

Potential impacts of acidic deposition on forest soils of Alberta

by

Doug G. Maynard
Natural Resources Canada
Canadian Forest Service
Northern Forestry Centre
5320 122nd Street,
Edmonton, Alberta T6H 3S5

Phone (403) 435-7309
Fax (403) 435-7359
Email Dmaynard@nofc.forestry.ca

Abstract

Acidic deposition, primarily as sulfur (S) compounds in Alberta, has been shown to impact ecosystems alone or in combination with other stress factors including natural disturbances. Chemical changes include decreased pH, nutrient imbalances, leaching of base cations, and the mobilization of potentially toxic elements such as aluminum (Al). In Alberta, regional acidic deposition to forest soils is low, relative to eastern North America and Europe. Any changes to soil properties would be subtle and difficult to differentiate from natural soil variability. Soil acidification has been observed in localized areas near sour gas processing plants deposited with elemental S (S^0) dust. A biomonitoring study near two sour gas processing plants in west-central Alberta was carried out between 1981 and 1991. Soil chemical properties in mature lodgepole pine stands exposed to various levels of acidic deposition were evaluated with respect to foliar nutrients, forest health conditions, and tree growth. Chemical changes as a result of acidification from the deposition of large amounts of S^0 were found to a depth of at least 40 cm. These soil changes resulted in the elimination of the understory vegetation and a reduction in tree growth. At sites receiving low levels of acidic deposition, small reductions in pH and calcium (Ca) and magnesium (Mg) concentrations of the surface organic horizon were observed, but it was not possible to determine a cause and effect relationship between higher S deposition and lower pH, Ca and Mg. There was no apparent acidification of forest soils near the two sour gas processing plants outside of the areas dusted with S^0 . Critical loads for acidic deposition of Alberta soils were estimated from the results of this study, the soil sensitivity maps for Alberta and the European guidelines. Based on our current knowledge and acidic deposition rates, the potential for soil acidification in Alberta forests from acidic deposition is low.

Introduction

Acidic deposition may directly or indirectly result in impacts on forest vegetation (Morrison 1984). The indirect mechanisms include synergistic relationships with other stress factors, soil mediated changes in nutrient availability, and plant-mediated changes in competitive stress and the microenvironment (Maynard and Fairbarns 1994). The indirect soil-mediated effects are considered one of the most important in the impact of acidic deposition on tree growth and health (Roberts et al. 1989; Cape 1993; Rodhe et al. 1995). Changes in soil chemistry most often associated with increasing soil acidity (decreasing pH) include: increased leaching of base cations, increased solubility of aluminum (Al) to toxic levels, nutrient imbalances, and impacts on microorganisms (Foster 1989; Binkley et al. 1989).

Regional acidic deposition to forest soils in Alberta is low, relative to eastern North America and Europe. Any changes to soil properties would be subtle and difficult to differentiate from natural soil variability. To date in Alberta, soil acidification as a result of acidic deposition has been limited to localized areas contaminated with elemental sulfur (S^0) dust from adjacent sour gas processing plants.

The overall objective of this paper was to assess the potential impact of acidic deposition on forest soils in Alberta based on the results of a ten year (1981-1991) biomonitoring program in west-central Alberta near two sour gas processing plants (part of a larger study, Maynard et al. 1994). Two objectives of this study reported here are:

- i) determine changes to selected soil chemical properties as a result of extreme soil acidification from S^0 contamination, and
- ii) determine the potential impact of low levels of acidic deposition ($0.25-0.50 \text{ kmol}_e \text{ H}^+$

ha⁻¹ yr⁻¹ on selected soil chemical properties.

The results of this study were also used to evaluate critical loads of acidic deposition for forest soils in Alberta.

Materials and methods

A summary of the methods used in this study is provided. Detailed descriptions of the materials and methods are given elsewhere (Maynard et al. 1994).

All sites were located in an area 30 × 30 km with two sour gas processing plants at the center (Figure 1). There were 31 sites in mature, even-aged lodgepole pine (*Pinus contorta* Dougl. var. *latifolia* Engelm.) dominated stands, 6 young lodgepole pine stands, and 3 sites in mature trembling aspen (*Populus tremuloides* Michx.) stands. The young pine stands and trembling aspen stands were established in 1991. The sites reflect the full range of S^o and SO₂ deposition levels in the forests around the two sour gas processing plants (Figure 2). The relative SO₂ deposition was based on the static sulfation stations using lead dioxide cylinders maintained by the gas companies.

Ten (50 × 2 m) quadrats were set up in the mature pine-dominated and mature trembling aspen stands. A smaller quadrat size (10 × 2 m) was used in the 6 young pine stands because of the much greater tree density relative to mature stands.

Soil chemical analyses were compared in 1981, 1985 and 1991 from the mature pine stands. Soil samples were collected using the same methods in all 3 sampling years. Five replicates from the surface organic horizon (LFH), and the top three mineral horizons were sampled. Samples were collected in plastic bags and stored at -20°C if not processed immediately. Chemical analysis followed the procedures of Kalra and Maynard (1991).

