Using steady-state mass balance model to determine critical loads of acidity for terrestrial ecosystems in Alberta

Alberta

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Table of Contents

List of Tables

List of Figures

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Executive Summary

In 1998, the federal, provincial and territorial Energy and Environment Ministers signed the Canada-Wide Acid Rain Strategy for Post 2000. For western Canada, the Strategy was initially designed to prevent an acidification problem from developing in areas identified as clean. Acidifying gases and particulate matter can be removed from the atmosphere to the surface of ecosystems through atmospheric deposition.

To facilitate the management of acidic deposition in Alberta, a framework for terrestrial ecosystems was developed by the Clean Air Strategic Alliance through a multi-stakeholder process and adopted in 1999 and revised in 2008 (Alberta Environment, 2008). The 2008 framework identifies 1° by 1° empirical acid deposition loads which are based on the soil buffering capacity. A 2011 review of the framework identified the need to move from a method that only considered soil buffering capacity to a more detailed approach to identify areas at risk of impact from acidic deposition. The steady-state mass balance (SSMB) model was selected as the method to determine critical loads of acidity for Alberta. The SSMB model considers both biological response and soil buffering capacity and has been used to support the development of regional and national scale emission management policies.

This document outlines the input data, methods and assumptions used to determine critical loads of acidity for terrestrial ecosystems in Alberta using the SSMB model, and fulfills the need identified by the 2011 review of the framework. Alberta soil characteristics, dominant vegetation cover, run off, biomass harvest and grazing, base cation deposition and the impact of wildfires are used to derive critical loads of acidity for sulphur and nitrogen. Critical loads of acidity calculated using the SSMB model will be used in the Alberta Acid Deposition Management Framework (ADMF), as noted in Section 2 of the ADMF document (Alberta Environment and Parks, 2022).

The critical loads approach simplifies the soil-vegetation health cycle system to facilitate the identification of a benchmark for the purpose of environmental management. The risk of acidifying impact of nitrogen and sulphur is defined by comparing atmospheric deposition of total nitrogen and sulphur to the critical loads. Determination of exceedances of various critical load values are part of management framework implementation (Alberta Environment, 2008) and thus outside the scope of this work.

Higher resolution critical load maps were developed to allow various levels of spatial averaging at a later time. Due to uncertainty associated with input data and the simplified SSMB model, the critical load values presented in this report are best used as a tool to identify potentially at risk areas. This information can then be used to plan further investigations, conduct monitoring and/or develop proactive management actions.

1 Introduction

Atmospheric emissions of sulphur dioxide and oxides of nitrogen are the predominant source of acidic deposition in Alberta. These compounds react with oxidizing agents in the atmosphere, such as hydroxyl radicals, to produce sulphur and nitrogen derived acids (Seinfeld, 1998). Such acids are removed from the atmosphere through wet and dry deposition. Wet deposition refers to the removal of atmospheric substances via precipitation. Dry deposition refers to the transfer of gases and particles from the atmosphere to the earth's surface by processes in the absence of precipitation, such as gravitational sedimentation or diffusion. Acidic deposition can degrade soil quality, and impact the health of terrestrial ecosystems.

In 1998, the federal, provincial and territorial Energy and Environment Ministers signed the Canada-Wide Acid Rain Strategy for Post 2000. For western Canada, the Strategy was initially designed to prevent an acidification problem from developing in areas identified as clean. To facilitate the management of acidic deposition in Alberta, a framework was developed and adopted in 1999 and revised in 2008 (Alberta Environment, 2008). The Alberta framework (2008) uses potential acid input to identify critical, target and monitoring acid deposition loads for three soil sensitivity classes that are mapped on a 1° by 1° resolution. A 2011 review of this framework identified the need to use a more detailed approach to detect areas at risk of impact from acidic deposition; and thus the steady-state mass balance (SSMB) model was selected as the method to determine critical loads of acidity for Alberta. This document outlines the data and methods used to determine critical loads of acidity for terrestrial ecosystems in Alberta using the SSMB model. The critical loads of acidity presented in this report are used in the 2022 version of the Alberta Acid Deposition Management Framework (Alberta Environment and Parks, 2022).

Critical loads based on soil sensitivity to the acidifying impact of sulphur and nitrogen are an alternative to judgement-based soil sensitivity classes. Critical loads may be derived from field experiments to determine dose-response relationships, SSMB model, or dynamic models. Whitfield & Watmough (2015) evaluated a range of models and identified the SSMB model as the most viable method for identifying "at risk" areas. Since the development of the current Alberta framework, the SSMB model has been increasingly used to identify critical loads of acidity (De Vries *et al*., 2015; Ouimet *et. al.*, 2006; McNulty *et. al.*, 2007; Skeffington *et. al.*, 2006; Mongeon *et al*., 2010; Reinds *et al*., 2008; Whitfield *et al*., 2010; Williston *et al*., 2016). The Convention on Long Range Transport of Air Pollution (CLRTAP, 2016, 2017) describes the steady-state mass balance model.

This work does not take into account the high nitrogen input and removal, and base cation removal associated with fertilizer application and agricultural production on cultivated agricultural land (Janzen *et. al.*, 2003). As a result, critical loads of acidity were not derived for cultivated agricultural land (hereafter referred to as cultivated land). Furthermore, areas identified as rock, exposed soil, water, ice or developed were also not included.

The best available input data known to the authors were used to develop this work. Where possible, Alberta specific input data have been used. Skeffington et al. (2006) conducted a review of propagation of input uncertainty to critical loads determined using SSMB and concluded the range of uncertainty in the final critical loads was relatively small due to compensation of errors. This being said, uncertainty in critical loads of acidity values arises from input data uncertainty and/or the SSMB approach itself. As a result, critical load values presented in this report are best used as a provincial scale tool to identify potential at risk areas. This information can be used to plan further investigations, monitoring and/or develop proactive management plans.

A further limitation of the SSMB model is that it does not estimate the length of time it takes for acidification effects to occur. Should areas at risk be identified using the SSMB model, a dynamic model (Hettelingh *et al*., 2007) and monitoring data may be used to refine our knowledge of impacts of acidic deposition on the identified area. Dynamic models are not widely used for provincial scale critical load assessments as they require a great deal of data. Alternatively, when an area at risk has been identified, an empirical study may be designed to examine ecosystem response to a changing deposition gradient in time or space (Laubhann *et al*., 2009; Solberg *et al*., 2009).

A number of the variables derived for critical loads of acidity may also be used to derive critical loads of nutrient nitrogen albeit there may be a need to adjust the assumptions. Derivation of Alberta based critical loads of nutrient nitrogen, limitations and need for improved data will be discussed in a separate document.

The model used to calculate the critical loads of acidity for water bodies differs from terrestrial ecosystems. As a result, critical load determination for water bodies will be conducted in a separate project.

2 Critical Loads of Acidity for Terrestrial Ecosystems

Critical loads of acidity establish maximum levels of deposition required to protect sensitive receptors from the adverse impacts of acidification. In terrestrial systems, critical loads of acidity are based on the sensitivity of plants to soil acidification. Interactions between soil and vegetation are complex; however, the SSMB approach to critical load estimation simplifies the soil-plant system. The SSMB combines a mass balance approach with climate and soil data and a critical threshold criterion to estimate when acid deposition reaches a critical threshold for plant growth.

Approaches for developing critical thresholds for soil acidity depend on soil characteristics. In mineral soils, acidic inputs lead to a drop in pH and an associated rise in dissolved aluminum ion (Al^{+3}) concentration, which can reach levels that are toxic to plants (Delhaize and Ryan, 1995). H^+ and Al^{+3} ions displace base cations (calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+) and sodium (Na^+)) from the exchange complex into the soil solution where they are subject to loss by leaching. Calcium, magnesium, and potassium are required for plant nutrition and if they are leached from the root zone in sufficient quantities, vegetation will be impacted by nutrient deficiencies. Base cations in the soil solution also mitigate the toxicity of aluminum, making the ratio of base cations to aluminum (BC:Al) a useful chemical criterion for evaluating negative effects of acid deposition on plants (Cronan and Grigal, 1995). In organic soil, the lack of a mineral component results in low soluble Al^{+3} concentrations. However, plant species distribution in bogs and fens are strongly related to pH and alkalinity (Vitt and Chee, 1990; Vitt *et al*., 1995). The relationships between plant growth and BC:Al and base cation to hydrogen ion (BC:H) ratios are similar, making BC:H a useful chemical criterion for organic soil (Sverdrup and Warfvinge, 1993).

The critical load function (illustrated in Figure 1, adopted from Mapping Critical Loads for Ecosystems (CLRTAP, 2017) is defined by three quantities: the maximum critical loads for sulphur ($CL_{max}(S)$) and nitrogen ($CL_{max}(N)$) deposition and the minimum critical load for nitrogen deposition ($CL_{min}(N)$). The maximum critical loads of sulphur ($CL_{max}(S)$) and nitrogen ($CL_{max}(N)$) represent the level at which the soil system cannot compensate for sulphur (S_{dep}) or nitrogen (N_{dep}) deposition when deposition of the other specie (nitrogen or sulphur) is zero. The minimum CL for nitrogen, $CL_{min}(N)$, marks the level below which N_{dep} does not contribute towards acidification (i.e., deposited nitrogen is taken up by plants and/or immobilized in the soil). The nutrient requirement for sulphur is low in forest ecosystems (Johnson, 1984); therefore, the minimum CL for sulphur ($CL_{min}(S)$) is set to zero.

The grey area in Figure 1 symbolizes total sulphur and nitrogen deposition combinations that do not exceed the critical load of acidity for mineral soils. Only $CL_{max}(S)_{org}$ is calculated for organic soil; nitrogen deposition is assumed to have negligible acidifying effect on organic soils. The hatched area below $CL_{max}(S)_{org}$ symbolizes total sulphur deposition that does not exceed the critical load of acidity for organic soils.

Derivation of the four quantities ($CL_{max}(S)$, $CL_{max}(S)_{org}$, $CL_{max}(N)$ and $CL_{min}(N)$) are presented in Sections 2.1 and 2.2.

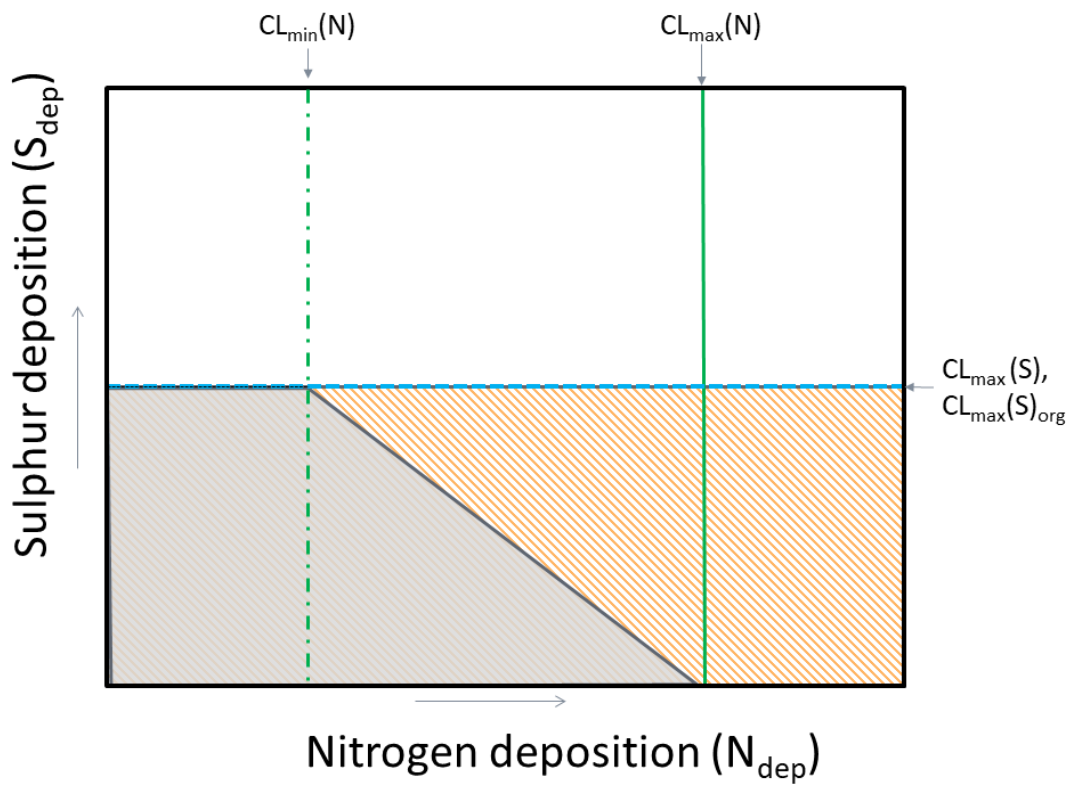

Figure 1. Critical Load Function for soils is defined by CLmax(S), CLmax(S)org, CLmax(N) and CLmin(N). The grey area denotes Sdep and Ndep pairs that do not exceed critical loads of acidity for mineral soils. The hatched area denotes Sdep that does not exceed critical load of acidity for organic soils. CLmax(S) and CLmax(S)org are not calculated in the same way.

2.1 Critical Loads of Sulphur

Critical loads of sulphur are calculated for mineral and organic soils. $CL_{max}(S)$ for mineral soil is calculated in Eq 1. $CL_{max}(S)$ for organic soil is adapted from Eq 1 by substituting porewater buffering for BC_W, as explained in Section 2.1.3.

$$CL_{max}(S)_{min} = BC_{dep} + BC_w - Cl_{dep} - Bc_u - ANC_{le,crit} \qquad \textbf{Eq 1}$$

The terms used in Eq 1 for mineral soils are described in Table **1**. The implementation of Eq 1 for organic soils is described section 2.1.3.

Table 1. Variables used in the deriviation Critical Loads of acidity for mineral soils

Variable	Description	Source/Derivation
$CL_{max}(S)_{min}$	Critical load of sulphur for mineral soils (eq ha^{-1} yr^{-1})	Eq 1
BC_{dep}	Non-marine annual base cation deposition, which is the sum of sodium (Na^+), calcium (Ca^{2+}), potassium (K^+), magnesium (Mg^{2+}) deposition (eq ha^{-1} yr^{-1})	Estimates of base cation deposition derived from monitoring stations for the years 1994 to 1998 are used to represent average annual deposition (Aherne J, 2008) See Section 2.1.1
BC_w	Release of base cations as a result of chemical dissolution from the soil mineral matrix; determined using a soil texture approximation method (eq ha^{-1} yr^{-1})	See Section 2.1.2
Cl_{dep}	Non-marine chloride deposition $Cl_{dep} = 0.29 \times BC_{dep}$	Non-marine chloride deposition is calculated using the molar ratio of chloride ion and base cation within precipitation collected at 4 regional sites between 2014-2018. The average molar ratio for the period indicated is 0.29.
Bc_u	The average base cation (Ca^{2+}, Mg^{2+}, K^+) removal in harvested biomass (eq ha^{-1} yr^{-1})	See Section 2.1.2.2
$ANC_{le,crit}$	Acid neutralizing capacity leaching (eq ha^{-1} yr^{-1})	See Section 2.1.2.3

2.1.1 Base Cation Deposition

The neutralizing capacity of the soils is determined in part by BC_{dep} (deposition). Long term average (>10 years) non-anthropogenic information would be the ideal representation of long term deposition (CLRTAP, 2016). However, since these data were not available, the estimated average total (wet and dry) base cation deposition for the years 1994 to 1998 was used to represent long term base cation deposition (K^+, Na^+, Ca^{2+} and Mg^{2+}) for the province (Figure **2**). These data were provided by Environment and Climate Change Canada on the Global Environmental Multiscale (GEM) grid at a resolution of 35 km x 35 km (see Aherne J, 2008). Base cation deposition was estimated using wet deposition, air concentration and modelled meteorological data. Meteorological data from GEM were used to estimate dry deposition velocities for each land use.

Fenn *et. al.* (2015) and Watmough *et. al.* (2014) reported base cation deposition rates in the oil sands region of Alberta that are notably higher than values represented in Figure **2**. An annual throughfall deposition 3 km from mining activities was reported as 3111 eq ha^{-1}. This deposition decreased exponentially with distance from source to an annual deposition of 235 eq ha^{-1} at about 113 km from major surface-mining facilities (Fenn *et al.*, 2015). The impact of high levels of base cation deposition on the ecosystem is an ongoing area of study (Mandre *et al*. 2008; Paal *et al*. 2013). Furthermore, due to mitigation efforts, base cation deposition from industrial activity may change over time. BC_{dep} from anthropogenic activities such as open pit mining in the oils sands may mask impacts from acid deposition (Fenn *et al*. 2015; Watmough *et al*. 2014) and should be

considered when ground truthing potential impact. This being said, elevated BC_{dep} associated anthropogenic activities cannot be used as a solution to acidic deposition. As an area of ongoing study, we do not yet fully understand the impact of increased BC_{dep} on an ecosystem. As a result, BC_{dep} from anthropogenic activities such as oil sands mining are not included in this work.

2.1.2 Critical Loads of Sulphur for Mineral Soils

2.1.2.1 Base Cation Weathering

In addition to buffering through base cation deposition, base cation weathering (BCw) also provides buffering capacity to mineral soils. The soil texture approximation method (CLRTAP, 2017) was used to determine base cation weathering. Using Eq 2, the median base cation weathering rate determined for European soils (de Vries et al., 1993) was adjusted for relative differences in weathering rate due to soil texture, pH and temperature (see Appendix B). The terms used in Eq 2 are described in **Table 2**.

$$BC_w = 500 \times (W_{class} - 0.5) \times depth_{root} \times 10^{\left(\frac{A}{281} - \frac{A}{273+T}\right)}$$ **Eq 2**

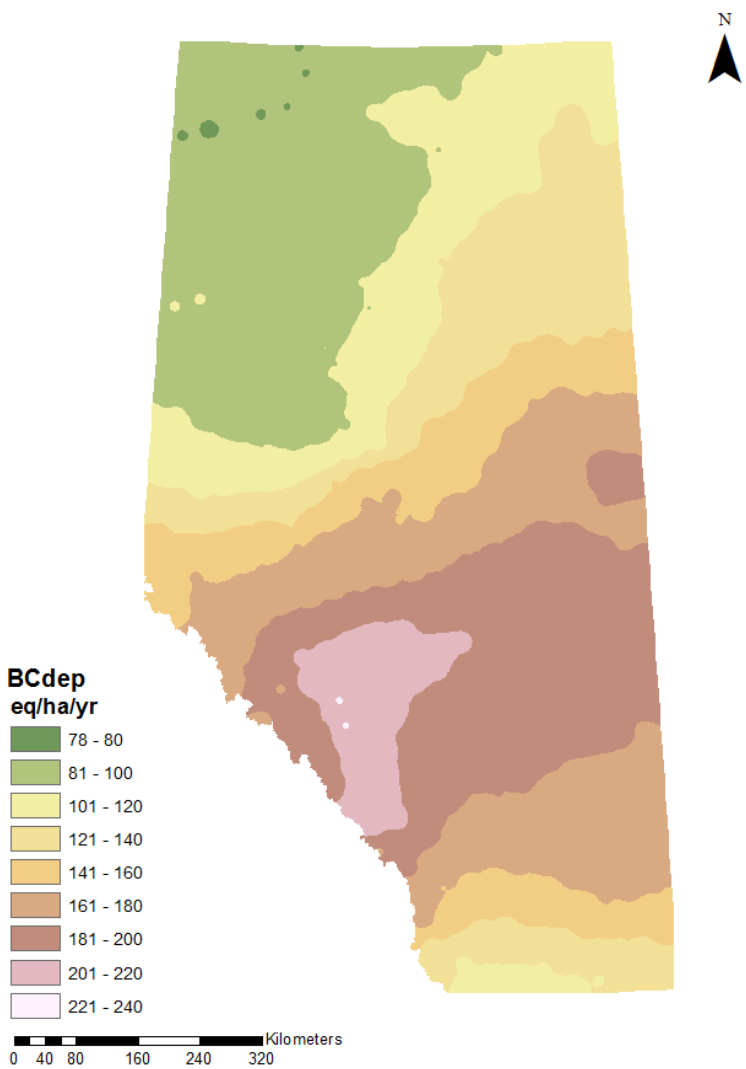

Figure 2. Long term average land use weighted base cation BC_{dep} deposition (1994 to 1998)

Table 2. Terms used to determine base cation weathering in Eq 2

Term	Description	Source
BC_w	Base cation (Ca, Mg, K, Na) weathering (eq $ha^{-1}yr^{-1}$)	
500	Median base cation weathering rate (eq $ha^{-1}yr^{-1}m^{-1}$)	de Vries *et al*., 1993
W_{class}	Weathering rate class based on soil texture (clay and sand content) and soil pH up to rooting depth.	
$depth_{root}$	Rooting depth set to 0.5 m	Watmough & Dillon, 2002
A	A temperature coefficient for soil weathering (3600 K)	Sverdrup *et al*. as cited in CLRTAP, 2017
T	Long term annual average soil temperature (K) mapped from modelled data	Zhang *et al*., 2005

In order to derive $ANC_{le,crit}$ (Section 2.1.2.3) the weathering rate for K^+, Ca^{2+} and Mg^{2+} (Bc_w, does not include Na^+) is needed. Bc_w can be approximated from BC_w (includes Na^+) by applying a factor related to soil texture (CLRTAP, 2017). The following factors were applied based on soil texture:

- 0.70 for poor sandy soils with sand ≥65%
- 0.85 for rich sandy soils with sand <65% and clay <35%
- 1.00 for soils where clay content ≥35%

Figure **3** illustrates the distribution of BC_w in the province.

