents either exceeded or were equivalent to observations across benchmark sites over both years, suggesting that exchangeable inorganic N pools are not a factor of concern at the current stage of reclamation. Assessing soil extractable ammonium and nitrate contents on a gravimetric basis, however, yielded skewed results, given that soil bulk density across wet and dry plots at SHF were 5-20 times greater than benchmark sites.

Benchmark Net Nitrogen Mineralization and Nitrification Patterns

Net N mineralization rates across benchmark sites were highly variable, especially for rich fen site types, but compare well with other Albertan studies that utilized the *in situ* buried bag approach (Humphrey and Pluth 1996; Bayley et al. 2005; Wood et al. 2015). On both areal and gravimetric metrics, mean net N mineralization rates were slowest across the *Sphagnum*-dominated poor fens and fastest across rich fen sites, corresponding with patterns previously observed by Updegraff et al. (1995), Bridgman et al. (1998), Bayley et al. (2005) and Fellman and D'Amore (2007), studies that compared net N

mineralization rates across a gradient of minerotrophy in peatlands. Although single-factor regression analysis revealed few factors that influenced net N mineralization, sites with the highest soil N contents and low C:N ratios generally exhibited the fastest N mineralization rates, corroborating ecological theory and findings from Damman (1988) and Schimel and Bennett (2004).

Net nitrification rates across the benchmark sites were quite slow on both areal and gravimetric normalization metrics, and provided minimal contributions to the net N mineralization total.

Observations of slow net nitrification rates are characteristic of most studies utilizing the *in situ* buried bag approach (Williams and Wheatley 1988; Humphrey and Pluth 1996; Devito et al. 1999; Bayley et al. 2005; Fellman and D'Amore 2007; Wray and Bayley 2008; Macrae et al. 2013; Andersen et al. 2013; Wood et al. 2015). In our study, there was a significant effect of year on nitrification, whereby 2014 rates were about four times faster than 2013 observations, suggesting that higher water tables in 2013 potentially drove down nitrification potentials by shrinking aerobic zones in the peat profile. In addition, nitrification rates across ERF and MRF1-3 were correlated with several factors, suggesting that nitrification is under tighter biotic and abiotic controls in moderate and extreme rich fens than N mineralization.

Sandhill Fen Net Nitrogen Mineralization and Nitrification Patterns

Assessing microbial-mediated processes across recently reclaimed or restored peatlands is important, given that peatland functionality is a product of several above and belowground feedbacks. If key microbial-mediated reactions are proceeding at unprecedented and/or insufficient rates, early diagnosis and rapid management responses by reclamation practitioners may alleviate these undesired observations from hindering reclamation success. In the first growing season, areal net N mineralization rates across SHF wet and dry plots were similar to three out of six benchmark fens (ERF and PF1-2). In the second year, net N mineralization rates in wet and dry plots increased nearly 2-fold from 2013

observations, and were equivalent to all benchmark sites, especially ERF and MRF1-2. On a gravimetric basis, our observations were quite similar to observations from Mackenzie and Quideau (2012) over both growing seasons. Because soil bulk density was markedly higher across SHF wet and dry plots, mean net N mineralization rates were slower than all benchmark sites, yet, statistically equivalent to all sites except MRF1-2. Surface-water EC and nitrate concentrations were relatively strong controllers of N mineralization in SHF wet plots, but, did not influence N mineralization rates across the benchmark fens. Because sample size was small for correlation analyses against surface-water parameters at SHF ($n=9$), it is possible these relationships manifested from chance alone. Soil extractable ammonium correlated with N mineralization across both wet (gravimetric $R^2=0.40$) and dry plots (gravimetric $R^2=0.25$), but again, did not affect rates across the benchmark fens. Given that nitrification contributed more to the net N mineralization total across both wet and dry plots at SHF, and ammonium pools are necessary for nitrification, we suspect relationships with soil extractable ammonium were autocorrelated with nitrification observations. Overall, the environmental factors controlling N mineralization at SHF were quite dissimilar from benchmark fens, highlighting that microbial mediated processes at SHF potentially are controlled by novel interactions at the current stage of reclamation.

Areal net nitrification rates across SHF wet plots were similar to benchmark sites (Fig. 5a). Across dry plots, however, areal nitrification rates were markedly faster than benchmark fens, and contributed over 50% to the net N mineralization total in both years. On a gravimetric basis, although mean nitrification rates were fastest in SHF dry plots, there were no significant differences across sites or between years. Nitrification rates across wet plots were tightly controlled by a myriad of factors that overlapped with ERF and MRF to some degree, namely, surface-water pH and soil extractable ammonium contents. Although bulk density failed to control nitrification at benchmark sites, the degree of soil compaction was important at SHF, and correlated with nitrification across both wet and dry plots (gravimetric $R^2= -0.27$ and -0.21 , respectively). Surprisingly, nitrification in wet plots correlated positively