Ten replicate samples from 1-year old lodgepole pine needles, 1-year old white and black spruce needles (*Picea glauca* [Moench] Voss and *Picea mariana* [Mill.] B.S.P.), twin-flower (*Linnaea borealis*), and feathermoss (*Pleurozium schreberi* [Brid.] Mitt.). Green alder (*Alnus crispa* [Ait.] Pursh) leaves were collected in 1981 and 1991 only. Foliage samples were analyzed for total elemental concentrations as outlined in Maynard et al. (1994).

Results and Discussion

Pollutants

The Ram River and Strachan sour gas processing plants process and recover about 3,000 t S^o d⁻¹. Incinerator stack emissions were about 70 t S d⁻¹ in 1981, declining to about 56 t S d⁻¹ in 1985 and to 25 t S d⁻¹ in 1991. The highest SO₂ emissions were measured during the first years of operation with the highest emission at about 120 t S d⁻¹ in 1973. The incinerator stacks at both plants disperse SO₂ over a wide area. Fugative emissions from the plant operations and flare stacks contribute to the overall gaseous emissions, particularly near the plants. The levels of these emissions, however, have not been quantified.

The S removed from the sour natural gas is recovered as S^o. Excess S^o that is not immediately shipped is stored in blocks at the plant. Dusting of the forest near the gas plants has resulted from the handling of the S^o during the shipping or storage of S^o. The amount and dispersion of S^o dust is a function of the amount shipped, handling methods used, including prilling towers, and weather conditions. In general, the majority of S^o was deposited within 1 km of the plants (accumulated S^o deposition was in the t S^o ha⁻¹ range), although S^o was measured in the soil up to 3 km from the source (Maynard et al. 1994).

Soil acidification from elemental S

The mature lodgepole pine sites were assigned to deposition classes based on the sulfation data from static sulfation stations and total S in the LFH (Figure 3). The high deposition class consisted of seven sites. Of these sites, six had measurable S^0 ($> 100 \text{ mg } S^0 \text{ kg}^{-1}$) and elevated S concentrations in the LFH.

Estimates of S^0 deposition were 263 and 70 kg S ha^{-1} at two sites 50 and 250 m (sites 1 and 2) east of the Strachan sour gas processing plant over a 6-month period between April and October 1983 (Maynard 1990). These sites are unique in the high amount of soil acidification that has occurred, but they provide valuable insight into the effect of soil acidification on soil chemical properties and impacts on tree growth and health.

The pH of the LFH and three mineral horizons of sites 1 and 2 are given in Table 1. The LFH of site 1, 50 m downwind of the S^0 block has been acidified to 2.5 for at least 10 years (the duration of the study). The pH of the LFH at site 2 initially decreased between 1981 and 1985 but increased between 1985 and 1991 almost to normal values. The mean pH of the LFH in the low deposition sites was 5.1 (Maynard et al. 1994).

The companies carry out a liming program in areas heavily contaminated by S^0 . Liming was excluded from these sites during the initial five years of the study; however, since 1985 these sites have been included in the regular liming program. The increase in LFH pH at site 2 was the result of lime application. Liming at the most heavily contaminated site had no effect on the LFH pH.

The pH of the mineral horizons decreased between 1981 and 1991 (Table 1). The decreases at site 1 were > 1 pH unit for the three horizons. At site 2, a 1 unit pH decrease was

measured only in the surface mineral horizon (A_{e1}). Acidification occurred first within the LFH horizon as the S^0 is deposited directly to the LFH and is oxidized to SO_4 . The acid produced must then be leached through the profile. Thus, there was a lagtime between acidification in the LFH and mineral horizons. The degree of acidification between sites 1 and 2 is a reflection of the amount of S^0 deposition and acid produced as a result of S^0 oxidation.

The decrease in pH by itself may not be the most important factor affecting forest health and the long-term productivity of these sites. The solubilization of Al and Mn and the accelerated leaching of base cations (calcium (Ca), magnesium (Mg), and potassium (K)) have been associated with soil acidification (Robarge and Johnson 1992). These changes can also occur without significant changes to soil pH (Johnson and Taylor 1989).

Extractable Al in the LFH and mineral horizons of sites 1 and 2 are given in Table 2. Extractable Al at sites 1 and 2 were already higher in the LFH at the initial sampling in 1981 compared to the mean extractable Al in the LFH of the low deposition sites (109 mg kg^{-1} ; Maynard et al. 1994). The Al concentration decreased significantly between 1981 and 1985 but did not change between 1985 and 1991. This was similar to SO_4 concentrations. The introduction of mobile, mineral acid anions, such as SO_4 , to the soil solution will result in increased Al solubility (Johnson and Taylor 1989). Thus, the high level of extractable Al in the LFH in 1981 occurred because of increased SO_4 and acidification. The continual input of SO_4 to the LFH through S^0 oxidation between 1981 and 1985 maintained high SO_4 concentrations (although lower than in 1981) and extreme acidic conditions. Thus, extractable Al was continually being leached from the LFH and resulted in a depletion of total Al from the LFH. Total Al decreased from $8,070$ to $1,750 \text{ mg kg}^{-1}$ at site 1 and from $7,100$ to $2,060 \text{ mg kg}^{-1}$ at site 2 between 1981

and 1991 (Maynard et al. 1995).

Leaching of Al from the LFH resulted in increased Al in the three surface mineral horizons (25-30 cm depth). These increases in Al coincided with decreasing pH and increasing SO_4 concentrations. Differences in the extractable Al concentrations of the mineral horizons between sites 1 and 2 were related to the differences in S^0 deposition.