2.1.2.2 Base Cation Uptake

Base cation uptake was based on removal of base cations by forest harvest and removal by grazing animals from native grassland. Bc_u for shrubland was assumed to be negligible. A similar long-term removal of base cations occurs in organic soils due to the slow accumulation of peat. A full examination of base cation sequestration in peat was beyond the scope of this project, but is recommended for future revisions.

An average base cation uptake due to forest harvesting was calculated for areas dominated by mineral soil in each natural subregion in Alberta. Subregions of Alberta are illustrated in Appendix E.

To calculate the amount of base cation removal due to forest harvest in each natural subregion, the percentage of each forest management unit within each natural subregion was determined. The annual median harvest volume (Alberta Agriculture and Forestry, 2015) for each management unit was calculated using information for the period of 2006 to 2015 (Figure **E-2**). If more than one forest management unit occurs within the natural subregion, base cation uptake was calculated for each forest management unit and the results combined for all management units. Base cations removed by harvest for each natural region were calculated using Eq 3.

$$Ca, Mg, KFH_{NR} = \frac{\sum_{FMU=a}^{n} \left(H \times \frac{P_{Ca,Mg,K}}{100} \times \rho \times \frac{P_{FMU}}{100} \right)_a}{Area_{NR}} \quad \textbf{Eq 3}$$

Where:

Ca,Mg,KFH_{NR}	=	base cations removed by forest harvest (kg ha^{-1} yr^{-1}). Base cation removal is calculated separately for Ca, Mg, and K.
FMU	=	forest management unit
n	=	number of forest management units in natural subregion
H	=	annual median harvest volume (m^3) for each forest management unit *a*
ρ	=	density of hardwood (390 kg m^{-3}) and coniferous (400 kg m^{-3}) stemwood (Gonzalez, 1990)
$P_{Ca,Mg,K}$	=	percent base (Ca, K and Mg) content of stem wood and bark for coniferous and hardwood trees (Eq 4)
P_{FMU}	=	percentage of each forest management unit within natural subregion
$Area_{NR}$	=	area of natural subregion

The general practice in timber harvest is stem-only removal; where leaves/needles and branches are handled in various ways (e.g., left in scattered piles in the harvested area). Therefore, base cation removal was estimated to include only the mass in tree stems. Base cation content was estimated using Eq 4 and:

- Relative proportion of stem and bark biomass. Data for mature lodgepole pine stands in Alberta (Monserud *et al*., 2006) was used for conifers and data for aspen poplar from British Columbia (Wang *et al*., 1995) was used for hardwoods.
- Nutrient content by tree component (Paré *et al*., 2013). Base cation content of aspen was used to represent hardwood forests and the average value of lodgepole and jack pine represented conifers.

$$P_{Ca,Mg,K} = \left(\frac{Stem_M}{Total_M} \times P_{SB} \right) + \left(\frac{Bark_M}{Total_M} \times P_{WB} \right) \quad \textbf{Eq 4}$$

Where:

$P_{Ca,Mg,K}$	=	percent Ca, K and Mg content of stem wood and bark for coniferous and hardwood trees
$Stem_M$	=	Mass of stemwood (kg ha^{-1})
$Bark_M$	=	Mass of bark (kg ha^{-1})
$Total_M$	=	Mass of stemwood and bark (kg ha^{-1})
P_{SB}	=	Percent base (Ca, Mg, K) content in stemwood
P_{WB}	=	Percent base (Ca, Mg, K) content in bark

The calculated percent base cation contents (P_{Bc}) for hardwood trees and coniferous are listed in Table **3**.

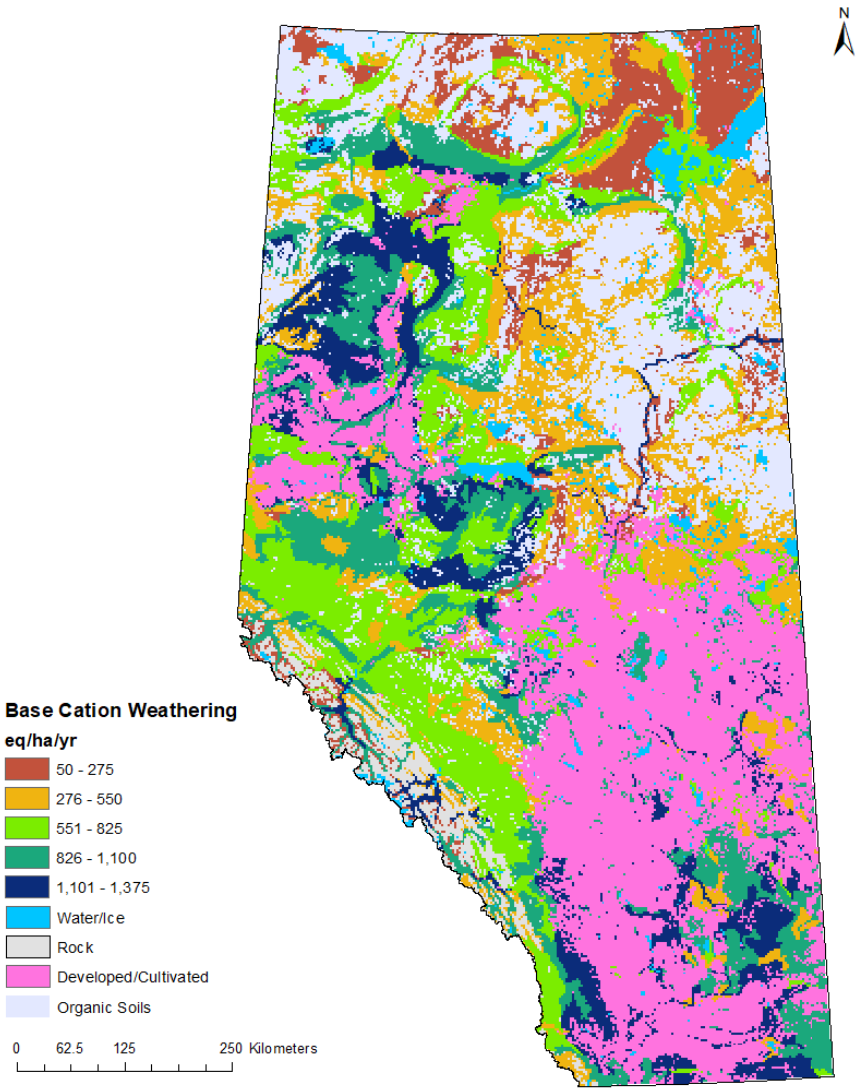

Figure 3. Base cation weathering (BC_w) for mineral soils determined using the soil texture approximation method

Table 3. Stem Nutrient content for Hardwood and Coniferous trees

Nutrient	**Hardwood**	**Coniferous**
	Nutrient concentration (%)	
Ca	0.40	0.16
Mg	0.05	0.02
K	0.14	0.05

Removal of Ca, Mg, and K by harvest is converted to an equivalent charge basis and added together to calculate total base cation removal using Eq 5.

$$Bc_uH_{NR} = \left[\left(Ca_R \times \frac{Eq_{Ca}}{MW_{Ca}}\right) + \left(Mg_R \times \frac{Eq_{Mg}}{MW_{Mg}}\right) + \left(K_R \times \frac{Eq_K}{MW_K}\right)\right] \times 1000 \qquad \textbf{Eq 5}$$

Where:

Bc_uH_{NR} = Base cation uptake and removal by harvesting (eq ha^{-1} yr^{-1})

Ca,Mg,K_RH_{NR} = Uptake and removal of Ca, Mg, and K by harvesting (kg ha^{-1} yr^{-1})

$MW_{Ca,\ Mg,\ K}$ = Molecular weight of Ca, Mg, and K (g/mole)

$Eq_{ca,\ Mg,\ K}$ = Equivalent charge per mole (Ca=2, Mg=2, K=1)

Like forested areas, native grasslands are typically managed without the addition of fertilizer. Base cations can be removed from native grasslands through grazing and subsequent removal of animals from the range. To account for this loss, Eq 6 was used to estimate the removal of base cations by grazing from areas of native grassland within the Dry Mixed Grass, Mixed Grass, Foothills Fescue, Northern Fescue, and Fescue Parkland natural subregions. Areas of native grassland were identified using the Grassland Vegetation Inventory (GVI) (Government of Alberta, 2011). The GVI classifies native grassland areas according to site type, which is determined by vegetation, soil, and hydrological regime. Removal of each base cation ($Ca,Mg,K\ GR_{ST}$) was estimated separately for each site type in the natural subregion using Eq 6. Ecologically sustainable stocking rates for each site type and a standard forage consumption rate per animal unit (AU) were obtained from Alberta Range Plant Community Guides (Adams *et al*., 2003, 2013a, 2013b; Adams *et al*., 2019; DeMaere *et al*., 2012). Calcium, magnesium, and potassium concentrations in native vegetation from Grings *et al*. (1996) were used to estimate base cation ingestion rates. Loss of base cations from the system due to grazing was assumed to be equal to the amount retained in grazing animal live weight gain (Whitehead, 2000). Bc_u was calculated by adding together the uptake estimates for calcium, magnesium, and potassium after converting them to equivalents, using Eq 5.

$$Ca,Mg,K\ GZ_{ST} = \frac{F_C \times Bc_F \times Bc_R}{ESSR_{ST} \times 10000} \qquad \textbf{Eq 6}$$

Where:

$Ca,Mg,K\ GZ_{ST}$ = Mass of calcium, magnesium, or potassium removed by grazing (kg ha^{-1} yr^{-1})

F_C = Forage consumption (340 kg AU^{-1} yr^{-1})

Bc_F = Calcium (0.23%), magnesium (0.1%), or potassium (0.7%) content of forage

Bc_R = Calcium (17%), magnesium (5.6%), or potassium (10%) retained in live weight gain (% of Ca, Mg, or K consumed)

$ESSR_{ST}$ = Ecologically sustainable stocking rate for each site type within a subregion (ha AU^{-1})

Central Parkland and Peace River Parkland subregions are extensively cultivated with relatively little area remaining in native grassland. Bc_u for these regions were ≤1 eq ha^{-1} yr^{-1}. The Peace Athabasca Delta subregion is composed largely of shallow lakes and wetlands, while the Kazan subregion is not harvested. These two regions were assigned $Bc_u = 0$ eq ha^{-1} yr^{-1}. Figure **E-1** illustrates Alberta natural subregions. The average base cation uptake by natural subregion are illustrated in Figure **4**.

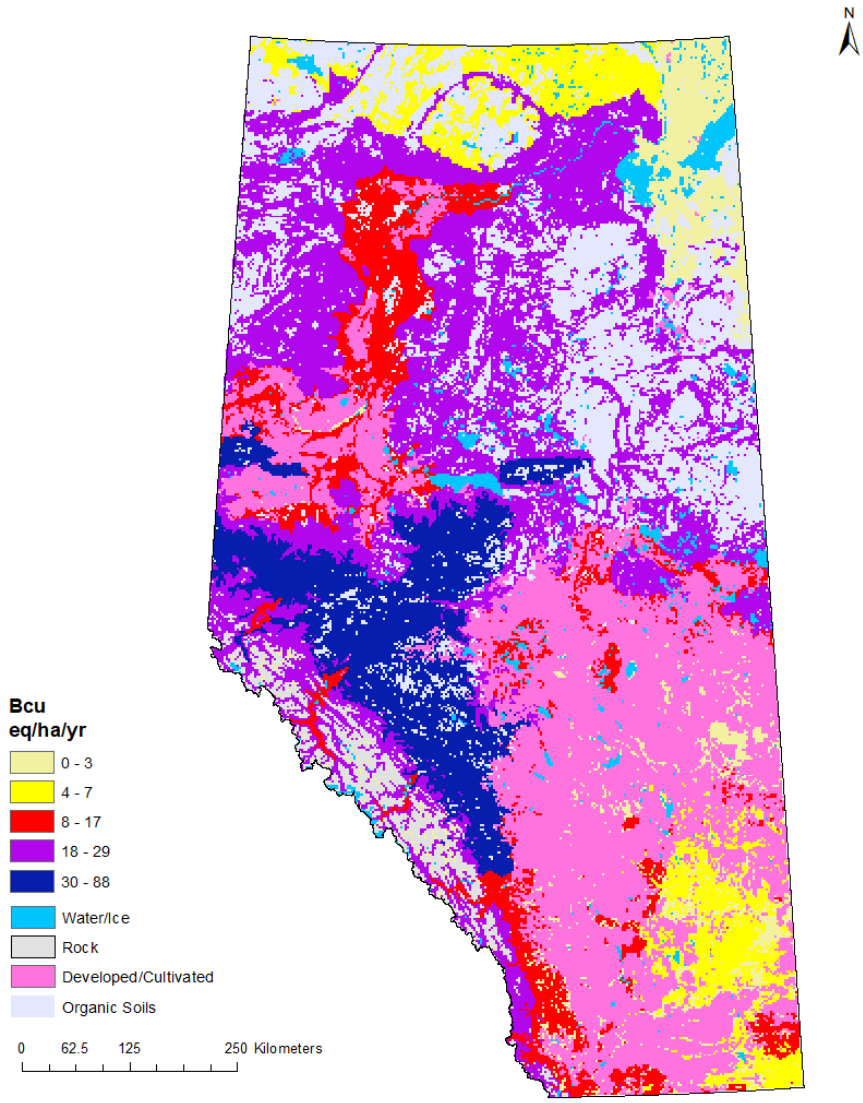

Figure 4. Average base cation uptake by natural subregion

2.1.2.3 Critical Acid Neutralizing Capacity Leaching for Mineral Soil

The critical acid neutralizing capacity leaching ($ANC_{le,crit}$) estimates the maximum removal of base cations the system can handle before biological impact is observed. Areas with high $ANC_{le,crit}$ indicate soils that have low tolerance for leaching.

For mineral soils the $ANC_{le,crit}$ is calculated with Eq 7. Equation variables are described in Table **4**.

$$ANC_{le,crit} = -Q^{2/3} \times \left(1.5 \times \left(\frac{Bc_{dep} + Bc_w - Bc_u}{K_{gibb} \times (Bc:Al)_{crit}}\right)\right)^{1/3} - \left(1.5 \times \left(\frac{Bc_{dep} + Bc_w - Bc_u}{(Bc:Al)_{crit}}\right)\right) \quad \textbf{Eq 7}$$

There is a minimum concentration in soil solution below which base cations cannot be taken up by vegetation (CLRTAP, 2016); as a result, the base cation leaching (Bc_{le}) is constrained as shown in Eq 8.

$$Bc_{le} = \left(Bc_{dep} + Bc_w - Bc_u\right) > 0 \qquad \textbf{Eq 8}$$

$$\therefore\ Bc_{le} = \max\{Bc_{dep} + Bc_w - Bc_u, Q[Bc]_{min}\}$$

Table 4. Variables used to derive acid neutralizing capacity for mineral soils

Term	Description	Source
$ANC_{le,crit}$	Critical acid neutralizing capacity leaching (eq ha^{-1} yr^{-1})	Eq 7
Q	Estimated runoff or soil percolation (m^3 ha^{-1} yr^{-1})	See Appendix C
Bc_{dep}	Non-marine base cation (Ca^{2+}, K^+, Mg^{2+}) deposition (eq ha^{-1} yr^{-1})	Observation-based estimates of base cation deposition derived from monitoring stations for the years 1994 to 1998 (eq ha^{-1} yr^{-1}) were used to represent average annual deposition (Aherne J., 2008).
$[Bc]_{min}$	0.01 eq m^{-3}	CLRTAP, 2016
Bc_w	For mineral soils - weathering rate used in ANC leaching and approximated from BC_w (eq ha^{-1} yr^{-1})	(See Section 2.1.2)
K_{gibb}	Aluminum dissolution constant for mineral soil with low (<5%) organic matter (300 m^6/eq^2)	(CLRTAP, 2017)
$(Bc{:}Al)_{crit}$	Critical base cation to aluminum ratio is the connection between soil chemical status and plant damage used to calculate critical alkalinity leaching for mineral soils. Land cover is used to assign chemical criteria.	Table **5**

For mineral soils the critical ratio of base cations to aluminum $(Bc{:}Al)_{crit}$ establishes the minimum ratio below which adverse effects to plants may occur.

The (Bc:Al)crit values (**Table 5**) were calculated by applying a weight of evidence approach (Canadian Council of Ministers of the Environment, 2006) to the data provided by Sverdrup and Warfvinge (1993). Sverdrup and Warfvinge provide Bc:Al ratios for a range of plant species that correlate with a 20% reduction in growth. The (Bc:Al)crit data for each species were sorted into the land cover types: coniferous forest, broadleaf forest, mixed forest, shrubland and grassland. Under the weight of evidence approach, (Bc:Al)crit data for each vegetation type were ranked and the 75th percentile was calculated. This value was checked against the list of vegetation species in Alberta to ensure that none of the (Bc:Al)crit ratios for significant Alberta species were greater than the 75th percentile value. If the value passed this check, the ratio was used as the critical value. If one or more significant Alberta species were above the 75th percentile, the critical value was set to protect those species. Further explanation on method used is found in Appendix I.

