Uncertainty-based terrestrial critical loads of nutrient nitrogen in northern Saskatchewan, Canada

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Atmospheric deposition of nitrogen (N) to boreal and taiga forests of Saskatchewan, Canada, is currently low, but there is concern over terrestrial eutrophication due to potential increases in deposition associated with industrial activities. Critical loads of nutrient N ($CL_{nut}N$) were determined for five upland forest ecoregions according to the Simple Mass Balance (SMB) model using an uncertainty-based approach. Median ecoregion $CL_{nut}N$ ranged from 88 to 123 mol$_c$ ha^{-1} yr^{-1} (1.2–1.8 kg N ha^{-1} yr^{-1}), with higher $CL_{nut}N$ in the south. Median $CL_{nut}N$ for a small number of sites in the southernmost ecoregion were exceeded under both fire and harvest disturbance regimes, according to current N deposition. During the 21st century, median $CL_{nut}N$ were predicted to increase by more than 20% in response to climate driven increases in runoff and temperature. Current atmospheric deposition levels in the region are uncertain, and there is an ongoing need to monitor deposition and exceedance risk.

Introduction

Nitrogen (N) is an important nutrient for plants and is often the limiting nutrient in terrestrial ecosystems (Aber *et al.* 1989). Consequently, where elevated inputs of N (e.g. from atmospheric deposition) occur, eutrophying effects including altered plant growth rates (Aber *et al.* 1989) and shifts in species composition (Nordin *et al.* 2005) can result. A number of studies have noted a significant decrease in species diversity under conditions of increased N deposition (Aerts and Berendse 1988, Bobbink *et al.* 1988,

Huenneke *et al.* 1990). The extent of the impact of increased N deposition has been found to decrease from grassland to forested sites, with coniferous forests being the least affected by increased N, but considerable variability exists due to differences in soil type, water table, and stand ages (Wamelink *et al.* 2009). Increased N inputs have also been linked to poor drought resistance, and higher sensitivity to interannual climatic variations (Tilman 1999).

In recent decades, ecosystem-based critical loads (CL) have been widely used in Europe (Hettelingh *et al.* 2001) to guide N deposition

management policies. The CL identifies the highest level of deposition that can be sustained over the long-term without detrimental impacts to sensitive biota (Nilsson and Grennfelt 1988). The CL of nutrient nitrogen ($CL_{nut}N$) is the amount of N deposition an ecosystem can support without causing eutrophication over the long-term. The $CL_{nut}N$ is generally determined according to one of two approaches. The empirical $CL_{nut}N$ is established based on documented detrimental effects to sensitive vegetation species for similar ecosystem types (e.g. Bobbink *et al.* 2010). This approach can be useful where observational data necessary to determine steady-state CL are limited; however, it relies on the underlying behaviour of analogous systems to be consistent and this can be difficult to know *a priori*. The steady-state $CL_{nut}N$ (Umweltbundesamt 2004) characterizes the equilibrium (long-term) state of an ecosystem (i.e. unchanging climate) according to a mass balance approach that quantifies sinks of N for the system or site of interest. Both approaches are subject to methodological limitations and uncertainties associated with data; accordingly, the $CL_{nut}N$ identified by these methods can be quite different.

In Canada, CL have rarely been used to address the issue of terrestrial eutrophication, despite a legacy of elevated N deposition in some regions. The empirical $CL_{nut}N$ was recently used to assess the risk of eutrophication in temperate regions of Ontario, Canada (Aherne and Posch 2013). In western Canada, growing emissions of N to the atmosphere associated with transportation and natural resource extraction, most notably the Athabasca oil sands region in northeastern Alberta, are a concern (Clair *et al.* 2015). Given current evidence suggesting retention of the limiting nutrient N in upland boreal forests of western Canada (Laxton *et al.* 2012), and indicators of early stages of N saturation in forests surrounding industrial activities (Laxton *et al.* 2010), eutrophication may be a risk for boreal ecosystems of the region. Taiga and boreal regions of the province of Saskatchewan (SK) are downwind of pollution sources associated with the oil sands industry, and fossil fuel reserves in northern SK may be developed in the future which could lead to further increases in N deposition in the region. Emissions sources in more southern parts of the province (resource extraction, electricity and heat generation, transportation, and agriculture) are also important (Government of Saskatchewan 2013) for the southern boreal forests of SK. For these reasons, it is important to define the $CL_{nut}N$ and identify exceedance (present and future) of the CL in order to minimize potential eutrophication of these ecosystems.

The central objective of this study was to estimate the steady-state $CL_{nut}N$ for upland forest ecoregions of northern SK. The hypothesis is that the ecoregions will exhibit different $CL_{nut}N$ owing to differences in N dynamics across the region. An uncertainty-based approach to account for parametric uncertainty in N sinks within the rooting zone of the forest soils was used and the major disturbance regimes in the region, forest harvesting and fire, were considered. The supporting objectives were to calculate exceedance of the $CL_{nut}N$ in order to identify where eutrophication is most likely to occur, and to investigate how anticipated changes in climate during the 21st century may affect the $CL_{nut}N$. The latter is important because CL estimations are premised on the assumption of an unchanging climate (Posch *et al.* 2002), but regions of northern SK have been identified as particularly sensitive to climate change (e.g. changes in the water balance in the 21st century; Ireson *et al.* 2015). Detailed steady-state $CL_{nut}N$ for this region will inform the risk of eutrophication because N availability across much of the region is very low and it remains unknown whether the empirical $CL_{nut}N$ established for the more impacted (via air pollution) boreal forests of northern Europe is suitable for taiga or boreal forests in the study region.

Data and methods

Study area

The study area was located in northern SK, a province in western Canada that is dominated by prairie in the south, and forest in the north (Fig. 1). The provinces northern and predominantly forested ecoregions (Selwyn Lake Upland (SLU), Tazin Lake Upland (TLU), Athabasca Plain (AP), Churchill River Upland (CRU), and