The accelerated leaching of base cations can result in nutrient imbalances and eventually tree decline. Results for Ca and Mg are confounded because these sites were limed with dolomite (Ca and Mg carbonate) in 1979, 1980, and 1985. Potassium concentrations, however, indicate to which degree these cations are depleted by acidification. Total and extractable K concentrations in the LFH decreased at both sites by 70-90%. No changes, however, were found in the mineral horizons.

The extreme soil acidification from S^0 and its oxidation to SO_4 resulted in the depletion of cations and Al from the LFH. Leaching of Al from the LFH appears to have accumulated in the surface (top 25 cm) of the mineral horizons. In contrast, cations leached from the LFH did not accumulate in the surface mineral soils. The effect of soil acidification on the vegetation has resulted in the complete elimination of most understory species in the areas most heavily contaminated (site 1) and the decreased abundance of species and cover in areas acidified but to a lesser degree (site 2). The effect on lodgepole pine was not as obvious.

The main impact on the lodgepole pine has been a reduction in the annual growth increment. There was a greater proportion of recently dead trees in the heavily S^0 contaminated sites; however, there was no evidence of an increase in insect and disease activity associated with the high S^0 sites. At an adjacent aspen site, there was a greater proportion of declining and

recently dead trees compared to a non-S° site. The increase in effected trees was associated with an increased incidence of stem and root diseases of aspen. There were only three aspen sites surveyed; consequently, any conclusions should be viewed with caution.

The most visual impact of the extreme soil acidification is the lack of understory. Two 400-m parallel transects were established downwind of the Strachan S° block. Understory vegetation was assessed at 1m × 1m quadrats every 20 m along the transects. Species richness (number of species) was most closely related to pH of the top mineral horizon ($r^2 = 0.43$ and 0.74 for the two transects) and S concentrations in the soil (Total S natural log transformed, $r^2 = 0.45$ and 0.51 for the two transects). Changes in the extractable cations and Al of the soil were not as closely related to species richness. Thus, the lower soil pH as a result of S° deposition and its oxidation to sulfuric acid (H_2SO_4) was probably the most important factor in killing the understory vegetation.

Potential impact of chronic low levels of acidic deposition

The medium deposition class sites had sulfation rates between 0.030 and 0.050 eq mg SO_3 100 cm⁻¹ d⁻¹ (equivalent to 0.30 - 0.46 kmol_e H⁺ ha⁻¹ yr⁻¹)¹ based on the static sulfation stations. Soil chemical properties were compared within sites over time.

The total S concentrations of the LFH increased for three of the medium deposition class sites between 1981 and 1991 (Table 3). No changes were observed in the other 10 medium deposition sites (data not shown). The total S concentrations of these sites increased by 50%

¹ The conversion of eq mg SO_3 100 cm⁻² d⁻¹ to kmol_e ha⁻¹ yr⁻¹ assumes 1 mole of S is equivalent to 2 moles of H⁺. Static sulfation stations have a higher affinity for SO_2 and other gases than soils and vegetation, therefore, the acidic deposition rate calculated from the sulfation data is probably overestimated.

between 1985 and 1991 indicating the deposition and retention of considerable S. Elemental S was measured at site 14 and possibly site 13 (only one of five samples contained measurable S⁰). There was no S⁰ detected at site 23. The large increases in total S (50%) were more typical of S⁰ deposition than gaseous S deposition. Large amounts of S⁰ were shipped between 1985 and 1989 and this may have resulted in S⁰ being deposited further away from the gas plants. Sulfur dioxide emissions from both gas plants decreased between 1985 and 1991 with a parallel decrease in sulfation rates. Thus, the higher S concentrations in the LFH of these three sites may have been due to S⁰; however, deposition of SO₂ cannot be completely ruled out.

Foliar S concentrations of the trees in 1981 and 1991 were similar; however, foliage collected in 1985, had S concentrations 20-40% lower than either 1981 or 1991. Climatic and moisture conditions were probably responsible for the decrease in S concentrations of the vegetation in 1985 (Maynard et al. 1994). Increased S concentrations between 1981 and 1991 were observed for feathermoss at three medium deposition class sites (sites 3, 13, and 14). No other foliage sampled had higher S concentrations in 1991 compared to 1981 in the medium deposition sites. This is consistent with the total S concentrations in the LFH and would support the hypothesis that S⁰ deposition may have been responsible for the increase in S concentrations at these 3 sites.

The LFH pH of the 13 medium deposition class sites did not decrease between 1981 and 1991 (Table 3, three sites). Similarly, extractable cations of the LFH did not change among sampling times at any of the medium deposition class sites.

Exchangeable Al of the LFH in the 13 medium deposition class sites generally decreased from 1981 to 1985 and then increased in 1991 to concentration similar to those measured in 1981

(Table 3). A similar decrease in exchangeable Al among years was observed in all the low deposition class sites. No changes were observed in the mineral horizons of these sites.