Table 5. $Bc{:}Al_{crit}$ values assigned based on land cover

Land Cover	$Bc{:}Al_{crit}$ assigned	Land Cover Description from the ABMI Land Cover Map Guide
Mixed Forest	6	Treed areas with at least a 10% crown closure of trees, where neither coniferous nor broadleaf trees account for 75% or more of crown closure.
Shrubland	6	At least 20% ground cover which is at least one-third shrub, with no or little presence of trees (<10% crown closure). Examples of plants belonging to this class are alder, willow, juniper, and sagebrush. Shrubby fens and other non-treed woody wetlands, usually associated with floodplains and the shores of lakes and streams, belong to this class. Includes cutblocks where trees are still < 2m in height, and recently burned forest areas
Broad Leaf forest	6	Treed areas with at least a 10% crown closure of trees, where broadleaf trees (trembling aspen, balsam poplar and white birch) are 75% or more of the crown closure. Providing crown closure is more than 10% and dominated by broadleaf trees, young plantations or regenerating cutblocks, and treed swamps along floodplains or wetlands are included in this class providing mean tree height exceeds 2 m.
Coniferous	2	Treed areas with at least a 10% crown closure of trees, where coniferous trees (spruce, pine, fir, larch) are 75% or more of the crown closure. Providing crown closure is more than 10% and dominated by conifers, young plantations or regenerating cutblocks, and treed wetlands (e.g. black spruce bogs and fens) are included in this class providing the mean tree height exceeds 2 m.
Grassland	40	Predominantly native grasses and other herbaceous vegetation with a minimum of 20% ground cover; may include some shrub cover (but less than a third of the vegetated area) or a few trees (but the tree cover cannot exceed 10%). Land used for range or native unimproved pasture (e.g., rough fescue) is included in this class. Alpine meadows fall into this class. Marshes and other non-woody wetlands with at least 20% vegetation cover (sedges, cattails, or moss) belong to this class. Note: A forestry cutblock harvested more than a year ago containing seedlings with less than 10% cover, belongs to this class. If the cutblock had no successful regeneration and is covered by more than 20% shrubs, it would belong to the 'Shrubland' class.

Critical values for coniferous forest and grassland were set at the 75th percentile. In the case of broadleaf trees, the $(Bc{:}Al)_{crit}$ for aspen poplar (6) was higher than the 75th percentile for all broadleaf trees (2). Aspen is an major species in Alberta's broadleaf and mixed wood forests, therefore the $(Bc{:}Al)_{crit}$ was set at 6 for these vegetation types, as well as for shrubland.

The (Bc:Al)crit values were used in conjunction with Alberta land cover information (ABMI, 2010) to assign spatially varying (Bc:Al)crit values (**Table 5**). ANCle,crit for Alberta ranged from -19 to >1000 eq ha-1yr-1, with lower values distributed in western regions and is illustrated in **Figure 5**.

2.1.3 Critical Loads of Sulphur for Organic Soils

2.1.3.1 Buffering Capacity for Organic Soils

Base cation weathering is minimal in organic soils, which contain very little mineral material. Instead, porewater alkalinity estimates are used to account for buffering capacity in organic soil.

Buffering capacity in fen systems occurs below the water table due to dissolved bicarbonate. Water table depths (W_t) of moderate and extreme rich fens were set to 10 cm (Zoltai *et al.*, 2000). Alkalinity buffering does not occur in bogs or poor fens because the pore water is too acidic. An average buffering capacity for moderate and extreme rich fens of 820 and 5460 eq ha^{-1} yr^{-1}, respectively, was used based on data from the Athabasca oilsands region (Abboud *et al*., 2002). A depth weighted average buffering capacity ($Buffer_w$) was calculated as follows:

$$Buffer_w = \left(1 - \frac{W_t}{depth_{root}}\right) \times Buffer_{fen} \qquad \textbf{Eq 9}$$

Where:

$Buffer_w$ = Depth weighted porewater buffering capacity (eq ha^{-1} yr^{-1})

W_t = Water table depth

$depth_{root}$ = Rooting depth (50 cm)

$Buffer_{fen}$ = Porewater buffering capacity (eq ha^{-1} yr^{-1})

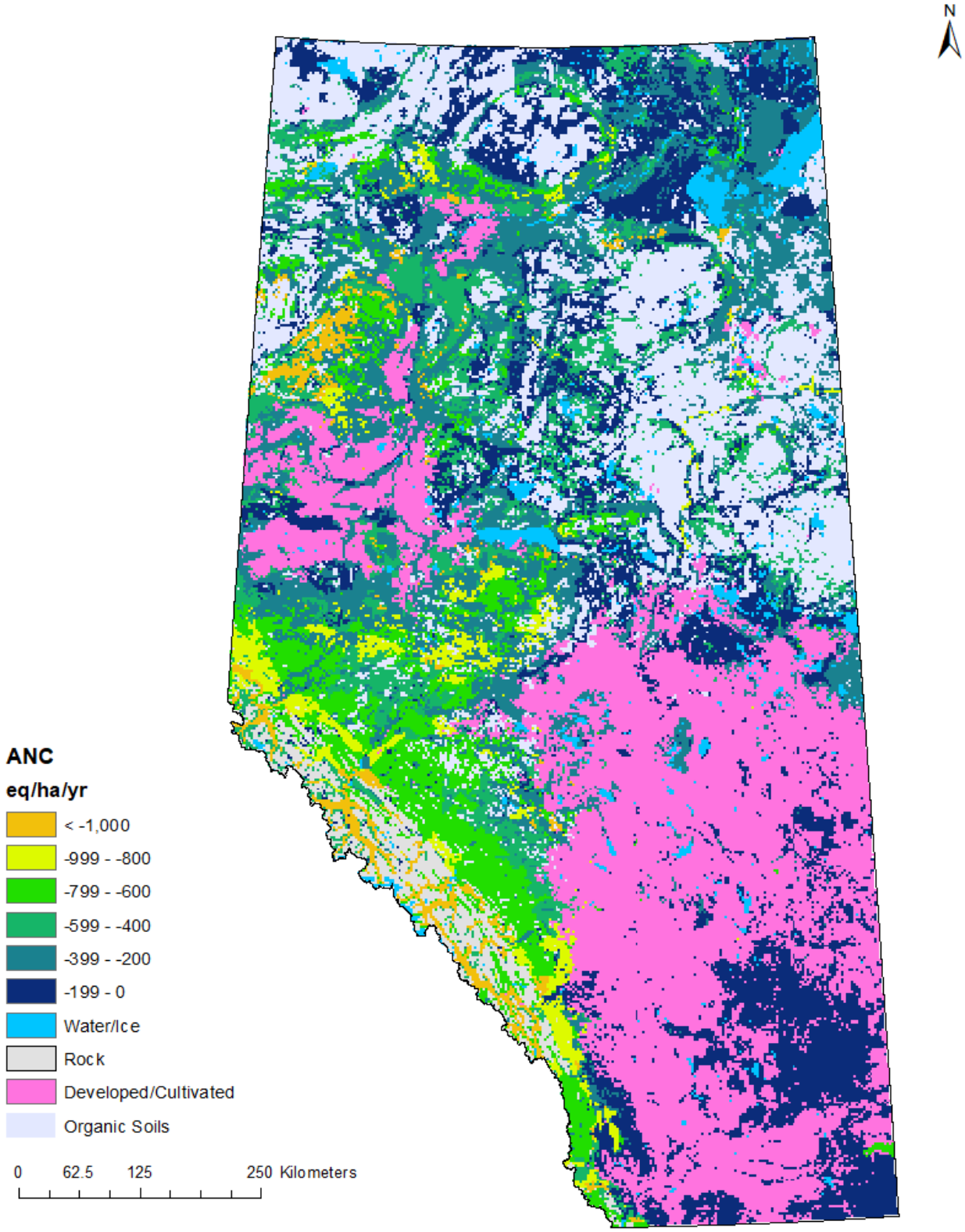

Figure 5. Critical leaching of soil acid neutralizing ($ANC_{le,crit}$) capacity for mineral soils

2.1.3.2 Critical Acid Neutralizing Capacity Leaching for Organic Soil

Acid neutralizing capacity leaching for organic soil ($ANC_{le,crit,org}$) calculated as:

$$ANC_{le,crit,org} = -0.5 \times \left(\frac{Bc_{dep} + Buffer_w}{(Bc:H)_{crit}}\right) \quad \textbf{Eq 10}$$

Where:

Bc_{dep} = base cation deposition (Section 2.1.1)

$Buffer_w$ = buffering capacity (Eq 9)

$Bc:H_{critical}$ = critical base cation to hydrogen ion ratio

Organic soils can have low exchangeable Al^{3+} levels (Thomas, 1976; Närhi, *et al*., 2013); thus, it has been suggested that $ANC_{le,crit,org}$ be calculated using the ratio of base cations to hydrogen ions $(Bc:H)_{crit}$ (Nordic Council of Ministers, 2000). The critical thresholds for $(Bc:H)_{crit}$ for bog and fen systems were derived using the relationship established in Abboud *et al*. (2002). Bc:H values used are listed in Table **6**. The $ANC_{le,crit,org}$ for fen systems ranged from -1.4 to 0.2 eq ha^{-1} yr^{-1}. For bog systems $ANC_{le,crit,org}$ were lower and ranged from -31 to -10 eq $ha^{-}1$ yr^{-1}. For both systems, values were notably higher than values for mineral soils, which means this type of soil can tolerate the least amount of leaching due to low buffering capacity. As illustrated in Figure **5**, organic soils are distributed in areas where adjacent mineral soils have the highest ANC values, which indicates a low tolerance for leaching due to low Bc_w.

Table 6. Critical Bc:H for the organic soil types

Organic soil type	**Critical pH**	$(Bc:H)_{crit}$
Extreme Rich Fen	6.5	12400
Moderate Rich Fen	5.5	780
Poor Fen	4.5	50
Bog	3.5	3

2.1.3.3 Maximum Critical Load for Sulphur

$CL_{max}(S)$ benchmarks the levels above which the soil system cannot compensate for S_{dep}. Figure **6** illustrates $CL_{max}(S)$ values for both mineral and organic soils. Figure **G-1** and Figure **G-2** illustrate $CL_{max}(S)$ for mineral and organic soils separately.

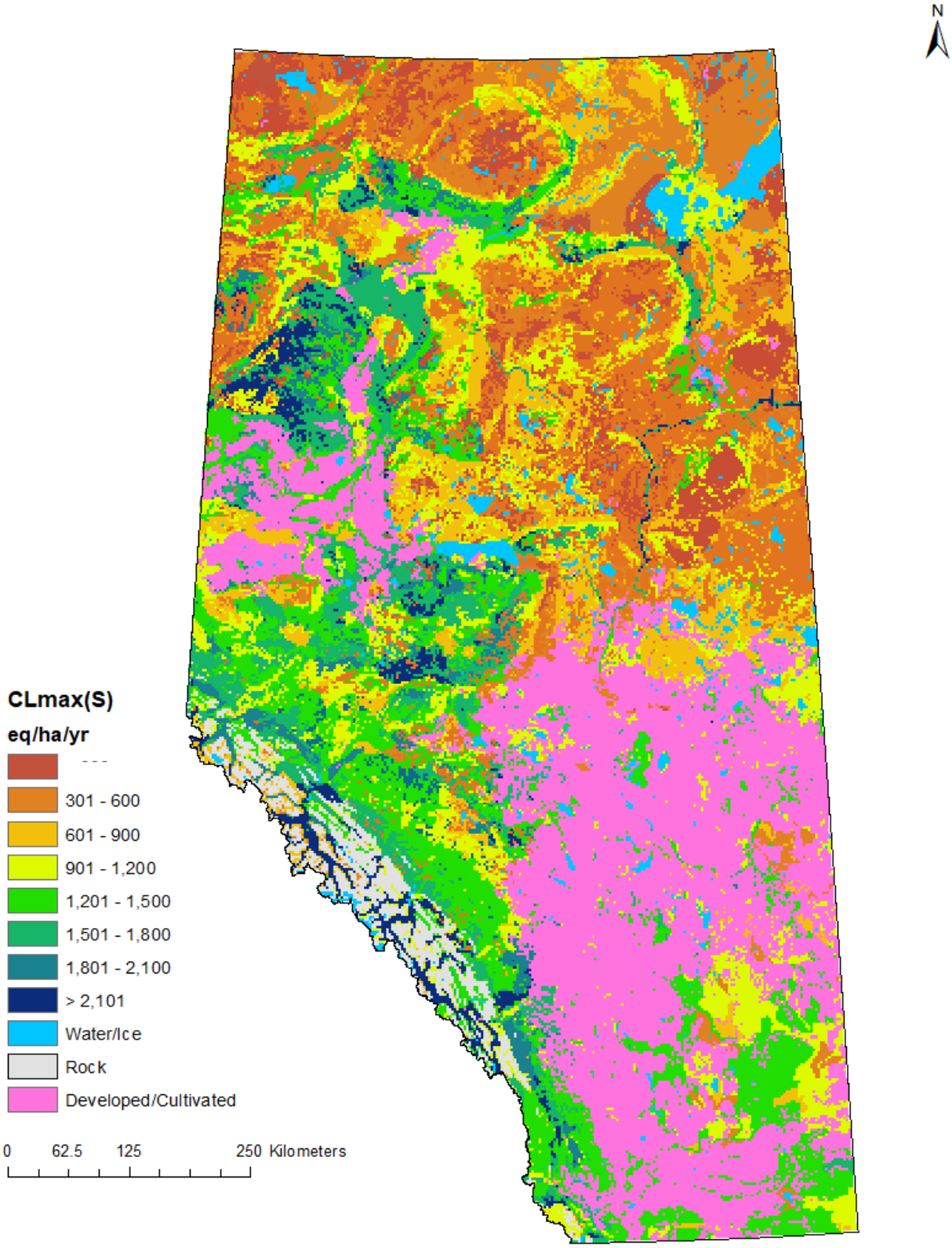

Figure 6. *CLmax(S)* for mineral and organic soils

2.2 Critical Loads of Acidifying Nitrogen

Like sulphur, loss of nitrogen is acidifying when it leaches from the soil system. However, calculation of critical loads for nitrogen is more complex than for sulphur and requires determination of both maximum ($CL_{max}(N)$) and minimum ($CL_{min}(N)$) critical loads for nitrogen (see Section 2). The maximum critical load for nitrogen is assumed to equal the maximum critical load for sulphur after accounting for the loss of nitrogen by denitrification, which is not acidifying, and permanent removal of nitrogen by the biological components of the ecosystem ($CL_{min}(N)$). $CL_{max}(N)$ is determined using Eq 11. The terms used in Eq 11 are listed and described in Table **7**.

$$CL_{max}(N) = CL_{min}(N) + \left(\frac{CL_{max}(S)}{(1 - f_{de})}\right) \qquad \textbf{Eq 11}$$

2.2.1 Minimum Critical Load for Nitrogen

The minimum CL for nitrogen, $CL_{min}(N)$, is the level below which N_{dep} is either removed from the system in biomass or immobilized in soil organic matter before it can contribute to acidification. Immobilization in soil organic matter (N_i) is considered a permanent removal, while the nitrogen removed by plant biomass is only permanent if the biomass is removed from the system by harvest or fire. Two methods of estimating $CL_{min}(N)$ are explored.

(1) The first method uses a long term net immobilization of nitrogen (N_i) value and removal of nitrogen from the ecosystem (Nu) due to the removal of biomass from managed forests or from grazing and subsequent removal of animals in grassland regions. This method is hereafter referred to as the Harvest regime **(**$CL_{min}(N)_h$**)**. $CL_{min}(N)_h$ is determined using Eq 12. The terms used in Eq 12 are listed in Table **7**.

$$CL_{min}(N)_h = Ni_{NSubR} + N_{u,h} \qquad \textbf{Eq 12}$$

(2) The second method considers Ni and the loss of nitrogen due to biomass harvest, but also includes the loss of nitrogen due to forest fires. Hereafter, this method is referred to as the **Harvest and wildfire regime (**$CL_{min}(N)_{hf}$**)**. This method assumes that modelled annual total deposition used to determine critical load exceedances adequately characterizes the emission, transport and deposition of nitrogen emissions from wildfire. $CL_{min}(N)_{hf}$ is determined using Eq 13. The terms used in Eq 13 are listed in Table **7**.

$$CL_{min}(N)_{hf} = Ni_{NSubR} + N_{u,comb} \qquad \textbf{Eq 13}$$

Table 7. Variables used in the deriviation of the Critical Loads of Acidifying Nitrogen

Variable	Description	Source/Derivation
f_{de}	Fraction of nitrogen that is removed from the soil by denitrification. Used to relate $CL_{max}(N)$ to $CL_{max}(S)$ Values are assigned as a function of soil drainage Very rapidly drained ---0 Well drained --- 0.1 Moderately well drained --- 0.2 Imperfectly drained ---0.4 Poorly drained ---0.7 Very poorly drained ---0.8 Bedrock – not considered	Soil Landscape of Canada (2011); CLRTAP, 2016
Ni_{NSubR}	Long term net immobilisation of nitrogen	See Section 2.2.1.1
$N_{u,h}$	Average removal of nitrogen from an ecosystem by harvest	See Sections 2.2.1.2
$N_{u,comb}$	Average removal of nitrogen from an ecosystem by harvest and fire combined	See Sections 2.2.1.2, 2.2.1.3, and 2.2.1.4

2.2.1.1 Estimation of Nitrogen Immobilisation

Nitrogen immobilization (*Ni*) refers to the net accumulation of nitrogen in soil organic matter since soil formation began after glaciation.

Alberta specific *Ni* for forested areas was calculated using nitrogen content of mineral soil horizons in the Shaw *et al*. (2018) forest soil database. *Ni* for grassland sites was calculated using a combination of literature data and monitoring data, as described below. An average *Ni* was calculated for each natural subregion (Ni_{NSubR}) (Eq 14) and then converted to equivalent charge basis. The distribution of soil sites and natural sub regions are illustrated in Figure H-1.

$$\boldsymbol{Ni_{NSubR} = \sum_{R=1}^{m} \frac{N_{soil,m}}{soil\ age}} \qquad \textbf{Eq 14}$$

Where:

Ni_{SubR} = the nitrogen immobilization (kg ha^{-1} yr^{-1}) estimated by natural subregion

$N_{soil,\ m}$ = the nitrogen content (kg ha^{-1}) for each soil site *m*

Soil age = the glacial age of Alberta mineral soil (*soil age*), estimated as 11,000 years (Christiansen, 1979).

The nitrogen content of mineral soil horizons ($N_{soil,m}$) was calculated using the method outlined in Appendix H.

There were no soil sample sites (Figure H-1) within the Boreal Subarctic. The soil orders for this region were similar to the Northern Mixedwood subregion. Both subregions were dominated by Orthic Gray and Gleyed Gray Luvisols developed on till with organic soils in wetland regions (Downing & Pettapiece, 2006). Therefore, the Northern Mixedwood *Ni* value of 36 eq $ha^{-1}yr^{1}$ was applied to the Boreal Subarctic subregion.

There were also no sample sites (Figure H-1) within the Foothills Fescue subregion, which contained mainly Black Chernozem soils (Downing & Pettapiece, 2006). The *Ni* value for this subregion was set to be equivalent to the Foothills Parkland (130 eq $ha^{-1}yr^{1}$) where soils were also predominantly Black Chernozems. A value of 130 eq $ha^{-1}yr^{1}$ was calculated for Black Chernozem *Ni*, based on nitrogen data from Li *et al.* (2012).

Nitrogen content for the Dry Mixedgrass subregion was initially calculated using Alberta Environment and Park's Ecological Recovery monitoring program soil data (Figure H-1). These soil samples were collected at two depths, 0-15 cm and 15-30 cm. The average *Ni* up to 30 cm was 30 eq $ha^{-1}yr^{1}$. Although nitrogen content in mineral soil decreased with depth, the nitrogen content below 30 cm may add a significant amount to the soil profile. For example, a *Ni* of 53 eq $ha^{-1}yr^{1}$ can be calculated using the nitrogen content of soil sampled to 90 cm in the Dry Mixedgrass subregion (Thomas, *et al*., 2017). To ensure adequate accounting of the nitrogen content in grassland soils, the *Ni* value of 53 eq $ha^{-1}yr^{1}$ was applied to the Dry Mixedgrass and Mixedgrass subregions.