Fig. 1. Location of the study sites (filled circles) in boreal and taiga ecoregions (SLU: Selwyn Lake Upland; TLU: Tazin Lake Upland; AP: Athabasca Plain; CRU: Churchill River Upland; MBU: Mid-Boreal Upland) of northern Saskatchewan, Canada. The thin black lines are ecoregion boundaries (note that MBU is discontinuous) and white areas are waterbodies (rivers or lakes).
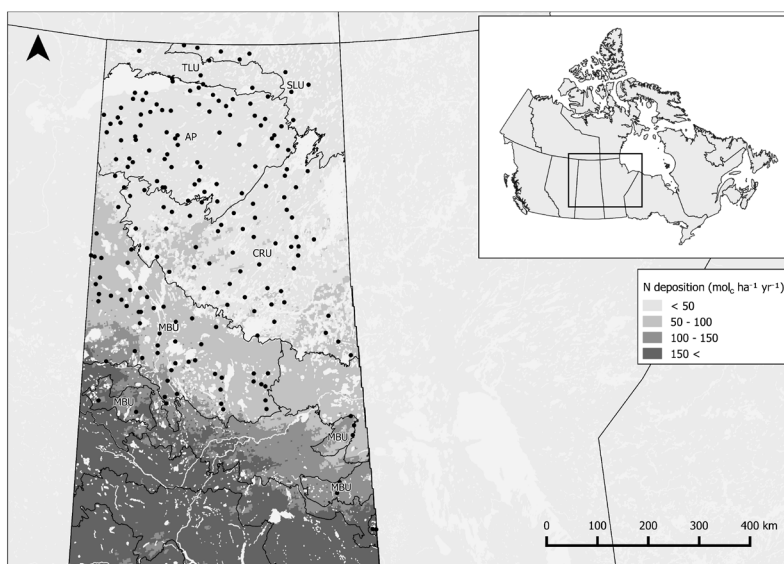
Mid-Boreal Upland (MBU)) were considered in this analysis. Ecoregions are a sub-division of ecozones, differentiated by their dominant plant and animal communities, soil type, and climate (Table 1). Saskatchewan experiences a semi-arid, humid, or subarctic continental climate depending on the region. The average winter and summer temperatures for the most northerly study region are –21.5 °C and 11 °C, respectively and –14.5 °C and 14.5 °C respectively for the southerly areas (Environment Canada 2015). The SLU and TLU are taiga forest, while the remaining three ecoregions are boreal forest. The MBU has historically been managed for commercial forest harvesting while the other ecoregions are remote with land disturbance limited to localized mining operations.

A total of 199 upland forest sites supporting a variety of canopy types (e.g. *Populus tremu-loides* (trembling aspen), *Pinus banksiana* (jack pine), *Picea glauca* (white spruce), and *Picea mariana* (black spruce), were sampled during 2011 and 2012 (Fig. 1). The focus was exclusively on upland forests, and wetland and riparian areas were not considered. Each ecoregion is comprised of subunits called ecodistricts; multiple sites within each ecodistrict were sampled to ensure that the ecoregions were adequately spatially represented. For the taiga ecoregions, this stratified sampling approach was not carried out owing to the remoteness and challenges associated with access (only selected ecodistricts were sampled). Sites were selected to be minimally impacted by disturbance (roads, and recent fire or harvesting). The study area is remote and as such, the distribution of sites was partly influenced by accessibility (via road or air). Additional details on site selection and soil sam-

Table 1. Dominant vegetation species and soil types, and average annual temperature (°C) of the five ecoregions. See text for full ecoregion names.

Ecoregion	Dominant vegetation	Dominant soil	Average temperature
SLU	black spruce	dystric brunisol	–5
TLU	black spruce	dystric brunisol	–5
AP	jack pine	dystric brunisol	–3.5
CRU	jack pine, black spruce	dystric/eutric brunisol, gray luvisol	–2.5
MBU	trembling aspen	gray luvisol, dystric brunisol	0

pling are described in Whitfield and Watmough (2012).

A single soil pit was excavated at each site and horizon-based sampling of the mineral soil layers within the rooting zone and subsoil (e.g. horizons A–C) was completed using a volumetric sampling ring and hammer corer to determine soil bulk density (ρ). Forest floor was sampled from a quadrat of known area. Composite samples were collected for each horizon, using soil from all sides of the pit. Rooting zone depth was estimated visually.

Soil analysis

Composite and ρ samples were air-dried and sieved to 2 mm preceding further analysis. Mineral soil ρ was determined by gravimetric analysis. All mineral horizons within the rooting zone were analysed for organic matter content (OM), and particle size. Organic matter content was determined by measurement of loss-on-ignition after heating oven-dried soils at 375 °C for 16 h in a muffle furnace (Ball 1964). The proportions of particles in the clay, silt and sand size classes ($\leq 2\ \mu$m, 2–60 μm and 0.06–2.0 mm, respectively) were measured in triplicate by laser ablation (Horiba Partica LA-950) and averaged for each soil horizon. Carbon and N contents were determined using an Elementar CNS analyzer for all horizon samples with OM > 2%; N was below detection at lower OM contents.

Vegetation and climate data

Species-specific aboveground biomass N content (kg ha^{-1}) was determined according to Rencz and Auclair (1977), Jokela *et al.* (1980), Perala and Alban (1982), Halliwell and Apps (1997), and Ruess *et al.* (2003). Ecoregion-specific basal area for each species (McLaughlan *et al.* 2010) was used to normalize aboveground biomass N from the above studies to northern SK forests (Table 2).

Gridded monthly climate normals (Mitchell *et al.* 2004) were used to estimate long-term average annual precipitation (P) and runoff (Q) each site. Recent modeled (2013 meteorological year) N deposition estimates (P.A. Makar unpubl. data) were used to calculate exceedance of the steady-state $CL_{nut}N$. Projected Q and temperature were downscaled (resolution: ~2.5 degree latitude by longitude) from the Canadian Earth System Model (CanESM2: Chylek *et al.* 2011). The relative concentration pathways (RCP) 2.6 and 8.5 were considered as they represent the 'best' and 'worst' case scenarios for the future, respectively. The former is a 'peak-and-decline' scenario, where greenhouse gas emissions and air pollutants are reduced significantly over time due to changes in energy use (van Vuuren *et al.* 2011) while the latter characterizes increasing greenhouse gas emissions over time as a result of rapid population growth and lower rates of technological and energy use change. Averages for

Table 2. Above-ground nitrogen (N) content, basal area, fire return period and stand age data used to calculate nitrogen uptake in fire and harvest disturbance regimes. Nitrogen content values in parentheses represent bole-only N content used for the harvesting regime in the Mid-Boreal Upland (MBU).

Region	Species	N content[a] (kg ha^{-1})	Basal area[a] (mean, m^2 ha^{-1})	Basal area (study area)[b] (mean ± SD, m^2 ha^{-1})	Fire return period (yr)	Stand age (harvest rotation) (mean ± SD, yr)
SLU & TLU	jack pine	196	30.4	8.2 ± 2.7	113	61 ± 7.8
SLU & TLU	black spruce	165	36.7	5.1 ± 1.7	113	105 ± 3.8
AP & CRU	jack pine	196	30.4	13.2 ± 3.6	102	74 ± 8.6
AP & CRU	black spruce	165	36.7	8.9 ± 2.2	102	90 ± 5.7
AP & CRU	trembling aspen	368	34.7	9.8 ± 3.1	102	67 ± 8.2
MBU	jack pine	196 (93)	30.4	20.3 ± 4.5	169	69 ± 8.3
MBU	black spruce	165	36.7	14.1 ± 1.7	169	91 ± 3.6
MBU	trembling aspen	368 (199)	34.7	19.9 ± 4.5	169	70 ± 8.3
MBU	white spruce	326 (98)	41.1	18.7 ± 2.3	169	85 ± 3.6

[a] From Perala and Alban (1982), Rencz and Auclair (1977), Ruess *et al.* (2003); and Jokela *et al* (1980).
[b] McLaughlan *et al* (2010).

the 2071–2100 period were used to characterize climate variables for these scenarios.