The pH and extractable S, Ca, Mg and K concentrations in the LFH by deposition class are given in Table 4. The S concentrations of the LFH were higher in the medium deposition class sites compared to the low deposition class sites. Increased S concentrations were probably due to S deposition either as gaseous SO₂ or particulate S°. Decreases in pH, and Ca and Mg concentrations of the LFH were also observed (6-10%); however, it was not possible to determine a cause and effect relationship between higher S concentrations in the medium deposition class sites and lower Ca and Mg concentrations. In contrast to Ca and Mg, extractable K concentrations in the LFH of the medium deposition class sites were higher than in low deposition sites (Table 4). In addition, there was no trend towards decreasing pH or base cation concentrations at the sites in the medium deposition class in the decade over which measurements were made. Thus, it is reasonable to conclude the decreases in pH, Ca and Mg were probably not associated with the elevated S concentrations but other edaphic factors.

Critical loads for soil acidification

The concept of critical loads² has been used extensively in Europe to assess the potential risk of soils to acidic deposition (Bull 1991; WHO 1994). Critical loads for acidic deposition can be estimated at three complexity levels. The least complex level is an empirical approach that assigns critical loads to soils based on existing soil survey information and current acidic deposition rates. This approach has been used to map critical loads for Great Britain, Ireland and

²A quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to our present knowledge (UN/ECE 1988 according to Brodin and Kuylenskierna 1992).

parts of the former Soviet Union (Kämäri et al. 1992; Hornung et al. 1995). The other more complex methods used to estimate critical loads are steady-state mass balance models and dynamic integrated models. In this paper, the critical loads for Alberta soils will be based on the least complex empirical approach.

In Alberta, soil sensitivity to acidic inputs and potential to neutralize acidic inputs were mapped using current soil survey information (Holowaychuk and Fessenden 1987). Maps for Alberta were prepared showing the distribution of soils relative to soil sensitivity to acidic deposition and the potential of the soils and geology to neutralize acidic inputs (Figure 4, potential to neutralize acid input adapted from Holowaychuk and Fessenden 1987). The maps were not the same because in some areas of the province sensitive topsoil material were underlain by calcareous parent material that had a high potential to neutralize acidic input (Holowaychuk and Fessenden 1987).

The WHO guideline for critical loads of soils in Europe recommended critical loads of $0.25 \text{ kmol}_e \text{ H}^+ \text{ ha}^{-1} \text{ yr}^{-1}$ for the most sensitive soils, $0.25\text{-}0.50 \text{ kmol}_e \text{ H}^+ \text{ ha}^{-1} \text{ yr}^{-1}$ for moderately sensitive soils and $> 0.50 \text{ kmol}_e \text{ H}^+ \text{ ha}^{-1} \text{ yr}^{-1}$ for the least sensitive soils. Combinations of parent material and texture were used in developing these guidelines (WHO 1994). Similar critical load ranges were used for Great Britain and recommended as interim critical loads for Alberta (Alberta Environment 1990) and western Canada (Interim Acid Deposition Critical Loadings Task Group 1990).

The majority of the soils in our study were moderately sensitive to acidic input and had medium to high potential to neutralize acidic input. The sulfation data from the static sulfation stations (11 years) provided an estimate of S deposition. Based on the sulfation data, the medium

deposition area of our study would have acidic deposition rates between 0.30 and $0.46 \text{ kmol}_e \text{ ha}^{-1} \text{ yr}^{-1}$ ($0.035 - 0.05 \text{ eq mg SO}_3 \text{ 100 cm}^{-2} \text{ d}^{-1}$). As discussed earlier, there were elevated S concentrations in the LFH and foliage of some understory and tree species at these sites; however, there was no decrease in pH of the LFH or mineral soils during the 10 years of our study. There was also no widespread forest decline within this deposition area. Based on these results, there were no significant harmful effects of acidic deposition on these soils except on soils heavily contaminated by S^0 .

Sulfur emissions have been declining in the central and southern parts of Alberta for the past 10 years and are expected to continue to decline. The majority of soils have either medium to high potential to reduce the acidity of acidic deposition. Thus, the overall risk of soil acidification due to acidic deposition would be low in most of central and southern Alberta.

The most sensitive soils, including organic soils, to acidic deposition are in northern Alberta. Northeastern Alberta is also the area of the highest S emissions in the province. Cheng et al. (1995) modeled current deposition rates of $0.18 \text{ kmol}_e \text{ H}^+ \text{ ha}^{-1} \text{ yr}^{-1}$ in the highest deposition areas. If the modeled emissions are accurate, then the critical load for the most sensitive ecosystems of $<0.25 \text{ kmol}_e \text{ H}^+ \text{ ha}^{-1}$ (WHO 1994) would not be exceeded.

Therefore, based on our current knowledge, using the results of our 10 year study near two point sources and the critical load guidelines from Europe and given the current rates of acidic deposition, the potential for soil acidification from acidic deposition in Alberta is low.

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Table 1. The pH of the surface organic horizon (LFH) and mineral horizons of sites 1 and 2 in 1981, 1985 and 1991. Values are means \pm 95% confidence limits (n = 5).