with increasing soil moisture ($R^2=0.34$). Gravimetric soil moisture across SHF wet plots (mean=73%) was lower than benchmark sites (>90%), and plots at the upper end of the soil moisture spectrum potentially fell below a threshold moisture level that inhibits nitrification. The fact that nitrification in wet plots were positively correlated with high soil C:N ratios is also puzzling, as previous studies show inorganic N production potentials decrease as soil C:N ratios rise above 30:1 (Damman 1988). Soil C:N ratios were somewhat deceiving at SHF considering some plots contained fractions of clay-mineral deposits (with low total soil carbon and nitrogen contents); an artifact of placing stockpiled peat across the wetland. In addition, observations of low soil organic matter contents characteristic of SHF soils suggest the reclamation site is more carbon limited at the current stage of reclamation in comparison to the benchmark sites. For SHF dry plots specifically, soil moisture levels were on average 10-15% lower than wet plots. In some dry plot areas, iron oxidation and subsequent rust formation was observed on surveying flags > 50 cm deep into the soil profile (data not shown), which certainly increased nitrification potentials across areas of SHF exposed above the water table throughout the entirety of the growing season. In addition, weedy annual plant species, as well as the perennial grass *Calamagrostis canadensis* have established and outcompeted desirable wetland species (*C. aquatilis*) across dry → wet transition zones. After two growing seasons, net N mineralization patterns, net nitrification patterns and above ground plant assemblages in dry plots are transitioning into ecological analogues similar to upland or riparian systems (Vitt et al. 2016).

Conclusions

Overall, the key characteristics observed across benchmark sites are 1) mean net N mineralization rates were faster in rich fens than poor fens, and 2) net nitrification was slow across all benchmark sites. After two years of reclamation, SHF exhibited several physical, chemical, and functional characteristics that were similar to the benchmark sites, although, many observations were unique to SHF alone. The

majority of these differences at SHF centered on soil characteristics that may change over relatively short time scales as wetland plants continue to add organic material to the rhizosphere via annual root production and litter deposition, in a similar fashion as the MRF1 benchmark site. Over both buried bag incubation periods (2013 and 2014), net N mineralization rates across SHF wet and dry plots were equivalent to several benchmark sites. This observation suggests the amount of plant available inorganic N remaining following microbial immobilization should not limit NPP rates at the current stage of reclamation, and the return of microbial activity at the reclamation site was quite resilient. Based on surface-water and N mineralization observations, plots localized in wet areas at SHF were most similar to ERF and MRF sites. Dry areas at SHF, however, exhibited few fen characteristics, and net N mineralization was dominated by nitrification, a situation characteristic of upland systems. Wetland reclamation success may be achieved across some wet plots at SHF, and many aspects of peatland functionality (i.e., carbon sink capacity) may be restored. Going forward, to improve the probability of success for future wetland reclamations on previously mined in-pits, managing water tables at appropriate levels during the early the stages of fen reclamation is key. Utilizing net N mineralization as an indicator for site performance can be a useful tool for evaluating the overall 'health' of the belowground environment, and provides useful information to improve future wetland reclamations.

Acknowledgments

We are grateful to Syncrude Canada, Ltd. for funding, site access, and logical support. Jessica Piercey and Carla Wytrykush (both Syncrude) provided enthusiastic guidance and were always available for assistance. Thanks to Dr. John D. Reeve (SIUC), who provided guidance on our statistical procedures. Special thanks to two reviewers and Dr. Nicole Fenton who provided constructive input, and greatly improved the current manuscript from a previous version.

References

- Andersen, R., Wells, C., Macrae, M., and Price, J. 2013. Nutrient mineralisation and microbial functional diversity in a restored bog approach natural conditions 10 years post restoration. *Soil Biol. Biochem.* **64**: 37-47. doi: 10.1016/j.soilbio.2013.04.004.
- Bartsch, I., and Moore, T.R. 1985. A preliminary investigation of primary production and decomposition in 4 peatlands near schefferville, quebec. *Can. J. Bot.-Rev. Can. Bot.* **63**(7): 1241-1248.
- Basiliko, N., Moore, T.R., Lafleur, P.M., and Roulet, N.T. 2005. Seasonal and inter-annual decomposition, microbial biomass, and nitrogen dynamics in a Canadian bog. *Soil Science* **170**(11): 902-912. doi: 10.1097/01.ss.0000196765.59412.14.
- Bayley, S.E., Thormann, M.N., and Szumigalski, A.R. 2005. Nitrogen mineralization and decomposition in western boreal bog and fen peat. *Ecoscience* **12**(4): 455-465.
- Booth, M.S., Stark, J.M., and Rastetter, E. 2005. Controls on nitrogen cycling in terrestrial ecosystems: A synthetic analysis of literature data. *Ecological Monographs* **75**(2): 139-157. doi: 10.1890/04-0988.
- Bridgham, S.D., Updegraff, K., and Pastor, J. 1998. Carbon, nitrogen, and phosphorus mineralization in northern wetlands (vol 79, pg 1545, 1998). *Ecology* **79**(7): 2571-2571.
- Campbell, C., Vitt, D.H., Halsey, L.A., Thormann, M.N., Bayley, S.E. 2000. Net primary production and standing biomass in northern continental wetlands. *Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-369.*

Damman, A.W.H. 1988. Regulation of nitrogen removal and retention in sphagnum bogs and other peatlands. *Oikos* **51**(3): 291-305.