Steady-state critical load calculation

A steady-state method was used to calculate the long-term $CL_{nut}N$. The calculation uses N sinks, acceptable losses to leaching, and losses to the atmosphere to define a maximum load of N (Umweltbundesamt 2004):

$$CL_{nut}N = N_i + N_u + N_{le(acc)}/(1 - f_{de}) \qquad (1)$$

where N_i is the long-term net immobilization of N in the rooting zone, N_u is the average long-term uptake of N by vegetation, $N_{le(acc)}$ is the acceptable N loss to leaching below the rooting zone, and f_{de} is the fraction of N deposition that is denitrified (Umweltbundesamt 2004). Consistent with standard practice, erosion, volatilization, and adsorption are assumed to be negligible (Umweltbundesamt 2004). For N terms based on canopy type (N_u, $N_{le(acc)}$), a single species was assumed herein according to observations of dominant tree species at each site.

An uncertainty-based approach was used to account for the limited observational data available to describe N behavior at each study site. Generally, a normal distribution [sampled as mean ± 1 standard deviation (SD)] where site or regional data were available, or a uniform distribution based on the range of available estimates from other regions, were used to quantify uncertainty in each term in equation 1 (Table 3). For each site, 1000 iterations were used to sample from the distributions and characterize the steady-state $CL_{nut}N$.

Nitrogen immobilization, the capacity to store N in the soil organic matter, was estimated by dividing the current N pool by the soil age (years since deglaciation):

$$N_i = \rho N_{content} z / \text{soil age} \qquad (2)$$

Site-specific data were used for ρ, N content ($N_{content}$) and rooting depth (z). Because N_i is estimated from $N_{content}$, it would include any N fixed by biota (microbes and plants). Soil age for each region was characterized from glacial retreat phase maps from Christiansen (1978), region-based soil ages were used as they most accurately represent the glacial retreat pattern of the study area. The N_i estimates were grouped by ecozone, and a truncated normal distribution for each region was used to characterize N_i, with one SD used to truncate the distribution. Where this approach yielded values outside typical guidelines (Umweltbundesamt 2004), the 5th or 95th percentile of observations for the region as a whole was used to constrain the lower or upper bound of the distribution, respectively.

Nitrogen uptake was calculated to reflect (1) high intensity fire and (2) harvesting disturbance regimes. Disturbance by high intensity, stand-replacing fire typical of the region (Weir *et al.* 2000), was determined for the entire study area. For simplicity, it was assumed that all N in aboveground tree compartments (bole wood, bark, branches, and foliage) represents an uptake flux lost during fire events. In contrast, disturbance by tree harvesting was considered only for the study sites with merchantable species (trembling aspen, jack pine, white spruce) located in the commercial forest zone (MBU sites). Common practice in SK is to limb trees onsite and spread the discarded branches and foliage (slash) over the cut-block to promote natural revegetation, thus only the bole wood and bark were included in the harvesting uptake flux. These disturbance

Table 3. Distribution type for each Simple Mass Balance parameter for $CL_{nut}N$ calculation, source of data used to describe distribution, and scale at which distribution applies.

Parameter	Distribution type	Data source	Scale
N_i	normal	this study	ecoregion
N_u	normal	this study	ecoregion
$N_{le(acc)}$	uniform	Umweltbundesamt (2004)	site
f_{de}	uniform	Umweltbundesamt (2004)	site

regimes were considered independently because fires are actively suppressed in the MBU while harvesting activities are limited in more northern ecoregions. Thus while fire and harvesting can co-occur, this analysis provides information on the dominant disturbance regimes. Fire disturbance was also considered for the MBU despite active efforts to suppress fires in this region as this would have been the natural disturbance mechanism prior to settlement of the region. Uptake was calculated by dividing the amount of N removed in biomass (via combustion or export of tree boles) by the fire return interval, or harvest rotation period, based on the formula presented in Umweltbundesamt (2004):

$$N_u = N_{removed}/period \qquad (3)$$

Region and species-specific mean basal area were used to characterize above ground N following the method described above, with 1 SD used to bound each distribution (Table 3). Region-specific fire return interval was based on fire statistics for the MBU, Boreal Shield (AP and CRU), and Taiga Shield (SLU and TLU), from Parisien *et al*. (2004), and supported by findings by Heinselman (1980) and Stocks *et al*. (1996). The ecoregion default ± 10% was used to characterize the uncertainty in the long-term fire return interval. Harvest rotation period was estimated using mature species-specific stand ages (± 1 SD; McLaughlan *et al*. 2010). Accordingly, unique N_u distributions were calculated for each canopy type in each ecoregion.

The acceptable N leaching rate was calculated on a site-specific basis using runoff (Q) and acceptable leaching concentration ($[N]_{acc}$) for waters draining forests of different types (Umweltbundesamt 2004):

$$N_{le(acc)} = Q[N]_{acc} \qquad (4)$$

To account for uncertainty, a range of $[N]_{acc}$ values for coniferous (0.0129–0.01573 mol_c m^3 (default value ± 10%)) and deciduous (0.0143–0.0276 mol_c m^3) stands were used (Umweltbundesamt 2004). A default uncertainty for Q was estimated as ± 10% owing to the coarse (gridded climate normal) rather than site-specific nature of the available data.

The denitrification fraction (f_{de}) based on soil drainage (Reinds *et al*. 2001) was approximated using soil texture. Silt and silt loam sites were characterized as having moderate drainage, loam and sand loam as good drainage, and sand and loamy sand as excessive drainage. Rather than a fixed value for f_{de} (Umweltbundesamt 2004) uniform distributions of 0–0.05, 0.05–0.10, and 0.1–0.2 were used to characterize sites classified as having excessive, good, and moderate drainage, respectively.

Exceedance and climate influence

Exceedance of the steady-state $CL_{nut}N$ was identified by subtracting the CL from modeled N deposition. Future changes in steady-state $CL_{nut}N$ associated with an alternate climate were identified by using Q from the climate scenarios to determine alternate values of $N_{le(acc)}$. The influence of future change in temperature on N_i was evaluated by assuming that a soil temperature-N_i relationship can be used to approximate future site-level changes. The distributions established for N_i (above) were adjusted according to the predicted changes in temperature for each region. Changes in steady-state $CL_{nut}N$ associated with climate induced effects on tree growth were not considered as calculations of N_u do not directly involve climate variables as calculated herein, and because tree growth responses to climate involve many factors and alone are insufficient to characterize N_u (*see* Discussion).

Results and discussion

Soil nitrogen content

Most sites were sandy: sand content between 11% and 100%, with silt content 0% and 84%. Few of the soils featured clay content greater than several percent (Table 4). Median OM content was 1.3%, and was typically below 5%; higher values were observed in rare instances were soil depth was limited and the shallow soils were more closely influenced by overlying organic horizons. The median mineral horizon molar C:N ratio across all sites was 22.1 (Table 4).