	LFH	IAe	Bm	IIAe
Site 1				
1981	2.5 \pm 0.2a ¹	4.1 \pm 0.6a	5.3 \pm 0.4a	5.1 \pm 0.4a
1985	2.5 \pm 0.5a	3.3 \pm 0.1b	4.2 \pm 0.0b	4.2 \pm 0.2b
1991	2.5 \pm 0.5a	2.8 \pm 0.4b	3.9 \pm 0.2b	3.9 \pm 0.2b
Site 2				
1981	4.3 \pm 1.1b	4.7 \pm 0.7a	5.7 \pm 0.5a	5.4 \pm 0.3a
1985	3.9 \pm 1.3b	3.9 \pm 0.6b	4.8 \pm 0.5b	5.0 \pm 0.2ab
1991	4.8 \pm 1.2a	3.7 \pm 0.3b	4.6 \pm 0.6b	4.8 \pm 0.1b

¹pH, for a given site and horizon, followed by the same letter do not differ significantly ($p \geq 0.05$).

Table 2. Extractable Al (1.0 M NH_4Cl) concentrations of the surface organic horizon (LFH) and mineral horizons of sites 1 and 2 in 1981, 1985, and 1991. Values are means \pm 95% confidence limits (n = 5).

Site	LFH	IAe	Bm	IIAe
----- mg kg ⁻¹ -----				
Site 1				
1981	859 \pm 493a	414 \pm 125b	191 \pm 95b	98 \pm 48c
1985	115 \pm 74b	634 \pm 166a	620 \pm 39a	420 \pm 141b
1991	133 \pm 78b	181 \pm 86c	796 \pm 518a	667 \pm 264a
Site 2				
1981	267 \pm 242a	300 \pm 229b	187 \pm 221a	80 \pm 69a
1985	50 \pm 65b	503 \pm 177a	373 \pm 326a	80 \pm 50a
1991	75 \pm 88b	627 \pm 180a	306 \pm 179a	160 \pm 32a

¹Extractable Al, for a given site and horizon, followed by the same letter do not differ significantly ($p \geq 0.05$).

Table 3. Total sulfur (S), elemental S (S^o), pH, and extractable Al in the surface organic horizon (LFH) of three medium deposition class sites in 1981, 1985, and 1991. Values are means \pm 95% confidence limits (n = 5).

	Total S	Elemental S	pH	Extractable Al
	----- mg kg ⁻¹ -----			mg kg ⁻¹
Site 13				
1981	920 \pm 503b	B. D. ¹	4.5 \pm 0.3ab	208 \pm 63a
1985	1044 \pm 174b	B. D.	4.4 \pm 0.3b	82 \pm 47b
1991	1666 \pm 104a	B. D. ²	5.0 \pm 0.5a	127 \pm 98ab
Site 14				
1981	1080 \pm 390b	B. D.	4.6 \pm 0.3a	151 \pm 61a
1985	1040 \pm 184b	B. D.	4.4 \pm 0.3a	61 \pm 58a
1991	1500 \pm 288a	100 \pm 11	4.7 \pm 0.6a	108 \pm 79a
Site 23				
1981	1265 \pm 239b	B. D.	4.2 \pm 0.2a	293 \pm 72a
1985	952 \pm 112c	B. D.	4.2 \pm 0.2a	135 \pm 88b
1991	1590 \pm 172a	B. D.	4.5 \pm 0.2a	253 \pm 46ab

¹B. D. = below detection limit of the method. For LFH material the detection limit is 100 mg S^o kg⁻¹ soil.

²One of the five replicates had detectable S^o in the LFH (138 mg kg⁻¹).

Table 4. pH, extractable sulfur (S), calcium (Ca), and magnesium (Mg) of the surface organic horizon (LFH) in 1991 by deposition class. Values are means \pm 95% confidence limits.

Deposition Class	pH	Extractable S	Extractable Ca	Extractable Mg	Extractable K
----- mg kg ⁻¹ -----					
High (7 sites)	5.3 \pm 0.1a	748 \pm 37a	10 400 \pm 267a	501 \pm 20c	596 \pm 18c
Medium (13 sites)	4.8 \pm 0.1c	158 \pm 27b	5110 \pm 196c	653 \pm 14b	848 \pm 13a
Low (10 sites)	5.1 \pm 0.0b	145 \pm 31c	5410 \pm 223b	716 \pm 16a	772 \pm 15b

Table 5. Soil sensitivity to acidic deposition and potential to neutralize acidic inputs and the critical loads in each class for mineral soils of Alberta.¹

Soil Sensitivity to acid deposition ²	Potential to neutralize acid deposition ³	Critical load ⁴ kmol _c H ⁺ ha ⁻¹ yr ⁻¹
High	Low	< 0.25
Medium	Medium	0.25 - 0.50
Low	High	> 0.50

¹Mapping units with high sensitivity to acid inputs are not always associated with mapping units with low potential to neutralize acidic inputs.

²Sensitivity to base loss, soil acidification and solubilization of Al from acidic inputs for Alberta soils (Holowaychuk and Fessenden 1987).

³Potential of soils and geology to neutralize acidic inputs was based on soil depth, exchangeable bases of soil, bedrock and parent material type, and soil drainage (Holowaychuk and Fessenden 1987).

⁴Guideline range for critical loads of soils for Europe based on a combination of parent material and texture of soils (WHO 1994).