Devito, K.J., Westbrook, C.J., and Schiff, S.L. 1999. Nitrogen mineralization and nitrification in upland and peatland forest soils in two Canadian Shield catchments. *Can. J. For. Res.-Rev. Can. Rech. For.* **29**(11): 1793-1804. doi: 10.1139/cjfr-29-11-1793.

Drexler, J.Z., and Bedford, B.L. 2002. Pathways of nutrient loading and impacts on plant diversity in a New York peatland. *Wetlands* **22**(2): 263-281. doi: 10.1672/0277-5212(2002)022[0263:ponlai]2.0.co;2.

Echols, B.S., Currie, R.J., and Cherry, D.S. 2010. Preliminary results of laboratory toxicity tests with the mayfly, *Isonychia bicolor* (Ephemeroptera: Isonychiidae) for development as a standard test organism for evaluating streams in the Appalachian coalfields of Virginia and West Virginia. *Environmental Monitoring and Assessment* **169**(1-4): 487-500. doi: 10.1007/s10661-009-1191-3.

Fellman, J.B., and D'Amore, D.V. 2007. Nitrogen and phosphorus mineralization in three wetland types in Southeast Alaska, USA. *Wetlands* **27**(1): 44-53. doi: 10.1672/0277-5212(2007)27[44:napmit]2.0.co;2.

Foote, L. 2012. Threshold considerations and wetland reclamation in Alberta's mineable oil sands. *Ecol. Soc.* **17**(1). doi: 3510.5751/es-04673-170135.

Government of Alberta, 2011. Alberta's oil sands: Facts about tailings management. Government of Alberta, Edmonton, Alberta, Canada. [online] URL:

http://www.oilsands.alberta.ca/FactSheets/Tailings_Management.pdf

Gorham, E. 1991. Northern peatlands - role in the carbon-cycle and probable responses to climatic warming. *Ecol. Appl.* **1**(2): 182-195.

Hart, S.C., Nason, G.E., Myrold, D.D., and Perry, D.A. 1994. Dynamics of gross nitrogen transformations in an old-growth forest: the carbon connection. *Ecology* **75**(4): 880-891. doi: 10.2307/1939413.

Hemstock, S.S., Quideau, S.A., and Chanasyk, D.S. 2010. Nitrogen availability from peat amendments used in boreal oil sands reclamation. *Canadian Journal of Soil Science* **90**(1): 165-175.

Humphrey, W.D., and Pluth, D.J. 1996. Net nitrogen mineralization in natural and drained fen peatlands in Alberta, Canada. *Soil Sci. Soc. Am. J.* **60**(3): 932-940.

Locky, D.A., and Bayley, S.E. 2010. Plant diversity in wooded moderate-rich fens across boreal western Canada: an ecoregional perspective. *Biodiversity and Conservation* **19**(12): 3525-3543. doi: 10.1007/s10531-010-9914-x.

Loisel, J., Yu, Z.C., Beilman, D.W., Camill, P., Alm, J., Amesbury, M.J., Anderson, D., Andersson, S., Bochicchio, C., Barber, K., Belyea, L.R., Bunbury, J., Chambers, F.M., Charman, D.J., De Vleeschouwer, F., Fialkiewicz-Kozziel, B., Finkelstein, S.A., Galka, M., Garneau, M., Hammarlund, D., Hinchcliffe, W., Holmquist, J., Hughes, P., Jones, M.C., Klein, E.S., Kokfelt, U., Korhola, A., Kuhry, P., Lamarre, A.,

Lamentowicz, M., Large, D., Lavoie, M., MacDonald, G., Magnan, G., Makila, M., Mallon, G., Mathijssen, P., Mauquoy, D., McCarroll, J., Moore, T.R., Nichols, J., O'Reilly, B., Oksanen, P., Packalen, M., Peteet, D., Richard, P.J.H., Robinson, S., Ronkainen, T., Rundgren, M., Sannel, A.B.K., Tarnocai, C., Thom, T., Tuittila, E.S., Turetsky, M., Valiranta, M., van der Linden, M., van Geel, B., van Bellen, S., Vitt, D., Zhao, Y., and Zhou, W.J. 2014. A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. *Holocene* **24**(9): 1028-1042. doi: 10.1177/0959683614538073.

MacKenzie, M.D., and Quideau, S.A. 2012. Laboratory-based nitrogen mineralization and biogeochemistry of two soils used in oil sands reclamation. *Canadian Journal of Soil Science* **92**(1): 131-142. doi: 10.4141/cjss2010-070.