Calculated N_i rates reflected the high degree of variability in soil N content, both across and within regions. Immobilization rates ranged from 1.1 to 180 mol$_c$ ha^{-1} yr^{-1}, with rates generally decreasing northward (Table 5). At sites in the AP, SLU and TLU, N_i was low and exhibited little variability. It should be noted that N content was below detection limits for many sites (Table 5), particularly in the north. Given shorter fire return intervals in northern ecoregions, immobilization rates used herein may underestimate true long-term rates, as there is potential for some OM in the mineral horizons to be lost during high intensity fire events. Accordingly $CL_{nut}N$ may be underestimated if the size of N sinks due to fire is also underestimated. Laxton (2009) suggested that N_i estimates may be conservative as they do not consider losses from fire disturbance or harvesting. Nitrogen immobilization was highest in the MBU and CRU which may be partly related to climate. Temperature and N_i were significantly positively related ($p <$ 0.01) and temperature range across the ecoregions is approximately 5 °C. Higher $CL_{nut}N$ will result where higher N_i is observed due to higher temperatures. Higher mean immobilization rates in the MBU may also reflect higher rates of fire suppression owing to the number of communities and amount of harvesting that takes place in the region compared with more northerly regions (Parisien *et al.* 2004).

In general, the estimated rates presented herein (1.1–180 mol$_c$ ha^{-1} yr^{-1}) were more variable than published rates for comparable boreal forest soils, despite strong N limitation. Rosen *et al.* (1992) found Swedish forest soil N_i of 14–36 mol$_c$ ha^{-1} yr^{-1}, and noted that N_i is often higher in warmer climates, suggesting that rates of 71 mol$_c$ ha^{-1} yr^{-1} and higher may also be rea-sonable. The higher variability in calculated N_i rates reported herein could reflect N fixation at some sites, as N_i was not measured directly, but inferred from N content of soils and N fixing species (e.g. *Alnus* sp.) are present in the study region. Where fixation occurs, it could increase the soil N pool and lead to overestimates of N_i, however this does not appear to be common among the study sites as N content was below detection at most sites, and instances of high N_i were rare (Table 5). The long-term net immobilization of N in the soil rooting zone is often highly variable, and at present there is no real consensus on sustainable long-term immobilization rates. These data add to that body of knowledge, and given the regional patterns and variability in N_i also serve as a warning against use of a single default value to characterize N_i for all sites across a region. This is particularly true in cases where the relative magnitude of N_i makes it an important component of the total N sink, or where differences in N_i among sites might be expected due to climatic influences. In such cases, use of a simplistic (default N_i) approach should be done with caution as it may risk

Table 4. Minimum, median, and maximum rooting zone depth and bulk density (ρ), and mineral horizon organic matter (OM), clay, silt, and sand fractions, and molar carbon to nitrogen ratio (C:N) ($n = 199$).

Variable	Unit	Minimum	Median	Maximum
Depth	m	0.04	0.33	0.85
ρ	g cm^{-3}	0.13	1.23	1.94
OM	%	0.12	1.33	19.9
Clay	%	0	1.8	19.5
Silt	%	0	24.7	83.5
Sand	%	11	73.1	100
C:N		6.1	22.1	39.4

Table 5. Estimated nitrogen immobilization rate for each ecoregion.

	SLU	TLU	AP	CRU	MBU
Minimum (mol$_c$ ha^{-1} yr^{-1})	4.7	4.2	4.8	1.1	21
Median (mol$_c$ ha^{-1} yr^{-1})	5.4	4.8	9.8	11	41
Mean (mol$_c$ ha^{-1} yr^{-1})	5.4	5.3	10	24	45
Maximum (mol$_c$ ha^{-1} yr^{-1})	6.2	7.2	16	180	87
n	2	4	5	17	18
Sites with N below detection (%)	78	33	92	68	71
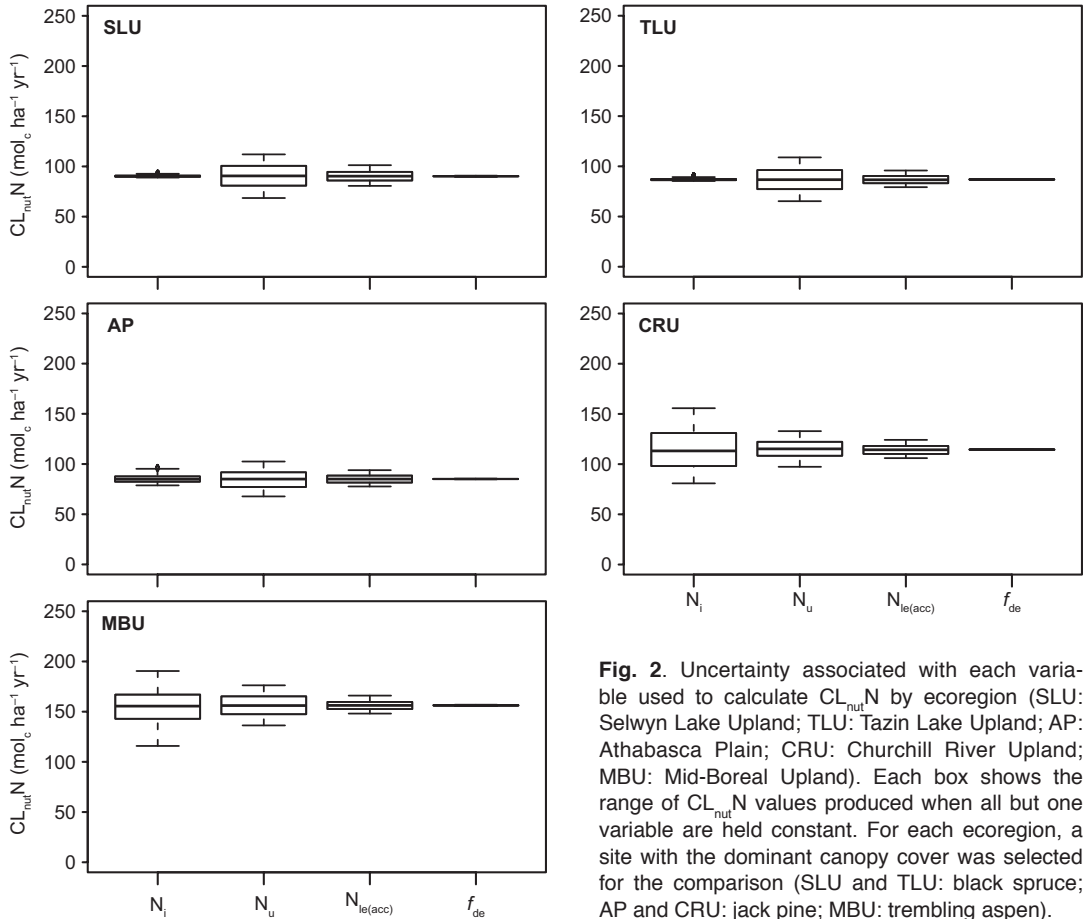
Fig. 2. Uncertainty associated with each variable used to calculate $CL_{nut}N$ by ecoregion (SLU: Selwyn Lake Upland; TLU: Tazin Lake Upland; AP: Athabasca Plain; CRU: Churchill River Upland; MBU: Mid-Boreal Upland). Each box shows the range of $CL_{nut}N$ values produced when all but one variable are held constant. For each ecoregion, a site with the dominant canopy cover was selected for the comparison (SLU and TLU: black spruce; AP and CRU: jack pine; MBU: trembling aspen).

over or underestimating the steady-state $CL_{nut}N$, owing to regional differences in immobilization rates.

