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INTENSE, NATURAL POLLUTION AFFECTS ARCTIC TUNDRA VEGETATION AT THE SMOKING HILLS, CANADA[1]

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Abstract. Long-term, natural emissions of sulfur dioxide and acidic aerosols have had an impact on remote tundra at the Smoking Hills. The emissions have caused plant damage by SO_2 toxicity, and have severely acidified soil and freshwater. At the most intensively fumigated locations closest to the sources of emission, pollution stresses have devegetated the terrestrial ecosystem. The first plants that are encountered along a spatial gradient of decreasing pollution stress are *Artemisia tilesii* and *Arctagrostis latifolia,* which dominate a characteristic, pollution-tolerant community. Farther away at moderately polluted sites there are mixed communities with floristic elements of both fumigated and reference, unfumigated tundra. This pattern of ecosystem response to a concatenation of stresses caused by natural air and soil pollution is qualitatively similar to the damage that occurs in the vicinity of anthropogenic point sources of air pollution, such as smelters.

Key words: air pollution; Arctagrostis latifolia; *arctic tundra;* Artemisia tilesii; *calcium; historical change; magnesium; pollution-tolerant plant community; Smoking Hills, N.W.T., Canada; soil acidification; sulfur dioxide.*

INTRODUCTION

The Smoking Hills are located at 70°14′ N, 127°10′ W, on the east coast of Cape Bathurst in the Northwest Territories of Canada, ≈250 km east of the delta of the Mackenzie River (Fig. 1). This remote, uninhabited area has been little influenced by human activity. At several locations along ≈30 km of this seacoast, exposures of bituminous shale in 100 m high seacliffs have spontaneously ignited. Plumes from these burns fumigate the tundra with sulfur dioxide, sulfuric acid mists and aerosols, and particulates (Fig. 2). As a result, local damage has been caused to terrestrial and aquatic tundra ecosystems (Hutchinson et al. 1978, Havas and Hutchinson 1983).

The area has a bedrock of Upper Cretaceous shale, covered by <10 m of calcareous glacial till and alluvial deposits. Interbedded in the bedrock matrix are layers of yellow jarosite (a hydrated sulfate of iron and potassium) and bituminous shale. Where the latter has been exposed to the atmosphere by erosion or slumping of the seacliff, pyritic sulfur undergoes an exothermic oxidation to sulfate. The accumulation of heat can be sufficient to ignite the bituminous material, and the fire then burns into the eroding seacliffs until the oxygen supply becomes insufficient to support further combustion. This mechanism of spontaneous combustion is similar to that proposed for the ignition of waste coal heaps and underground coal mines, where pyrites are also exposed to atmospheric oxygen (Sussman and Mulhearn 1964, Mathews and Bustin 1984).

The earliest documented sighting of the Smoking Hills by Europeans was by the Franklin-Richardson expedition of 1826, but the burns are undoubtedly more ancient than this. The area was probably not covered by ice as late as 18 000 BP during the most recent Wisconsinan glaciation (Prest 1984, Ritchie 1987), and the large piles of ash and oxidized shale at the bottom of the seacliffs suggest that the burns are ancient.

During the short growing season, the prevailing wind direction at the Smoking Hills is from the sea, and the sulfurous plumes are carried inland where they can be tracked visually as far as 40–50 km. Previously published studies have demonstrated the impacts of atmospheric pollution on the chemistry of freshwater and soil. The most important effect is a severe acidification, including the occurrence of very low pH, and also secondary effects such as the solubilization of toxic metals

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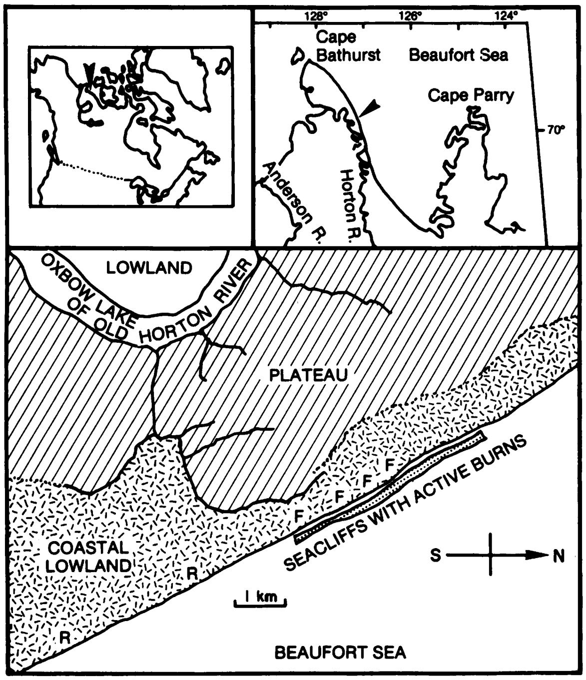

FIG. 1. The study area. All sampling transects started at the edge of the seacliff, and ran inland perpendicular to the coastline. The starting points of transects in fumigated habitat are denoted by "F," and by "R" in reference habitat.

such as aluminum, and the leaching of calcium and magnesium (Hutchinson et al. 1978, Gizyn 1980, Havas and Hutchinson 1983). Effects of the toxic conditions on the biota of freshwater tundra ponds have also been described (Havas 1980, Havas and Hutchinson 1983).

The air and soil pollution at the Smoking Hills area has also damaged the terrestrial vegetation. In terms of degree, but not scale, these effects rival or exceed the worst anthropogenic cases of ecological damage by SO_2 and acidic deposition, for example, the damage that has been caused in the vicinity of some metal smelters (Freedman and Hutchinson 1980*a, b,* Freedman 1989). Moreover, because the exposure has lasted much longer than the comparable anthropogenic stresses, studies of damage at the Smoking Hills can contribute to the prediction of the longer term consequences to ecosystems and biota of sulfur dioxide pollution and the deposition of acidifying substances. The purpose of the present report is to describe how the terrestrial vegetation of the Smoking Hills has been affected by a long-term exposure to sulfurous fumigations and the concomitant soil effects.