Macrae, M.L., Devito, K.J., Strack, M., and Waddington, J.M. 2013. Effect of water table drawdown on peatland nutrient dynamics: implications for climate change. *Biogeochemistry* **112**(1-3): 661-676. doi: 10.1007/s10533-012-9730-3.

McGill, W.B., and Cole, C.V. 1981. Comparative aspects of cycling of organic C, N, S and P through soil organic-matter. *Geoderma* **26**(4): 267-286. doi: 10.1016/0016-7061(81)90024-0.

McMillan, R., Quideau, S.A., MacKenzie, M.D., and Biryukova, O. 2007. Nitrogen mineralization and microbial activity in oil sands reclaimed boreal forest soils. *Journal of Environmental Quality* **36**(5): 1470-1478. doi: 10.2134/jew2006.0530.

Price, J.S., McLaren, R.G., and Rudolph, D.L. 2010. Landscape restoration after oil sands mining: conceptual design and hydrological modelling for fen reconstruction. *Int. J. Min. Reclam. Environ.* **24**(2): 109-123. doi: 10.1080/17480930902955724.

Pouliot, R., Rochefort, L., and Graf, M.D. 2012. Impacts of oil sands process water on fen plants: Implications for plant selection in required reclamation projects. *Environ. Pollut.* **167**: 132-137. doi: 10.1016/j.envpol.2012.03.050.

Purdy, B.G., MacDonald, S.E., and Lieffers, V.J. 2005. Naturally saline boreal communities as models for reclamation of saline oil sand tailings. *Restor. Ecol.* **13**(4): 667-677. doi: 10.1111/j.1526-100X.2005.00085.x.

Regina, K., Nykanen, H., Silvola, J., and Martikainen, P.J. 1996. Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity. *Biogeochemistry* **35**(3): 401-418.

Rezanezhad, F., Andersen, R., Pouliot, R., Price, J.S., Rochefort, L., and Graf, M.D. 2012. How fen vegetation structure affects the transport of oil sands process-affected waters. *Wetlands* **32**(3): 557-570. doi: 10.1007/s13157-012-0290-z.

Robertson, G.P., Wedin, D., Groffman, P.M., Blair, J.M., Holland, E.A., Nadelhoffer, K.J., and Harris, D. 1999. Soil carbon and nitrogen availability - Nitrogen mineralization, nitrification and soil respiration potentials. *Standard Soil Methods for Long-Term Ecological Research* **2**: 258-271.

Rooney, R.C., and Bayley, S.E. 2011. Setting reclamation targets and evaluating progress: Submersed aquatic vegetation in natural and post-oil sands mining wetlands in Alberta, Canada. *Ecol. Eng.* **37**(4): 569-579. doi: 10.1016/j.ecoleng.2010.11.032.

Schimel, J.P., and Bennett, J. 2004. Nitrogen mineralization: Challenges of a changing paradigm. *Ecology* **85**(3): 591-602.

Sjörs, H. 1950. On the relation between vegetation and electrolytes in North Swedish mire waters. *Oikos* **2**(2): 241-258.

Trites, M., and Bayley, S.E. 2009. Organic matter accumulation in western boreal saline wetlands: A comparison of undisturbed and oil sands wetlands. *Ecol. Eng.* **35**(12): 1734-1742. doi: 10.1016/j.ecoleng.2009.07.011.

Trites, M., and Bayley, S.E. 2009. Vegetation communities in continental boreal wetlands along a salinity gradient: Implications for oil sands mining reclamation. *Aquat. Bot.* **91**(1): 27-39. doi: 10.1016/j.aquabot.2009.01.003.

Trowbridge, W.B. 2007. The role of stochasticity and priority effects in floodplain restoration. *Ecological Applications* **17**(5): 1312-1324. doi: 10.1890/06-1242.1.

Updegraff, K., Pastor, J., Bridgham, S.D., and Johnston, C.A. 1995. Environmental and substrate controls over carbon and nitrogen mineralization in northern wetlands. *Ecol. Appl.* **5**(1): 151-163.

Verhoeven, J.T.A., Maltby, E., and Schmitz, M.B. 1990. Nitrogen and phosphorus mineralization in fens and bogs. *J. Ecol.* **78**(3): 713-726.

Vitt, D.H., and Andrus, R.E. 1977. Genus *Sphagnum* in Alberta. *Canadian Journal of Botany-Revue Canadienne De Botanique* **55**(3): 331-357.

Vitt, D.H., and Chee, W.L. 1990. The relationships of vegetation to surface-water chemistry and peat chemistry in fens of Alberta, Canada. *Vegetatio* **89**(2): 87-106. doi: 10.1007/bf00032163.

Vitt, D.H., Bayley, S.E., and Jin, T.L. 1995. Seasonal-variation in water chemistry over a bog-rich fen gradient in continental western Canada. *Canadian Journal of Fisheries and Aquatic Sciences* **52**(3): 587-606. doi: 10.1139/f95-059.

