ELSEVIER

Ecological Indicators 6 (2006) 369–387

ECOLOGICAL INDICATORS

This article is also available online at:
www.elsevier.com/locate/ecolind

An assessment of pollution impacts due to the oil and gas industries in the Pechora basin, north-eastern European Russia

Tony R. Walker [a,*], Peter D. Crittenden [a], Scott D. Young [a], Tatyana Prystina [b]

[a] *School of Biology, The University of Nottingham, University Park, Nottingham NG7 2RD, UK*
[b] *Institute of Biology, Komi Science Centre, Russian Academy of Sciences, Syktyvkar, Komi Republic, Russian Federation*

Accepted 29 March 2005

Abstract

The chemical composition of terricolous lichens, top-soil and abundance and diversity of lichen communities were assessed at eight sites in the Pechora basin during the summer of 2000 and 2001 to assess local impacts of oil and gas operations. Sites close to industrial areas were compared with areas considered to be pristine. The broad objective of the study was to identify changes in the chemistry of a suite of lichens that could be caused by pollutants from the industrial sites. In particular, increases in the ratios of K:Mg and K:Ca + Mg are indicators of acid deposition and increased N indicates increased N deposition. Lead was also measured in lichens. Other elements measured were Ba, Ca, Cd, Cu, K, Mg, Na, Ni, Pb, Sr and Zn in top-soils. The Pechora basin, north-eastern European Russia, includes the north and east of the Komi Republic and a major portion of the Nenets Autonomous region. It is bounded by the Ural mountains to the east and by the Timan range to the west. The area has extensive natural resources, both renewable (e.g. forests) and non-renewable (e.g. minerals, coal, oil and gas). There were limited modifications in the chemistry of lichens and top-soils and lichen diversity close to an oil and gas industrial complex. Here, there were elevations of lead and nitrogen concentrations in lichen apices and in the apical:basal nitrogen ratio in *Flavocetraria cucullata*, with lower lichen diversity in epigeal and epiphytic lichens. Elevated concentrations of Ba and Ca were found in soil-ash over the 0–5 cm horizon, probably as a result of local emissions from construction activity and gas flaring, rather than from long-range transport. Virtually all other sites remained unmodified and reflected background concentrations. The ecological impacts of the measured pollution loads were low, as elemental concentrations were generally below detection limits, except for one industrial site, where there were signs of an early indicator of industrial activity.

Keywords: Russian Arctic; Pechora basin; Lichen and soil chemistry; Lichen biodiversity; Bio-indicator; Oil and gas industries

1. Introduction

The Russian Arctic is a fragile environment and is generally regarded as a vast sparsely populated wilderness (Rovinsky et al., 1995). These environments are susceptible to man-induced stresses, such as

---

* Corresponding author at: Department of Oceanography, Dalhousie University, Halifax, Nova Scotia, Canada B3H 4J1.
Tel.: +1 902 494 2759; fax: +1 902 494 3877.
E-mail address: trwalker@dal.ca (T.R. Walker).

1470-160X/$ – see front matter © 2005 Elsevier Ltd. All rights reserved.
doi:10.1016/j.ecolind.2005.03.015

pollution and climate change (Press et al., 1998). Despite its daunting size, some regions are beginning to display signs of degradation, such as reduced plant biodiversity and increases in heavy metal contamination of soil, rivers and precipitation, as a consequence of resource-exploitation, e.g. the mining and metallurgical industries in Norilsk and Kola Peninsula (Reimann et al., 1999). These areas are notorious sources of acid and metal emissions (Tuovinen et al., 1993).

Until recently, terrestrial pollution has received little attention within Russia and environmental data are limited (Ryaboshapko et al., 1998). Nonetheless, there is evidence of major environmental problems, some of which are in the vicinity of Russia's western borders, e.g. the impact of air pollution around the metallurgical complexes on the Kola Peninsula (Tuovinen et al., 1993). Several authors have classified the region as severely damaged and certain areas have been described as 'industrial deserts' (e.g. Reimann et al., 2000). Historically, exploitation of coal has been a major element of industrial activity in the Russian Arctic (Revich, 1995), but more recently, gas and oil recovery has shown signs of expansion (Ziegler, 1987; IUCN, 1993; Locatelli, 1999), e.g. the gas development on the Yamal Peninsula (Klein, 2000), and oil and gas exploration of the Timan-Pechora province (Lausala and Valkonen, 1999). According to Pelley (2001) the Timan-Pechora province is the third most important oil-producing region in Russia and is reputed to contain some of the World's richest deposits. Industrialization at high latitudes has resulted in the development of many large Arctic towns, increasing the risk of environmental pollution (Pryde, 1991; Alexeeva-Popova et al., 1995). Russia is the single largest contributor to SO_2 and NO_x emissions in Europe (Ryaboshapko et al., 1998). Impacts of acid pollution have been seen close to emission sources in industrialized regions where there have been many documented examples of environmental damage due to acidification and eutrophication (Jónsdóttir et al., 1995; Virtanen et al., 2002). However, it should be emphasised that vast areas of the Russian Arctic appear close to pristine (Rovinsky et al., 1995).

The Pechora river basin has abundant natural resources, both renewable (e.g. timber) and non-renewable (e.g. minerals, coal, oil and gas) (Lausala and Valkonen, 1999). Coal combustion for electric power generation has historically been the principal source of SO_2 and heavy metal pollution in the Pechora basin (Vinogradova, 2000; Solovieva et al., 2002), but other sources include gas and oil extraction (State of the Environment of the Komi Republic, 1992–1998). Exploitation of Pechora coal is in decline due to its poor quality and associated high transportation costs (Lausala and Valkonen, 1999). Recently, however, the oil and gas industries have boomed and are expected to expand further, bringing about significant risks of environmental pollution, e.g. from gas flaring and oil spills (Vilcheck and Tishkov, 1997).

Changes in the Russian Arctic environment, due to pollution, not only have local impacts, but also have consequences for global climate change (Krankina et al., 1997). The Pechora region may be affected by global warming, as general circulation models predict temperature increases will be pronounced at high latitudes (IPCC, 2001). Briffa et al. (1995) has shown that the northern Urals has experienced a pronounced warming trend during the last 100 years. Warming accelerates permafrost collapse, causing rupture of oil pipelines, thereby increasing the risk of environmental pollution (Vilcheck and Tishkov, 1997). Pelley (2001) estimates there are two major oil and gas pipeline spills every day, equating to 15–20 million tonnes of oil lost through spills annually in Russia.

Until now there have been few detailed studies of pollution loads in the Pechora region (Rusanova, 1995), which like the rest of the Russian Arctic is vast and largely pristine but with some notable pollution hotspots. The rapid development of gas and oil industries has potential to increase pollution in the Russian Arctic and the Pechora river basin is one such region where this rapid exploitation is taking place. The objective of our project was to see whether pollution impacts are already detectable and to seek evidence of environmental impacts in the vicinity of a booming petrochemical industry in the Pechora region. This data provides a baseline against which the extent of pollution in the future can be gauged. This investigation of potential terrestrial pollution complimented work carried out by other groups within the region. This included studies on aquatic pollution based on analysis of surface waters and lake sediments (Solovieva et al., 2002). This was achieved by quantifying the chemical status of terricolous mat-forming lichens and top-soil, and by measuring alpha (α) biodiversity of epigeal and epiphytic lichens in close proximity to both putative pollution 'hot spots' and unpolluted 'reference' sites with broadly comparable community structures.

1.1. Analysis of accumulators

Problems exist when monitoring pollution deposition in remote regions as air and precipitation chemistries show high temporal and spatial variation (Reimann et al., 1999). Elemental concentrations are usually low and sample contamination is a significant problem. Alternative approaches include: (i) analysis of ecological materials that accumulate contaminants (e.g. soils and lichens); and (ii) use of bio-indicators, the distribution of which become modified by exposure to pollution.