Critical loads

Uncertainty associated with $N_{le(acc)}$ and f_{de} were a minor contribution to total uncertainty (Fig. 2). These terms were calculated using data with relatively narrow and well established distributions (Umweltbundesamt 2004); sandy upland soils typical of the study region have low potential for denitrification. In contrast, greater uncertainty was linked to N_i and N_u. The uncertainty associated with N_i, mainly in the CRU and MBU, can largely be attributed to the large range in soil N contents in these regions, as described in the previous section; however, this parameter is difficult

to measure with accuracy, as low OM means that N (and C) content is often below detection limits. In the northernmost ecoregions, the largest source of uncertainty in the CL was attributed to N_u. The range of basal area values used to normalize N_u was principally responsible. Nitrogen uptake was also highly variable in the MBU.

Median steady-state $CL_{nut}N$ ranged from 88 to 123 mol_c ha^{-1} yr^{-1} across the study ecoregions under the fire regime. There was a northward decrease in steady-state $CL_{nut}N$, with MBU and CRU generally exhibiting the highest values and taiga ecoregions being lower and exhibiting less variability (Fig. 3). The northern boreal shield (AP) was intermediate and demonstrated considerable variability in $CL_{nut}N$, although the median value for this ecoregion was comparable to those of the SLU and TLU. This latitudinal trend of northward decreasing steady-state $CL_{nut}N$ can

be explained by a number of underlying factors. Trembling aspen was the dominant tree species for the MBU, and $N_{le(acc)}$ is higher for deciduous species, which contributes to higher steady-state $CL_{nut}N$. Coniferous forests by comparison, which are dominant in the AP, SLU and TLU generally have lower steady-state $CL_{nut}N$ on account of lower $N_{le(acc)}$. Immobilization also tended to be higher at CRU and MBU sites. The analysis also featured higher N_u for more southern sites, consistent with expectations for forest growth under more favourable climate and solar radiation conditions. It should be noted here, however, that as a necessary simplification of the modeling approach, a single (dominant) tree species and complete combustion of above-ground biomass were used to characterize N_u. In reality some sites feature a multi-species canopy and N_u may be underestimated in some instances. Likewise, combustion efficiency of high intensity fires may not always be complete, potentially leading to a small overestimate of N_u under a fire regime, although where biomass combustion is incomplete N may be lost in subsequent fires. Consequently, while these assumptions will to some degree offset one another, the analysis could produce a small underestimate or overestimate of N_u.

The harvest disturbance regime resulted in greater variability in MBU steady-state $CL_{nut}N$ (median: 90–195 mol$_c$ ha^{-1} yr^{-1} at individual sites) than the fire regime (Fig. 4). In theory, a smaller export of N (as would occur with bole-only harvesting compared with a stand-replacing fire event) should result in an ecosystem that is more sensitive to N deposition. Nonetheless the disturbance regime is also important. In the MBU, the time between disturbances was generally lower for the harvest rotation period than the forest fire return interval. This led to higher steady-state $CL_{nut}N$ at the majority of sites under a harvesting regime (Fig. 4). Both jack pine and trembling aspen have higher steady-state $CL_{nut}N$ in the harvest than fire disturbance regime, as the shorter harvesting disturbance interval had a stronger influence than a reduction in the N flux associated with bole-only biomass removal from these sites. White spruce stands in contrast had lower steady-state $CL_{nut}N$ in the harvest regime than fire, mainly due to considerably lower N
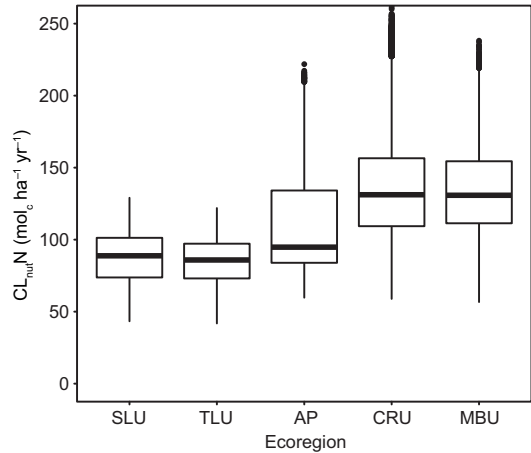

Fig. 3. Variability in critical loads of nutrient N ($CL_{nut}N$) for the five ecoregions studied (SLU: Selwyn Lake Upland; TLU: Tazin Lake Upland; AP: Athabasca Plain; CRU: Churchill River Upland; MBU: Mid-Boreal Upland) under a forest fire disturbance regime.

removal in the harvest regime owing to their comparatively low proportion of above-ground N in the bole. This result highlights the potential relationship between silvicultural practices (e.g. tree species $N_{le(acc)}$ and harvest rotation period) and the $CL_{nut}N$ in regions where forest harvesting is important.