Vitt, D.H., Halsey, L.A., Bauer, I.E., and Campbell, C. 2000. Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. *Canadian Journal of Earth Sciences* **37**(5): 683-693. doi: 10.1139/e99-097.

Vitt, D.H., Wieder, R.K., Scott, K.D., and Faller, S. 2009. Decomposition and peat accumulation in rich fens of boreal Alberta, Canada. *Ecosystems* **12**(3): 360-373. doi: 10.1007/s10021-009-9228-6.

Vitt, D.H., and House, M. 2015. Establishment of bryophytes from indigenous sources after disturbance from oil sands mining. *Bryologist* **118**(2): 123-129. doi: 10.1639/0007-2745-118.2.123.

Vitt, D.H., House, M.K., Hartsock, J.A. 2016. Sandhill Fen, an initial trial for wetland species assembly on in-pit substrates: Lessons after three years. *Botany (in press)*. doi: 10.1139/cjb-2015-0262.

Wiedermann, M.M., Gunnarsson, U., Ericson, L., and Nordin, A. 2009. Ecophysiological adjustment of two *Sphagnum* species in response to anthropogenic nitrogen deposition. *New Phytol.* **181**(1): 208-217. doi: 10.1111/j.1469-8137.2008.02628.x.

Wild, U., Kamp, T., Lenz, A., Heinz, S., and Pfadenhaeur, J. 2001. Cultivation of *Typha* spp. in constructed wetlands for peatland restoration. *Ecological Engineering* **17**(1): 49-54. doi: 10.1016/s0925-8574(00)00133-6.

Woo, I., and Zedler, J.B. 2002. Can nutrients alone shift a sedge meadow towards dominance by the invasive *Typha x glauca*? *Wetlands* **22**(3): 509-521. doi: 10.1672/0277-5212(2002)022[0509:cnasas]2.0.co;2.

Wood, M.E., Macrae, M.L., Strack, M., Price, J.S., Osko, T.J., and Petrone, R.M. 2015. Spatial variation in nutrient dynamics among five different peatland types in the Alberta oil sands region. *Ecohydrology*: n/a-n/a. doi: 10.1002/eco.1667.

Wray, H.E., and Bayley, S.E. 2008. Nitrogen dynamics in floating and non-floating peatlands in the Western Boreal Plain. *Canadian Journal of Soil Science* **88**(5): 697-708.

Wytrykush, C., Vitt, D.H., McKenna, G., and Vassov, R. 2012. Designing landscapes to support peatland development on soft tailings deposits Syncrude Canada Ltd.'s Sandhill Fen Research Watershed initiative. *Restoration and Reclamation of Boreal Ecosystems: Attaining Sustainable Development*: 161-+.

Yu, Z.C., Loisel, J., Brosseau, D.P., Beilman, D.W., and Hunt, S.J. 2010. Global peatland dynamics since the Last Glacial Maximum. *Geophys. Res. Lett.* **37**. doi: L1340210.1029/2010gl043584.

Zedler, J.B., and Kercher, S. 2004. Causes and consequences of invasive plants in wetlands: Opportunities, opportunists, and outcomes. *Critical Reviews in Plant Sciences* **23**(5): 431-452. doi: 10.1080/07352680490514673.

Draft

Table 1. Surface-water characteristics for Sandhill Fen and benchmark fen sites (SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = poor fen).

Year	Peatland	pH	Reduced EC ($\mu\text{S cm}^{-1}$)	$\text{NH}_4^+\text{-N}$ (mg L^{-1})	$\text{NO}_3^-\text{-N}$ (mg L^{-1})
2013	SHF Wet	9.1 (0.5) ^{ab}	472.5 (224.3) ^b	0.8 (0.4) ^c	0.3 (0.35)
	ERF	8.9 (0.6) ^{ab}	307.3 (51.9) ^c	1.1 (0.3) ^{bc}	0.3 (0.28)
	MRF 1	7.6 (0.7) ^{cd}	88.6 (34.4) ^e	1.5 (1.2) ^b	0.5 (0.69)
	MRF 2	7.9 (0.4) ^c	51.4 (17.8) ^{ef}	1.2 (0.4) ^b	0.4 (0.32)
	MRF 3	6.2 (0.2) ^f	28.6 (12.3) ^{fg}	1.6 (1.0) ^a	0.5 (0.60)
	PF 1	4.5 (0.2) ^g	19.7 (9.4) ^g	1.7 (0.8) ^a	0.5 (0.52)
	PF 2	4.5 (0.3) ^g	26.8 (15.9) ^{fg}	1.9 (1.0) ^a	0.5 (0.68)
2014	SHF Wet	8.2 (0.6) ^b	1714.2 (595.72) ^a	1.1 (0.2) ^c	0.2 (0.03)
	ERF	8.7 (0.8) ^a	406.0 (68.08) ^{cd}	1.0 (0.1) ^{bc}	0.2 (0.08)
	MRF 1	7.3 (0.6) ^{de}	86.8 (30.03) ^e	1.0 (0.1) ^b	0.2 (0.04)
	MRF 2	7.2 (0.2) ^e	69.8 (31.01) ^{ef}	1.1 (0.4) ^b	0.2 (0.05)
	MRF 3	6.2 (0.2) ^f	39.8 (12.45) ^{e-g}	1.8 (0.8) ^a	0.5 (0.26)
	PF 1	4.4 (0.1) ^g	30.6 (6.10) ^{fg}	1.4 (0.4) ^a	0.3 (0.13)
	PF 2	4.6 (0.4) ^g	34.7 (11.13) ^{fg}	2.0 (0.7) ^a	0.5 (0.24)