Mat-forming terricolous lichen communities are ecologically important at high latitudes (Ahti and Oksanen, 1990), where they contribute to primary production and nutrient cycling (Chapin and Bledsoe, 1992). Lichens are primarily dependant on atmospheric sources for key nutrients such as N and P (Crittenden, 1998; Ellis et al., 2003) and also readily accumulate atmospheric contaminants, such as metals (Nash and Gries, 1995). The use of lichens as indicators of air pollution due to the accumulation of metals has been extensively reviewed by Conti and Cecchettib (2001). The high tolerance of lichens to most metal contaminants and their slow growth are among the main factors that make them effective accumulators (Nash, 1989). Accordingly, spatial variation in the chemical composition of lichens has been frequently used to map atmospheric deposits (Nimis et al., 1990; Pakarinen et al., 1983; Gonzalez and Pignata, 1997; Riget et al., 2000).

The principal mat-forming lichen species found in lichen-dominated ground cover in the Pechora region are members of the genera *Cladonia* (sub-genus *Cladina*), *Cetraria*, *Flavocetraria*, *Alectoria* and *Stereocaulon*. Elevated concentrations of ions in polluted precipitation, therefore, has an impact on lichen chemical composition (Hyvärinen and Crittenden, 1998a). Nitrogen pollution is generally positively correlated with acid loads because nitrate is a component of acid deposition. The N content of lichen thalli is related to N deposition and has been used in pollution monitoring (Søchting, 1995). Hyvärinen and Crittenden (1998b) argued that mat-forming lichens are particularly good bio-indicators of air pollutant loads because they typically occur in open situations intercepting rainfall directly and largely unmodified by vascular plant canopies, which can confound relationships between the chemistry of epiphytic lichens and the atmosphere (Farmer et al., 1991). They are relatively independent of the chemical influence of the substratum owing to accumulation of basal necromass which tends to isolate the living parts of the thalli from underlying soil (Ellis et al., 2003). Hyvärinen and Crittenden (1996) also examined relationships between rainfall acidity and lichen chemistry and demonstrated that K^+:Mg^{2+} ratio in apices of *C. portentosa* varied among heathland sites in the British Isles. This variation was highly correlated with H^+ concentration in rainfall, therefore this ratio might provide a sensitive chemical marker for rainfall acidity in pollution surveys.

Analysis of the underlying soil can be used to evaluate long-term accumulation of conserved pollutants acting as a major sink for atmospheric deposits (Boyd et al., 1997). In particular, the topmost few centimetres of the soil profile becomes a sensitive accumulator in the terrestrial environment (Niskavaara et al., 1997). In highly polluted areas near industrial centres, physical and chemical properties of top-soils can become radically modified. Soil contamination due to alkaline emissions from coal combustion has been widely reported (Larssen and Carmichael, 2000; Walker et al., 2003b). Two approaches have been used to sample heavy metal contamination in soils. Either the uppermost, organic horizon is sampled, or soil is removed to a standard depth (Reimann et al., 1997). Depth-defined samples can be taken anywhere, independently of the soil type or depth of the organic layer. Analysis of soil samples allows for assessment of aerial deposition by comparison of concentrations per unit area, provided sampling is carried out to sufficient depth to audit all the analyte of interest. Numerous investigations have used variation in soil chemistry to map pollution deposition, particularly in the case of heavy metals (Reimann et al., 1997; Haapala et al., 2001).

1.2. Impacts on communities—lichen biodiversity

Due to their susceptibility to SO_2 and other phytotoxic air pollutants, epiphytic lichens have been used for decades to monitor air pollution (Hawksworth and Rose, 1970). In boreal forests, localised decline in abundance and diversity of the epiphytic lichens, *Alectoria*, *Usnea* and *Bryoria* has been attributed to increasing air pollution and acid rain (Thor, 1998). Diversity is measured as species richness or the number of species in an area and varies with spatial scale and alpha diversity is defined as, ‘the number of species within a small area that is relatively uniform’ (Wilson, 1988). Methodologies for assessing lichen diversity and abundance have been reviewed by Will-Wolf et al. (2002).

2. Methods

2.1. Study area

These ecological assessments were undertaken at eight sampling sites, four of which were selected close to industrial ‘hot spots’ (‘industrial’ sites, $F1_i$, $F5_i$, $F3_i$ and $F7_i$), and four were remote from industrial activity (‘reference’ sites, $F2_r$, $F6_r$, $F4_r$ and $F8_r$) (Fig. 1, Table 1).

The region is unique in continental Europe with extensive lowland tundra and permafrost in the North and the largest continuous area of ‘old growth’ taiga (boreal forest) to the south in the Urals. There are a variety of human impacts in the region with relatively uninhabited ‘pristine’ areas and a few densely populated regions.

Sampling sites were chosen to quantify environmental impacts in close proximity to perceived pollution ‘hot spots’ and in unpolluted reference sites of broadly comparable community structure. Across the Pechora basin, ‘industrial’ sites (e.g. $F1_i$) and unpolluted ‘reference’ sites (e.g. $F2_r$), comprised of the following: Ukhta area ($F1_i$), Belaya Kedva river ($F2_r$), Ortina river ($F3_i$), Neruta river ($F4_r$), Svetly Vuktyl ($F5_i$), Maly Patok ($F6_r$), Upper Kolva ($F7_i$) and Moreyu river ($F8_r$) (Fig. 1). Details of industrial developments and pristine sites are given in Table 1.

Field-work was carried out during spring in 2000 ($F3_i$ and $F4_r$) and 2001 ($F1_i$, $F2_r$, $F5_i$, $F6_r$, $F7_i$ and $F8_r$). Helicopters, small boats and overland tracked vehicles were used to reach remote field sites and sub-sites were reached on foot. Sites close to towns were accessed by car, although sub-sites were again reached on foot and chosen some distance away from roads.

2.2. Lichen sampling

At least one species of terricolous mat-forming lichen of the genera *Cladonia* [*C. stellaris* (Opiz) Pouzar and Vezda, or *C. arbuscula* (Wallr.) Flot.] or *Flavocetraria* [*F. cucullata* (Bellardi) Kärnefelt and Thell] was collected, subject to availability. They were chosen to provide biomarkers for atmospheric deposition and because of their ubiquity and abundance in forest tundra. At each site three sub-sites were selected, 500–1000 m apart, at which six replicate samples of lichen material were collected at distances 10–20 m apart in open areas subject to minimal tree canopy effects; these were usually inter-tree positions in open forest, or in open tundra. Lichen samples were air-dried, sealed in LDPE containers and stored at 4 °C until analysis. Powder-free LDPE gloves were worn when handling lichens both in the field and in the laboratory to minimise contamination.

2.3. Soil

Soil samples were collected from the same sub-sites as lichen samples. Samples were collected 5 m from the base of trees, in open areas subject to minimal tree canopy effects. Collection involved excavating organic and mineral top-soil from a 20 cm × 20 cm plastic quadrat with a stainless steel hand trowel, which was cleaned between samples to avoid cross contamination. Samples were passed through a 4-mm stainless steel sieve in the field, double-wrapped in polythene bags and, were stored at −20 °C prior to analysis. The thickness of the surface organic and litter layers were recorded for all quadrats. Due to variation in the depth of the organic layer (O-horizon) it was decided to sample to a standard depth of 5 cm following Niskavaara et al. (1997). Powder-free polythene gloves were used at all times both in the field and in the laboratory to minimise the risk of contamination.