METHODS

Vegetation studies

Sample sites.—Field data on the vegetation of the Smoking Hills were collected in July of 1986. Fifty sample sites were chosen nonrandomly to represent locations within gradients of pollution from the fumigations, including unpolluted, reference habitats occurring in the general vicinity (Fig. 1).

i) *Reference mesic–xeric sites.*—At an unfumigated location, we sampled a transect of four sites at distances as far as 500 m from the coast. There were also two other coastal, mesic–xeric, reference sites and several inland sites, the farthest being 4.2 km inland. All of these reference stands were dominated by *Dryas integrifolia* and *Salix arctica.*

ii) *Fume-damaged mesic–xeric sites.*—Four transects were sampled in areas of obvious fumigation damage. Each transect extended inland from the edge of the seacliff, from a point where a discrete plume wafted inland with the prevailing, roughly southwesterly, wind from the sea. The sample sites were located at increasing distances from the cliff, and represented a gradient of pollution. The maximum transect length (650 m) was determined by the width of the coastal lowland. Further inland, there is an upland plateau with different plant communities in mesic and hydric habitats. The 32 sites along the fumigated transects

FIG. 2. A slump of the seacliff at the Smoking Hills has exposed bituminous shale to the atmosphere, resulting in a spontaneous ignition, and fumigation of the nearby tundra with sulfur dioxide, and sulfuric acid mists and aerosols. This has caused a severe acidification and other damage to terrestrial and aquatic ecosystems.

ranged in character from totally devegetated, to having vegetation with floristic elements representative of both reference and fumigated sites.

iii) *Reference hydric sites.*—There were four reference wet-meadow sites, all located at least 2 km inland. All were dominated by graminoids and bryophytes.

iv) *Fumigated hydric sites.*—Three fumigated wet-meadow sites were located along a transect extending inland through a plume, with vegetation dominated by graminoids and bryophytes. In addition, we sampled two former wet meadows that had been devegetated by fumigations.

Collection of vegetation data.—At each sample site, 10 1 × 1 m quadrats were placed at random locations along a sampling line that extended parallel to the seacoast. In each quadrat, the cover of each plant species, litter, and bare ground was estimated as the percentage of surface obscured, i.e., the sum of all cover values totalled 100%. If cover was $<1\%$, then the species was noted as "present" (P). The taxonomy used for the identification of plants was; (i) Hulten (1968) for vascular plants; (ii) Conard (1956) and Crum (1976) for bryophytes; and (iii) Fink (1960) and Hale (1969) for lichens.

Analysis of vegetation data.—To identify the statistically associated groups of stands on the basis of their species composition and cover, the data matrix of stands × species cover was subjected to the divisive clustering procedure, two-way indicator species analysis (TWINSPAN). TWINSPAN is based on a reciprocal averaging algorithm, and divisions are made on the basis of stand attributes (Hill 1979*a*, Gauch 1982). Note that totally devegetated sites were not clustered or ordinated (see below), because they lacked vegetation data.

The results of the cluster analyses were used to assign the stands among plant "communities." For each community, the average cover of plant species was calculated from the original stand data, as were the average values of measured soil variables (see Methods: Environmental Measurements [Soil Chemistry]).

To explore the multivariate relationships among stands on the basis of their composition and abundance of plant taxa, the same data matrix was subjected to an ordination by detrended correspondence analysis (DCA, using the DECORANA program [Hill 1979*b*]), which calculates eigenvectors by a reciprocal averaging procedure. The detrending of second and subsequent axes prevents a quadratic dependence on previous axes (i.e., the arch effect and involution) and reduces end-compression of the axes (Hill 1979*b*, Gauch 1982). Because DECORANA only calculates the first four axes, the eigenvalues are relative rather than absolute measures of the variation accounted for by each axis (Whittaker 1987). To index the variation accounted for by particular axes, we computed the relative eigenvalue, i.e., its magnitude divided by the sum of the eigenvalues of all four DECORANA axes (Whittaker 1987).

Environmental measurements

Soil chemistry.—To characterize the chemistry of the surface (0–5 cm) soil or organic substrate of each site, five samples of at least 250 g were collected in 1986 at random locations along the vegetation sampling lines. Each sample was analyzed as follows:

i) *Loss-on-ignition.*—A sample dried at 40°C was ignited at 430°, and the mass loss was expressed as a percentage of the initial mass.

ii) *pH.*—Water was added to a sample to create a paste, the preparation was left for 2 h, and pH was measured with a combination electrode.

iii) *Elemental concentration.*—An air-dried soil sample was sieved at 2 mm. To determine sulfur concentration, a subsample of 0.20 g was analyzed using a Leco Model SC32 Sulfur Determinator. To determine the concentrations of calcium and magnesium, a 0.20-g subsample was digested for 16 h at 100° in 3 mL of concentrated nitric acid, diluted to 50 mL with distilled-deionized water, filtered at 0.45 μm, and analyzed by inductively coupled plasma emission spectrometry (ICP). The concentrations of various other elements (Al, As, B, Ba, Be, Cd, Co, Cr, Cu, Fe, Mn, Mo, Ni, P, Pb, Sb, Se, Si, V, Zn) were also determined by this multi-elemental analytical procedure, but only Ca and Mg are reported here as their concentrations were most strongly influenced by pollution at the Smoking Hills.