Note: Values are means $\pm$ standard deviations in parentheses. For each factor measured the approximate number of observations was N=360. For each parameter, differences were determined by a 2×7 factorial ANOVA, differing letters or numbers represent significantly different groups ($\alpha = 0.05$, Tukey's HSD).

Table 2. Soil characteristics (0-10 cm cores) for Sandhill Fen and benchmark fen sites for years 2013 and 2014 (SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = poor fen).

Year	Peatland	Gravimetric	Bulk density (mg/cm ⁻³)	Organic matter (%)	Total N (%)	Total C (%)	C:N ratio	Soil temperature (°C)
		soil moisture (%)						
2013	SHF Wet	73.0 (7.6) ^b	262.9 (108.3) ^a	35.4 (7.5)	1.3 (0.4) ^c	27.5 (9.5) ^{1,d}	20.6 (2.73) ^d	17.2 (5.8) ^{ab}
	SHF Dry	62.5 (10.7) ^c	250.5 (93.0) ^a	35.2 (6.9)	1.5 (0.5) ^c	30.1 (10.1) ^{1,d}	20.7 (2.10) ^{cd}	15.0 (4.7) ^{a-c}
	ERF	92.5 (1.3) ^a	44.3 (14.8) ^b	77.9	1.9 (0.4) ^a	41.8 (2.8) ^{1,b}	22.2 (3.63) ^{cd}	18.0 (2.9) ^a
	MRF 1	92.3 (2.8) ^a	39.1 (18.8) ^b	84.2	1.6 (0.30) ^b	37.6 (6.6) ^{1,c}	24.6 (3.82) ^c	16.2 (3.6) ^{a-c}
	MRF 2	92.2 (1.8) ^a	47.5 (17.0) ^b	71.7	2.2 (0.3) ^a	43.3 (4.1) ^{1,bc}	20.2 (3.79) ^d	14.1 (2.5) ^{b-d}
	MRF 3	92.8 (2.6) ^a	46.0 (19.1) ^b	72.1	2.0 (0.4) ^a	40.3 (7.5) ^{1,bc}	20.2 (2.34) ^d	15.4 (3.2) ^{a-c}
	PF 1	95.7 (0.8) ^a	21.4 (4.6) ^c	94.9	0.9 (0.1) ^d	48.0 (0.7) ^{1,a}	54.6 (7.67) ^b	15.9 (3.2) ^{a-c}
	PF 2	96.0 (1.3) ^a	15.6 (3.0) ^c	98.7	0.8 (0.1) ^d	48.7 (0.6) ^{1,a}	66.7 (9.83) ^a	14.3 (2.5) ^{a-c}
2014	SHF Wet	73.8 (22.0) ^b	283.0 (161.2) ^a	34.2 (5.3)	1.2 (0.5) ^c	24.9 (12.3) ^{2,d}	20.4 (5.1) ^d	15.5 (4.4) ^{a-c}
	SHF Dry	58.5 (15.2) ^c	279.6 (109.1) ^a	33.1 (6.4)	1.2 (0.5) ^c	25.7 (11.5) ^{2,d}	20.8 (2.7) ^{cd}	12.7 (2.5) ^{cd}
	ERF	93.2 (0.7) ^a	44.0 (11.3) ^b	*	2.0 (0.3) ^a	41.6 (1.8) ^{2,b}	20.8 (2.8) ^{cd}	17.3 (4.7) ^{ab}
	MRF 1	91.9 (1.7) ^a	45.4 (15.8) ^b	*	1.6 (0.1) ^b	36.0 (5.8) ^{2,c}	22.5 (2.6) ^c	9.2 (2.8) ^{ef}
	MRF 2	91.6 (2.6) ^a	55.5 (23.9) ^b	*	2.1 (0.4) ^a	38.9 (5.2) ^{2,bc}	19.8 (6.8) ^d	10.2 (3.0) ^{d-f}
	MRF 3	90.6 (2.0) ^a	54.3 (17.7) ^b	*	1.9 (0.4) ^a	38.3 (6.8) ^{2,bc}	20.1 (3.0) ^d	9.5 (3.9) ^f
	PF 1	94.7 (1.5) ^a	19.0 (7.3) ^c	*	0.9 (0.1) ^d	47.5 (0.6) ^{2,a}	55.0 (6.7) ^b	12.8 (3.1) ^{c-e}
	PF 2	96.3 (0.8) ^a	16.8 (2.9) ^c	*	0.8 (0.1) ^d	48.4 (0.8) ^{2,a}	62.5 (7.9) ^a	15.4 (3.7) ^{a-c}

Note: Values are means ± standard deviations in parentheses. For each factor measured the approximate number of observations was N=390 (except for loss-on-combustion organic matter content; N=48). For each parameter, differences were determined by a 2 × 8 factorial ANOVA, differing superscript letters indicate site differences, differing superscript numbers indicate year differences ($\alpha = 0.05$, Tukey's HSD). *Represents missing data points for benchmark fens that were sampled for organic matter content in 2013 only.