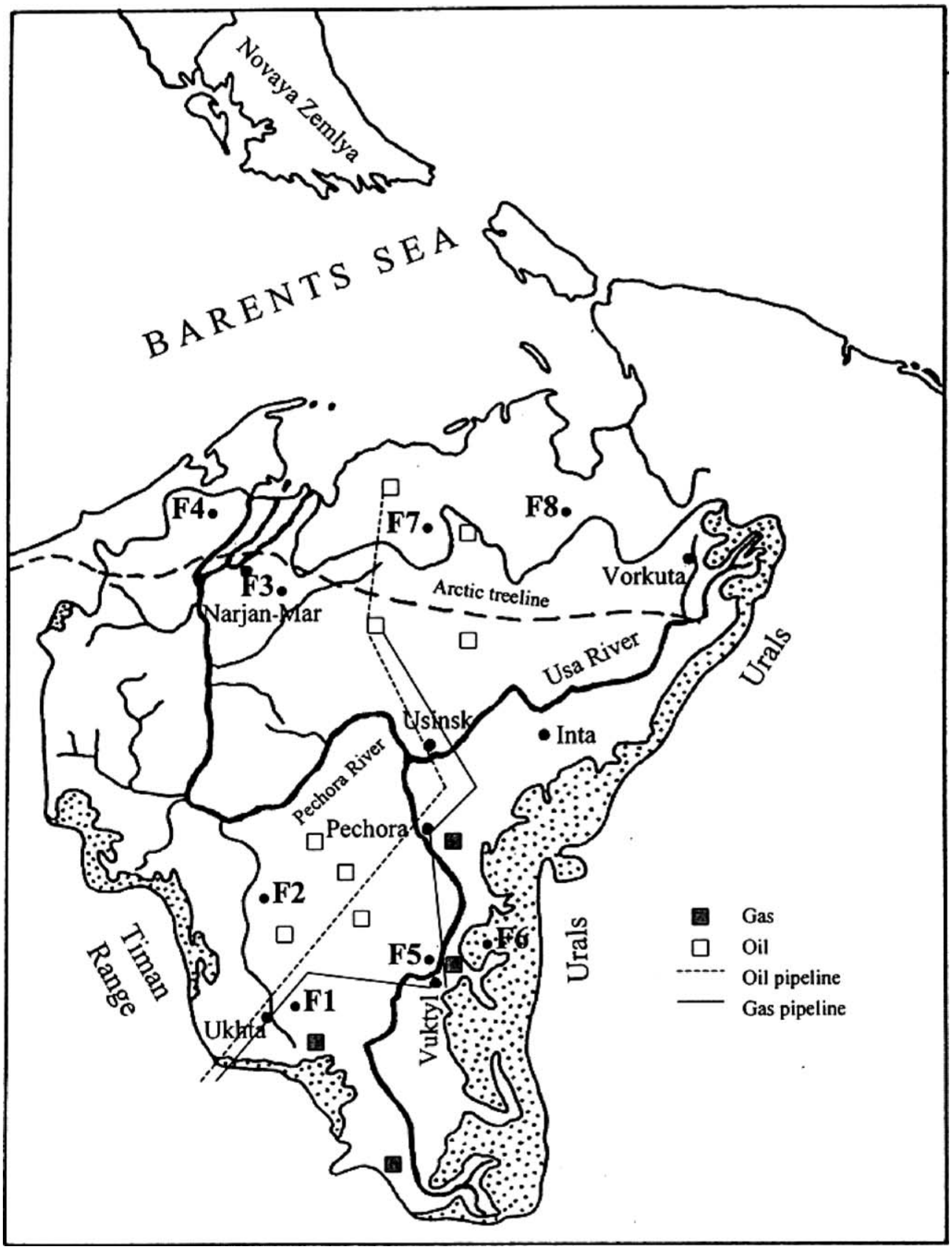

Fig. 1. Sampling sites in the Pechora basin in north-eastern European Russia, with major towns, industrial areas and natural ecotones.

2.4. Lichen biodiversity determinations

Lichen abundance and species diversity was determined on epigeals in the tundra and epiphytes in the taiga. The purpose of this lichen biodiversity study was to determine differences in lichen flora and percentage cover on trunks of *Picea obovata* and in sample plots in the tundra to determine the extent to which lichens serve as indicators of air purity.

Table 1
Summary of the main sampling sites within the Pechora region

Site no.	Name of site	Co-ordinates	Type of industry and status of site
$F1_i$	Izhma river (near Suz'u river mouth), Ukhta area	63°44′10″N, 53°42′57″E	Oil and gas recovery, forestry, fragmented lowland taiga
$F2_r$	Belaya Kedva river	64°19′27″N, 53°03′39′E	Pristine lowland taiga
$F3_i$	Ortina river, Narjan-Mar area	67°55′59″N, 54°02′33″E	Oil and gas recovery, tundra delta area
$F4_r$	Neruta river, Malozemelsk tundra	68°00′10″N, 52°24′16″E	Reindeer herding, pristine tundra, partly protected, coastal
$F5_i$	Svetly Vuktyl river, Vuktyl area	63°47′42″N, 57°32′36″E	Gas industry, fragmented lowland taiga
$F6_r$	Maly Patok river	64°18′54″N, 59°04′40″E	Pristine taiga in Ural foothills, protected Yugyd Va national park
$F7_i$	Kolva river (near the Kharayaha river mouth)	67°08′22″N, 56°41′41″E	Large oil industrial complex, tundra
$F8_r$	The Moreyu river (near the Syamayu river mouth)	67°52′51″N, 59°43′21″E	Reindeer herding, pristine tundra, partly protected, coastal

2.4.1. Taiga sampling sites

Eight to nine mature spruce trees (*P. obovata*) were selected based on accessibility of trunks and representative nature of the stand from each forested study site. Trees were selected ≥100 m apart. An estimate of abundance and cover of each lichen species was made on trunks and branches, up to a height of 1.7 m. Estimates were made according to the scale used by Kauppi and Halonen (1992): 7 = >50%; 6 = 26–50%; 5 = 11–25%; 4 = 3–10%; 3 = poor cover; <3%; 2 = little, many specimens, but not constituting any real cover; 1 = extremely little, only one or two specimens per trunk. After some training this scale proved easy and rapid to use. All the macrolichen and crustose lichen species were identified in the field according to Dobson (1979), Moberg and Holmåsen (1982), Goward et al. (1994), McCune and Geiser (1997). Where this was not possible, unidentified specimens were collected and returned to the Institute of Biology, Komi Science Centre, Syktyvkar for detailed examination and identifications made by Tatyana Prystina and Olga Lavrinenko.

2.4.2. Tundra sampling sites

Five quadrats (2 m × 2 m), separated by >100 m, were placed subjectively in lichen rich sub-sites which were dominated by dwarf birch (*Betula nana* L.) at each of the tundra study sites. The number and percentage cover of epiphytic and epigeal lichen species were again recorded on Kauppi and Halonen's (1992) 1–7 point scale.

2.5. Chemical analysis of lichens

The variables measured in lichens were: total N, determined by Kjeldahl digestion and distillation; major cations: Mg^{2+} by flame-atomic absorption spectrophotometry (F-AAS); Ca^{2+} and K^+ by flame emission spectrophotometry (FES) and Pb by graphite furnace atomic absorption spectrophotometry (GF-AAS). Lichens were rehydrated overnight by exposure to water-saturated air, over de-ionized (DI) water in a dessicator at 4 °C, then, fully saturated by spraying twice with DI water. The rehydrated material was cleaned of extraneous debris using forceps. Thalli were then cut horizontally at 5, 40 and 50 mm from the apex using a razor blade. The horizontal strata 0–5 mm (apices) and 40–50 mm (thallus base) were retained for analysis and oven dried at 80 °C overnight. Analysis performed on each species is given in Table 2.

2.5.1. Total [Ca], [K], [Mg], [Zn] and [Pb] in lichen tissue

Approximately 100 mg of dried apical tissue was digested to dryness in 1 mL of concentrated HNO_3 (Aristar, Fisher Chemicals) at 175 °C in digestion tubes. The residue was dissolved in 10 mL 1 M HNO_3 and appropriate quantities of ionisation suppressant and releasing agent ($CsCl_2$, $LaC1_2$) added. Concentrations of $[Pb]_{lichen}$ were determined on the top 5 mm of apical thalli for *C. stellaris* and *C. arbuscula* and *F. cucullata*. Magnesium and Pb were determined by GF-AAS and Ca and K by FES.