The Central Parkland and Peace River Parkland subregions are largely agricultural areas; however, there are pockets of natural areas that are included in critical load determination. As there were no soil sample sites for these regions, *Ni* values were assigned to be equivalent to value(s) in adjacent subregions with comparable soil types. *Ni* for the Central Parkland subregion was set to be equivalent to the Foothills Parkland subregion (130 eq $ha^{-1}yr^{1}$) and Ni for the Peace River Parkland subregion to that of the Dry Mixedwood subregion (56 eq $ha^{-1}yr^{1}$). The Peace Athabasca Delta subregion, which consists of 40 % water bodies with additional areas of wetland, also did not have soil sample sites. Ni for this subregion was set to be equivalent to that of the Athabasca Plain subregion (18 eq $ha^{-1}yr^{1}$).

The average *Ni* by natural subregion ranged from 18 eq $ha^{-1}yr^{1}$ for the Athabasca Plain to 130 eq $ha^{-1}yr^{1}$ for the Foothills Parkland (Figure **7**). The upper range of *Ni* values was obtained for grasslands with Black Chernozem soil. The provincial average *Ni*, which includes both forested and grassland, was 48 eq $ha^{-1}yr^{1}$.

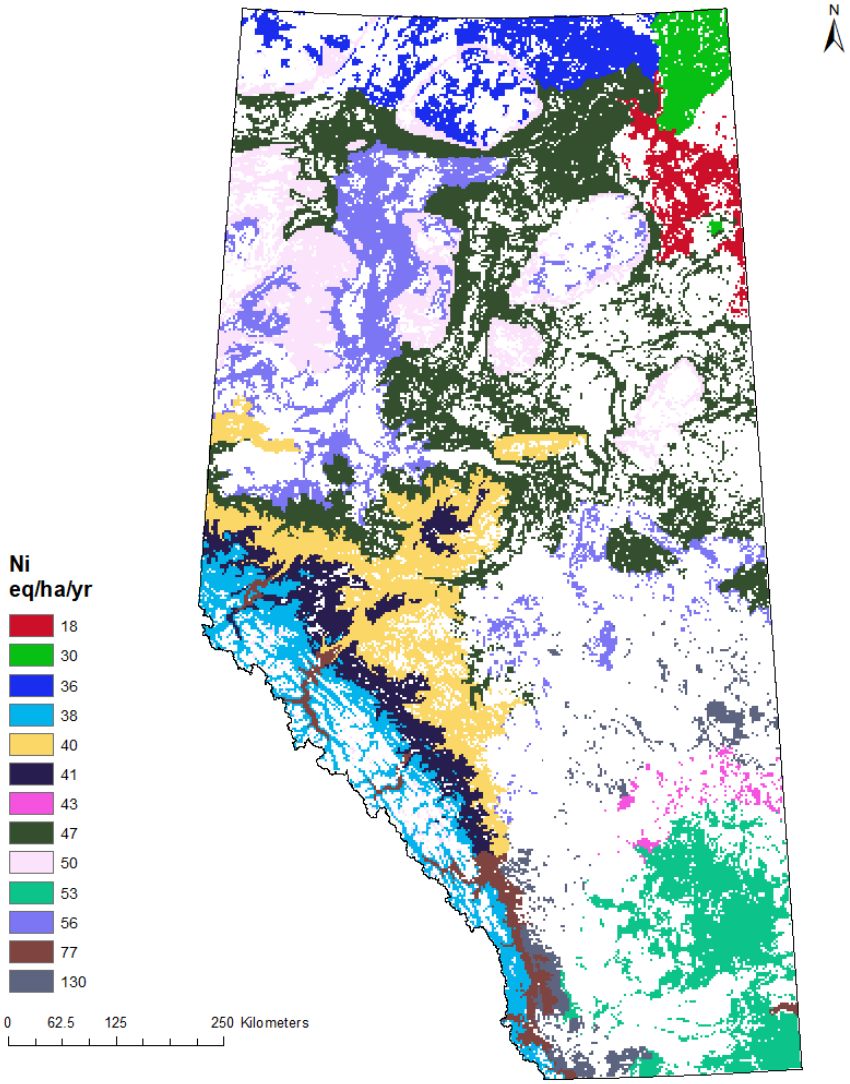

Figure 7. Alberta specific nitrogen immobilization (Ni). Areas coloured white have been excluded from calculations.

2.2.1.2 Nitrogen Removal due to Harvest and Grazing

Nitrogen uptake (N_u) was calculated for managed forests and native grassland. N_u for shrubland was assumed to be negligible.

An average nitrogen uptake due to forest harvesting is calculated for areas dominated by mineral soil in each natural subregion of Alberta. Subregions of Alberta are illustrated in Figure **E-1**.

To calculate the amount of nitrogen removal due to forest harvest in each natural subregion, the percentage of each forest management unit within each natural subregion was determined. The annual median harvest volume (Alberta Agriculture and Forestry, 2015) for each management unit was calculated using harvest information for the period of 2006 to 2015 (Figure **E-2**). If more than one forest management unit occurs within the natural subregion, nitrogen uptake was calculated for each forest management unit and the results combined for all management units. Nitrogen removed by harvest for each natural region was calculated using Eq 15.

$$N_{u,h} = \frac{\sum_{FMU=a}^{n} \left(H \times \frac{P_N}{100} \times \rho \times \frac{P\_FMU}{100} \right)_a}{Area_{NR}} \qquad \textbf{Eq 15}$$

Where:

$N_{u,h}$	=	the average removal of nitrogen (kg ha^{-1} yr^{-1}) from a natural subregion by harvest
H	=	the annual median harvest volume (m^3 yr^{-1}) (Figure **E-2**) for each forest management unit *a* within the natural subregion,
P_N	=	the percent nitrogen in the harvested biomass calculated in Eq 16
P	=	density of hardwood (390 kg m^{-3}) and coniferous (400 kg m^{-3}) stemwood (Gonzalez, 1990),
P_FMU	=	percentage of each forest management unit within the natural subregion
$Area_{NR}$	=	area (ha) of the natural subregion

The general practice in timber harvest is stem-only removal; where leaves/needles and branches are handled in various ways (e.g., left in scattered piles in the harvested area). Therefore, nitrogen removal was estimated to include only the mass in tree stems. Nitrogen content was estimated using:

- Relative proportion of stem and bark biomass. Data for mature lodgepole pine stands in Alberta (Monserud *et al*., 2006) was used for conifers and data for aspen poplar from British Columbia (Wang *et al*., 1995) was used for hardwoods.
- Nutrient content by tree component (Paré *et al*., 2013). Nitrogen content of aspen was used to represent hardwood forests and the average value of lodgepole and jack pine represented conifers.

Percent nitrogen in the harvested biomass was calculated using Eq 16, with data for stem bark and stem wood.

$$P_N = \left(\frac{Bark_{bm}}{Total} \times P_{Nb} \right) + \left(\frac{Wood_{bm}}{Total} \times P_{Nw} \right) \qquad \textbf{Eq 16}$$

Where:

$Bark_{bm}$ and $Wood_{bm}$	=	the relative proportion of stem bark and stemwood
$Total$	=	the sum of stem bark and stemwood biomass
P_{Nb} and P_{Nw}	=	the nitrogen concentration (%) for stemwood and bark

The calculated percent nitrogen content (P_N) for hardwood trees was 0.19% and 0.07% for coniferous trees.

Nutrient nitrogen can also be removed from grassland through grazing and subsequent removal of animals from the range. To account for this loss, N_{GZ} was estimated using Eq 17 for areas of native grassland within the Dry Mixedgrass, Mixedgrass, Foothills Fescue, Northern Fescue, and Fescue Parkland natural subregions. Areas of native grassland were identified using the GVI (Government of Alberta, 2011). The GVI classifies native grassland areas according to site type, which is determined by vegetation, soil, and hydrological regime. Ecologically sustainable stocking rates for each site type and a standard forage consumption rate per animal unit (AU) were obtained from Alberta Environment and Parks Range Plant Community Guides (Adams *et al*., 2003, 2013a, 2013b; Adams *et al.*, 2019; DeMaere *et al*., 2012). A standard nitrogen concentration in native vegetation obtained from Smoliak and Bezeau (1967) was used to estimate nitrogen ingestion rates. Loss of nitrogen from the

system due to grazing and subsequent removal of the animals was assumed to be equal to the amount retained in grazing animal live weight gain (Whitehead, 2000). N_{GZ} calculated by Eq 20 was then converted to equivalent charge basis.

$$N_{GZ} = \frac{F_C \times N_F \times N_R}{ESSR_{ST} \times 10000} \qquad \textbf{Eq 17}$$

Where:

N_{GZ} = Nitrogen removed by grazing (kg ha^{-1} yr^{-1})

F_C = Forage consumption (4100 kg AU^{-1} ha^{-1} yr^{-1})

N_F = Nitrogen content of forage (1.4 %)

N_R = Nitrogen retained in live weight gain (10% of nitrogen consumed)

$ESSR_{ST}$ = Ecologically sustainable stocking rate for each site type within a subregion (ha AU^{-1})

Figure 8 illustrates nitrogen removal due to harvest and grazing ($N_{u,h}$). $N_{u,h}$ ranged from < 1 eq ha^{-1} yr^{-1} in the Kazan Uplands and Central Parkland to >35 eq ha^{-}1 yr^{-1} in the Lower and Upper Foothills. The Kazan Uplands subregion with stunted Jack pine and Alaska birch and sparse understory (Downing & Pettapiece, 2006) has negligible harvesting of timber or grazing. The Central Parkland subregion is highly cultivated and does not contain extensive native grasslands or actively harvested forest management units (Figure **E-2**). The vegetation within Foothills natural regions includes hardwood and mixedwood forests (Downing & Pettapiece, 2006) and as illustrated in Figure **E-2** has among the highest annual median harvest volume.

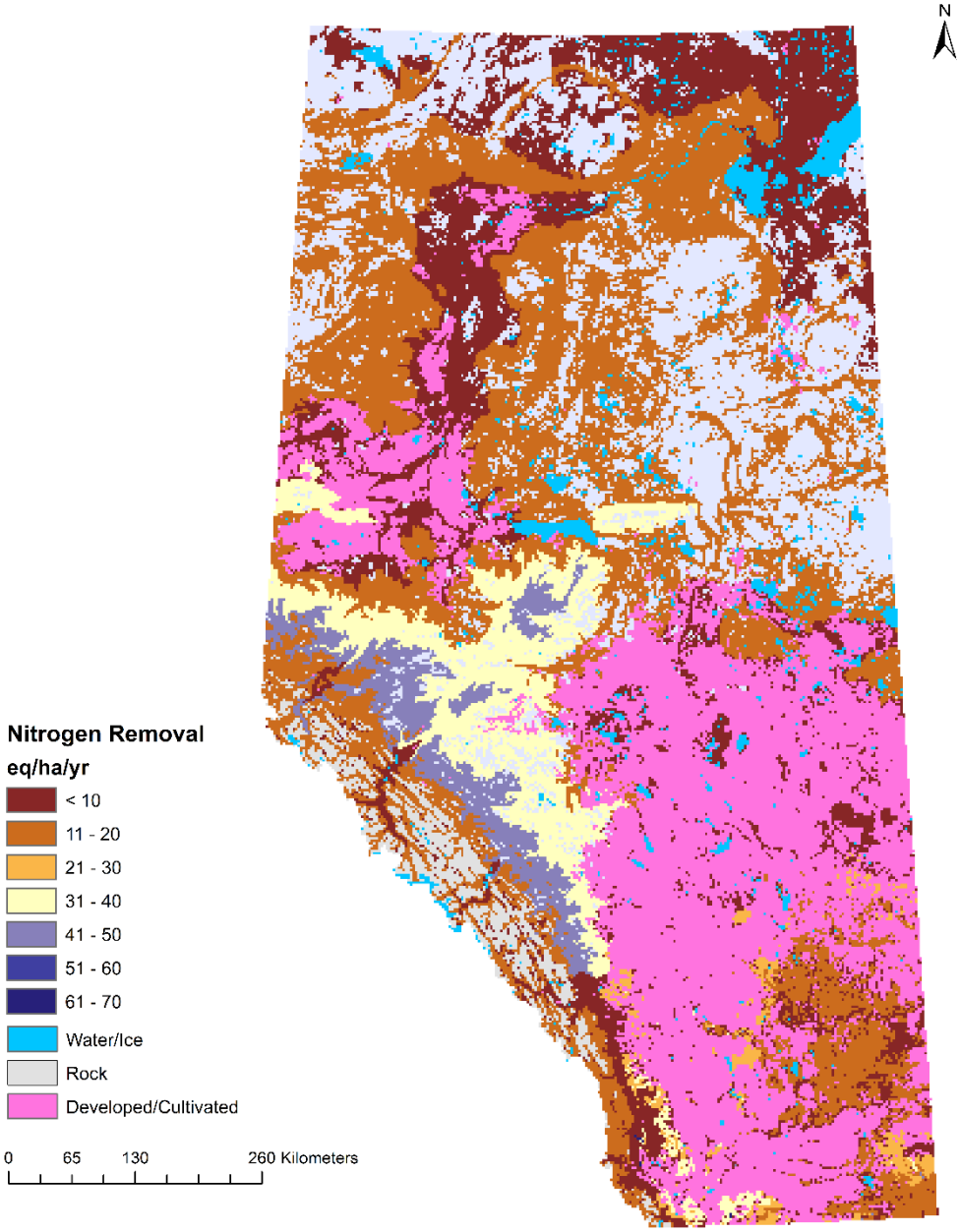

Figure 8. Nitrogen removal due to harvest and grazing. The white areas are areas not included in calculations and include water bodies, developed, agricultural land and areas of organic soil

2.2.1.3 Nitrogen Removal due to Wildfire

Wildfire greatly influences the life cycle of boreal forest ecosystems (Flannigan *et al*., 2006). To account for the influence of fire, nitrogen uptake from the ecosystem was adjusted by including nitrogen lost due to fire from forest biomass and soil organic horizons. All grassland natural subregions and shrubland subregions are excluded from these calculations. Only a small proportion of the grassland natural subregions experiences fire on an annual basis (Tymstra *et al.,* 2005) and nitrogen losses are low relative to N_i (Redmann ,1991). Shrubland is excluded from calculations as associated data needed to calculate nitrogen losses during fire are lacking. The method used does not account for fire severity and is a first order estimation of impact from wildfire.

Estimates of the nitrogen content of forest biomass and organic soil horizons, average area burnt, and proportion of nitrogen in the system that is lost to the atmosphere during a fire were used to determine fire loss of nitrogen from forest ecosystems.

Nitrogen content of trees within a natural subregion was determined using Eq 18.

$$N_{Tree} = \frac{P_{NT}}{100} \times \frac{AGB_{NR}}{Area_{NR}} \qquad \textbf{Eq 18}$$

Where:

N_{Tree} = nitrogen content of trees (kg ha^{-1})

P_{NT} = nitrogen content of tree (%)

ABG_{NR} = aboveground biomass (kg) in natural subregion

$Area_{NR}$ = area (ha) of natural subregion

P_{NT} was determined using the distribution of coniferous and hardwood trees (ABMI, 2010) and nitrogen content of trees. Nitrogen content of hardwood and coniferous trees were estimated using a similar method to nitrogen removal by harvest (Section 2.2.1.2); however, for wildfire removal all of the tree components (foliage, branch and stem) was included. Nitrogen content of 0.15% was determined for coniferous forests and 0.23% for hardwood forests (Paré *et al*., 2013). The average of these two values (0.19%) was used to represent mixed woods. Aboveground biomass in each natural subregion was obtained from Zhang *et al*. (2014). Average nitrogen content of trees (N kg ha^{-1}) in each natural subregion is illustrated in Figure **J-1**.

Total nitrogen content (kg ha^{-1}, N_{cont}) for each subregion is the sum of nitrogen content of soil organic horizons ($N_{soil,m}$) and tree biomass(N_{Tree}) (Eq 19).

$$N_{cont} = N_{Tree} + N_{soil,m} \qquad \textbf{Eq 19}$$

Nitrogen content of organic soil horizons ($N_{soil,m}$) was calculated using the method outlined in 0 and is an average for each natural subregion.

No soil data were available for the Boreal Subarctic and Peace-Athabasca Delta subregions. Sixty percent or greater of the Peace-Athabasca Delta and Boreal Subarctic subregions are wetlands and water bodies and these subregions were excluded from wildfire impact (Downing & Pettapiece, 2006). The Central Parkland and Peace River Parkland subregions are largely agricultural areas and thus were excluded from these calculations.

Total nitrogen content in forest tree biomass and soil organic horizons by subregion is illustrated in Figure J-2. The observed nitrogen content was predominantly due to N content of the soil organic horizon, forming 82-99% of the nitrogen content. The highest nitrogen content values were observed for the Northern Mixedwood subregion, an area largely covered by wetlands, with organic rich soils where black spruce is a dominant tree species (Downing & Pettapiece, 2006) and the Alpine subregion, a region identified as least likely to experience wildfire (Figure J-3). It should be noted that the Northern Mixedwood subregion value is based on limited data points (n=2), additional data for this subregion is needed to better characterize soil nitrogen content in this region.

Nitrogen lost from the ecosystem due to wildfire was determined using the percent of subregion burnt annually, nitrogen content of biomass and organic soil horizon within a subregion, and percent of nitrogen from biomass and organic soil horizon lost to the atmosphere due to wildfire. The proportion of each natural subregion burnt per year was estimated based on the area burnt within each subregion for the period between 1961 and 2002 (Tymstra *et al.*, 2005) and is illustrated in Figure J-3. Nitrogen combustion loss (P_N_{loss}) from vegetation and organic soil horizons was estimated to be 33% for broadleaf and mixed wood stands and 40% for coniferous stands (Harden *et al*., 2002). Nitrogen lost from the ecosystem due to wildfire was calculated using Eq 20 and converted to equivalent charge basis.

$$N_{fire} = P\_Area_{burn} \times N_{cont} \times P\_N_{loss}$$ **Eq 20**

Where:

N_{fire} = nitrogen released from soil and vegetation by wildfire (kg ha^{-1} yr^{-1})

P_Area_{burn} = proportion of natural subregion burned annually (ha ha^{-1} yr^{-1})

N_{cont} = nitrogen content of trees and soil organic horizons (kg ha^{-1}), calculated in Eq 19

P_N_{loss} = proportion of nitrogen contained in soil-plant system lost during wildfire (kg kg^{-1})

The distribution N_{fire} is illustrated in Figure **9**. Average annual nitrogen loss due to wildfire ranged from negligible for unmanaged areas with minimal wildfire impact to 378 eq ha^{-1} yr^{-1} for some areas of northern Alberta.