Atmospheric chemistry.—Studies of atmospheric sulfur were made in 1977 at locations on two of the same transects along which we collected vegetation data in 1986 (Gizyn 1980). One of these transects ran through the largest of the plumes at the Smoking Hills, and the other was in unfumigated reference habitat. The following measurements were made.

i) *Measurement of longer term SO_2.*—Sulfation is a simple and reliable method for the estimation of atmospheric SO_2 in remote field situations. Sulfation plates contain PbO_2, which reacts with atmospheric SO_2 to form $PbSO_4$, which is analyzed turbidimetrically (Huey 1968, Thomas and Davidson 1971, American National Standards Institute 1976*a*). During the period 22 June–6 July 1977, four replicate 4.8 cm diameter sulfation plates were exposed upside-down at 2 m above the ground surface at each sampling station. After exposure, the plates were sealed airtight, transported to the laboratory, the sulfate content was extracted and analyzed by the barium chloride turbidimetric method (American National Standards Institute 1976*b*), and the sulfation data (as SO_3^{-2} were standardized to $mg/(100\ cm^2 \cdot d)$. Sulfation rate is a direct index of the concentration of SO_2; a conversion factor of 0.035 has been suggested for the calculation of time-averaged atmospheric concentrations of SO_2 (in μL/L) from field-collected sulfation measurements (Huey 1968, Liang et al. 1973, Rider et al. 1977).

ii) *Measurement of shorter term SO_2.*—To measure the concentration of atmospheric SO_2 during fumigation episodes, a limited number of short-term measurements were made in 1977 using a hand-held, battery-operated air sampling pump together with a SO_2-in-air test kit (La Motte Chemical Company). The determination of SO_2 is based on the colorimetric West-Gaeke method (West and Gaeke 1956). The sampling time was 10 min when sampling at an obviously polluted location, and 30–40 min otherwise.

iii) *Moss-bag sulfur.*—Moss bags sample gaseous and particulate sulfur in the atmosphere, and they provide an index of time-averaged pollution during field studies in remote situations. Each sampler contained ≈1.5 g dry mass of *Sphagnum* moss collected at a remote location in southern Ontario, acid-washed in 0.5 mol/L HNO_3 and rinsed in distilled water, and placed into bags made of 1.7-mm mesh fiberglass screening (No. 15 mesh). The moss bags were exposed at a height of 1.5 m aboveground during 20 June–7 July 1977, and were then collected, dried, and placed into sealed plastic bags. In the laboratory, the moss was ground, digested in hot 5:2 nitric : perchloric acid, and analyzed for sulfur by a turbidimetric procedure (American National Standards Institute 1976*b*).

iv) *Bulk-collected sulfur deposition.*—Bulk-collected deposition includes wet inputs that occur during events of precipitation, plus any dry deposition of particulates or gaseous materials that occurs during the lengthy time periods between precipitation events. Bulk deposition was collected during 22 June–6 July 1977, using duplicate polypropylene containers with a surface area of 200 cm^2. After exposure, any collected water was poured into a clean Nalgene bottle, the sampler was rinsed several times with distilled-deionized water with the rinse saved, and the total solution volume was measured. Sulfate concentration was measured turbidimetrically (American National Standards Institute 1976*b*), and SO_4^{-2} deposition was expressed as $g \cdot m^{-2}(30\ d)^{-1}$.

OBSERVATIONS

Vegetation

The cluster analysis identified five plant "communities" among the sample sites that had vegetation (Fig. 3). These five communities as well as the two types of devegetated sites are briefly described below:

Stands on mesic–xeric sites.—The character of the vegetation of mesic-xeric sites was strongly influenced by the intensity of pollution (Table 1).

i) Dryas–Salix *Reference stands (MX-REF).*—This cluster includes stands on typical, unfumigated, mesic–xeric sites on the coastal lowland and the inland plateau. The total plant cover averaged ≈33%, with 70% of this total contributed by dicotyledonous plants, 11% by monocots, and 17% by lichens. The most prominent taxa were *Dryas integrifolia, Salix arctica,* and lichens such as *Thamnolia subuliformis* and *Cornicularia divergens.* The surface soil at these sites was nonacidic (average pH 6.8) and base-rich, with a calcium concentration of 1.7% and magnesium 1.0%.

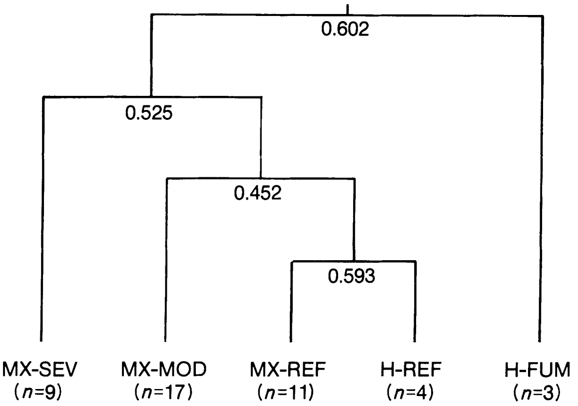

FIG. 3. Results of divisive cluster analysis by TWINSPAN. The values are the eigenvalues for the divisions. See Observations: Vegetation for community acronyms.

ii) *Mixed stands with moderate pollution damage (MX-MOD).*—This group includes stands at sites that have been somewhat affected by fumigations. The total plant cover (33%) was similar to that of the MX-REF stands, but only 28% of the relative cover was contributed by dicots, while monocots contributed 28% and lichens 41%. The plant community was mixed, in that it contained elements typical of both reference (e.g., *Salix arctica* and *Dryas integrifolia*) and severely damaged mesic–xeric sites (e.g., *Arctagrostis latifolia* and *Artemisia tilesii*). On average, the surface soil was acidic (pH 3.2), with a more impoverished base status than the MX-REF sites, i.e., calcium 0.9% and magnesium 0.6%. Some of the MX-MOD stands were visually distinct grasslands dominated by *Arctagrostis latifolia,* but these did not cluster separately from the other stands in the MX-MOD group.

iii) Artemisia tilesii *stands in areas of severe fumigation (MX-SEV).*—This group includes stands on the coastal lowland that have been severely affected by pollution. The total plant cover averaged only 7%, almost all of which was contributed by *A. tilesii,* together with a few graminoids such as *Phippsia algida* and *Arctagrostis latifolia.* The surface soil was of a similar acidity (pH 3.5) and contained concentrations of calcium (1.1%) and magnesium (0.5%) similar to the MX-MOD sites.

iv) *Devegetated sites (MX-DEV).*—Three of the mesic–xeric sites that were sampled on the coastal lowland have been so severely affected by pollution that they are devoid of plants. The surface soil at these sites was on average very acidic (pH 3.1), with a smaller con-

TABLE 1. Average percent cover of plant taxa and soil chemistry in the groups of mesic–xeric stands that resulted from the cluster analysis. Refer to Observations: Vegetation for a description of the community acronyms. The species acronyms refer to the ordination of species in Fig. 5.