Table 2
Terricolous species of *Cladonia* and *Flavocetraria* sampled across the Pechora basin at 'industrial' sites (e.g. F1$_i$) and unpolluted 'reference' sites (e.g. F2$_r$), together with elemental analyses performed ($n = 18$)

Sites	Species sampled	$[N]_{apices}$	$[N]_{base}$	[Pb]	$[Ca^{2+}]$	$[K^{+}]$	$[Mg^{2+}]$
F1$_i$	*Cladonia arbuscula*	+	−	+	+	+	+
Izhma river	*C. stellaris*	+	−	+	+	+	+
F2$_r$	*C. arbuscula*	+	−	+	+	+	+
Kedva river	*C. stellaris*	+	−	+	+	+	+
F3$_i$	*C. arbuscula*	+	−	−	+	+	+
Ortina river	*F. cucullata*	+	+	−	+	+	+
F4$_r$	*C. arbuscula*	+	−	−	+	+	+
Neruta river	*F. cucullata*	+	+	−	+	+	+
F5$_i$	*C. arbuscula*	+	−	+	+	+	+
Svetly Vuktyl							
F6$_r$	*C. arbuscula*	+	−	+	+	+	+
Malay Patok	*C. stellaris*	+	−	+	+	+	+
F7$_i$	*C. arbuscula*	+	−	+	+	+	+
Upper Kolva	*F. cucullata*	+	+	+	+	+	+
F8$_r$	*C. arbuscula*	+	−	+	+	+	+
Mareyu river	*F. cucullata*	+	+	+	+	+	+

(+) Analysis performed; (−), no analysis.

2.5.2. Total nitrogen concentration ($[N]_{lichen}$) in lichen thalli

Total N concentration was measured in both apical (0–5 mm) and basal (40–50 mm) tissue. Total $[N]_{lichen}$ was determined following the Kjeldahl digestion and distillation method of Bremmer and Breitenbeck (1983).

2.6. Soil analysis

Frozen soil samples were allowed to thaw overnight at room temperature and a representative sub-sample (5–10 g) was oven dried (105 °C) in acid washed silica dishes. Samples were ashed in a muffle furnace at 450 °C overnight, allowed to cool in a dessicator and weighed for loss on ignition (LOI). A sub-sample (1 g) of the soil-ash was digested in 10 mL of concentrated HNO_3 (Aristar, Fisher Chemicals). The digest residue (~1 mL) was diluted and filtered through Whatman no. 42 ashless filters then made up to 100 mL using ultra-pure water to give a final matrix of 1% HNO_3. Analysis of Ba, Ca, Cd, Cu, K, Mg, Na, Ni, Pb, Sr and Zn was undertaken using atomic absorption spectrophotometry (GF-AAS and F-AAS). Soil pH was measured on moist soil samples, suspended in DI water at a solid:solution ratio of approximately 1:2.5 following equilibration for 1 h. Drift in pH was recorded to determine when equilibrium was reached.

2.7. Statistical analysis

Unless otherwise indicated, significant differences between sites were determined by one-way ANOVA followed by Tukey's test ($n = 18$).

3. Results

3.1. Lichen material collected

Terricolous mat-forming lichens in the genera *Cladonia* [*Cladonia arbuscula* (Wallr.) Flot. or *C. stellaris* (Optiz) Pouzar and Vezda] or *Flavocetraria* [*F. cucullata* (Bellardi) Kärnefelt and Thell] were widespread and locally abundant in the Pechora basin. At least one species was collected at each site (Table 2). Cover was generally poor in the tundra due to grazing and trampling by reindeer, except for site F7$_i$. The most complete data set for lichen chemistry was obtained for *C. arbuscula*, which was collected at all sites in this study. However, only small thalli were found in the tundra, due to heavy grazing and it was only possible to measure [N] in the apical 5 mm. In the case of *Flavocetraria cucullata*, thalli were of sufficient length to measure [N] at different strata.

3.2. Total [Ca^{2+}], [K^{+}], [Mg^{2+}] and [Pb] in lichen tissue

Fig. 2 shows variation in the concentrations and concentration ratios of cations in the apices of *C. arbuscula*, *C. stellaris* and *F. cucullata*. There were some marked and significant differences between comparable 'industrial' and 'reference' sites, e.g. site F7_i compared to site F8_r in both *C. arbuscula* and *F. cucullata*, and F5_i compared to F6_r in *C. arbuscula*. However, there was no consistent relationship between differences in the cation values and proximity to perceived pollution sources. There was however, strong covariation in lichen chemistry among the three species (Fig. 3a–e), suggesting that differences among sites were due to different environmental circumstances as opposed to random variation within the data. Overall, potassium concentrations were generally higher in *F. cucullata* than *C. arbuscula* at all tundra sites.

Lead was selected as an additional analyte because it was one of several trace metals found to contaminate snow and soils locally in the tundra around the Vorkuta industrial complex (Walker et al., 2003b). Whilst absolute concentrations are low in all species there may be localised elevation of $[Pb]_{apices}$ in *C. arbuscula* and *F. cucullata* at F7_i (0.098 ± 0.008 and 0.143 ± 0.015 μmol g^{-1}, respectively) where concentrations were three to four times greater than at F8_r (0.025 ± 0.004 and 0.04 ± 0.005 μmol g^{-1} for *C. arbuscula* and *F. cucullata*, respectively) (Fig. 4). There was a strong relationship between $[Pb]_{apices}$ in *C. arbuscula* and concentrations of Pb in soil-ash ($[Pb]_{soil\text{-}ash}$) ($r^2 = 0.834$, $n = 6$).

3.3. Total nitrogen concentration in lichen thalli

Values of $[N]_{apices}$ in *Cladonia arbuscula* varied from 0.39 ± 0.03 mmol g^{-1} ($n = 18$) at F3_i in the tundra, to 0.60 ± 0.07 mmol g^{-1} ($n = 18$) at F5_i in the taiga (Fig. 5a). Values were significantly higher at taiga than at tundra sites ($P < 0.001$). There was little evidence of elevated $[N]_{apices}$ in *C. arbuscula* at 'industrial' sites with the exception of F7_i which had the highest values for the tundra sites and significantly greater than the value for F8_r. Nitrogen concentration in the basal stratum of *C. arbuscula* was not measured because thalli were generally of insufficient length. Total N in the apices of *C. stellaris* was measured at three taiga sites (Fig. 5b) and found to be similar at each. Values ranged from 0.62 mmol g^{-1} (±0.02) at F2_r to 0.65 mmol g^{-1} (±0.01) at F6_r.

Values of total N in the apical (0–5 mm) and basal strata (35–40 mm) of *F. cucullata* are shown in Fig. 6a. While the highest value of $[N]_{apices}$ was recorded at F7_i, the lowest value was also recorded at an industrial site (F3_i). There were significant differences in $[N]_{apices}$ and $[N]_{base}$ in *F. cucullata* between 'industrial' and 'reference' sites, but these did not provide any coherent evidence of pollution. Again the highest value of $[N]_{base}$ was recorded at F4_r whilst the lowest value was recorded at F7_i. There was a moderate degree of covariance between *F. cucullata* and *C. arbuscula* in $[N]_{apices}$ values ($r^2 = 0.692$, $n = 4$). The ratio of $[N]_{apices}$:$[N]_{base}$ in *F. cucullata* was significantly higher at F7_i than at the other tundra sites (Fig. 6b).

3.4. Soil analysis

The depth of the surface organic layer in sample quadrats varied between 0.5 and >5 cm. Top-soil pH (0–5 cm) at all sites was uniformly low, with mean values ranging from 4.03 (±0.02), at F7_i (Kolva) to 4.88 (±0.11) at F2_r (Kedva) (Fig. 7). Generally, top-soils at tundra sites had lower pH values than those at taiga sites. The lowest recorded mean top-soil pH (0–5 cm) was 4.03 (±0.02), at F7_i; this value was significantly lower than the value of 4.48 (±0.09) at F8_r (one-way ANOVA, followed by Tukey's test, $P < 0.001$, $n = 18$).