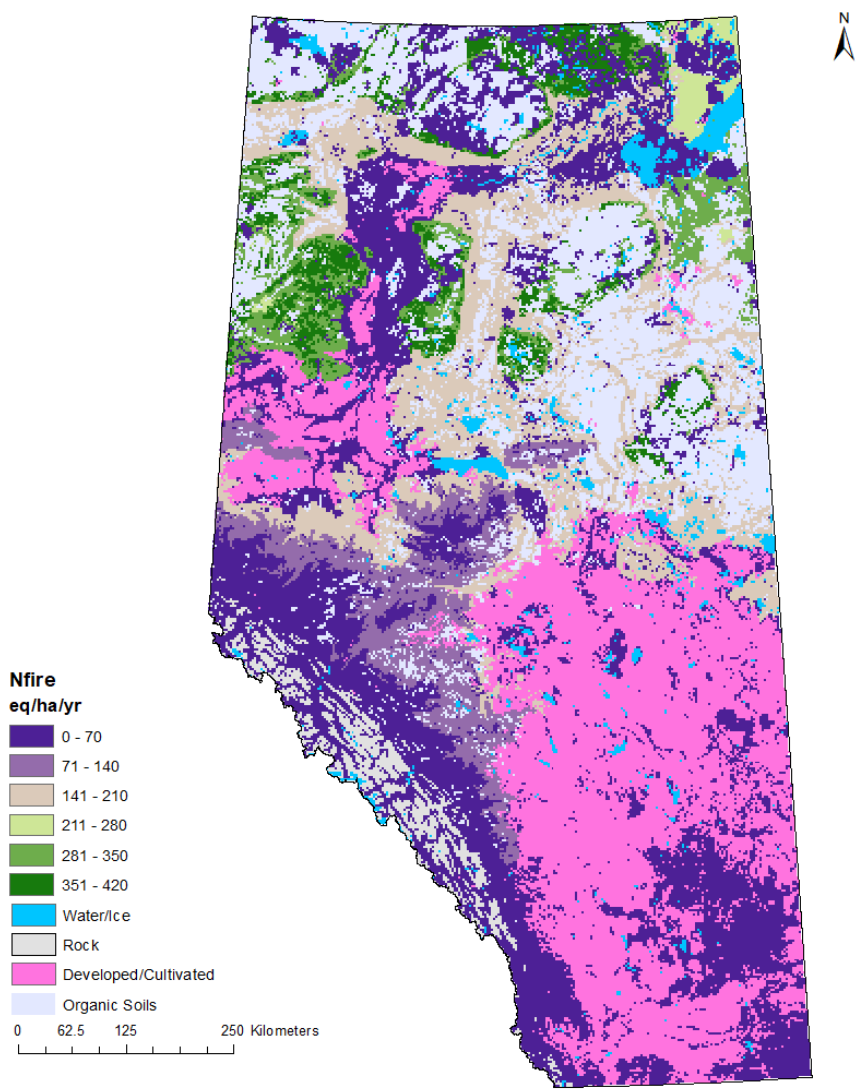

Figure 9. Nitrogen removal due to wildfire

2.2.1.4 Nitrogen Removal due to Harvest, Grazing and Wildfire

Combined nitrogen removal due to harvest, grazing and wildfire is calculated as illustrated in Eq 21. The derivation of the terms in Eq 21 is discussed in Sections 2.2.1.2 and 2.2.1.3.

$$N_{u,comb} = \left\{ \begin{matrix} N_{u,h} + N_{fire} & if forest \\ N_{GZ} & if\ grassland \end{matrix} \right\} \qquad \textbf{Eq 21}$$

The resulting $N_{u,comb}$ is illustrated in Figure 10. Lower $N_{u,comb}$ were determined for areas of unmanaged forests where timber harvesting is low, nitrogen removal is limited to grazing and/or the incidence of wildfire is low. Higher $N_{u,comb}$ were observed for northern Alberta, where the incidence of wildfire is high and nitrogen in soil organic horizons is susceptible to loss by fire. The relative contribution of $N_{u,h}$ and N_{fire} to $N_{u,comb}$ varied across the province. $N_{u,h}$ dominated in the Kazan Uplands where nitrogen removal due to wildfire was thought to be negligible; however, N_{fire} dominated most parts of northern Alberta. $N_{u,h}$ contribution to $N_{u,comb}$ ranged from 20-40% in the Lower and Upper Foothills subregions and from 60-80% in the Subalpine subregion. $N_{u,h}$ contribution is the highest in grassland and shrubland areas where removal due to wildfire was assumed to be small.

2.2.1.5 Minimum Critical Load for Nitrogen – Harvest Regime

Figure 11 illustrates the distribution of minimum CL for nitrogen calculated under the harvest regime. For the most part $CL_{min}(N)_h$ was less than 100 eq $ha^{-1}yr^{-1}$, albeit there are small areas with $CL_{min}(N)_h$ up to 175 eq $ha^{-1}yr^{-1}$ largely distributed in the grassland subregions. Areas with $CL_{min}(N)_h$ greater than 70 eq $ha^{-1}yr^{-1}$ contain managed forests in the foothills and grassland which have nitrogen removal rates associated with biomass harvest or grazing. Some areas are not substantially impacted by wildfire; for this reason, minimum critical load for nitrogen-harvest and fire regime (Section 2.2.1.6) is equivalent to that of the harvest only regime.

2.2.1.6 Minimum Critical Load for Nitrogen – Harvest and Fire Regime

Figure 12 illustrates the distribution of $CL_{min}(N)_{hf}$. Including nitrogen loss from the ecosystem due to wildfire notably increased $CL_{min}(N)_{hf}$, especially for northern Alberta. This area of the province had higher soil nitrogen and experiences more frequent fires. Large areas of northern Alberta had $CL_{min}(N)_{hf}$ higher than 200 eq $ha^{-1}yr^{-1}$. The lowest $CL_{min}(N)_{hf}$ are for areas of shrubland (north) and National Parks that experience minimal wildfires.

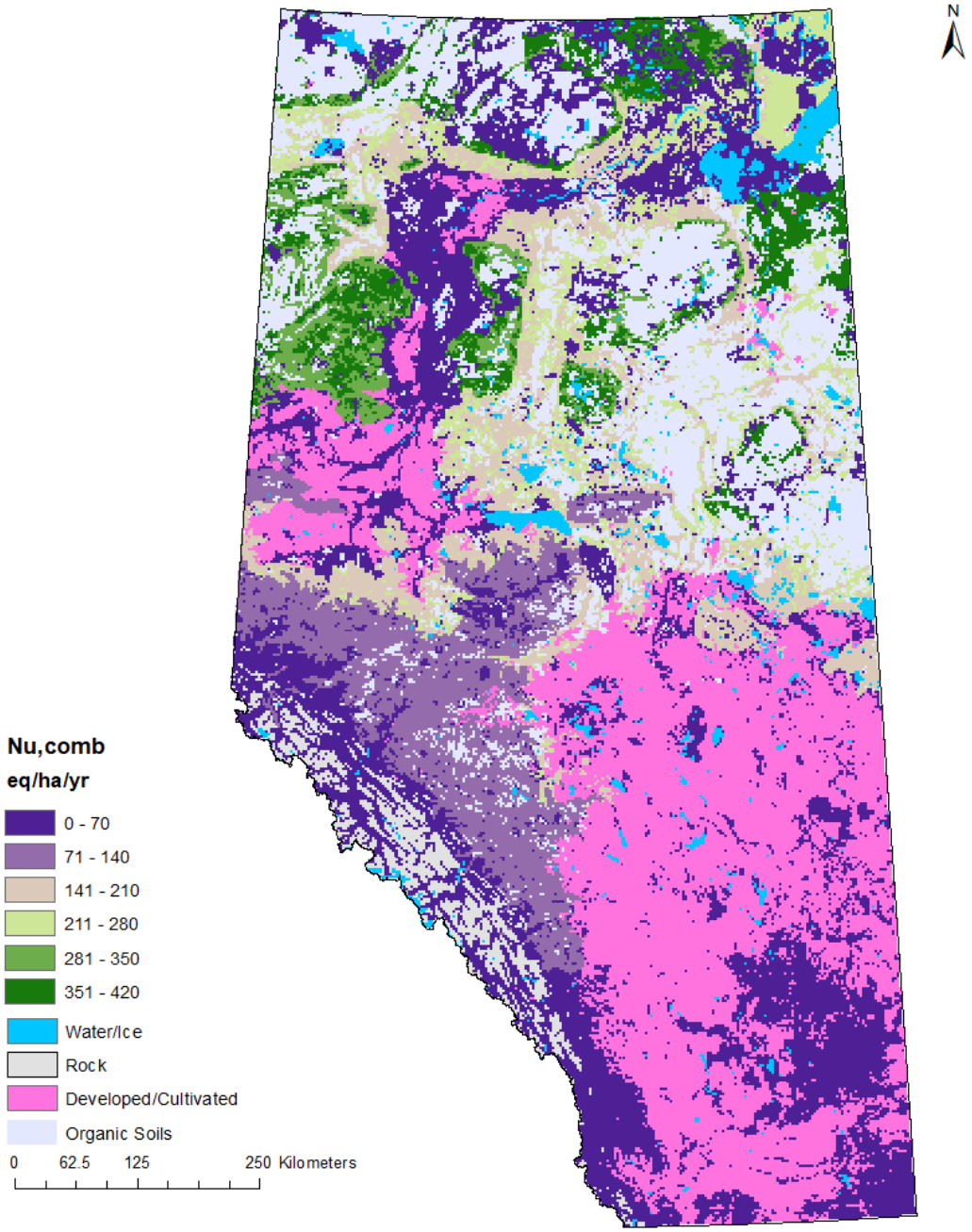

Figure 10. Average annual nitrogen uptake and loss due to wildfire, harvest and grazing. White areas include developed and agricultural land, rock, water bodies and areas of organic soil.

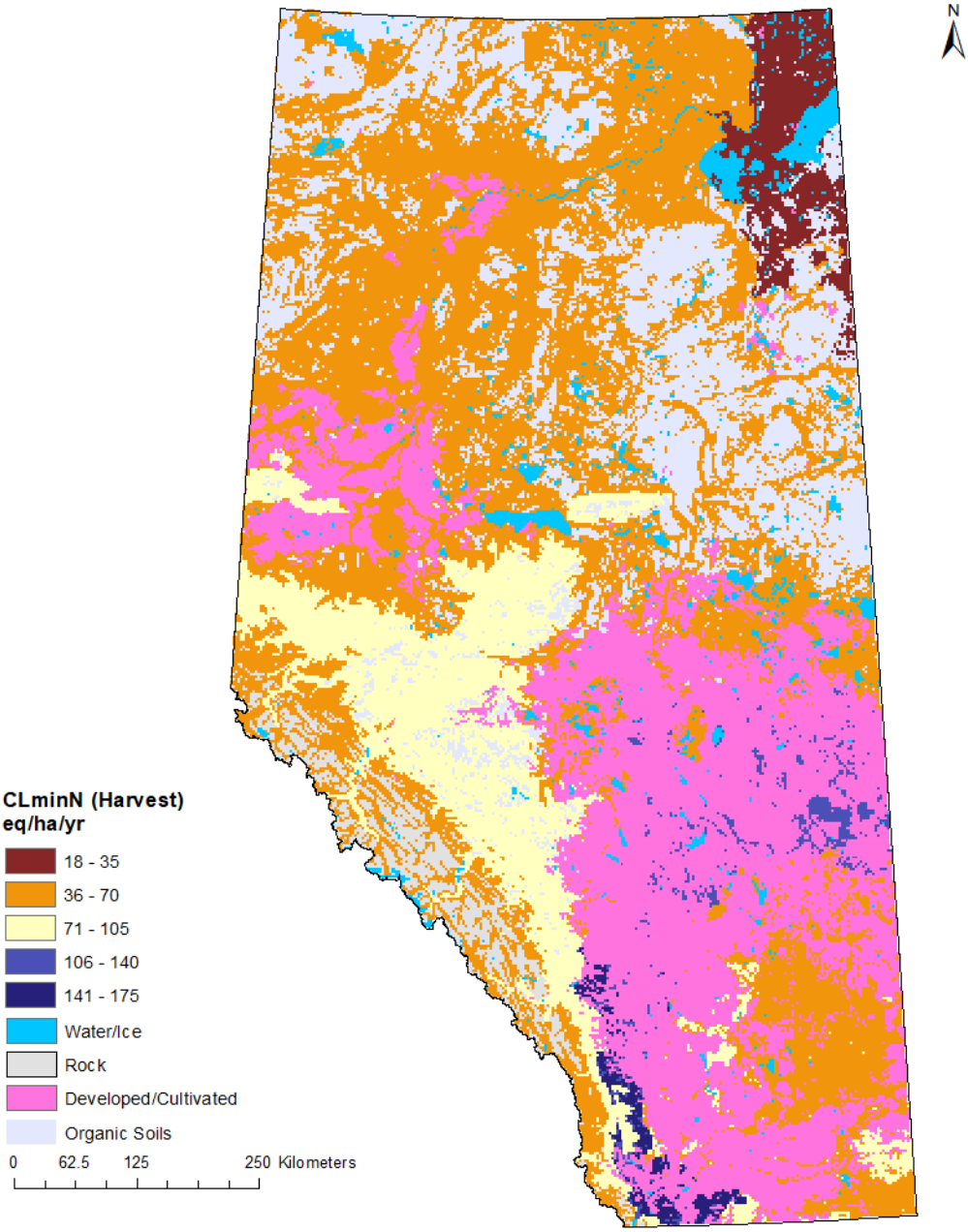

Figure 11. Minimum Critical Load for Nitrogen (Harvest Regime). Critical load maps were developed with cell size of 2.5 km by 2.5 km. Nitrogen removal due to biomass harvest only.

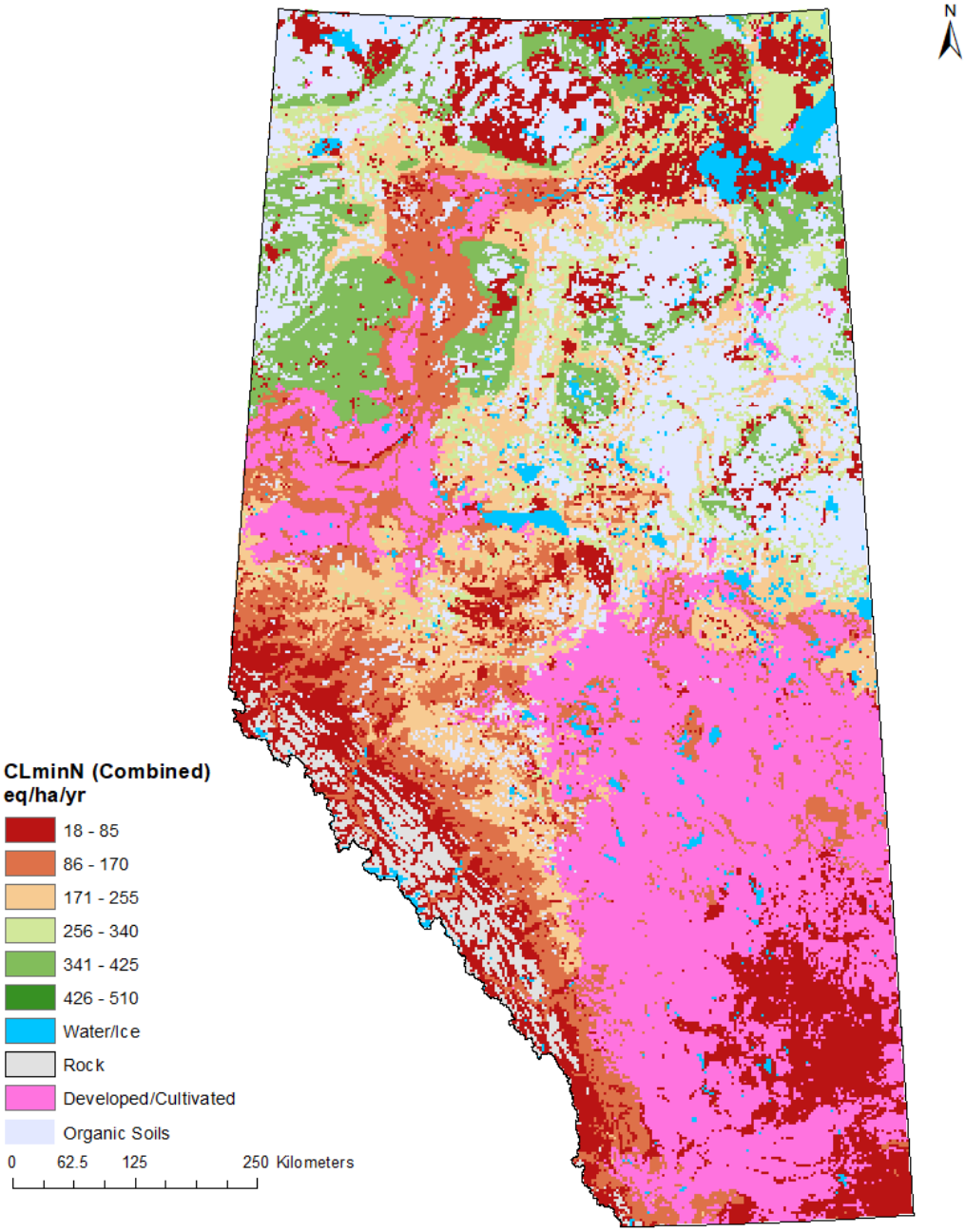

Figure 12. Minimum Critical Load for Nitrogen (Harvest and Wildfire Regime). Critical load maps were developed with cell size of 2.5 km by 2.5 km. Nitrogen removal due to biomass harvest, grazing and wildfires.

2.2.2 Maximum Critical Load for Nitrogen

As illustrated in Eq 11, $CL_{max}(N)$ is related to $CL_{min}(N)$ and $CL_{max}(S)$; however, with the exception of small pockets, $CL_{min}(N)$ is notably lower than $CL_{max}(S)$. Thus $CL_{max}(S)$ has a greater influence on the resulting $CL_{max}(N)$. Figure 13 illustrates $CL_{max}(N)$ calculated under the Harvest and Wildfire regime. The highest $CL_{max}(N)_{hf}$ values (>3000 eq ha^{-1} yr^{-1}) were observed for western regions of the province. These regions also had the highest $CL_{max}(S)$ values. The lowest $CL_{max}(N)_{hf}$ values (≤500 eq ha^{-1} yr^{-1}) were determined for some areas in northern Alberta. The ratio between the Harvest regime and values for the harvest and wildfire regime are shown in Figure 13B. For a large part of the province the ratio was greater than 0.8 indicating minimal influence of wildfire regime on $CL_{max}(N)$. A ratio less than 0.6 was observed for small areas in the northeast part of the province. These areas had low $CL_{max}(S)$ values that were comparable or lower than $CL_{min}(N)_{hf}$.