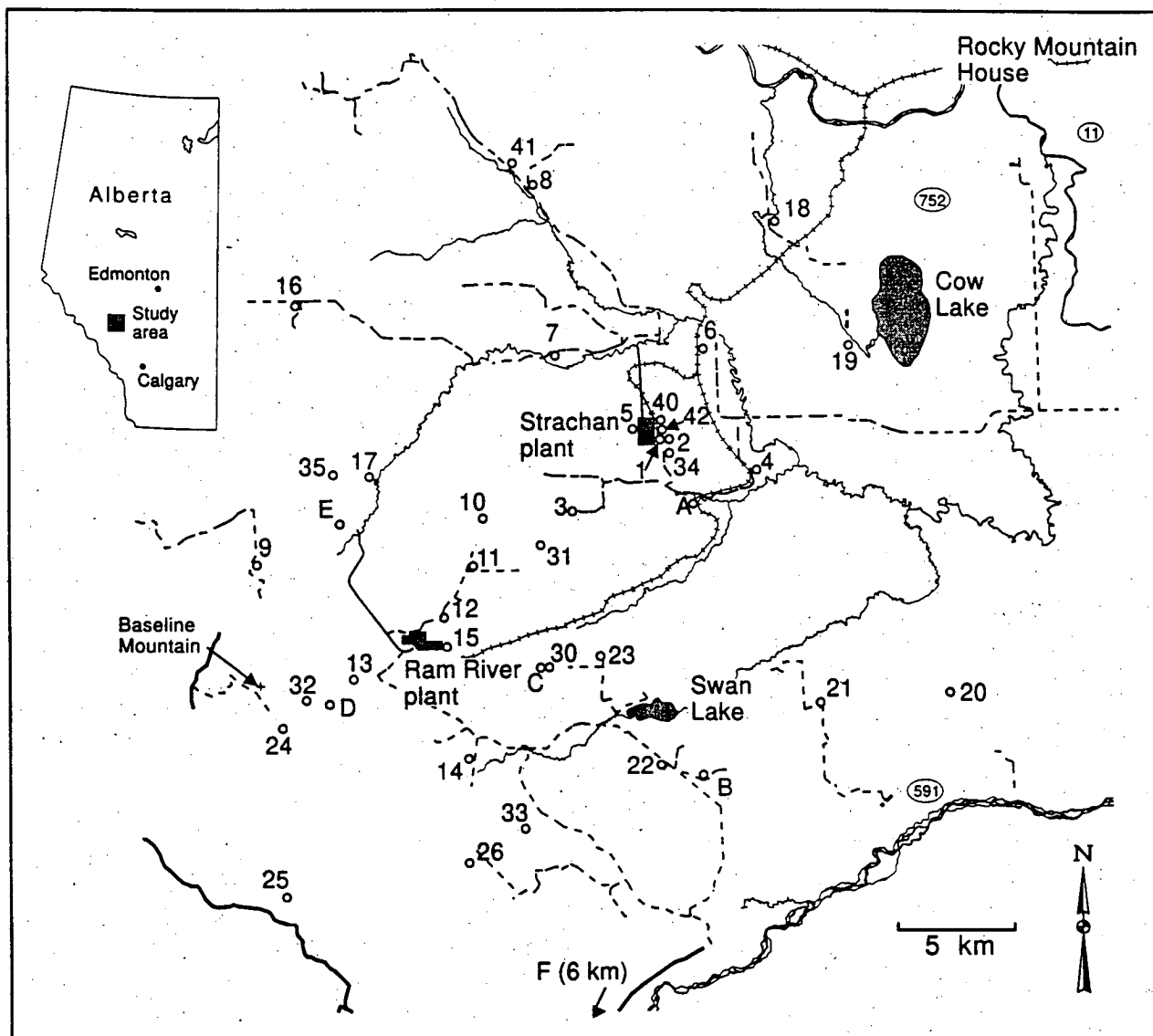


Figure 1. Location of biomonitoring plots (numbered and lettered) and sour gas processing plants in west-central Alberta. Reprinted from Information Report NOR-X-334.