Table 3. Single-factor regression equations for net nitrogen mineralization (0-10 cm cores) correlations with surface-water and soil properties across the major peatland site-types: Sandhill Fen (Wet and Dry), an extreme rich fen, moderate rich fens, and poor fens for years 2013 and 2014 (SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = poor fen).

Process/ property	Fen type	Equation	<i>N</i>	<i>F</i>	<i>P</i>	<i>R</i> ²
Areal N mineralization (Aer Nmin)						
Surface-water (SW) NO ₃ ⁻	SHF wet	$\log \text{Aer Nmin} = 1.54 + 1.86 \times \log \text{SW NO}_3^-$	9	6.9	0.034	0.50
Extractable NO ₃ ⁻	MRF	$\log \text{Aer Nmin} = 1.71 - 0.11 \times \log \text{Ext NO}_3^-$	47	10.1	0.003	-0.18
pH	MRF	$\log \text{Aer Nmin} = 0.76 + 0.11 \times \text{pH}$	46	7.5	0.009	0.15
Soil temperature (°C)	MRF	$\log \text{Aer Nmin} = 0.55 + 0.53 \times \log ^\circ\text{C}$	47	4.6	0.040	0.09
Gravimetric water content (GWC)	PF	$\log \text{Aer Nmin} = -20.18 + 10.85 \times \log \text{GWC}$	32	12.3	0.002	0.29
Gravimetric N mineralization (Grv Nmin)						
Surface-water electrical conductivity (EC)	SHF wet	$\log \text{Grv Nmin} = 0.72 + 0.18 \times \log \text{EC}$	9	9.3	0.019	0.57
Surface-water NO ₃ ⁻	SHF wet	$\log \text{Grv Nmin} = 1.27 + 0.95 \times \log \text{SW NO}_3^-$	9	6.6	0.037	0.48
Extractable NH ₄ ⁺	SHF wet	$\log \text{Grv Nmin} = 0.95 + 0.18 \times \log \text{Ext NH}_4^+$	19	11.5	0.003	0.40
Extractable NH ₄ ⁺	SHF dry	$\log \text{Grv Nmin} = 1.05 + 0.22 \times \log \text{Ext NH}_4^+$	20	6.0	0.025	0.25
pH	MRF	$\log \text{Grv Nmin} = 0.82 + 0.07 \times \text{pH}$	46	11.7	0.001	0.21
Surface-water NO ₃ ⁻	MRF	$\log \text{Grv Nmin} = 1.34 - 0.37 \times \log \text{SW NO}_3^-$	46	5.2	0.028	-0.11

Note: Units for net nitrogen mineralization rates are mg N m⁻² d⁻¹ (areal basis) and mg N kg⁻¹ d⁻¹ (gravimetric basis). Units for soil extractable ammonium and nitrate are mg N m⁻² (areal basis) and mg N kg⁻¹ (gravimetric basis). Units for surface-water ammonium and nitrate are mg L⁻¹.

Table 4. Single-factor regression equations for net nitrification (0-10 cm cores) correlations with surface-water and soil properties across the major peatland site-types: Sandhill Fen (Wet and Dry), Rich fens (ERF and MRF1-3) and Poor fens (PF1-2) for years 2013 and 2014 (SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = poor fen).