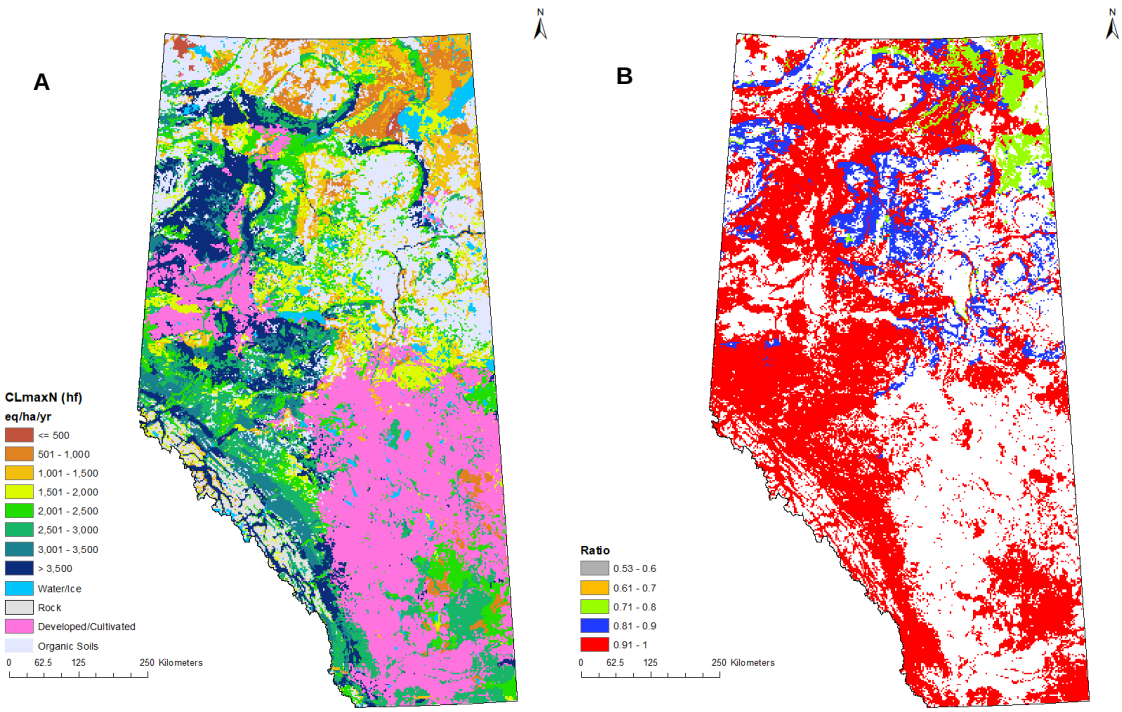

Figure 13. (A) Maximum Critical Load for Nitrogen for Harvest and Wildfire regime. (B) Ratio of CLmax(N) Harvest to Harvest and Wildfire regimes. Critical load maps were developed with cell size of 2.5 km by 2.5 km.

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Appendices

Appendix A Identifying Organic and Mineral soils

Organic soils are identified using the distribution and prevalence of bogs and fens within a 2.5 km x 2.5 km area. Any 2.5 x 2.5 km grid with the combined fen and bog fraction of 0.5 or greater is designated as organic. Bog and fen area coverage is identified using the Alberta Merged Wetland Inventory (Alberta Environment and Parks, 2018) and the Canadian Wetland Inventory (Amani *et al*., 2019). The Canadian wetland inventory is used for areas within national parks where there are no data in the Alberta Merged Wetland Inventory. The fraction of organic soil (F_{org}) within each grid is calculated using Eq A-1.

$$\boldsymbol{F_{org}} = \frac{\boldsymbol{Area_{fen}} + \boldsymbol{Area_{bog}}}{\boldsymbol{Area_T}} \qquad \text{Eq A-1}$$

Where:

$Area_T$ = the total area within a grid cell

$Area_{fen}$ and $Area_{bog}$ = the total bog and fen area within the same grid

The distribution of soils identified as organic or mineral is illustrated in Figure A-1.

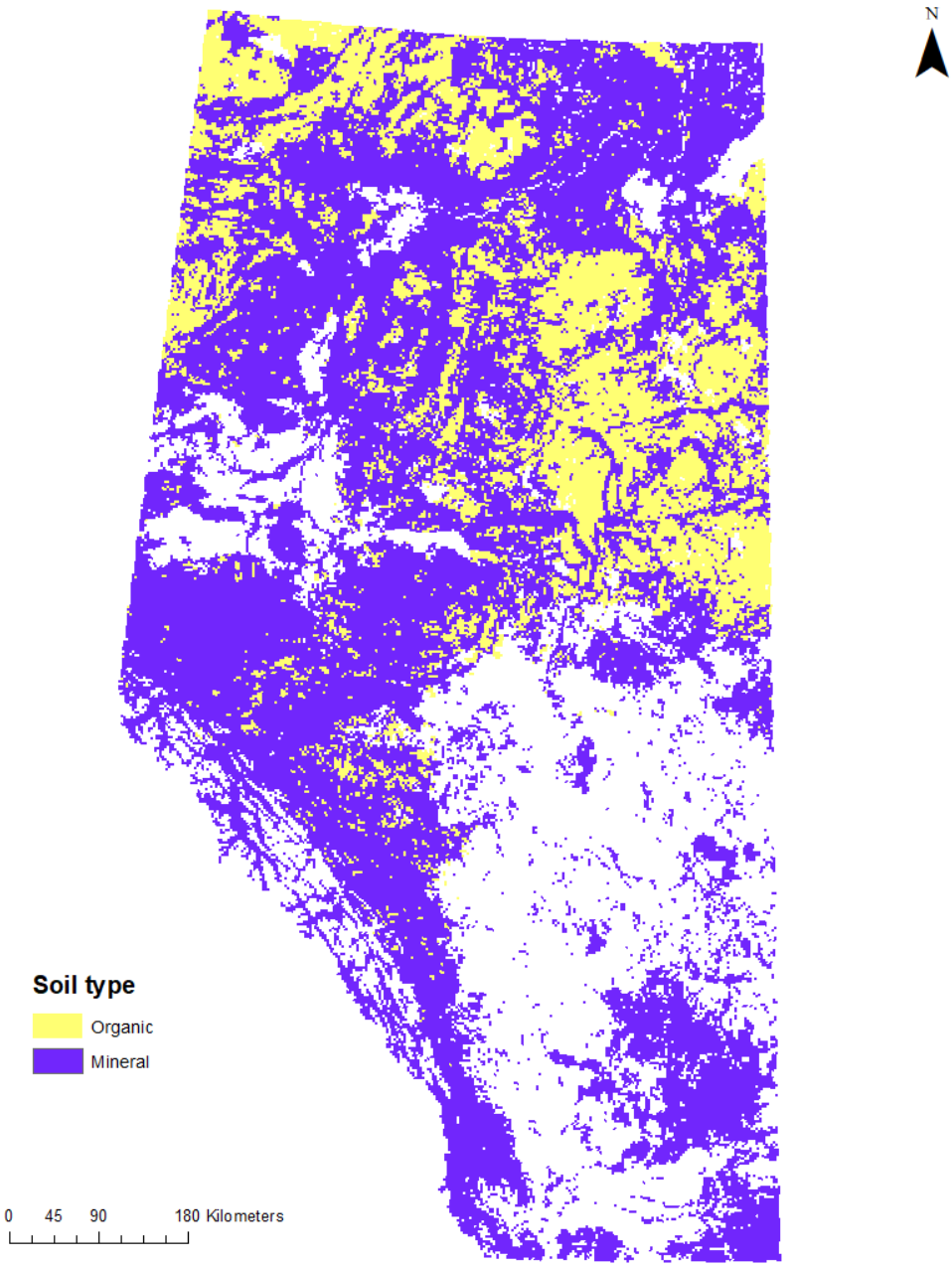

Figure A-1. Distribution of organic and mineral soils in Alberta as identified using Eq A-1. 25% of the soil in natural areas was determined to have organic soil

Appendix B Estimating Weathering Rate Class

Each area of the map (polygons in GIS) is described by soil landscape components (attributes), including soil series, using data from Soil Landscape of Canada (SLC, version 3.2, released March 2011). SLC provides digital (mapped) land and soil characteristics for Canada. The data are based on 1:1,000,000 scale soil survey maps. The map is divided into polygons and a standard set of attributes are used to describe these areas on the map. Polygon sizes reflect the heterogeneous nature of the soil landscape and range from 21 - 8500 km^2. A single polygon may contain one or more distinct soil series; however the location of these attributes within a polygon are not identified. As a result, weighted average values are used to determine dominant characterization of each polygon. The SLC database contains information for Canada; a subset of these data pertaining to Alberta was extracted for this work.

Weathering rate classes for each polygon are determined as follows. Weathering rate classes are assigned to mineral soil series on the basis of their clay and sand content and pH. The SLC has horizon information for each soil series. Each soil series has a number of horizons (i) with information on soil properties ($Property_i$) such as percent clay, percent sand, and pH. The data for each horizon were combined to provide a weighted average for each soil series ($Property_{ss}$) as follows:

$$Property_{ss} = \sum_{i=1}^{i=n} \left(Property_i \times \frac{thickness_i}{depth_{root}} \right) \qquad \textbf{Eq B-1}$$

The weighted average clay and sand contents determined using Eq B-1 are used to determine texture class (Table V.14 CLRTAP 2017) for each soil series as follows:

$$T_{ss} = \begin{cases} 1 & if\ Clay_{ss} < 18\%\ and\ Sand_{ss} \geq 65\% \\ 2 & \begin{matrix} if\ Clay_{ss} < 35\%\ and\ 15\% \leq Sand_{ss} < 65\% \\ or \\ if\ 18\% \leq Clay_{ss} \leq 35\%\ and\ Sand_{ss} \geq 65\% \end{matrix} \\ 3 & if\ Clay_{ss} < 35\%\ and\ Sand_{ss} < 15\% \\ 4 & if\ 35\% \geq Clay_{ss} < 60\% \\ 5 & \text{if}\ Clay_{ss} \geq 60\% \end{cases} \qquad \textbf{Eq B-2}$$

Soil series texture class T_{ss} from Eq B-2 and soil pH are used in Table **B-2** to determine soil series weathering class $(Wclass)_{ss}$.

Table B-1. Weathering class (*Wclass)* as a function of soil texture and pH

Soil pH	**Texture class (from** Eq B-2**)**				
	1	**2**	**3**	**4**	**5**
<5.5	1	3	3	6	6
5.5-6.5	2	4	4	6	6
>6.5	2	5	5	6	6

adopted from CLTRAP 2017

$Wclass_{poly}$ for each polygon is derived by taking an area-weighted average of $(Wclass)_{ss}$ using the percent of the polygon occupied by each soil series (ext_{ss}), as follows:

$$Wclass_{poly} = \sum_{ss=1}^{m} (Wclass)_{ss} \times \frac{ext_{ss}}{100} \qquad \textbf{Eq B-3}$$

This is repeated for all mapped polygons (1028 in total) for Alberta. The spatially allocated $Wclass_{poly}$ is used in Section 2.1.2.1 to determine base cation weathering.

Table **B-2** lists and describes the variables used in equation Eq B-1 to Eq B-3.

Table B-2. Variables used to determine weathering class (Eq B-3)

Variable	Description	Source/Derivation
$Property_{ss}$	the horizon thickness weighted average soil series property (percent clay, percent sand and pH converted to $[H^+]$) for each soil series	Soil Landscapes of Canada Eq B-1
$property_i$	the ith horizon property (clay, sand, or pH converted to $[H^+]$)	Soil Landscapes of Canada
$Thickness_i$	the thickness (m) of the ith mineral horizon	Soil Landscapes of Canada
n	the number of horizons within a soil series to a maximum depth of 0.5 m	Soil Landscapes of Canada
$depth_{root}$	set to 0.5 m approximate rooting zone of forest vegetation The forest floor layer (LFH) is not included in the 0.5 m rooting depth	Watmough & Dillon, 2002
m	the number of soil series within a polygon	Soil Landscapes of Canada
$Clay_{ss}$	Percent clay content (wt%) of the ss^{th} soils series	Soil Landscapes of Canada
$Sand_{ss}$	Percent sand content (wt%) of the ss^{th} soils series	Soil Landscapes of Canada
T_{ss}	Soil texture class for the ss^{th} soils series	Eq B-2
$(W_{class})_{ss}$	Weathering rate class for soil series ss	Eq B-3
ext_{ss}	the proportional (%) extent of the ss^{th} soil series	Soil Landscapes of Canada

Appendix C Annual Runoff

Runoff estimates for the province were obtained from Annual Unit Runoff in Canada (Agriculture and Agri-food Canada, 2013). Annual unit runoff is a measure of runoff volume in cubic decameters per square kilometer (dam^3/ km^2). This provides a general overview of runoff patterns. The annual unit runoff is calculated using historical data (at least 24 years) from hydrometric gauging stations. The database provides runoff values as probability of exceedance given as iso-lines. This means that if an annual runoff is estimated at 20 dam^3/km^2 at 50% probability, for any given year, there is 50% probability that runoff at that location will exceed 20 dam^3/km^2. For this work, 50% probability of exceedance was used. It is expected to provide a representation of runoff in a “typical” year.

The distribution of annual runoff for Alberta is illustrated in Figure **C-1**. The highest runoff rates were observed for the Rocky Mountains, associated foothills and the Swan Hills area. These areas also have higher precipitation rates. Central-west and southeast Alberta receive among the lowest annual precipitation in the province and have low runoff rates (<500 m^3 ha^{-1}). The Peace-Athabasca delta area (northwest Alberta) in the Canadian Shield region has relatively lower precipitation but moderate runoff rates (500-1700 m^3 ha^{-1}).

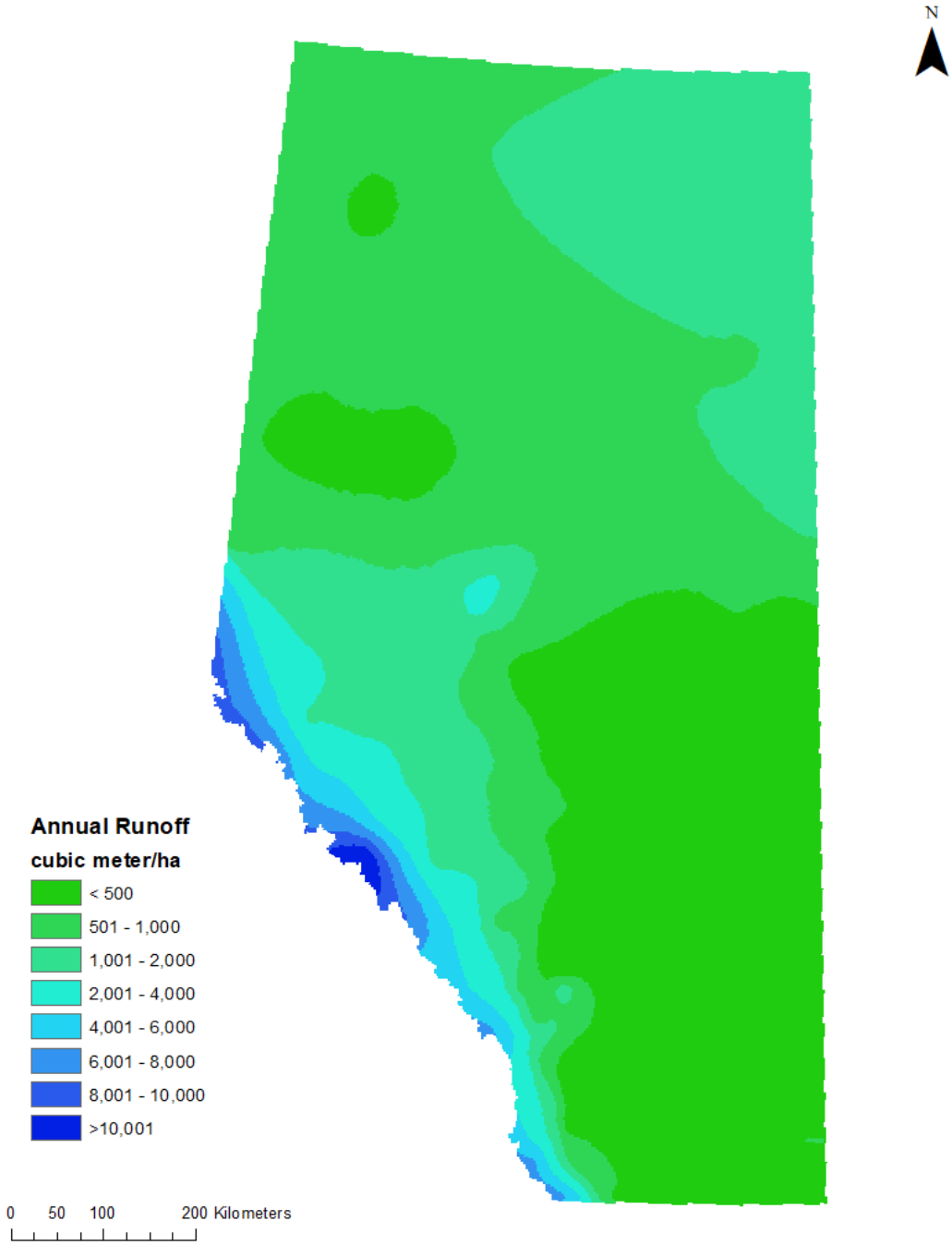

Figure C-1. Annual runoff (m^3 ha^{-1}) determined from 50% probability values for Alberta

Appendix D Long Term Average Soil Temperature

Long term annual average soil temperature was calculated using simulated soil temperature for the years 1953-2002 at 50 cm depth (Zhang *et al*., 2005). These temperature data were provided at 0.5 degree resolution and the average simulated soil temperature is illustrated in Figure **D-1**. This work has used long term temperature data and did not consider the impact of climate change.