Plant taxa	Species acronym	MX-REF ($n = 11$)	MX-MOD ($n = 17$)	MX-SEV ($n = 9$)	MX-DEV ($n = 3$)
A) Lichens					
Alectoria jubata	Ale jub	p*			
Alectoria nigricans	Ale nig	0.1			
Cetraria cucullata	Cet cuc	p			
Cetraria islandica	Cet isl	0.1			
Cetraria nivalis	Cet niv	0.2			
Cladonia spp.	Cla spp	0.5	3.6		
Cornicularia divergens	Cor div	0.3	0.1		
Thamnolia subuliformis	Tha sub	0.9			
Other lichens	Oth lic	3.4	10.0		
B) Bryophytes					
Aulacomnium spp.	Aul spp	p			
Pohlia spp.	Poh spp	p	0.4		
Pogonatum spp.	Pog spp	0.2	0.5		
Other bryophytes	Oth bry	0.7	p		
C) Monocotyledonae					
Alopecurus alpinus	Alo alp	p	p		
Arctagrostis latifolia	Arc lat	0.3	6.5	0.2	
Carex bigelowii	Car big	p	p		
Carex lugens	Car lug	0.2			
Carex nardina	Car nar	0.5			
Carex rupestris	Car rup	0.6			
Festuca rubra	Fes rub	0.7	p		
Hierochloe alpina	Hie alp	p	1.2		
Luzula arctica	Luz arc	0.9	1.0		
Luzula confusa	Luz con	p			
Phippsia algida	Phi alg	0.2		1.3	
Poa arctica	Poa arc	0.2	0.6	0.1	
D) Dicotyledonae					
Androsace chamaejasme	And cha	p			
Artemisia tilesii	Art til	0.1	1.8	4.8	
Astragalus alpinus	Ast alp	0.3	p		
Cassiope tetragona	Cas tet	1.2	p		
Dryas integrifolia	Dry int	12.6	1.4		
Ledum palustre	Led pal	p	0.1		
Oxytropis maydelliana	Oxy may	0.3	p		
Oxytropis nigrescens	Oxy nig	0.4	p		
Pedicularis kanei	Ped kan	p	p		
Petasites frigidus	Pet fri	p	p		
Phlox sibirica	Phl sib	0.1			
Polygonum viviparum	Pol viv	0.1	p		
Potentilla pulchella	Pot pul	0.1	0.1	p	
Ranunculus hyperboreus	Ran hyp	p			
Salix arctica	Sal arc	6.9	5.8	0.1	
Salix reticulata	Sal ret	0.1	p		
Saxifraga oppositifolia	Sax opp	p	p		
Senecio atropurpureus	Sen atr	0.1	p		
Stellaria edwardsii	Ste edw	0.2	0.1	p	
Vaccinium uliginosum	Vac uli	p	0.1		
Vaccinium vitis-idaea	Vac vit	0.4	0.1		
Total plant litter		26.8	20.5	3.2	0.0
Bare ground		40.3	46.1	90.3	100.0
Total lichens		5.5	13.7	0.0	0.0
Total bryophytes		0.9	0.9	0.0	0.0
Total monocots		3.6	9.3	1.6	0.0
Total dicots		22.9	9.5	4.9	0.0
Total plant cover		32.9	33.4	6.5	0.0
Soil					
Calcium (%)		1.7	0.92	1.1	0.45
Magnesium (%)		1.0	0.55	0.53	0.55
Sulfur (%)		0.94	1.2	1.5	0.88
Loss-on-ignition (%)		18.3	17.3	12.6	9.4
pH		6.8	3.2	3.5	3.1

* p denotes presence at <1% cover.

centration of calcium (0.45%) than the MX-MOD or MX-SEV sites.

Stands on hydric sites.—The cluster analysis identified two groups of stands on hydric sites. Also, in severely fumigated areas hydric sites were devoid of vegetation (Table 2).

i) *Reference stands on hydric sites (H-REF).*—This group included stands on unfumigated hydric sites. The total plant cover averaged 42%, with almost all of the relative cover contributed by sedges, especially *Carex aquatilis.* The surface organic layer was moderately acidic (pH 5.2), and had a relatively large concentration of calcium (0.64%), and a small concentration of sulfur (0.24%).

ii) *Stands on hydric sites with moderate fumigation damage (H-FUM).*—This group included stands on the coastal lowland that have been affected by pollution. The total plant cover of 24% was smaller than that of unfumigated H-REF wet meadows, but *Carex aquatilis* was still prominent. On average, the surface organic layer was strongly acidic (pH 3.3) and, compared with the H-REF meadows, it had a relatively small concentration of calcium (0.25%) and a large concentration of sulfur (0.94%).

iii) *Devegetated hydric sites (H-DEV).*—Several of the hydric sites that were sampled on the coastal lowland have been devegetated by pollution. The surface organic layer was strongly acidic, with a pH of 3.0, and compared with the H-REF and H-FUM meadows there were relatively small concentrations of calcium (0.17%) and magnesium (0.15%), while the concentration of sulfur (0.87%) was again large.

Ordination of stands and species

The eigenvalues for axes 1–4 of the detrended correspondence analysis were 0.843 (accounting for 48% of the variation within the four DCA axes), 0.487 (28%), 0.309 (18%), and 0.102 (6%). Because there is no obvious environmental interpretation for axes 3 and 4, they are not dealt with here. The first two axes of the ordination of stands and species are illustrated in Fig. 4. To assist the interpretation of the ordination, the stands are coded by the "communities" to which they were assigned, based on the results of the cluster analysis (Fig. 3).

The stands on mesic–xeric sites are situated on the left-hand side of axis 1, while hydric stands are situated on the right-hand side. Therefore, in terms of the species composition and relative abundance of plants, moisture status of the site appears to have a stronger influence than pollution stress in the ordination of the tundra vegetation at the Smoking Hills.