Concentrations of metal elements in soil-ash were generally low at all sites in the values for several elements (Ba, Ca, K, Mg, Na, Pb and Zn) varied markedly between sites. However, only Ba, Ca and possibly Ni, were elevated at an industrial site: concentrations of these elements were significantly greater at F7_i than at F8_r (one-way ANOVA, followed by Tukey's test; $P < 0.001$, $n = 18$). Mean concentrations of Ba, Ca and Ni in soil-ash were 459 (±93), 4225 (±1074) and 43 μg g^{-1} (±3.1), respectively, at F7_i, whereas, background concentrations for the 'reference' F8_r were 41 (±4.6), 138 (±16) and 21 μg g^{-1} (±3.3), respectively.

Concentrations of Ba and Ca in soil-ash, were highly positively correlated ($r = 0.997$, $P < 0.001$, $n = 8$), whereas concentrations of Cu and Pb were only

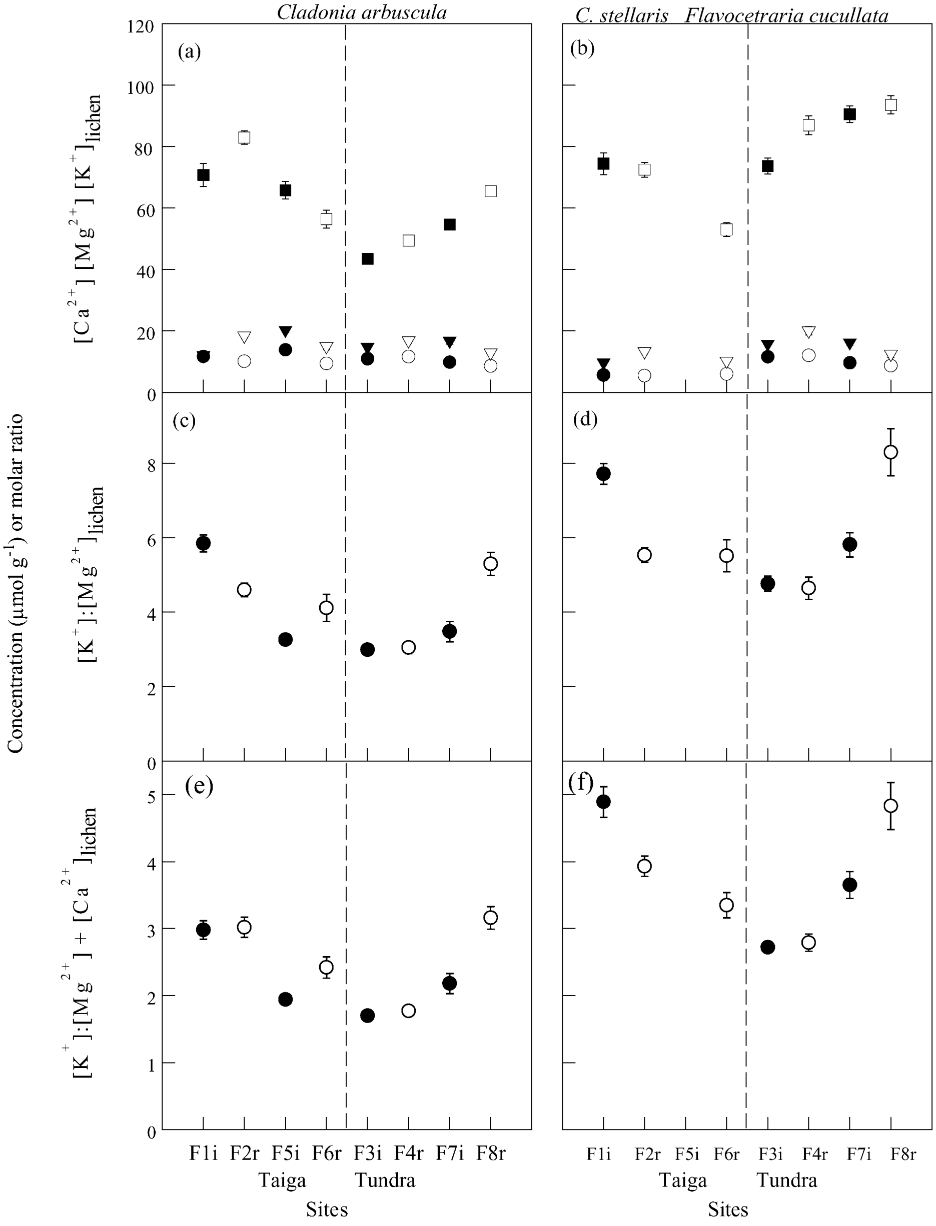

Fig. 2. Variation in $[Ca^{2+}]_{apices}$ (●, ○), $[Mg^{2+}]_{apices}$ (▼, ▽) and $[K^{+}]_{apices}$ (■, □) (a, b) and the ratios $[K^{+}]:[Mg^{2+}]_{apices}$ (c, d) and $[K^{+}]:([Ca^{2+}] + [Mg^{2+}])_{apices}$ (e, f) in *C. arbuscula*, *C. stellaris* and *F. cucullata*. Filled and open symbols indicate 'industrial' and 'reference' sites, respectively. Plotted values are mean ± 1 S.E. ($n = 18$).

weakly correlated ($r = 0.748$, $P > 0.01$, $n = 8$), perhaps as a result of a single outlying data point (data not shown). Sodium occurred in high concentrations at $F3_i$ and $F4_r$, probably due to the close proximity of these sites to the Barents coast, resulting in marine inputs from sea-spray.

3.5. Lichen biodiversity

The abundance and diversity of lichens differed significantly between sites (Fig. 8). Among the taiga sites differences were small, with 'reference' sites having the lower values. Complete listings of