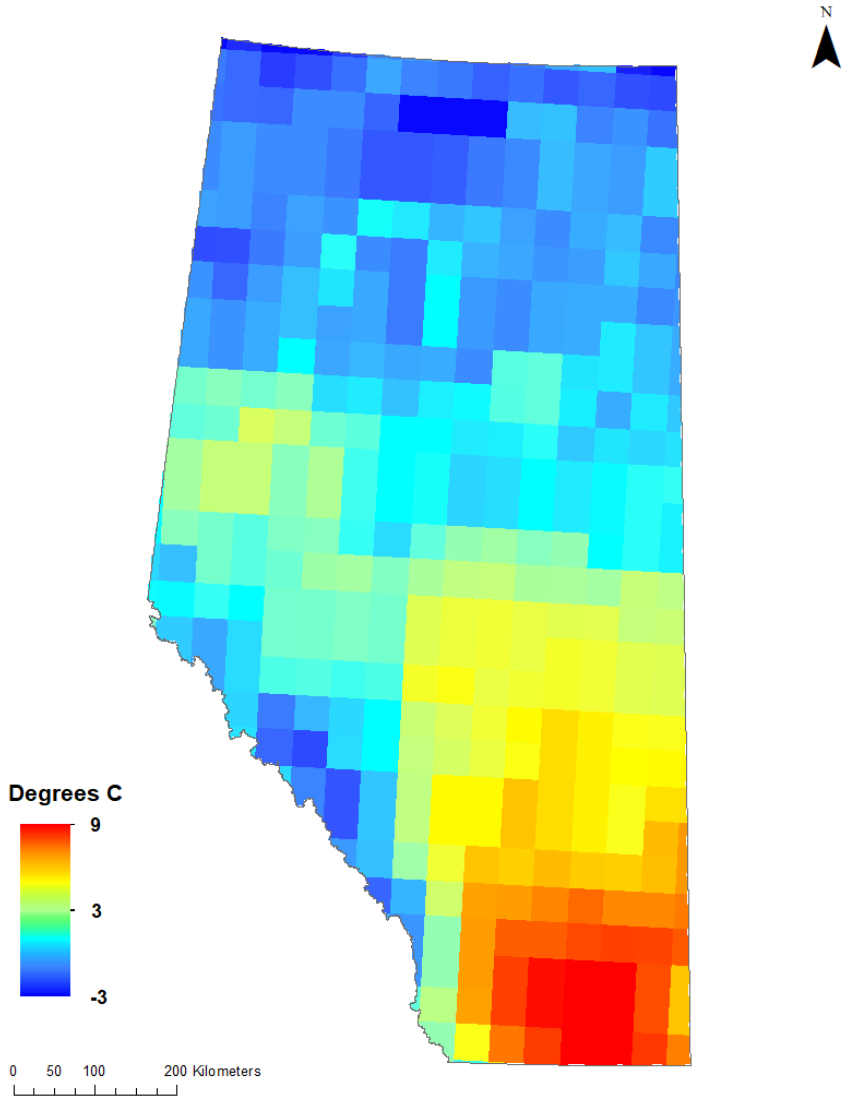

Figure D-1. Long term average soil temperature

Appendix E Natural Subregions and Annual Forest Harvest in Alberta

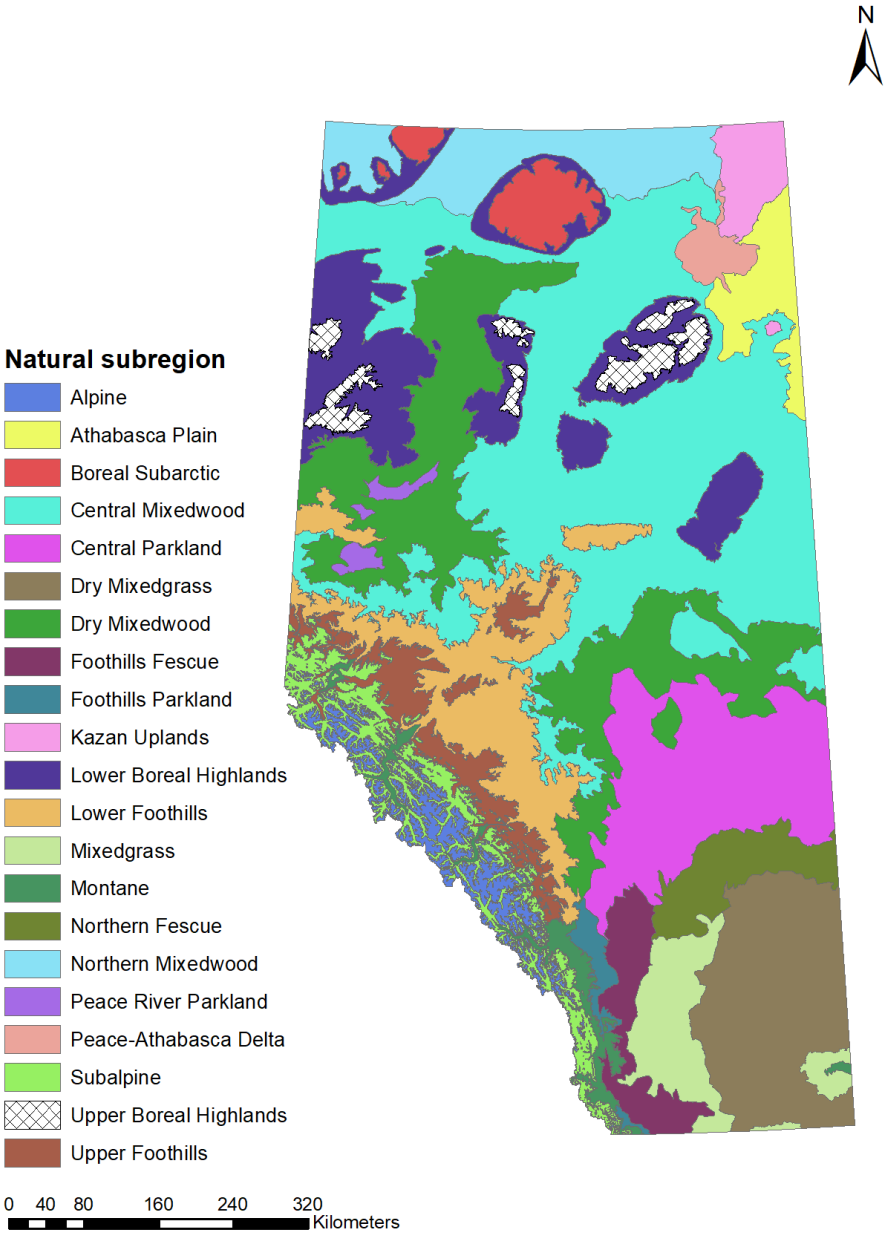

Figure E-1. Natural subregions of Alberta

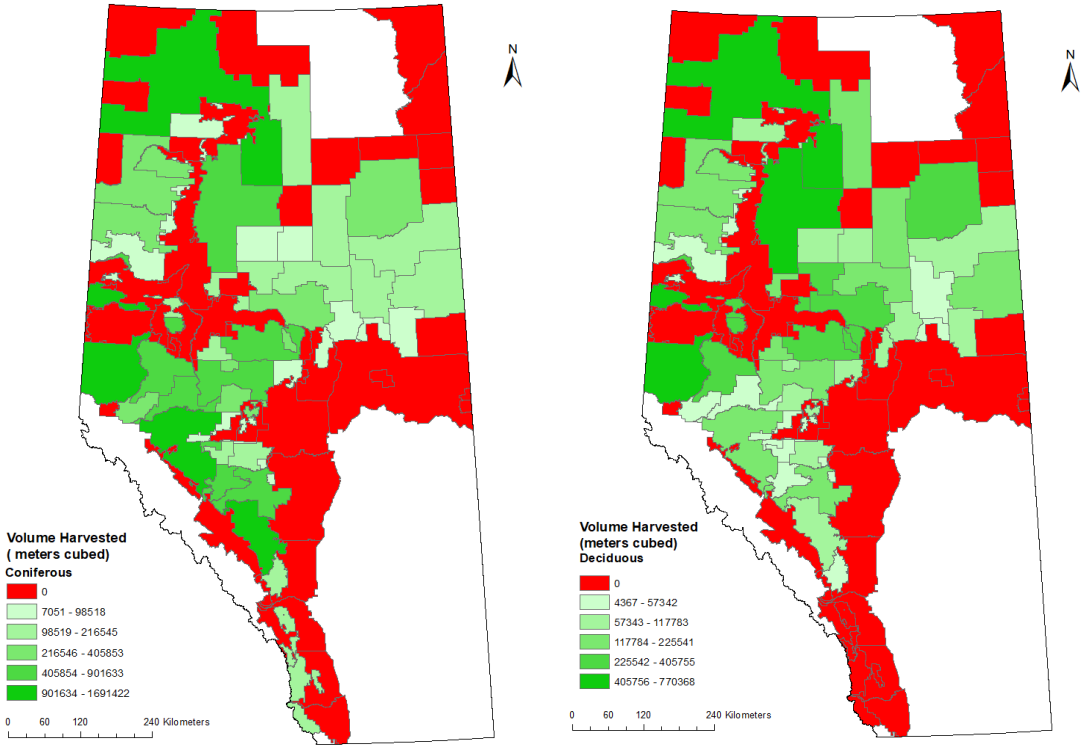

Figure E-2. Coniferous and hardwood annual median harvest (m^3)

Appendix F Classifying Fen systems

As indicated in Section 2.1.3.2, areas of bog and fen are identified using the Alberta Merged Wetland Inventory (Alberta Environment and Parks, 2018) and the Canadian Wetland Inventory *(Amani et al*., 2019). Neither inventory classifies fens into different types, as required for estimating critical loads for peatland systems. Therefore, fen types (extreme rich, moderate rich and poor) were spatially allocated using professional judgement (Larry Turchenek, personal communication, January, 2019) based on the Physiographic Regions of Alberta (Land Resources Research Centre, 1986). Physiographic classification includes factors such as climate, surface geology, and elevation that are important determinants of wetland type.

Table F-1. Fen classification by Physiographic Regions of Alberta

Region (colors indicated are shown in Figure **F-1**)	**Assignment**
Northern Alberta uplands (light green) and groups J1-J4 of Southern Alberta uplands (Dark green)	Poor fens
Southern Alberta uplands groups J5, J6, J7, J8, J9, J10 and J11 (Dark Green)	Extreme rich fens
Kazan upland and Athabasca plains groups A and B (Pink)	Poor fens
Groups D2 (Yellow)	Poor fens
C2.1 (Athabasca Delta)	Extreme rich fens
All other regions	Moderate rich fens

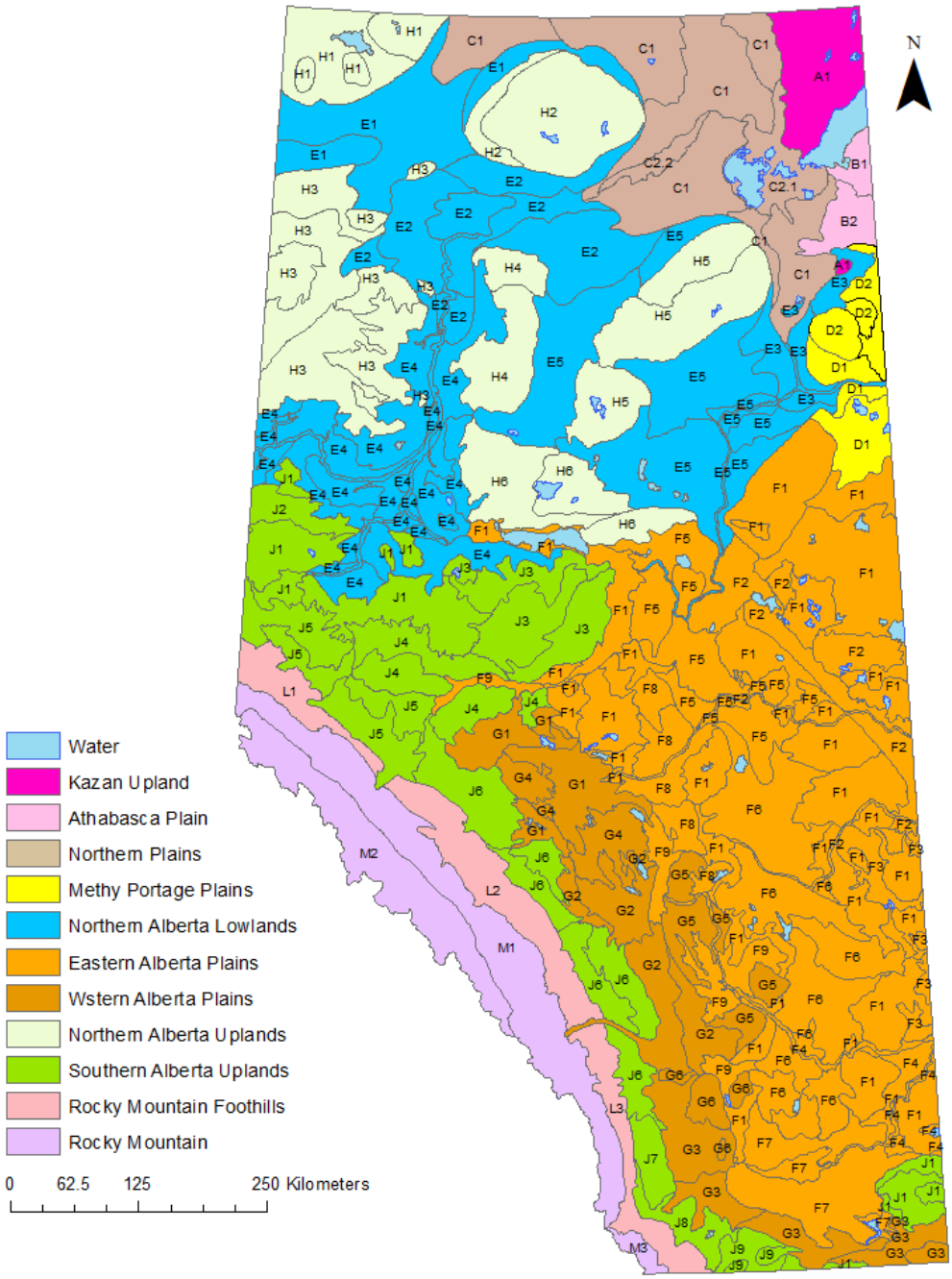

Figure F-1. Physiographic subdivisions of Alberta adopted from (Land Resources Research Centre, 1986)

Appendix G CLmax(S) Mineral and Organic Soils

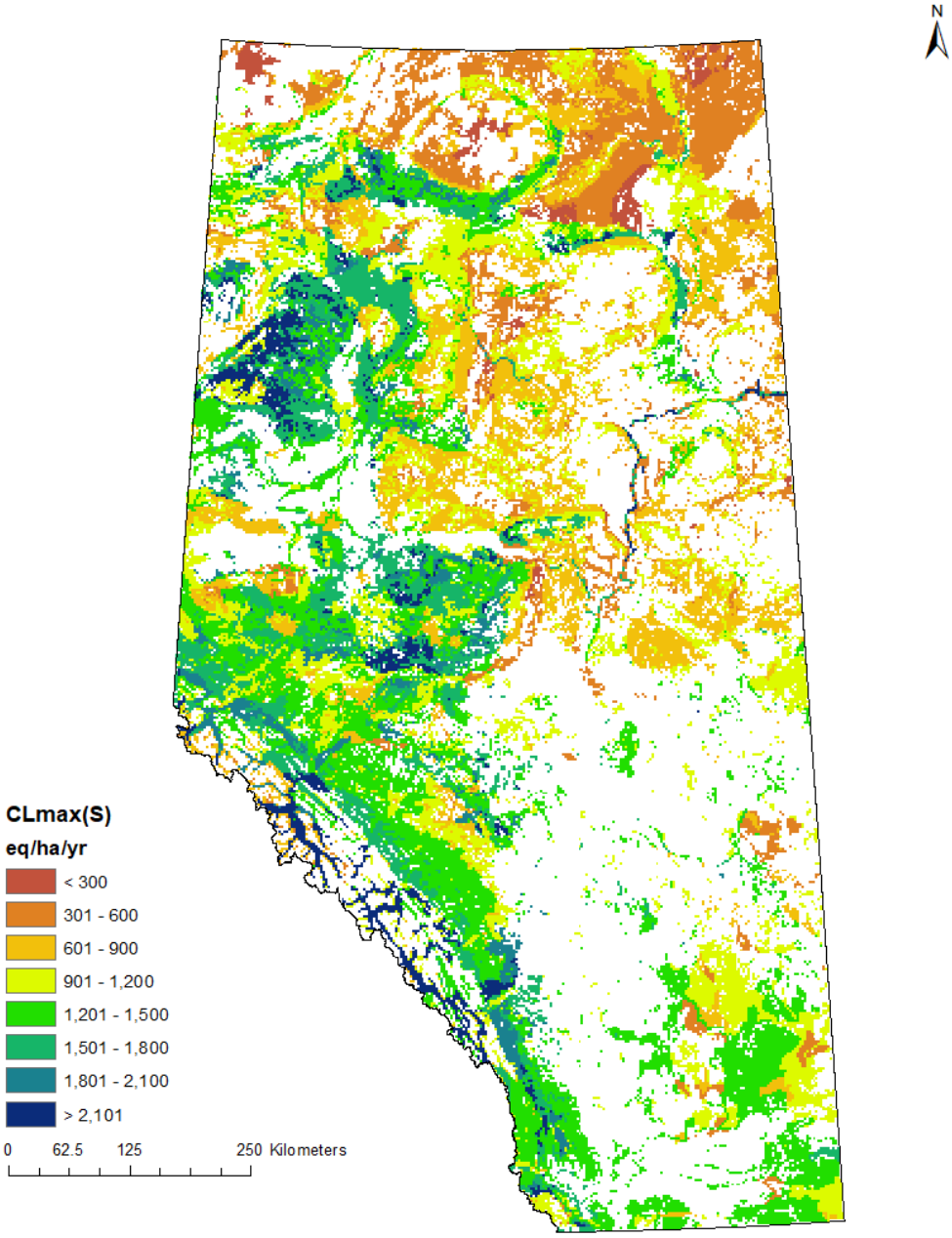

Figure G-1. *$CL_{max}(S)$* for mineral soils

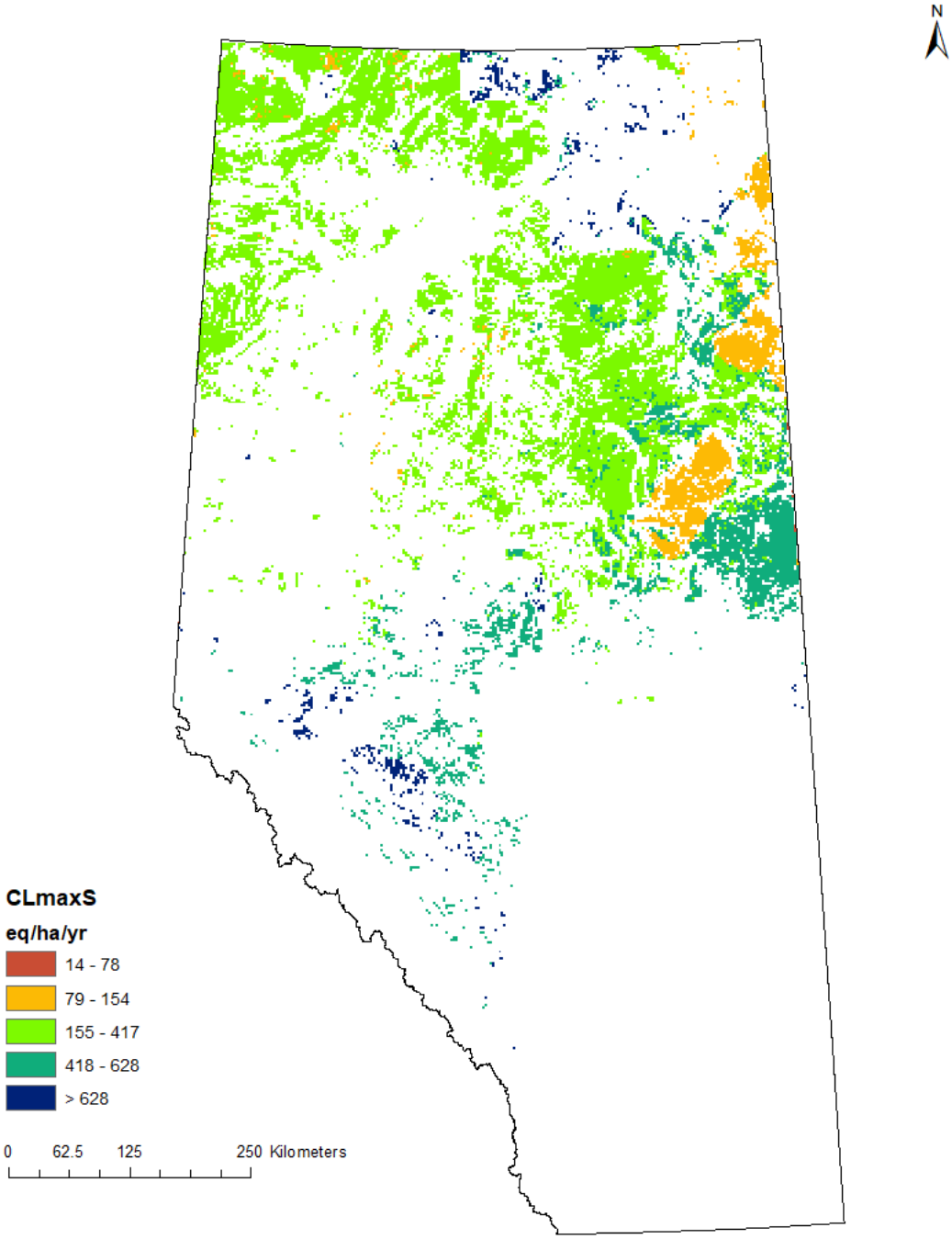

Figure G-2. $CL_{max}(S)$ for organic soils

Appendix H Estimating Nitrogen Content of Soils

The following were used to determine the pool of soil nitrogen throughout the province:

- Soil properties, i.e., percent nitrogen content, bulk density and horizon thickness from upland forest sites (Shaw *et al*., 2018)
- Soil properties for grassland using Alberta Environment and Park's Ecological Recovery monitoring program soil data (initial calculations only, see Section 2.2.1.1)
- Grassland soil property in literature (Li *et al*., 2012; Thomas *et al.*, 2017)

The distribution of upland forest and Ecological Recovery monitoring soil sites is shown in Figure H-1. Kilograms of nitrogen per hectare at each sample site ($N_{soil,\,m}$) is used to determine nitrogen immobilization and loss due to fire using Eq H-1.

$$N_{soil,m} = \sum_{h=1}^{n} \left(\frac{N_h}{100} \times \rho_{bulk\ h} \times D_h \times 10000 \right) \qquad \text{Eq H-1}$$

Where:

N_h = the percent nitrogen content of horizon *h*

D_h = the horizon thickness in m

$\rho_{bulk\ h}$ = bulk density at horizon *h* in kg m^3

n = the number of horizons

h applies to mineral soil horizons when calculating nitrogen immobilization and organic soil horizons when calculating nitrogen lost due to wildfires. Most forested sites (92%) had a single organic horizon with an average thickness of 7 cm (range of 1–46 cm).