The influence of pollution is largely accounted for in the somewhat weaker axis 2. The most severely affected mesic–xeric sites with vegetation (i.e., MX-SEV) ordinate with large scores on axis 2, as do the apparently fumigation-tolerant taxa *Artemisia tilesii, Phippsia algida, Arctagrostis latifolia,* and *Poa arctica.* Reference mesic–xeric stands (i.e., MX-REF) ordinate at the bottom of axis 2, as do taxa that are characteristic of unfumigated mesic–xeric stands, such as the lichens *Alectoria jubata, Cetraria nivalis,* and *Thamnolia subuliformis,* and the vascular plants *Saxifraga oppositifolia, Carex nardina,* and *C. rupestris.* The mesic–xeric stands that have been moderately affected by pollution (i.e., MX-MOD) ordinate intermediate to these extremes on axis 2.

Hydric stands ordinate at the right-hand side of axis 1, as do the characteristic species of wet meadows such as the moss *Drepanocladus* sp., and the vascular plants *Ranunculus gmelini, Arctophila fulva, Eriophorum scheuschzeri, Carex aquatilis, C. membranacea, Dupontia fischeri,* and *Saxifraga hirculis* (Fig. 4). The pollution-affected H-FUM stands tend to ordinate with larger scores on axis 1 than do the reference H-REF wet meadows, but the separation of these types is incomplete, reflecting their overlap in species composition and relative abundance of plants.

Effects of pollution on vegetation and soil

The sulfurous fumes at the Smoking Hills have caused profound but local changes in vegetation. These effects can be directly illustrated by comparing the patterns of change of vegetation and selected soil constituents along transects through reference and fumigated terrain (Fig. 5).

Vegetation damage was most intense in the vicinity of the largest plume that was studied. There was complete devegetation (i.e., zero plant cover) of sites close to the cliff-edge along a transect of mesic–xeric sites running inland through this plume (Fig. 5A). The first occurrence of living plants was at ≈150 m from the edge of the seacliff. The only species at this site was the perennial dicot herb *Artemisia tilesii,* which had a small average cover (1.1%) and frequency (20%). This species is clearly tolerant of the stresses associated with atmospheric SO_2, and extreme soil acidity and its associated metal toxicity. At sites farther inland, total cover and species richness of the vegetation increased progressively. At 200 m two species dominated pollution-stressed sites (*A. tilesii* with 3.4% cover and the grass *Arctagrostis latifolia* with 0.2%; Fig. 5B). At 250 m these were joined by the grass *Hierochloe alpina* and the dicots *Salix arctica* and *Stellaria edwardsii* (all $<1\%$ mean cover), but most of the cover was *Arctagrostis* (3.3%) and *Artemisia* (2.5%). At 350 m the total mean cover had increased to 11.5%. The most abundant taxa were *Arctagrostis* (5.9%), *Salix* (3.0%), *Hierochloe* (2.1%), and various lichens, especially *Cladonia bellidiflora* (total cover 8.9%). The final three sample sites were located 450–650 m inland along this transect, and had a total vegetation cover similar to that of mesic–xeric reference stands (Fig. 5A). However, vegetation at these sites was dominated by a mixture of vascular species characteristic of fumigated (e.g., *Arctagrostis latifolia, Artemisia tilesii, Hierochloe alpina*) and ref-

TABLE 2. Average percent cover of plant taxa and soil chemistry in the groups of hydric stands that resulted from the cluster analysis. Refer to Observations: Vegetation for a description of the community acronyms. The species acronyms refer to the ordination of species in Fig. 5.

Plant taxa	Species acronym	H-REF ($n = 4$)	H-FUM ($n = 3$)	H-DEVEG ($n = 2$)
A) Lichens				
Cetraria cucullata	Cet cuc	p*		
Cladonia spp.	Cla spp	0.4	0.1	
Cornicularia divergens	Cor div	0.1		
Other lichens	Oth lic	1.1		
B) Bryophytes				
Aulacomnium spp.	Aul spp	0.5		
Drepanocladus spp.	Dre spp	p	3.0	
Pogonatum spp.	Pog spp	0.4	3.7	
Mnium spp.	Mni spp	0.3		
Other bryophytes	Oth bry	8.2	p	
C) Monocotyledonae				
Alopecurus alpinus	Alo alp	0.7	0.2	
Arctagrostis latifolia	Arc lat	1.7		
Carex aquatilis	Car aqu	10.5	11.4	
Carex bigelowii	Car big		2.5	
Carex lugens	Car lug	0.1		
Carex membranacea	Car mem	0.4		
Dupontia fisheri	Dup fis	2.0		
Eriophorum angustifolium	Eri ang	1.3	0.4	
Eriophorum scheuschzeri	Eri sch	p	2.9	
Eriophorum vaginatum	Eri vag	1.9		
Festuca rubra	Fes rub	p		
Hierochloe alpina	Hie alp	0.1		
Luzula arctica	Luz arc	1.2		
Luzula confusa	Luz con	0.1		
Poa arctica	Poa arc	1.9		
Trisetum spicatum	Tri spi	0.2		
D) Dicotyledonae				
Cardamine bellidifolia	Car bel	0.1		
Oxyria digyna	Oxy dig	1.1		
Pedicularis kanei	Ped kan	0.3		
Pedasites frigidus	Pet fri	1.1		
Plantago canescens	Pla can	0.1		
Polemonium boreale	Pol bor	0.2		
Polygonum viviparum	Pol viv	0.1		
Ranunculus gmelini	Ran gme	p	0.1	
Ranunculus nivalis	Ran niv	0.6		
Ranunculus pygmaeus	Ran pyg	0.3		
Rumex arcticus	Rum arc	1.0	p	
Salix arctica	Sal arc	1.8		
Salix glauca	Sal gla	0.5		
Saxifraga cernua	Sax cer	0.6		
Saxifraga hirculis	Sax hir	0.1		
Stellaria edwardsii	Ste edw	0.3		
Vaccinium vitis-idaea	Vac vit	0.8		
Total plant litter		38.5	41.2	100.0
Bare ground		19.4	34.5	0.0
Total lichens		1.6	0.1	0.0
Total bryophytes		9.4	6.7	0.0
Total monocots		22.1	17.4	0.0
Total dicots		9.0	0.1	0.0
Total plant cover		42.1	24.3	0.0
Soil				
Calcium (%)		0.64	0.25	0.17
Magnesium (%)		0.32	0.25	0.15
Sulfur (%)		0.24	0.94	0.87
Loss-on-ignition (%)		36.8	42.0	53.3
pH		5.2	3.3	3.0

* p denotes presence at <1% cover.