Exceedance

Modelled N deposition in SK generally decreases northward across the boreal and taiga ecoregions of the province (Fig. 1) with the highest deposition (~180 mol$_c$ ha^{-1}) occurring in the southern MBU where industrial and agricultural activities to the south contribute to higher deposition. There is also a notable influence of industrial activities (fossil fuel extractions) to the west, with higher deposition in the west relative to the same latitude in the east. While both N deposition and steady-state $CL_{nut}N$ follow a similar spatial pattern, N deposition is consistently less than the median CL, with the exception of some MBU sites. Exceedance of the median steady-state $CL_{nut}N$ was identified for 7% of MBU sites (~30% exceedance of 5th percentile $CL_{nut}N$) under a fire disturbance regime (Fig. 4). There were fewer instances of exceedance of the
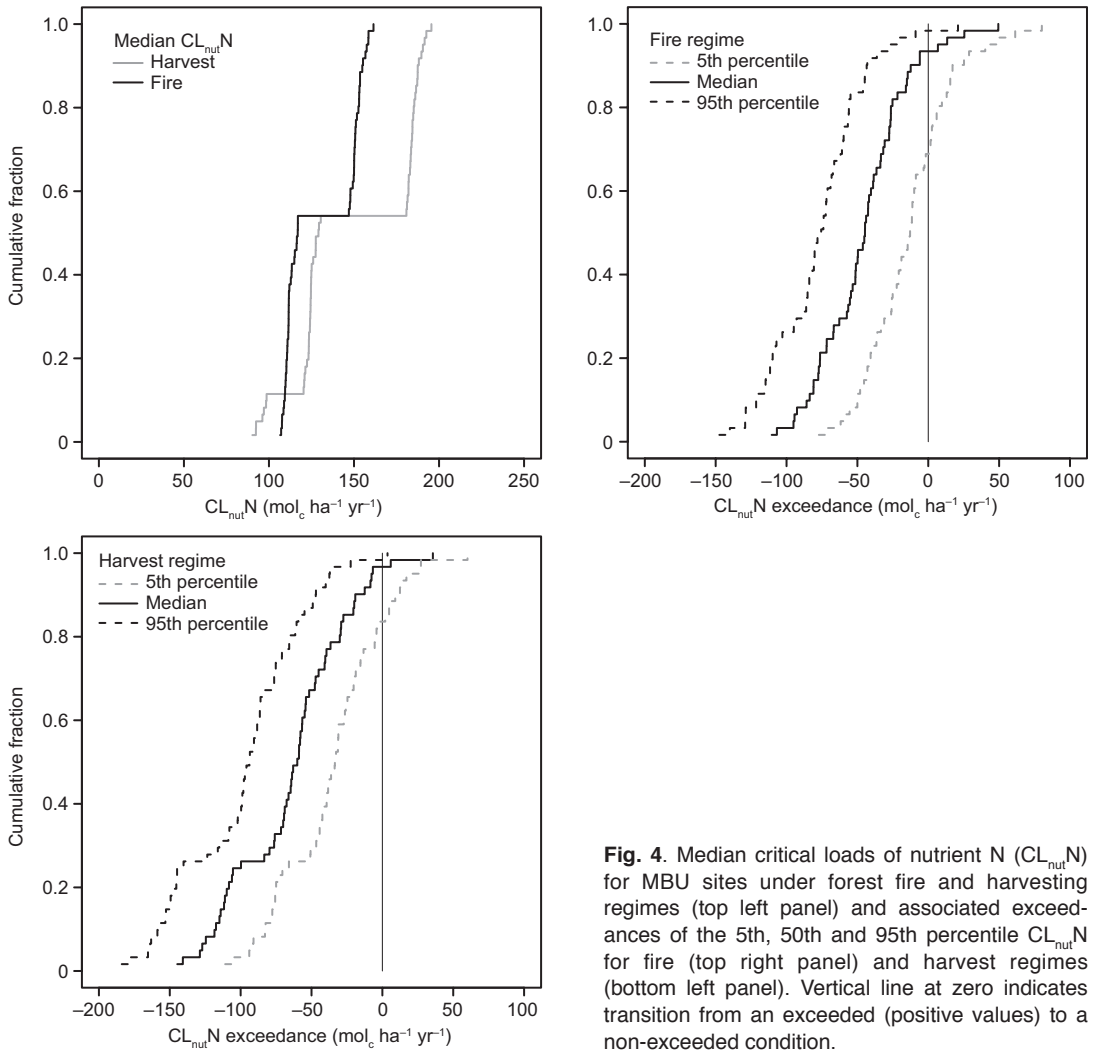
Fig. 4. Median critical loads of nutrient N ($CL_{nut}N$) for MBU sites under forest fire and harvesting regimes (top left panel) and associated exceedances of the 5th, 50th and 95th percentile $CL_{nut}N$ for fire (top right panel) and harvest regimes (bottom left panel). Vertical line at zero indicates transition from an exceeded (positive values) to a non-exceeded condition.

median steady-state $CL_{nut}N$ (3%) under harvesting conditions, despite some sites being more sensitive to N deposition.

There appears to be a relatively low risk of eutrophication effects in the study area at current deposition levels. Nonetheless, for many sites in both the fire and harvest regimes, modeled deposition and the steady-state $CL_{nut}N$ are in close balance as indicated by the steep nature of the cumulative distribution (Fig. 4). Future increases in N deposition on the order of 50 mol$_c$ ha^{-1} yr^{-1} could result in CL exceedance for a notable proportion of the sites due to the steep nature of the cumulative distribution (Fig. 4). Nevertheless this increase in deposition is small, the systems are currently N limited, and any effects may be

very slow to manifest. Likewise as N deposition is modelled, an underestimate of the level of N deposition on the order of several tens of mol$_c$ ha^{-1} yr^{-1} could lead to understatement of the risk of eutrophication. Deposition levels remain an important consideration as different atmospheric deposition models and different emissions inventories used in the (same) models can lead to notable differences in atmospheric deposition estimates for the study region. As atmospheric deposition monitoring for the region is sparse, current N deposition estimates remain uncertain.

The empirical $CL_{nut}N$, determined from observations of detrimental responses of an ecosystem or ecosystem component to observed N deposition flux, can be used as an alter-

native to steady-state $CL_{nut}N$. For boreal forests, the suggested empirical $CL_{nut}N$ is 360–710 mol_c ha^{-1} yr^{-1} (Bobbink *et al.* 2010, Pardo *et al.* 2011); higher than the estimates identified herein using the SMB. Further, recent deposition estimates to the study sites are considerably lower than the empirical $CL_{nut}N$ (N deposition: 18–180 mol_c ha^{-1} yr^{-1}), suggesting no risk of eutrophication at present according to this alternate approach.

Overall, this analysis suggests limited risk of ecosystem damage through eutrophication according to recent estimates of atmospheric N deposition. A key difference between the current study and many other CL studies in Canada was the inclusion of nutrient uptake. In the absence of removal of N via harvesting exports or combustion of above-ground biomass in forest fires, as may be the case for lands that are protected as parks, $CL_{nut}N$ will be lower and consequently the risk of exceedance could be higher than suggested by this analysis.

Climate change

The acceptable leaching of N is based on the threshold concentration of N in the leachate and Q (Eq. 4). Therefore, where Q changes in response to climate, a linear response in $CL_{nut}N$ is expected. Simulations of future climate scenarios according to CanESM2 indicate increasing Q for most sites, with increases averaging 16% and 17.7%, respectively, according to RCP 2.6 and RCP 8.5. This increase is not uniform and varies across and within regions; under RCP 8.5, increases in Q were generally greater in the SLU and TLU ecoregions, while for RCP 2.6 greater increases were noted for sites in the MBU. At higher Q, increases in steady-state $CL_{nut}N$ of several percent were predicted on average for RCP 2.6 and RCP 8.5.

Nitrogen dynamics may also exhibit a response to climate, for instance through changes in temperature. In the current study, N_i was strongly related to temperature. Predicted increases in temperature averaged 2.5 °C for RCP 2.6 and 7.3 °C for RCP 8.5, with larger increases at more northerly sites. Considering the potential increase in N_i associated with warming
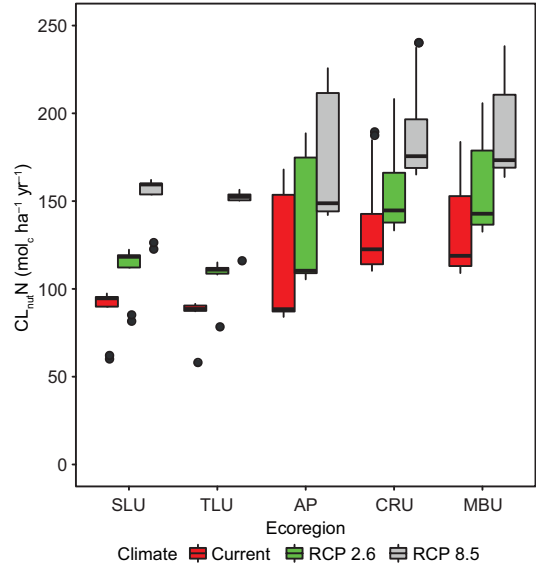
Fig. 5. Median $CL_{nut}N$ (fire disturbance regime) for all sites in each ecoregion (SLU: Selwyn Lake Upland; TLU: Tazin Lake Upland; AP: Athabasca Plain; CRU: Churchill River Upland; MBU: Mid-Boreal Upland) under three climate scenarios, current, RCP 2.6, and RCP 8.5.