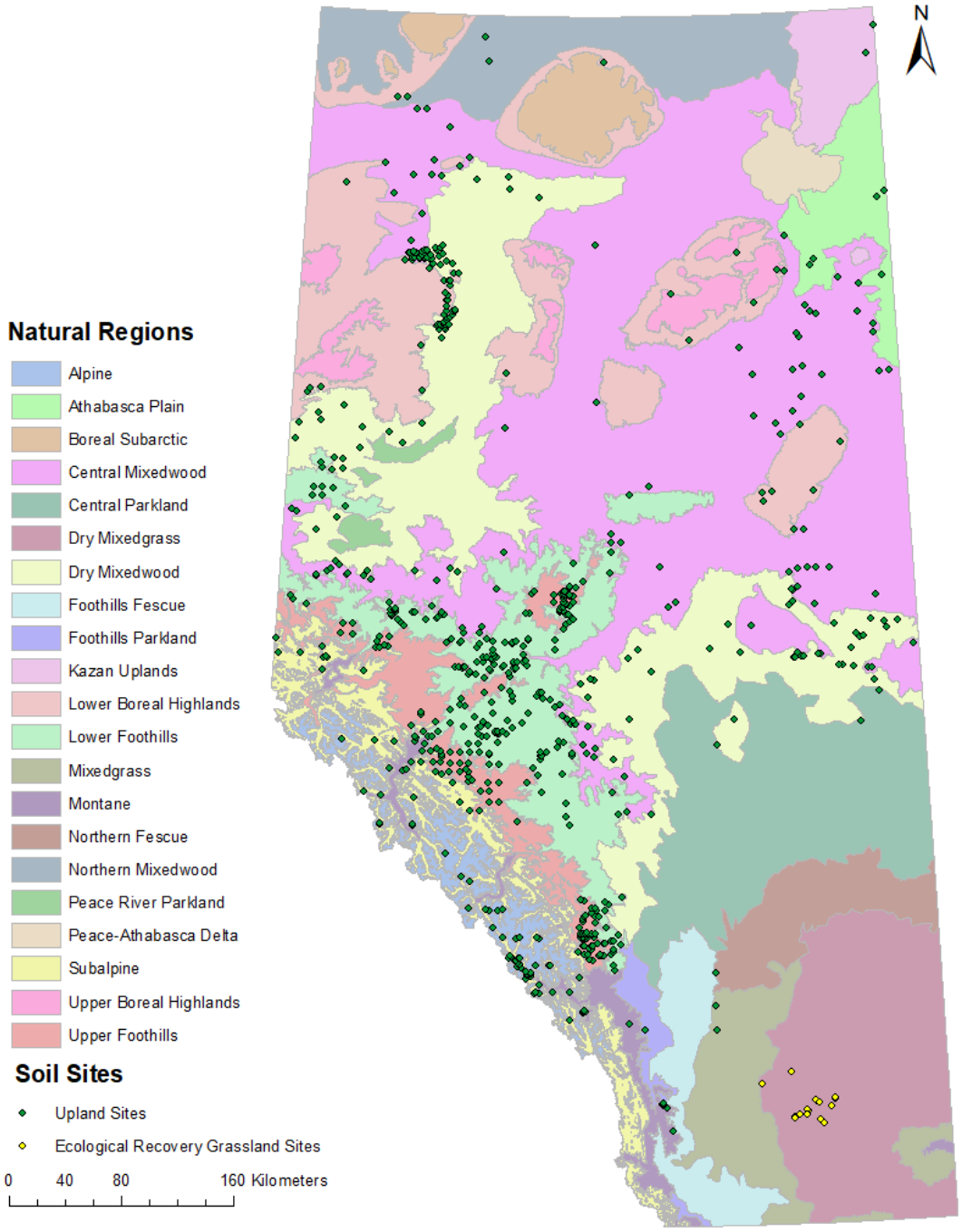

Figure H-1. Upland forest (green circles) and Ecological Recovery monitoring grassland (yellow circles) site location by natural subregions

Appendix I Method used to Estimate Critical Value to Protect Tree Species

Figure I-1 illustrates the weight of evidence approach used to derive the $BC{:}Al_{crit}$ for each land cover type. Each point in the graph represents the BC:Al ratio at which growth of a particular species was reduced by 20%. Data was obtained from a compilation by Sverdrup & Warfvinge (1993). The dataset has limited representation of Alberta species; therefore, species that are not found in Alberta are included as surrogates to better capture the full range of plant sensitivities.

In the example below for hardwood forests, a BC:Al ratio of 2 falls on the 75th percentile of data. However, aspen, which is an important forest species in Alberta, is more sensitive with a $BC{:}Al_{crit}$ of 6. Accordingly, the $BC{:}Al_{crit}$ for hardwood forests is set at 6 to ensure the critical load is sufficiently protective.

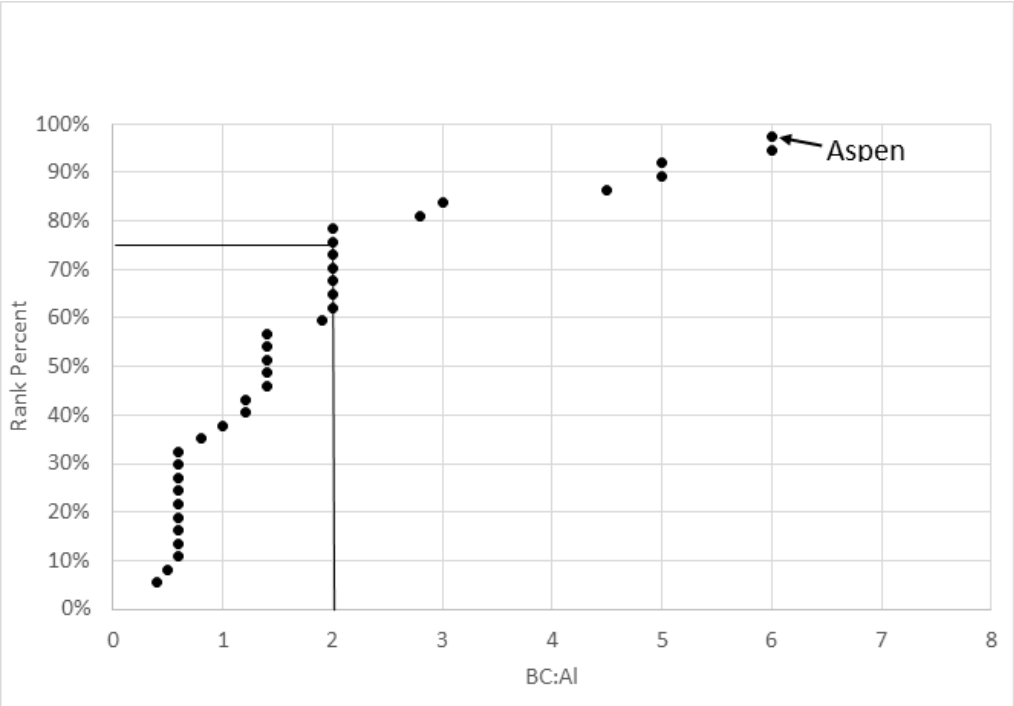

Figure I-1. Derivation of Base Cation:Al critical threshold for hardwoods (example). Data from Sverdrup and Warfvinge (1993) with Bc:Al ratio for coffee (BC:Al = 75) removed from the figure.

Appendix J Supplementary Figures

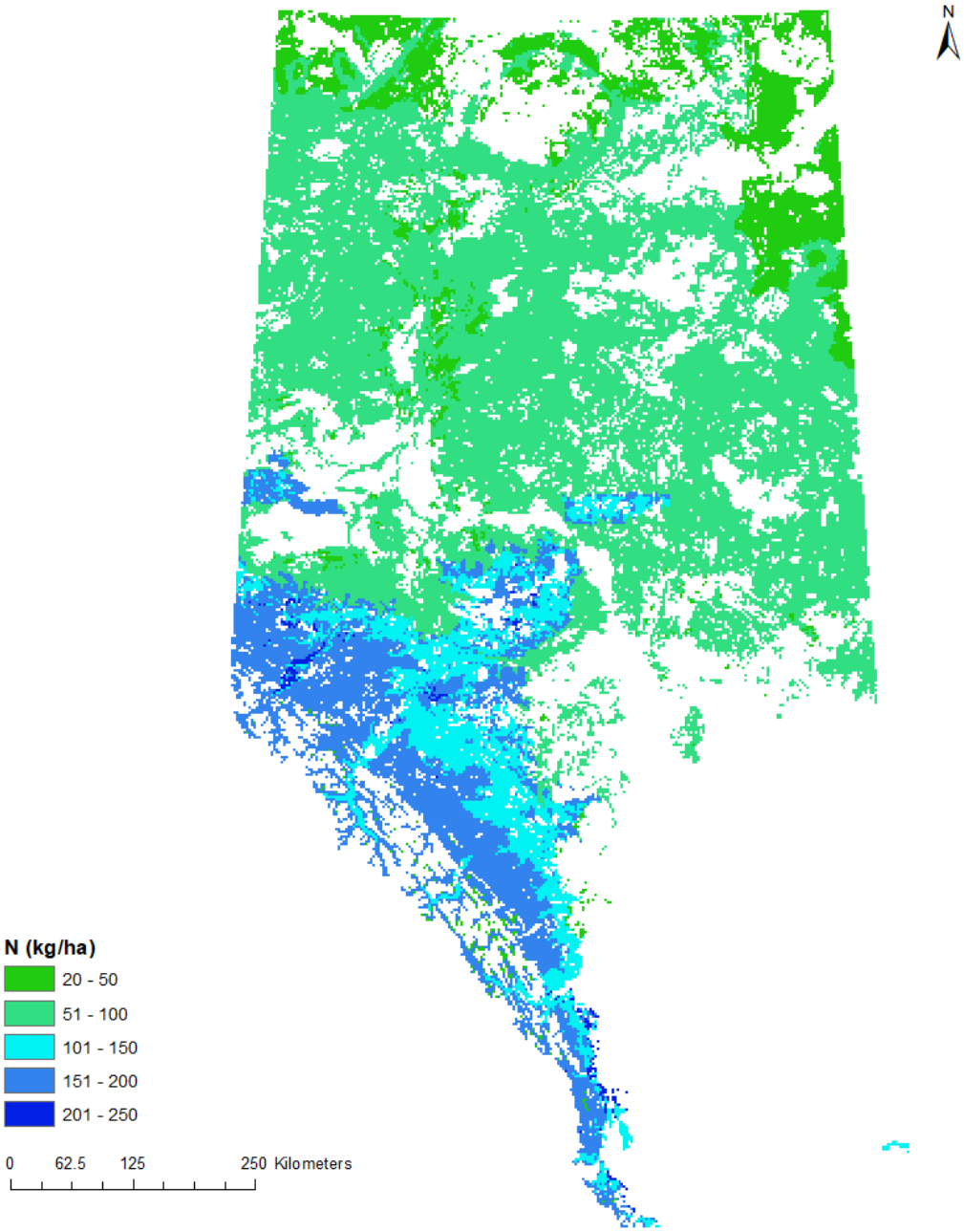

Figure J-1. Aboveground nitrogen (nitrogen content in trees)

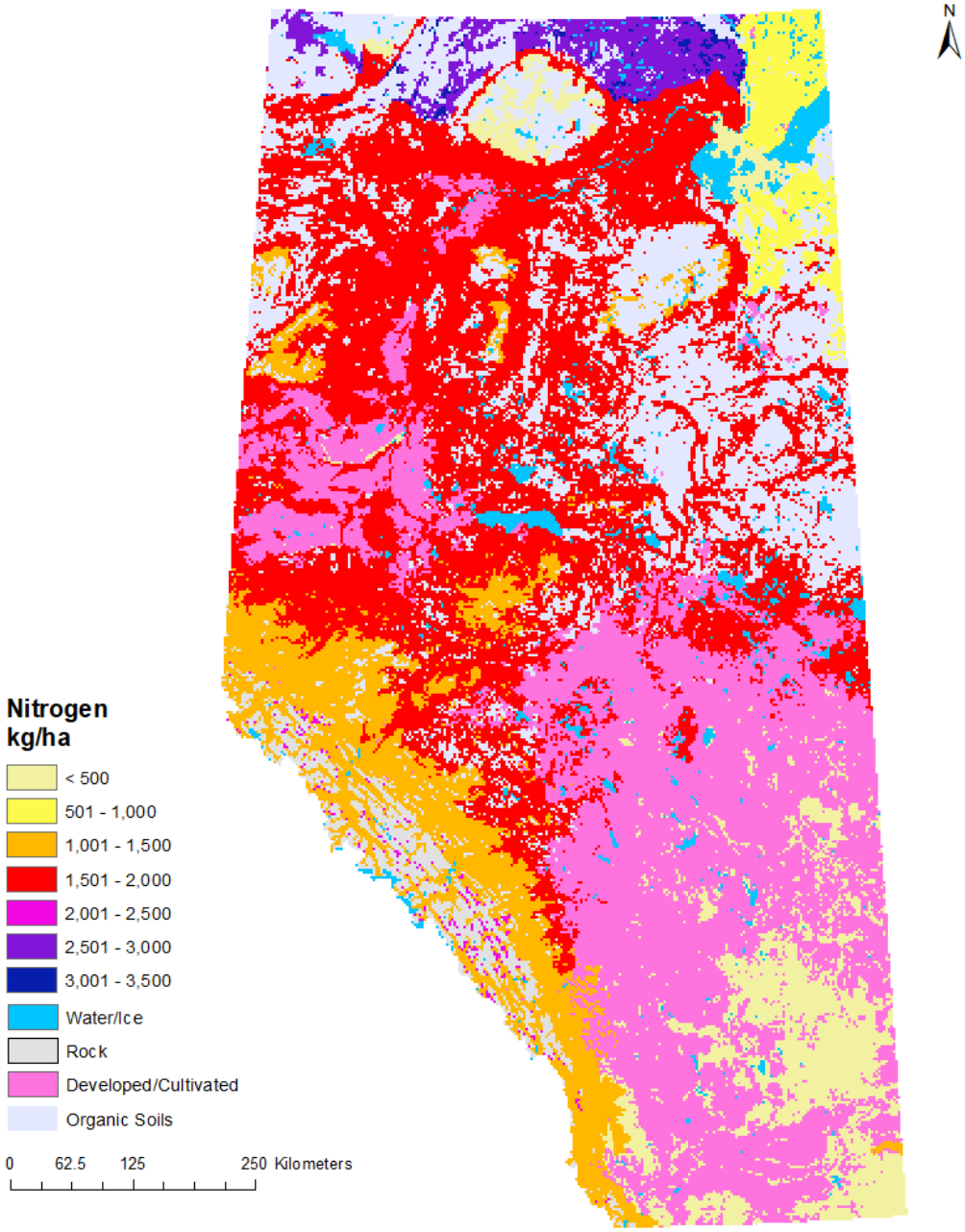

Figure J-2. Tree and soil organic horizon nitrogen content

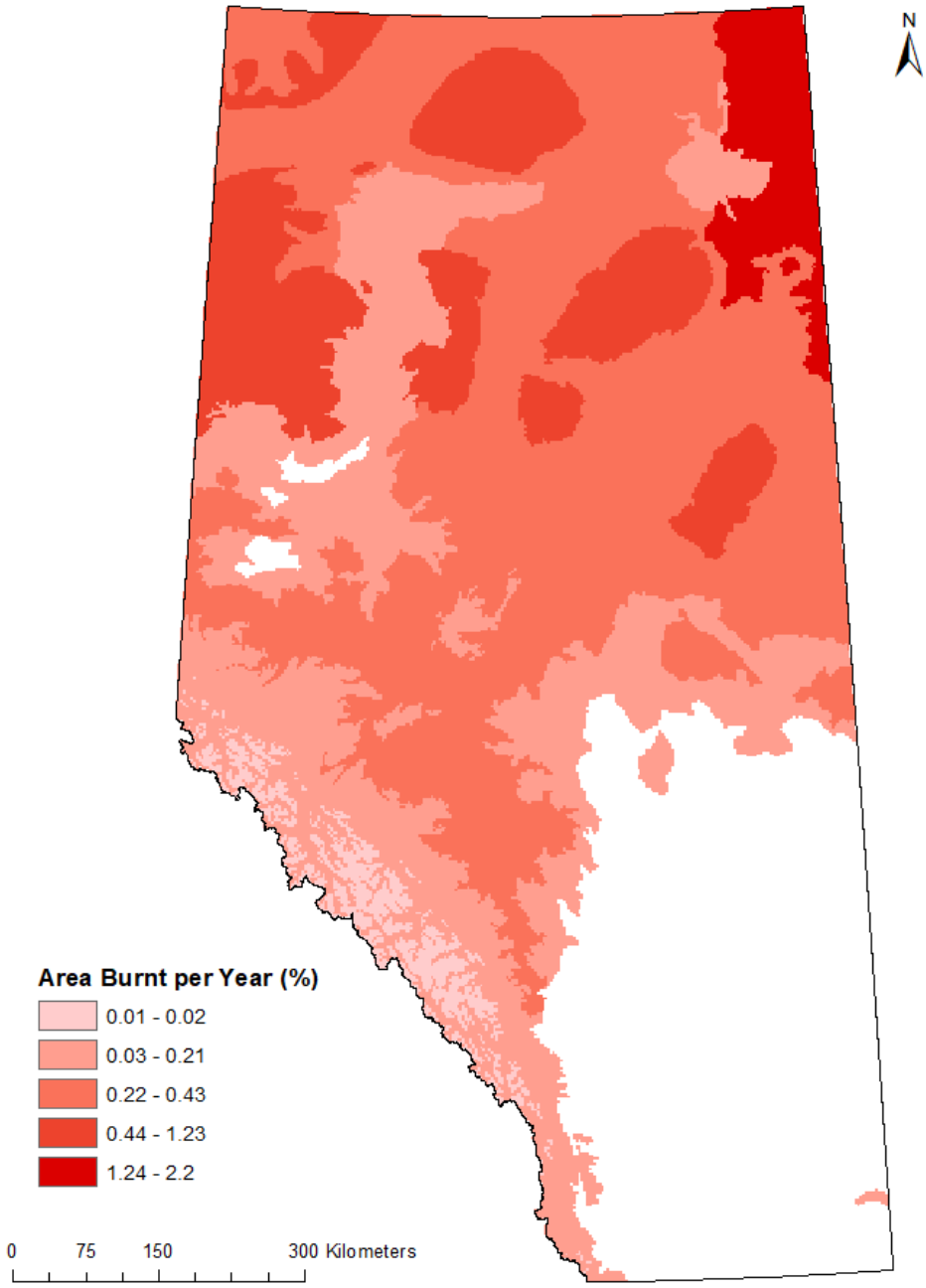

Figure J-3. Percent area burnt per year (Tymstra et al., 2005) determined using 41 years of data (1961-2002)