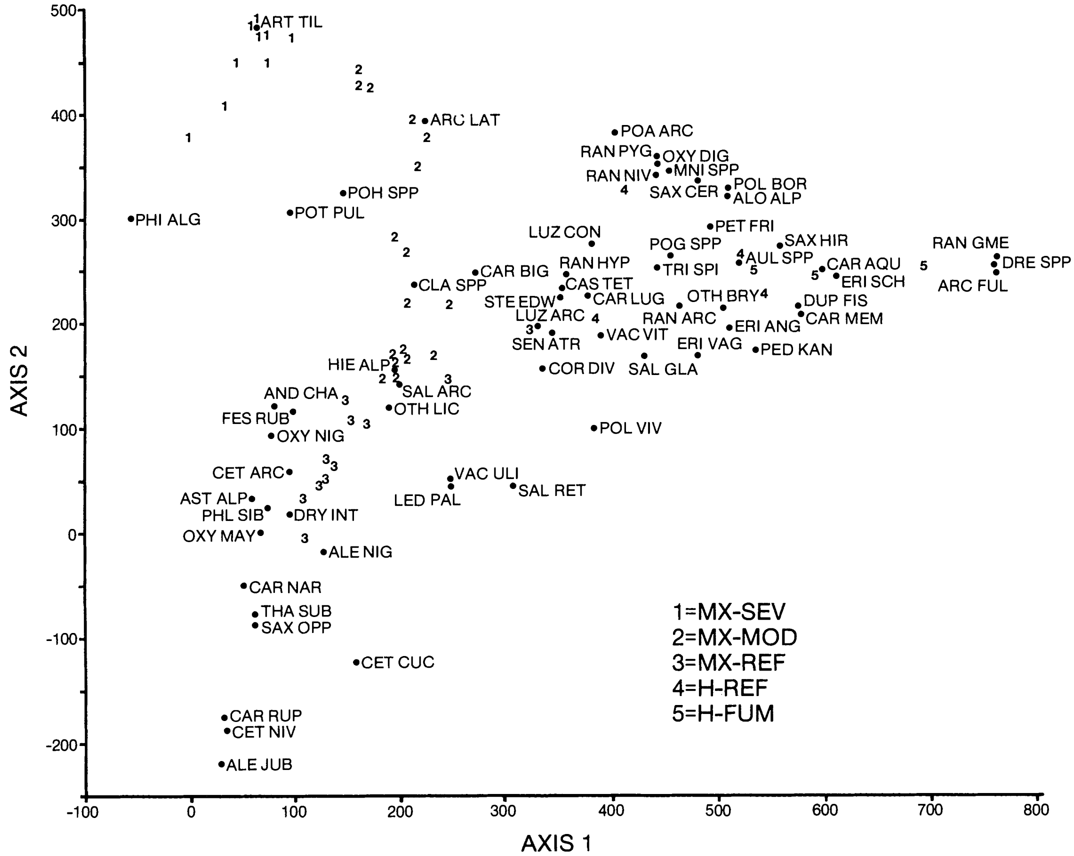

FIG. 4. Ordination of stands and plant species by detrended correspondence analysis. Only taxa with ≥0.5% average cover in at least one community are plotted. See Tables 1 and 2 for explanation of species acronyms, and Observations: Vegetation for community acronyms.

erence sites (e.g., *Dryas integrifolia, Luzula arctica, Salix glauca*), and there was a relatively large cover of crustose lichens and certain bryophytes (e.g., *Pogonatum* sp., *Pohlia* sp.). In contrast to the large changes in species along the fumigated transect, there was relatively little change in vegetation along the reference transect, and most cover was contributed by *Dryas integrifolia* and *Salix arctica* (Fig. 5C).

Surface soil chemistry did not change markedly among mesic–xeric sites along the transect through the gradient of pollution damage. At devegetated sites close to the edge of the seacliff, the soil was very acidic (pH 2.8–3.3) with small concentrations of calcium and magnesium (Fig. 5E–G). Further inland in damaged but progressively better vegetated stands, the surface soil was similarly acidic (pH 2.7–3.6) and depauperate in these base cations. In contrast, there was a large and obvious decrease in the concentration and deposition of atmospheric sulfur along this gradient of vegetation change (Fig. 5H–T). Short-term measurements of SO_2 by the West-Gaeke method during a fumigation indicated a concentration of 1–2 μL/L within ≈100 m of the cliff-edge, and rapidly decreasing concentrations farther inland. A similar pattern was shown by longer term measurements of atmospheric SO_2 by the sulfation method (Fig. 5H), which indicated a concentration of ≈0.6–0.8 μL/L within 80 m of the cliff-edge and rapidly decreasing concentrations farther away, over a 2-wk period with frequent fumigations.

There were not enough hydric sites to examine a gradient of pollution damage in the vicinity of the plumes at the Smoking Hills. However, at sites closest to the largest plume there were several devegetated wet meadows, with very acidic (pH 3.0), black peat with relatively small concentrations of calcium and magnesium, and a large concentration of sulfur (Table 2). Less damaged stands located farther inland had a smaller total cover than reference wet meadows, but were similarly dominated by the sedge *Carex aquatilis*. These fumigated but vegetated stands had highly acidic peat (average pH 3.3), with a smaller concentration of calcium and a larger concentration of sulfur than in the reference meadows (Table 2).