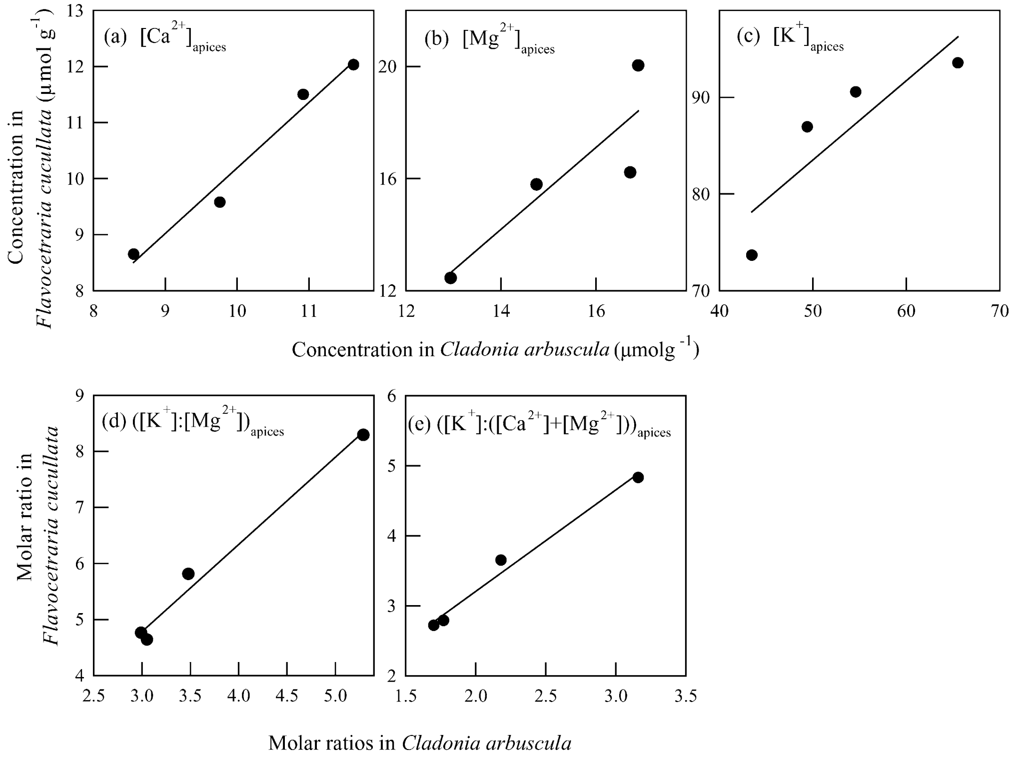

Fig. 3. Relationships between *C. arbuscula* and *F. cucullata* in cation concentrations and molar ratios at tundra sampling sites: (a) $[Ca^{2+}]_{apices}$ ($r = 0.988$, $P < 0.01$, $n = 4$); (b) $[Mg^{2+}]_{apices}$ ($r = 0.876$, $P > 0.05$, $n = 4$); (c) $[K^{+}]_{apices}$ ($r = 0.877$, $P > 0.05$, $n = 4$); (d) $([K^{+}]:[Mg^{2+}])_{apices}$ ($r = 0.992$, $P < 0.005$, $n = 4$); (e) $([K^{+}]:([Ca^{2+}] + [Mg^{2+}]))_{apices}$ ($r = 0.992$, $P < 0.005$, $n = 4$). Plotted values are mean ($n = 18$).

epiphytic lichen species assessed on the trunks and branches of *P. obovata* for $F1_i$ are given in Table 3. The first record of *Ramalina obtusata* in the Komi Republic was made at the Izhma river ($F1_i$). Among the tundra sites, $F7_i$ had a markedly lower abundance and diversity of epigeals and lower abundance of epiphytes. Complete listings of epigeal lichen species and epiphytic lichen species for $F8_r$ are given in Tables 4 and 5. The effect of reindeer grazing and trampling were marked to severe at $F3_i$, $F4_r$ and $F8_r$, with $F8_r$ perhaps suffering the worst effects based our observations.

Mean abundance of epigeal species at $F7_i$ and $F8_r$ were compared (data not shown) for the purpose of highlighting species with markedly different abundance values. *Alectoria nigricans*, *A. ochroleuca*, *Bryocaulon divergens* and *Sphaerophorus globosus* were common at $F8_r$ but were poorly represented at $F7_i$. With the exception of *Alectoria ochroleuca*, these species were also well represented at $F4_r$, although this site was not chosen as a comparable ‘reference’ site for $F7_i$. *Cetraria nigricans* and *Ochrolechia frigida* were common at $F8_r$ but were poorly represented at $F7_i$. *Ochrolechia frigida* was also common at $F4_r$.

4. Discussion

4.1. Chemical impacts on lichens and top-soil

The objectives of the study were to seek changes in lichen chemistry at ‘industrial’ sites that might be caused by air pollution. In particular, increases in the ratios $[K^{+}]:[Mg^{2+}]_{apices}$ and $[K^{+}]:([Ca^{2+}]+[Mg^{2+}])_{apices}$ were sought as evidence of acid deposition and increased $[N]_{apices}$ values were sought as evidence of N deposition. In general, variation in lichen chemistry was either small or, if large, unrelated to industrial activity. For example, the ratio $[K^{+}]:[Mg^{2+}]_{apices}$ in *C. arbuscula* varied across all

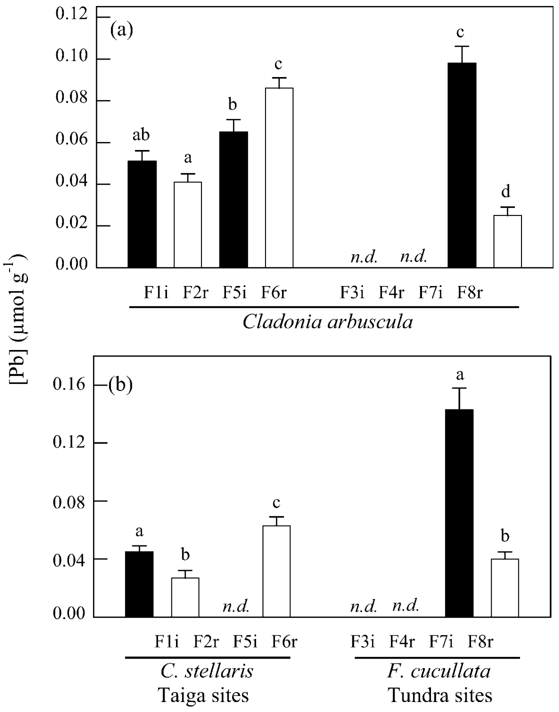

Fig. 4. Mean concentrations of lead in lichen apices. (a) *C. arbuscula*; (b) *C. stellaris* and *F. cucullata*. Filled and open columns indicate ‘industrial’ and ‘reference’ sites, respectively. Plotted values are mean ± 1 S.E. ($n = 18$); n.d., not determined. Significant differences were determined by one-way ANOVA followed by Tukey’s test, sites attributed with the same letters were not significant and sites with different letters were significantly different at the $P < 0.05$ level.

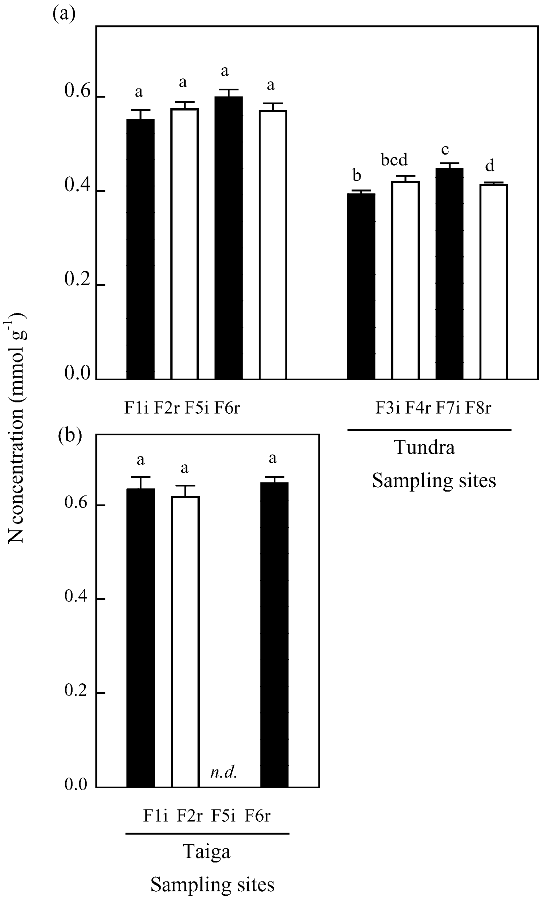

Fig. 5. Mean values of $[N]_{apices}$ in *C. arbuscula* (a) and *C. stellaris* (b). Filled and open columns indicate ‘industrial’ and ‘reference’ sites respectively. Plotted values are mean ± 1 S.E. ($n = 18$); n.d., not determined. Significant differences were determined by one-way ANOVA followed by Tukey’s test; values assigned the same letter were not significantly different at the $P < 0.05$ level.

eight sites by a factor of 1.9 and in *F. cucullata* amongst the four tundra sites by a factor of 1.8. Hyvärinen and Crittenden (1996) found that the ratio $[K^+]:[Mg^{2+}]_{apices}$ in *C. portentosa* varied by a factor of ca. 4 amongst sites in the British Isles. In the present work the strong covariation in the chemical indices between species suggests that these differences in cation contents reflected inter-site differences in environmental circumstances, as opposed to random variation. However, there were examples of both high and low ratios at both ‘industrial’ and ‘reference’ sites, suggesting that air pollution was not a factor contributing significantly to variation.