during the 21st century suggests that $CL_{nut}N$ will increase during this period. In fact, most of the change in steady-state $CL_{nut}N$ predicted under future climate conditions is attributed to N_i, as the median steady-state $CL_{nut}N$ increased by 20% and 48% for RCP 2.6 and RCP 8.5, respectively, when the influence of Q and temperature was considered together (Fig. 5). It is important to note that the relationship between N_i and temperature used in this analysis may include underlying influences of other factors (e.g. soil type, fire return interval), and consequently increases in N_i associated with anticipated increases in temperature could be overestimated.

In this study, $CL_{nut}N$ were strongly influenced by N_u; however, changes in N_u in response to future climate have not been explicitly quantified herein. Forest growth may respond to climate driven changes in precipitation, soil moisture, temperature, and possibly carbon dioxide. Where these factors act synergistically under future climate to promote growth, N_u, and consequently $CL_{nut}N$, can increase. Disturbance interval may also be important and can respond in concert with changes in forest growth (e.g. harvest rota-

tion period contraction in response to increased growth rate). Perhaps more important than the potential for changes in growth of vegetation in the region is the potential shift in species distribution that may occur as a consequence of climate change. The western boreal forest is predicted to contract at its southern extent as grasslands move northward in response to climate, although disturbance regime is expected to play an important role in determining ecoregion boundaries (Schneider *et al.* 2009). Where a change in dominant tree species occurs due to shifts in range, changes in $CL_{nut}N$ may be as much the result of changes in vegetation type as changes in growth rate of the species that dominate the ecoregions at present.

Conclusions

Given increases in N emission to the atmosphere in western Canada, the potential for eutrophication of N-limited terrestrial systems in the region should be considered. Nitrogen sinks are often poorly described in CL analyses, however the uncertainty-based approach used herein is comprehensive and characterizes in detail the potential range in steady-state $CL_{nut}N$ for the region. Critical loads of nutrient nitrogen followed a well-defined latitudinal gradient, with the highest loads identified for the CRU and MBU, and lower loads for more northern ecoregions. Nitrogen uptake and N_i contributed greater uncertainty than either $N_{le(acc)}$ or f_{de}. According to current modeled N deposition levels there is low risk of eutrophication according to steady-state $CL_{nut}N$ under natural conditions represented by occurrence of (high-intensity) forest fire. Likewise, if forest regeneration occurs due to anthropogenic disturbance in the form of bole-only harvesting, limited eutrophication is expected in the study region. While not evaluated in the current study, if forest fires are effectively suppressed and harvesting does not occur, as may be the case for provincial or national parks in the region, $CL_{nut}N$ would be lower than indicated herein. Given the similarity between modelled N deposition and $CL_{nut}N$ for many sites in the southern boreal forest, revised steady-state $CL_{nut}N$ for regions where this no-disturbance management approach

is employed are warranted. Furthermore, atmospheric deposition levels are poorly understood in the study region; few measurements are available and modelled deposition estimates vary by model. Consequently, because only the most recent estimate of N deposition was used, the uncertainty around exceedance calculations illustrated herein is understated. Efforts to quantify and reduce uncertainty around modelled N deposition are necessary for improved confidence in exceedance calculations. Nonetheless, steady-state $CL_{nut}N$ may increase in future due to a predicted increase in rooting zone temperature and water flux. Because N_u was an important influence on the $CL_{nut}N$, but also relatively uncertain, quantifying additional influences of climate (e.g. on species distribution, N_u, fire return interval) would help clarify future risk of eutrophication. Likewise, additional consideration of how N deposition might change in future across the MBU is necessary given the close balance between $CL_{nut}N$ and modelled N deposition in this ecoregion.

References

Aber J.D., Nadelhoffer K.J., Steudler P. & Melillo J.M. 1989. Nitrogen saturation in northern forest ecosystems: excess nitrogen from fossil fuel combustion may stress the biosphere. *Bioscience* 39: 378–386.

Aerts R. & Berendse F. 1988. The effect of increased nutrient availability on vegetation dynamics in wet heathlands. *Vegetation* 76: 63–69.

Aherne J. & Posch M. 2013. Impacts of nitrogen and sulphur deposition on forest ecosystem services in Canada. *Current Opinion in Environmental Sustainability* 5: 108–115.

Arp P.A., Oja T. & Marsh M. 1996. Calculating critical S and N loads and current exceedances for upland forests in southern Ontario, Canada. *Canadian Journal of Forest Research* 26: 696–709.

Ball D.F. 1964. Loss-on-ignition as an estimate of organic

matter and organic carbon in non-calcareous soils. *European Journal of Soil Science* 15: 84–92.

Black C.A. 1965. *Methods of soil analysis: part I physical and mineralogical properties*. American Society of Agronomy, Madison, WI, USA.

Bobbink R., Hicks K., Galloway J., Spranger T., Alkemade R., Ashmore M., Bustamante M., Cinderby S., Davidson E., Dentener F., Emmett B., Erisman J.W., Fenn M., Gilliam F., Nordin A., Pardo L. & De Vries W. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological Applications* 20: 30–59.

Bobbink R., Bik L. & Willems J.H. 1988. Effects of nitrogen fertilization on vegetation structure and dominance of *Brachypodium pinnatum* (L.) Beauv. in chalk grassland. *Acta Botanica Neerlandica* 37: 231–242.

Chylek P., Li J., Dubey M.K., Wang M. & Lesins G. 2011. Observed and model simulated 20th century Arctic temperature variability: Canadian Earth System Model CanESM2. *Atmospheric Chemistry and Physics Discussions* 11: 22893–22907.

Clair T.A., Davies M., Pucket K. & Percy K. 2015. *Introduction to the report: assessing forest health in the Athabasca Oil Sands Region*. WBEA Technical Report.

Christiansen E.A. 1978. The Wisconsinan deglaciation of southern Saskatchewan and adjacent areas. *Canadian Journal of Earth Science* 16: 913–938.

Environment Canada 2015. *Canadian climate normal 1981–2010 station data*. Available at http://climate.weather.gc.ca/climate_normals/index_e.html.

Government of Saskatchewan 2013. *The Saskatchewan 2013 state of the environment report*. Saskatchewan Ministry of Environment, Regina, SK. [Available at http://environment.gov.sk.ca/2013soereport].

Halliwell D.H. & Apps M.J. 1997. *BOReal ecosystem–atmosphere study (BOREAS) biometry and auxiliary sites: overstory and understory data*. Natural Resources Canada, Canadian Forest Service, Northern Forestry Centre, Edmonton, Alberta.