Process/ property	Fen type	Equation	N	F	P	R ²
Areal nitrification (Aer nit)						
Extractable NH ₄ ⁺	SHF wet	log Aer nit = -1.65 + 3.47 × log Ext NH ₄ ⁺	19	22.8	<0.001	0.57
Gravimetric soil moisture	SHF wet	log Aer nit = 0.10 + 0.42 × log GSM	19	8.7	0.009	0.34
Soil C:N ratio	SHF wet	log Aer nit = 0.63 + 0.20 × log C:N	19	6.9	0.017	0.29
Bulk density (BD)	SHF wet	log Aer nit = 1.17 - 0.12 × log BD	19	6.2	0.024	-0.27
Bulk density (BD)	SHF dry	log Aer nit = 1.16 - 0.12 × log BD	20	4.8	0.043	-0.21
Extractable NH ₄ ⁺	ERF	log Aer nit = 0.54 + 0.13 × log Ext NH ₄ ⁺	16	38.4	<0.001	0.73
Extractable NO ₃ ⁻	ERF	log Aer nit = 0.82 + 0.05 × log Ext NO ₃ ⁻	16	20.6	0.001	0.60
Surface-water NH ₄ ⁺	ERF	log Aer nit = 1.02 + 0.17 × log SW NH ₄ ⁺	16	14.0	0.002	0.50
pH	ERF	log Aer nit = -0.08 + 0.10 × pH	16	9.0	0.010	0.39
pH	MRF	log Aer nit = 1.13 - 0.04 × pH	46	18.9	<0.001	-0.30
Surface-water NO ₃ ⁻	MRF	log Aer nit = 1.27 + 0.95 × log SW NO ₃ ⁻	46	4.3	0.045	0.09
Surface-water NO ₃ ⁻	PF	log Aer nit = 0.91 - 0.58 × log SW NO ₃ ⁻	32	4.4	0.044	-0.13
Gravimetric nitrification (Grv nit)						
Extractable NH ₄ ⁺	SHF wet	log Grv nit = 1.13 + 0.01 × log Ext NH ₄ ⁺	19	18.5	0.001	0.52
pH	SHF wet	log Grv nit = 1.21 - 0.01 × pH	9	6.6	0.037	-0.49
Surface-water electrical conductivity (EC)	SHF wet	log Grv nit = 1.13 + 0.01 × log EC	9	6.2	0.042	0.47
Extractable NH ₄ ⁺	ERF	log Grv nit = 1.12 + 0.02 × log Ext NH ₄ ⁺	16	64.4	<0.001	0.82
Extractable NO ₃ ⁻	ERF	log Grv nit = 1.15 + 0.01 × log Ext NO ₃ ⁻	16	24.0	<0.001	0.63
pH	ERF	log Grv nit = 0.99 + 0.02 × pH	16	23.8	<0.001	0.63
Surface-water NH ₄ ⁺	ERF	log Grv nit = 1.17 + 0.02 × log SW NH ₄ ⁺	16	18.0	0.001	0.56
pH	MRF	log Grv nit = 1.19 - 0.01 × pH	46	17.2	<0.001	-0.28
Extractable NO ₃ ⁻	MRF	log Grv nit = 1.15 + 0.003 × log Ext NO ₃ ⁻	47	12.1	0.001	0.21
Extractable NH ₄ ⁺	MRF	log Grv nit = 1.14 + 0.004 × log Ext NH ₄ ⁺	47	6.4	0.015	0.12

Note: Units for net nitrification rates are mg N m⁻² d⁻¹ (areal basis) and mg N kg⁻¹ d⁻¹ (gravimetric basis). Units for soil extractable ammonium and nitrate are mg N m⁻² (areal basis) and mg N kg⁻¹ (gravimetric basis). Units for surface-water ammonium and nitrate are mg L⁻¹.

Fig. 1. Fort McMurray climate graphs for the 2013 and 2014 summer growing seasons (May-September) highlighting (a) cumulative precipitation (mm); and (b) cumulative degree-day temperatures above 0°C. Vertical dashed lines represent the incubation period for *in situ* net nitrogen mineralization buried bags.

Fig. 2. a) Areal and (b) gravimetric soil extractable ammonium contents for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

Fig. 3. a) Areal and (b) gravimetric soil extractable nitrate contents for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

Fig. 4. a) Areal and (b) gravimetric net nitrogen mineralization rates for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). Because differences were observed across sites only on a gravimetric basis (with no site x year interaction), 2013 and 2014 observations were pooled for each site, resulting in only 1 set of bars. SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

Fig. 5. a) Areal and (b) gravimetric net nitrification rates for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich

fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

Draft

Fig. 1. Fort McMurray climate graphs for the 2013 and 2014 summer growing seasons (May-September) highlighting (a) cumulative precipitation (mm); and (b) cumulative degree-day temperatures above 0°C. Vertical dashed lines represent the incubation period for *in situ* net nitrogen mineralization buried bags

119x164mm (300 x 300 DPI)

Fig. 2. a) Areal and (b) gravimetric soil extractable ammonium contents for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

126x185mm (300 x 300 DPI)

Fig. 3. a) Areal and (b) gravimetric soil extractable nitrate contents for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

126x185mm (300 x 300 DPI)

Fig. 4. a) Areal and (b) gravimetric net nitrogen mineralization rates for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). Because differences were observed across sites only on a gravimetric basis (with no site x year interaction), 2013 and 2014 observations were pooled for each site, resulting in only 1 set of bars. SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

129x193mm (300 x 300 DPI)

Fig. 5. a) Areal and (b) gravimetric net nitrification rates for Sandhill Fen (wet and dry plots) and benchmark fen sites (0-10 cm cores). SHF = Sandhill Fen; ERF = extreme rich fen; MRF = moderate rich fen; PF = Poor fen. Error bars represent standard errors. Significant differences across sites and between years are indicated by different letters ($\alpha=0.05$).

129x195mm (300 x 300 DPI)