Amongst tundra sites, values of $[N]_{apices}$ were greatest at $F7_i$ and the value of $[N]_{apices}:[N]_{base}$ in *F. cucullata* was significantly higher here than at other sites. Hyvärinen and Crittenden (1998a), demonstrated a strong relationship between N deposition and [N] in the common heathland mat-forming lichen *C. portentosa*. Their study in the British Isles, revealed that [N] in the apical 5 mm of thalli ($[N]_{apices}$) and in a deeper stratum 35–50 mm from the apices ($[N]_{base}$) were both significantly positively correlated, and that the ratio $[N]_{apices}:[N]_{base}$ was negatively correlated, with N deposition. Since Hyvärinen and Crittenden (1998a) found the concentration ratio $[N]_{apices}:[N]_{base}$ to be negatively correlated with N deposition in *C. portentosa*, the higher value of the ratio at $F7_i$ is difficult to interpret; it may reflect a subtle physiological perturbation or simply random variation.

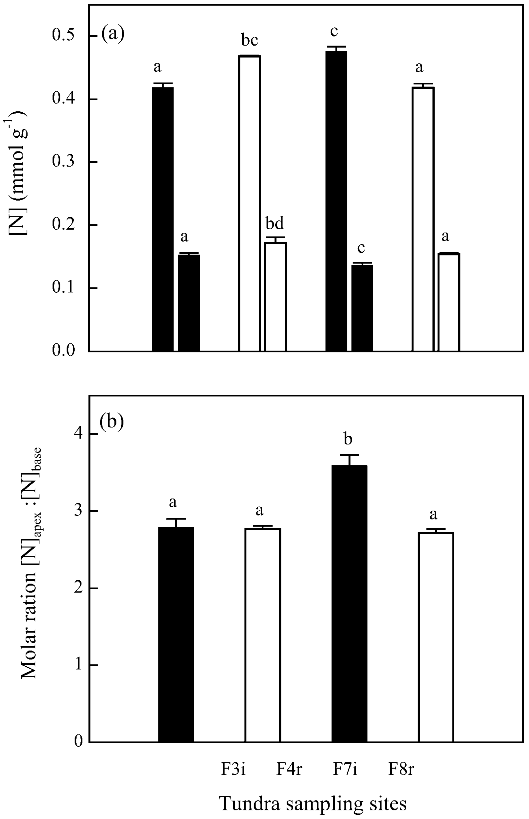

Fig. 6. Nitrogen concentration in *F. cucullata* (a) values of $[\mathrm{N}]_{\mathrm{apices}}$ (0–0.5 cm, values > 0.4 mmol g^{-1}) and $[\mathrm{N}]_{\mathrm{base}}$ (3.5–4 cm, values < 0.2 mmol g^{-1}); (b) the molar ratio $[\mathrm{N}]_{\mathrm{apices}}$:$[\mathrm{N}]_{\mathrm{base}}$. Filled and open columns indicate 'industrial' and 'reference' sites, respectively. Plotted values are mean ± 1 S.E. ($n = 18$). Columns with the same letters are not significantly different at the $P < 0.001$ level ($[\mathrm{N}]_{\mathrm{apices}}$ and $[\mathrm{N}]_{\mathrm{apices}}$:$[\mathrm{N}]_{\mathrm{base}}$) or $P < 0.05$ level ($[\mathrm{N}]_{\mathrm{base}}$).

Hyvärinen and Crittenden (1998a) reported that $[\mathrm{N}]_{\mathrm{apices}}$ in *C. portentosa* varied by a factor of 1.9 among sites in the British Isles and Søchting (1990) found [N] in unspecified strata of *Cladonia* sp. (subspecies *Cladina*) to vary by a factor of 3.4 among several European sites. In the present study, $[\mathrm{N}]_{\mathrm{apices}}$ varied by factors of 1.1–1.2 within each of the two biomes and 1.3–1.5 across all sites.

Coherent evidence of pollution was also generally absent in the soil metal concentration data. Elemental concentrations in soil-ash were generally low. Exceptions were marked elevations in Ba and Ca, and a small elevation in Ni, at $\mathrm{F7_i}$. Contamination of Ba and Ca might arise from dust blown from gravel roads or building work where cement is handled. However, concentrations of Ba in soil-ash taken from sites close to Vorkuta were six times greater than those measured at $\mathrm{F7_i}$ (Walker et al., 2003b). Other apparent chemical anomalies at $\mathrm{F7_i}$ include high $[\mathrm{Pb}]_{\mathrm{apices}}$ and low soil pH.

Nitrogen concentration in *C. arbuscula*, *C. stellaris* and *F. cucullata* at most sites were typical of background levels at comparable sites in the Usa basin (Walker et al., 2003a). For example, values of $[\mathrm{N}]_{\mathrm{apices}}$ in *C. arbuscula* and *F. cucullata* at tundra sites were between 0.4 and 0.5 mmol g^{-1} and in *C. stellaris* values of $[\mathrm{N}]_{\mathrm{apices}}$ at $\mathrm{F6_r}$ were 0.5–0.6 mmol g^{-1}. Values of $[\mathrm{N}]_{\mathrm{apices}}$ in *C. arbuscula* varied little within each of the tundra and taiga locations, but were consistently lower in the tundra. This reflects the trend observed in *C. stellaris* collected in the Usa basin where $[\mathrm{N}]_{\mathrm{apices}}$ decreased with increasing latitude and decreasing annual winter precipitation (Walker et al., 2003a). There are no data for N deposition around $\mathrm{F7_i}$, but gas flaring was observed at the industrial complex. Jaffe et al. (1995) reported significant NO_x emissions from gas flaring operations at Prudhoe Bay and accordingly elevated values of $[\mathrm{N}]_{\mathrm{apices}}$ in *C. arbuscula*, and *F. cucullata*, at $\mathrm{F7_i}$ might result from local NO_x emissions due to waste gas combustion at this site.

Hyvärinen and Crittenden (1996) showed that $([\mathrm{K}^+]:[\mathrm{Mg}^{2+}])_{\mathrm{apices}}$ in *C. portentosa* in the British Isles was positively correlated with precipitation acidity, while $([\mathrm{K}^+]:([\mathrm{Ca}^{2+}]+[\mathrm{Mg}^{2+}]))_{\mathrm{apices}}$ was not. Around the city of Vorkuta, Walker et al. (2003a) found $([\mathrm{K}^+]:[\mathrm{Mg}^{2+}])_{\mathrm{apices}}$ to be elevated in *C. arbuscula*, while $([\mathrm{K}^+]:([\mathrm{Ca}^{2+}]+[\mathrm{Mg}^{2+}]))_{\mathrm{apices}}$ to be reduced. These trends were interpreted as responses to alkaline ash deposition. In the Pechora region $([\mathrm{K}^+]:([\mathrm{Ca}^{2+}]+[\mathrm{Mg}^{2+}]))_{\mathrm{apices}}$ at $\mathrm{F7_i}$ was lower than at $\mathrm{F8_r}$ in both *C. arbuscula* and *F. cucullata*. Thus, at $\mathrm{F7_i}$, the marker ratio $([\mathrm{K}^+]:([\mathrm{Ca}^{2+}]+[\mathrm{Mg}^{2+}]))_{\mathrm{apices}}$ might have responded to deposition of Ca-rich particulates from construction activities. Potassium concentrations were higher in *F. cucullata* than *C. arbuscula* at tundra sites, comparing favourably with data from pristine sites in the Usa basin (Walker et al., 2003a).

Lead concentrations in the apices of *C. arbuscula*, *C. stellaris* and *F. cucullata* were low and compare favourably with values recorded in terricolous mat-forming lichens at unpolluted tundra sites in the Usa basin (T. Prystina, personal communication, 2001).