Heinselman M.L. 1981. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. In: Mooney H.A., Bonnicksen T.M., Christensen N.L., Lotan J.E. & Reiners W.A. (eds.), *Proceedings of the Conference Fire Regimes and Ecosystem Properties*, USDA Forest Service, pp. 7–57.

Hettelingh J.-P., Posch M. & de Smet P.A.M. 2001. Multi-effect critical loads used in multi-pollutant reduction agreements in Europe. *Water, Air, and Soil Pollution* 130: 1133–1138.

Huenneke L.F., Hamburg S.P., Koide R., Mooney H.A. & Vitousek P.M. 1990. Effects of soil resources on plant invasion and community structure in Californian serpentine grassland. *Ecology* 71: 478–491.

Ireson A.M., Barr A.G., Johnstone J.F., Mamet S.D., Van der Kamp G., Whitfield C.J., Michel N.L., North R.L., Westbrook C.J., DeBeer C., Chun K.P., Nazemi A. & Sagin J. 2015. The changing water cycle: the Boreal Plains ecozone of Western Canada. *Wiley Interdisciplinary Reviews: Water*. 5: 505–521.

Jokela E.J., Shannon C.A. & White E.H. 1980. Biomass and nutrient equations for mature *Betula papyrifera* marsh. *Canadian Journal of Forest Research* 11: 860–860.

Laxton D.L. 2009. *Nitrogen cycling in the Athabasca Oil Sands Region*. MSc. thesis, Trent University.

Laxton D.L., Watmough S.A., Aherne J. & Straker J. 2010. An assessment of nitrogen saturation in *Pinus banksiana* plots in the Athabasca Oil Sands Region, Alberta. *Journal of Limnology* 69(Suppl. 1): 171–180.

Laxton D.L., Watmough S.A. & Aherne J. 2012. Nitrogen cycling in *Pinus banksiana* and *Populus tremuloides* stands in the Athabasca Oil Sands Region, Alberta, Canada. *Water, Air, and Soil Pollution* 223: 1–13.

McLaughlan M.S., Wright R.A. & Jiricka R.D. 2010. *Field guide to the ecosites of Saskatchewan's provincial forests*. Saskatchewan Ministry of Environment, Forest Service. Prince Albert, Saskatchewan.

Mitchell T.D., Carter T.R., Jones P.D., Hulme M. & New M. 2004. *A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed record (1901–2000) and 16 scenarios (2001–2100)*. Paper 55, Tyndall Centre for Climate Change Research Working, Norwich, UK.

Nilsson J. & Grennfelt P. 1988. *Critical loads for sulfur and nitrogen*. Nordic Council of Ministers, Miljorapport.

Nordin A., Strengbom J., Witzell J., Näsholm T. & Ericson L. 2005. Nitrogen deposition and the biodiversity of boreal forests: implications for the nitrogen critical load. *Ambio* 34: 20–24.

Pardo L.H., Fenn M.E., Goodale C.L., Geiser L.H., Driscoll C.T., Allen E.B., Baron J.S., Bobbink R., Bowman W.D., Clark C.M., Emmett B.A., Gilliam F.S., Greaver T.L., Hall S.J., Lilleskov E.A., Liu L., Lynch J.A., Nadelhoffer K.J., Perakis S.S., Robin-Abbott M.J., Stoddard J.L., Weathers K.C. & Dennis R.L. 2011. Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. *Ecological Applications* 21: 3049–3082.

Parisien M.-A., Hirsch K.G., Lavoie S.G., Todd J.B. & Kafka V.G. 2004. *Saskatchewan fire regime analysis*. Information Report NOR-X-394, Canadian Forest Service, Northern Forestry Centre.

Perala D.A. & Alban D.H. 1982. Biomass, nutrient distribution and litterfall in *Populus*, *Pinus*, and *Picea* stands on two different soils in Minnesota. *Plant and Soil* 64: 177–192.

Posch M. 2002. Impacts of climate change on critical loads and their exceedances in Europe. *Environmental Science & Policy* 5: 307–317.

Reinds G.J., Posch M. & De Vries W. 2001. *A semi-empirical dynamic soil acidification model for use in spatially explicit integrated assessment models for Europe*. Alterra Report 084, Alterra Green World Research, Wageningen, The Netherlands.

Rencz A.N. & Auclair A.N. 1977. Biomass distribution in a subarctic *Picea mariana–Cladonia alpestris* woodland. *Canadian Journal of Forest Research* 8: 168–176.

Rosen K., Gundersen P., Tegnhammar L., Johansson M. & Frogner T. 1992. Nitrogen enrichment of nordic forest ecosystems: the concept of critical loads. *Ambio* 21: 364–368.

Ruess R.W., Hendric R.L., Burton A.J., Pregitzer K.S., Sveinbjornsson B., Allen M.F. & Maurer G.E. 2003. Coupling fine root dynamics with ecosystem carbon cycling in black spruce forests for interior Alaska. *Ecological Monographs* 73: 643–662.

Saxton K.E. & Rawls W.J. 2006. Soil water characteristic estimates by texture and organic matter for hydrologic solutions. *Soil Science Society of America Journal* 70: 1569–1578.

Schneider R.R., Hamann A., Far D., Wang X. & Boutin S. 2009. Potential effects of climate change on ecosystem distribution in Alberta. *Canadian Journal of Forest Research* 39: 1001–1010.

Stocks B.J., Lee B.S. & Martell D.L. 1996. Some potential carbon budget implications of fire management in the boreal forest. In: Apps M.J. & Price D.T. (eds.), *Forest ecosystems, forest management and the global carbon cycle, NATO ASI series, subseries 1 – global environmental change,* Springer-Verlag, Berlin, Germany, pp. 89–96.

Tilman D. 1999. The ecological consequences of changes in biodiversity: a search for general principles. *Ecology* 80: 1455–1474.

Umweltbundesamt 2004. *Manual on methodologies and criteria for modelling and mapping critical loads & levels and air pollution rffects, risks and trends.* Umweltbundesamt, Berlin.

van Vuuren D., Stehfest E., den Elzen M., Kram T., van Vliet J., Deetman S., Isaac M., Goldewijk K., Hof A., Beltran A., Oostenrijk R. & Ruijven B. 2011. RCP2.6: exploring the possibility to keep global mean temperature increase below 2 °C. *Climatic Change* 109: 95–116.

Wamelink G.W.W., van Dobben H.F., Mol-Dijkstra J.P., Schouwenberg E.P.A.G., Kros J., De Vries W. & Berendse F. 2009. Effect of nitrogen deposition reduction on biodiversity and carbon sequestration. *Forest Ecology and Management* 258: 1774–1779.

Wier J.M.H., Johnson E.A. & Miyanishi K. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. *Ecological Applications* 10: 1162–1177.

Whitfield C.J. & Watmough S.A. 2012. A regional approach for mineral soil weathering estimation and critical load assessment in boreal Saskatchewan, Canada. *Science of the Total Environment* 437: 165–172.