DISCUSSION

The "natural" combustion of pyrite-bearing bituminous shale at the Smoking Hills has caused a severe contamination of the atmosphere by sulfur dioxide and

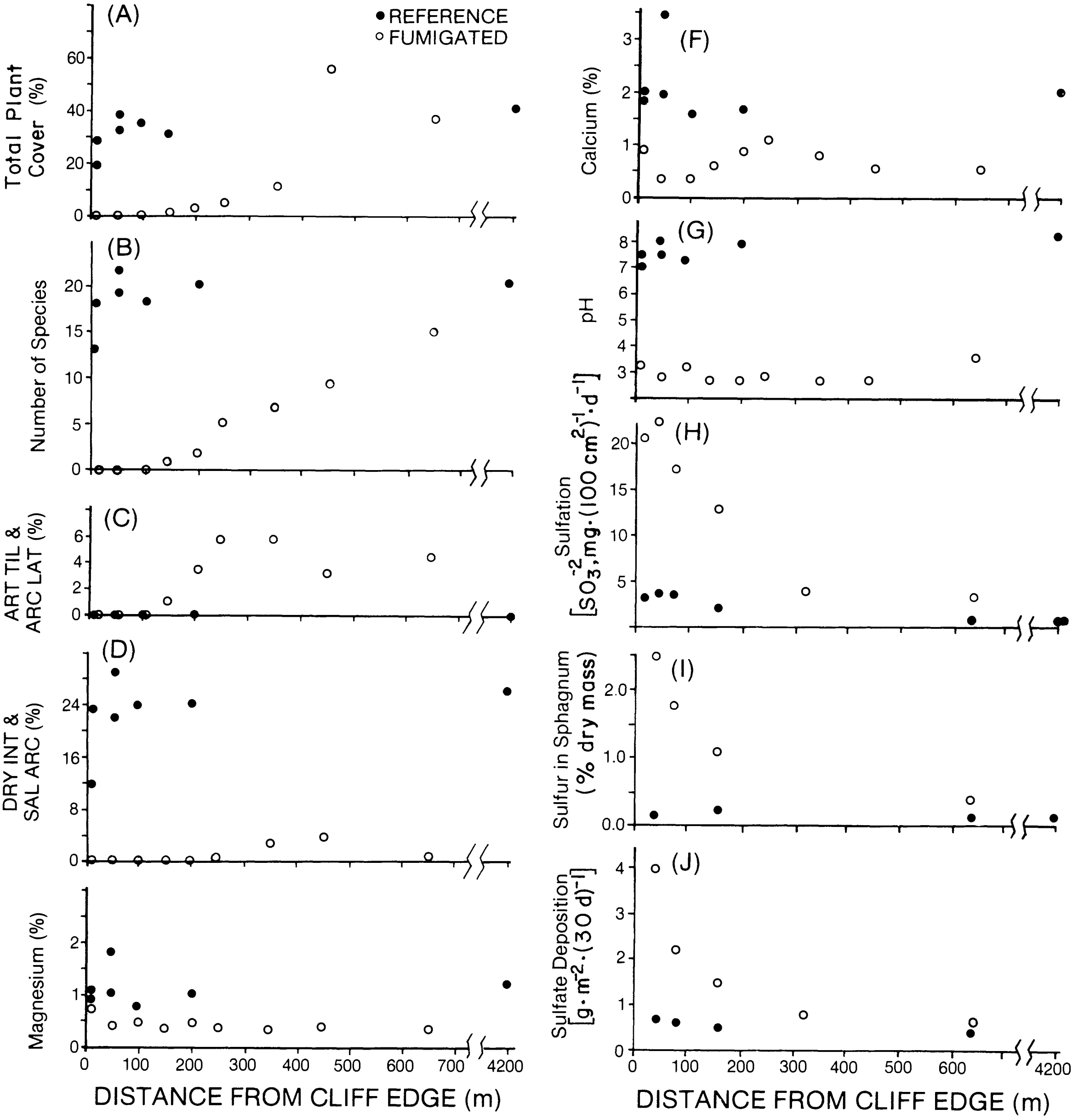

FIG. 5. Comparison of selected vegetation and environmental data along reference and fumigated transects extending inland from the seacoast through mesic–xeric habitat. (A) total plant cover; (B) species richness; (C) cover of *Artemisia tilesii* plus *Arctagrostis latifolia,* the dominant species of fumigated habitat; (D) cover of *Dryas integrifolia* plus *Salix arctica,* the dominant species of reference habitat; (E) concentration of magnesium in surface soil; (F) concentration of calcium in surface soil; (G) pH of surface soil; (H) sulfation, an index of atmospheric SO_2; (I) atmospheric sulfur accumulated by *Sphagnum* moss bags; and (J) bulk-collected sulfate deposition.

sulfuric acid mist and aerosol. The deposition of these pollutants has caused a severe acidification of soil and water. At sites where the atmospheric pollution by sulfur compounds is particularly intense, all terrestrial vegetation has been killed. As the intensity of stress associated with atmospheric sulfur decreases inland from the point of origin of the plumes, a pollution-tolerant vegetation occurs, which eventually intergrades with the more typical vegetation of reference tundra habitats.

The most important phytotoxic pollutant at the Smoking Hills is probably sulfur dioxide. During the growing season, fumigations of the tundra can be continuous for days on end, and during such events SO_2 can occur at a short-term (10 min) concentration of 1–2 μL/L and a longer term (2 wk) concentration of 0.6–

0.8 μL/L within ≈100 m of the point of origin of the largest plume. Most plants suffer acute toxicity from a short-term (i.e., 1 h) exposure of ≈0.2 μL/L, while a longer term exposure of only 0.1 μL/L can cause "hidden injury" by depressing yield without causing acute injury (IERE 1981, Roberts 1984). For hypersensitive plants the thresholds for acute and chronic injury are even lower (IERE 1981, Roberts 1984).

In addition to the regular occurrence of phytotoxic concentrations of SO_2, soils at frequently fumigated sites have been severely acidified by the depositions of SO_2 and sulfuric acid mist and aerosol from the atmosphere. Mesic–xeric sites within the fumigation zones typically have a surface soil with a pH of 2.7–3.6, while wet meadows have a pH of 3.0–3.3 in surface peat (Tables 1 and 2, Fig. 5). These compare with reference pHs in nonfumigated sites of ≈6.8 and 5.2, respectively. The very acidic pHs that occur in surface soil in the fumigation zones are comparable to the worst cases of anthropogenic acidification, such as may occur in the vicinity of coal-waste disposal areas, in acid sulfate soils, and in the vicinity of smelters such as those at Sudbury, Ontario (Gorham and Gordon 1960, Dost 1973, Freedman and Hutchinson 1980*a,* Rorison 1980).