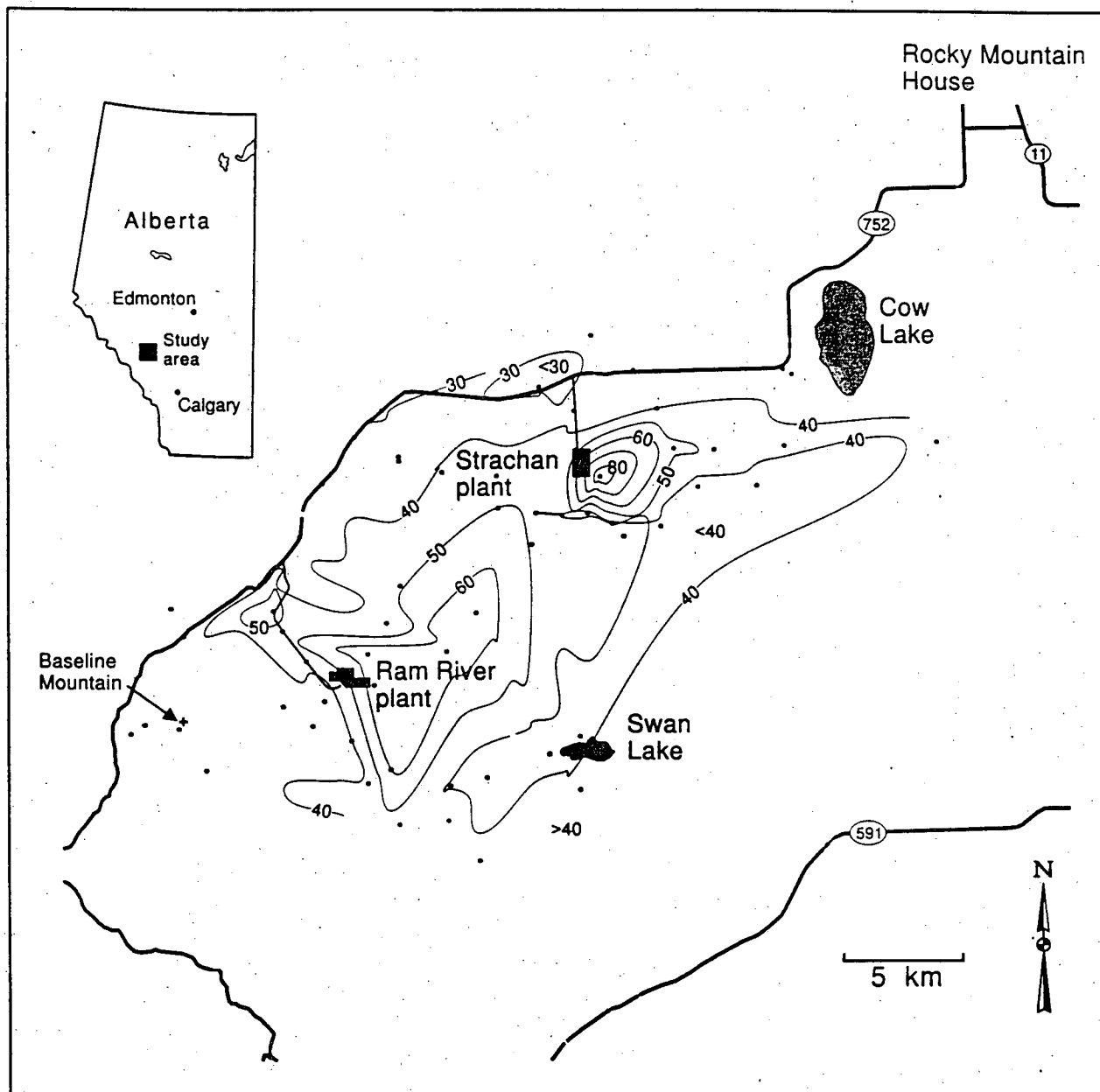


Figure 2. Isopleths of average sulfation rates near sour gas processing plants in west-central Alberta, 1975–86. Contour units are equivalent $\text{mg SO}_3 \text{ } 100 \text{ cm}^{-2} \text{ d}^{-1} (\times 1000)$. Reprinted from Information Report NOR-X-334.