In mineral soil, acidification causes important secondary effects, especially (i) toxicity due to the mobilization of ionic forms of aluminum, manganese, and other metals, and (ii) nutrient impoverishment and imbalance caused by the leaching of base cations under acidic conditions, particularly calcium and magnesium (Rorison 1980, Freedman and Hutchinson 1986). However, we believe that the toxic, longer term effects of soil acidification at the Smoking Hills have had a smaller influence than atmospheric SO_2 on the vegetation within the fumigation zones. Our rationale is that (i) devegetated sites occur closest to the points of origin of the plumes, where the concentrations of SO_2 are largest, while (ii) within the fumigation zones the surface soil of devegetated sites is no more acidic than that of damaged but vegetated sites. Regardless of the relative importance of particular pollutants, the concatenation of stresses associated with SO_2, acidic aerosol and mist, and severe acidification is clearly toxic to terrestrial plants at severely fumigated sites at the Smoking Hills. The zone of devegetation extends back to ≈150 m from the seacliff adjacent to the largest plume. *Artemisia tilesii* is apparently the most tolerant species of the combined pollution stresses, as it is the first plant to occur inland, where it dominates the MX-SEV "community." At somewhat less toxic sites there are additional stress-tolerant plants, most notably the grasses *Arctagrostis latifolia* and *Phippsia algida,* the bryophytes *Pogonatum* sp. and *Pohlia* sp., and the lichens *Ochrolechia frigida* and *Cladonia* spp., especially *C. bellidiflora.* These are prominent in the MX-MOD mixed community of moderately polluted sites that occurs inland of the MX-SEV type.

All of these species regularly occur in this pollution-stressed habitat, but we have also observed them as minor (i.e., infrequent) components of the vegetation of reference tundra habitat. They are often present at sites where disturbance by erosion is frequent, for example, near seacliffs and streambanks. Interestingly, a grassland dominated by *A. latifolia* in association with *A. tilesii, Pogonatum* sp., and *Cladonia* sp., was present in an unfumigated area ≈4 km inland at several sites with sandy, acidic (pH 3.5) soil beside a stream. These stands were sufficiently similar to the MX-MOD stands from the fumigation zone that they aggregated together in the cluster analysis.

Since all of the pollution-tolerant plant species at the Smoking Hills are also present in nonfumigated tundra habitat, they may be physiologically preadapted to tolerate the chemical stresses of sites in the fumigation zone. Their relatively greater abundance in communities at fumigated compared with reference sites, could be a result of reduced competition in the pollution-stressed habitat. The evolution of specific ecotypes for the fumigated areas would not be necessary in such a scenario. Grime and Hodgson (1969) described such a "catholic" distribution, with respect to tolerance of extremes of pH and aluminum toxicity, for the grass *Festuca ovina* in Britain. Similarly, McNaughton et al. (1974) described populations of *Typha latifolia* that were tolerant of elevated concentrations of several heavy metals, both in areas where soils were high in the toxic elements, as well as at uncontaminated sites. Ecotypes were not described in these studies. At the Smoking Hills, leaves of *Artemisia tilesii* and *Arctagrostis latifolia* collected from fumigated and nonfumigated sites have a similar ability to neutralize acidic droplets (Adams et al. 1984, Adams and Hutchinson 1984, T. C. Hutchinson, *personal observation*), and this surface-neutralizing ability was much greater than for species that are only present in unfumigated sites. Unfortunately, there are no data concerning the relative tolerances to gaseous SO_2 by populations of these plants collected from fumigated and nonfumigated sites at the Smoking Hills.

At the Smoking Hills, there are large changes in the species composition and relative abundance of plants along the fumigation gradients (Fig. 5). Similar patterns of vegetation change have been reported along gradients of sulfur dioxide pollution elsewhere, especially around smelters. One such example is the effect of the Sudbury nickel–copper smelter emissions on the surrounding landscape. At Sudbury, air pollution from three large metal smelters has caused severe damage to terrestrial and aquatic ecosystems. Previously forested sites have been devegetated by the intense pollution stress (Gorham and Gordon 1960, Watson and Richardson 1972, Freedman and Hutchinson 1980*a*). Soil at these devegetated sites has been rendered toxic to plants and microorganisms by severe acidification (to pH 3.0), by the concomitant solubilization of aluminum from soil minerals, and by toxic concentrations of smelter-emitted metals such as nickel and copper, which accumulate in the soil after deposition from the atmosphere (Costescu and Hutchinson 1972, Hutchinson and Whitby 1974, Freedman and Hutchinson 1980*b, c*). Close to the smelters, the toxic stresses are too severe to allow vegetation to persist, as at the Smoking Hills. At intermediate distances there is an impoverished plant community that is dominated by stress-tolerant and/or ruderal plants (Gorham and Gordon 1960, Freedman and Hutchinson 1980*a*). Such a pattern is equivalent to the *Artemisia tilesii* and *Arctagrostis latifolia*-dominated areas at the Smoking Hills. At greater distances from the Sudbury point sources there is a progressive increase in the biomass, species richness, and diversity of terrestrial plant communities, until at distances greater than $\approx$30 km a typical coniferous–angiosperm forest occurs. The spatial compression of the pattern of zonation at the Smoking Hills is probably a function of the ground-level emissions of relatively smaller quantities of SO_2, in comparison with the much larger releases from tall smokestacks at Sudbury.

Other examples where a similar pattern of ecological zonation has been caused by emissions of pollutants from a point source include a lead–zinc smelter at Trail, British Columbia, Canada (Katz 1939), an iron-sintering plant at Wawa, Ontario, Canada (Gordon and Gorham 1963, Scale 1982), a copper smelter at Ducktown, Tennessee, USA (Smith 1981), and a copper smelter near Superior, Arizona, USA (Wood and Nash 1976). In all of these cases, SO_2 and particulate deposition combined to damage vegetation in well-defined concentric zones, and caused the accumulation and solubilization of toxic elements in soil.

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