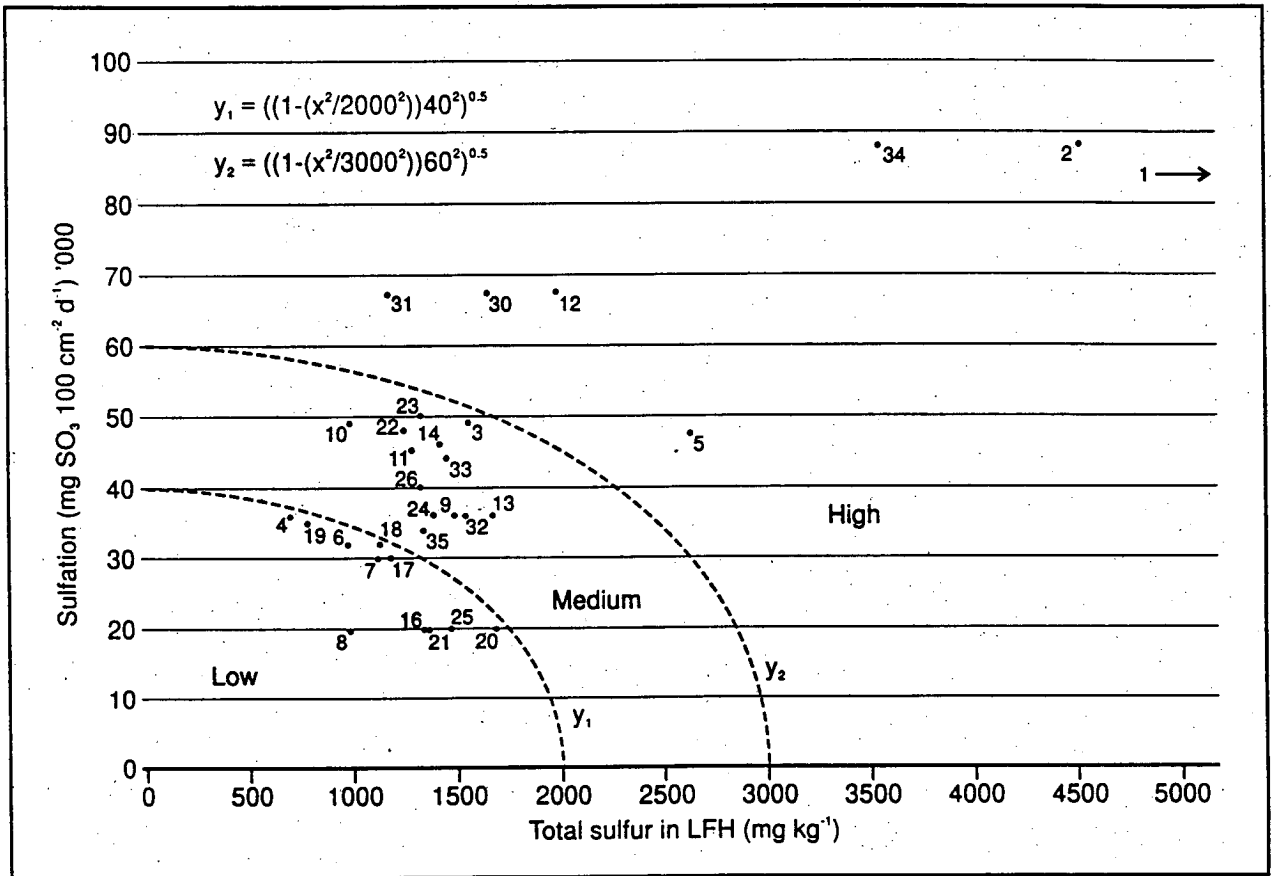


Figure 3. Classification of mature lodgepole pine study sites on the basis of sulfation and total sulfur in the LFH. The y_1 function curve separates the low and medium deposition classes and the y_2 curve separates the medium and high impact classes. Reprinted from Information Report NOR-X-334.

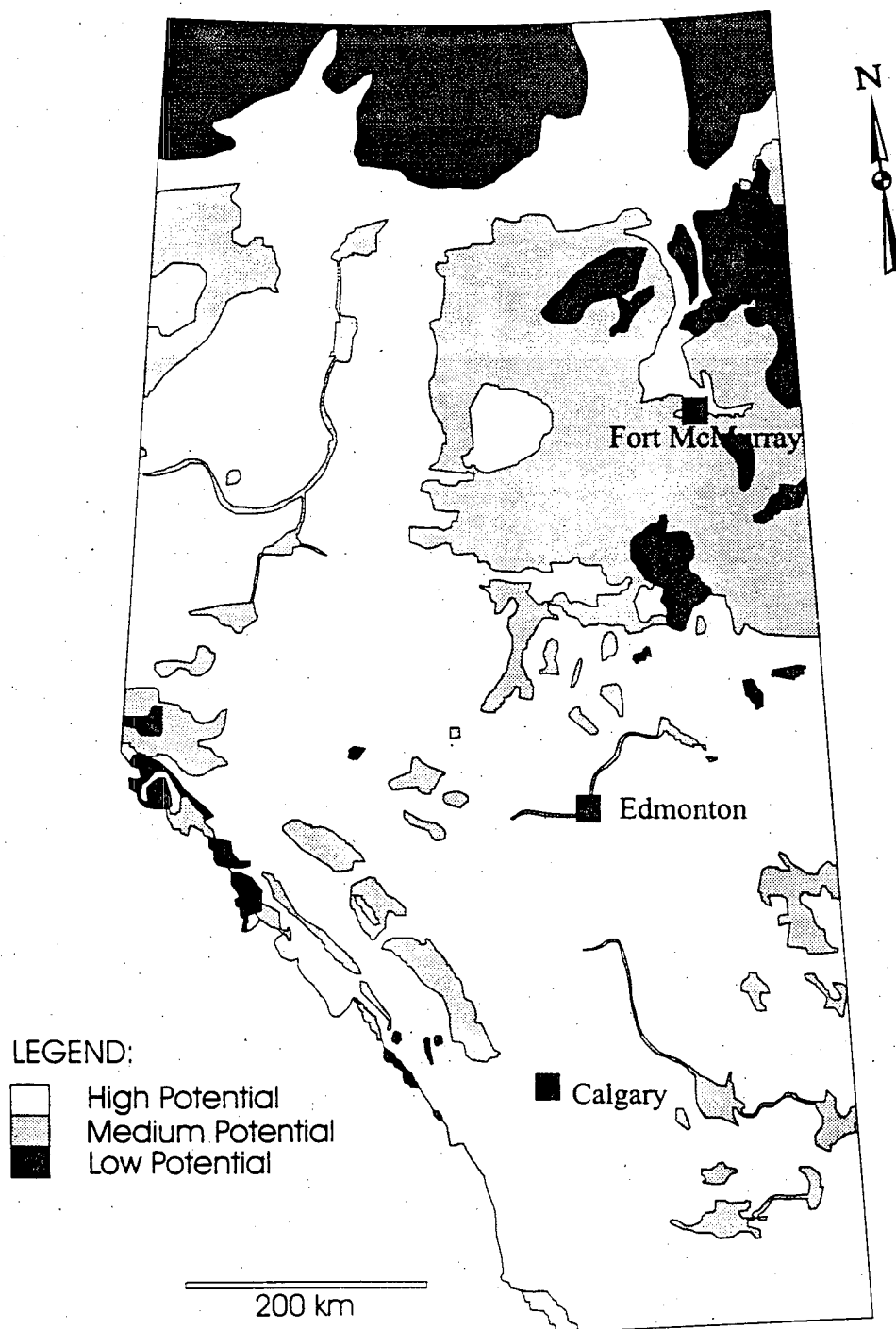


Figure 4. Potential of Alberta soils and geology to reduce the acidity of atmospheric deposition. (adapted from Holowaychuk and Fessenden 1987)