Table 3
An example of epiphytic lichen species and their cover abundance recorded on *P. obovata* at site F1$_i$ (Izhma river)

Species	F1$_i$ Izhma river								Mean (S.E.)
	1	2	3	4	5	6	7	8	
Epiphytes									
Bryoria capillaris[a]	2	5	4	3	4	2	5	4	**3.6**
B. furcellata[a]	2	2	0	1	1	1	1	0	**1.0**
B. fuscescens[a]	6	4	4	3	3	5	5	6	**4.5**
B. fremontii[a]	0	0	1	0	0	0	0	0	**0.1**
B. nadvornikiana[a]	0	1	0	0	0	1	0	0	**0.3**
Evernia mesomorpha	3	1	1	3	3	1	1	1	**1.8**
E. prunastri	0	0	0	1	1	0	0	0	**0.3**
Hypogymnia bitteri	0	0	0	0	0	2	1	1	**0.5**
H. physodes	4	6	4	6	5	4	4	3	**4.5**
H. tubulosa	3	2	2	1	1	3	2	2	**2.0**
Imshaugia aleurites	1	1	0	0	0	0	1	0	**0.4**
Melanelia exasperatula	0	0	1	2	0	1	0	0	**0.5**
M. olivacea	2	1	2	2	3	2	0	1	**1.6**
Parmelia sulcata	3	4	3	1	1	4	3	3	**2.8**
Parmeliopsis ambigua	2	1	1	0	0	1	1	2	**1.0**
P. hyperopta	2	0	1	1	1	0	1	2	**1.0**
Plastimatia glauca	2	2	0	0	0	2	1	1	**1.0**
Ramalina dilacerata	1	2	2	3	0	1	1	1	**1.4**
R. obtusata[b]	0	0	0	1	0	0	0	0	**0.1**
R. roesleri	0	0	0	2	1	0	0	0	**0.4**
R. thrausta	0	1	1	1	0	0	2	1	**0.8**
Tuckermanopsis chlorophylla	0	3	3	2	0	3	2	1	**1.8**
T. sepincola	3	0	0	0	0	0	0	0	**0.4**
Usnea filipendula[a]	3	3	2	3	4	3	3	3	**3.0**
U. globrescens[a]	0	0	0	0	0	0	0	1	**0.1**
U. lapponica[a]	0	0	0	1	3	0	1	2	**0.9**
U. subfloridana[a]	2	2	2	1	5	2	0	2	**2.0**
Vulpicida pinastri	2	1	1	1	1	1	1	0	**1.0**
Sum of abundance	43	42	35	39	37	39	36	37	**38.5 (1.00)**
Number of species	17	18	17	20	15	18	18	18	**17.6 (0.50)**

Other sites recorded in this way were F2$_r$, F5$_i$ and F6$_r$.

[a] Epiphytic species which are pollution sensitive according to Hawksworth and Rose (1970).

[b] First record of this species in the Komi Republic.

They are below concentrations measured at background sites elsewhere in the Arctic (Nash and Gries, 1995; Riget et al., 2000; Bargagli and Mikhailova, 2002). However, slightly elevated $[\mathrm{Pb}]_{\mathrm{apices}}$ and $[\mathrm{Pb}]_{\mathrm{soil\text{-}ash}}$ values were recorded at F7$_i$, which is known to derive from oil and gas industries. It is noteworthy that Garty et al. (1998) found elevated concentrations of Pb in *Ramalina lacera* at a site close to an oil combustion plant.

The mean pH values of top-soil in this study were all uniformly low and were comparable with values reported at pristine sites in the Usa basin (Rusanova, 1995). At most sites metal concentrations in soil-ash were typical of those found at pristine sites in the Russian Arctic, measured by Walker et al. (2003b) in the Usa basin and Alexeeva-Popova et al. (1995) on the Chukotka Peninsula. One exception was Na, for which concentrations were elevated at F3$_i$ and F4$_r$, possibly as a result of marine inputs from the Barents coast as sea-spray. Concentrations of Ba, Ca and Ni in top-soil were highest at F7$_i$, but Ba measured at this locality was still lower than elevated concentrations measured at sites close to Vorkuta and Inta in the Usa basin (Walker et al., 2003b). Elevated concentrations of Ni in the vicinity of F7$_i$ were probably the result of fuel oil combustion and gas flaring, which occurs around the Kolva site (Vilcheck and Tishkov, 1997). In a study by Genoni et al. (2000), elevated concentra-

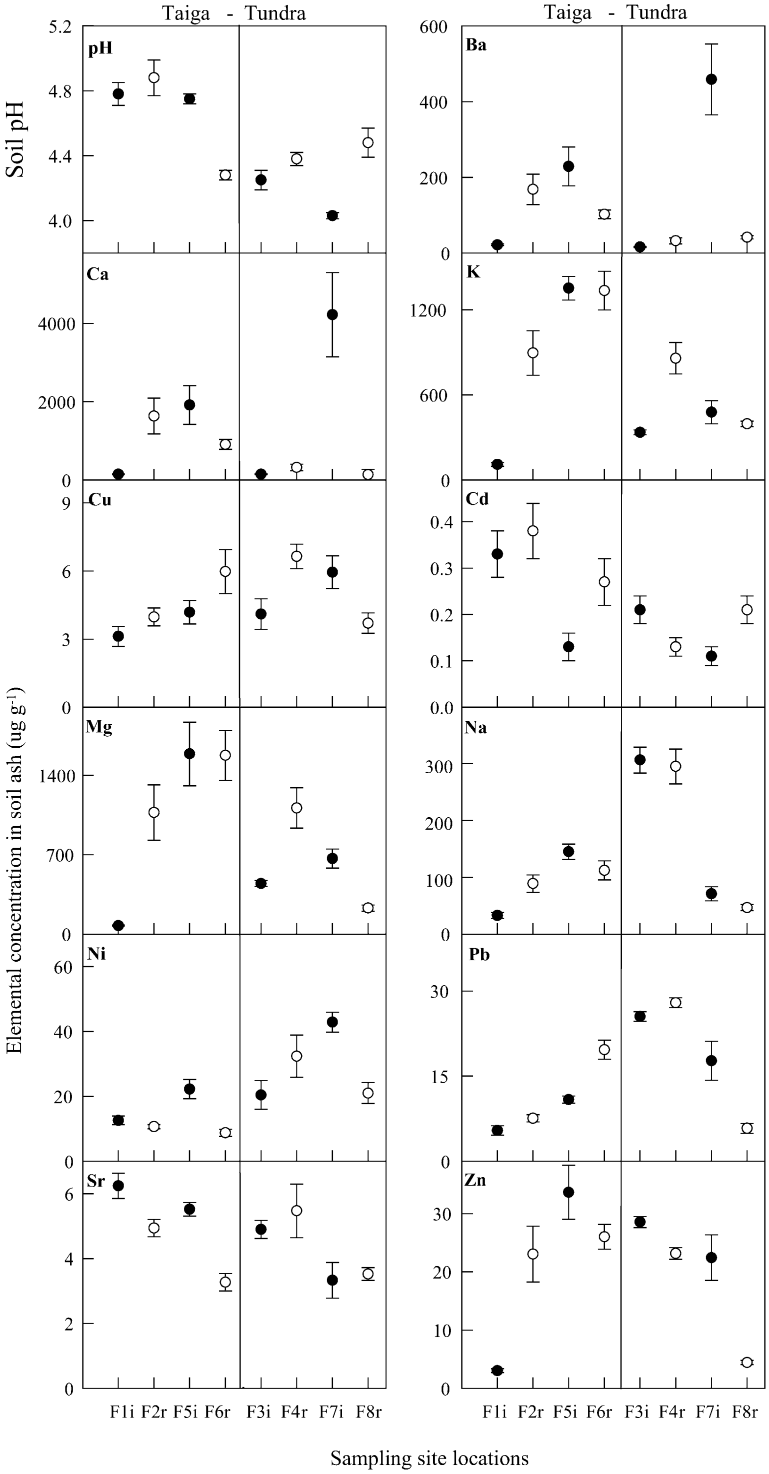

Fig. 7. Values of pH in top-soil (0–5 cm) and concentrations of metallic elements in soil-ash. Filled and open symbols indicate 'industrial' and 'reference' sites, respectively. Plotted values are mean ± 1 S.E. ($n = 18$).

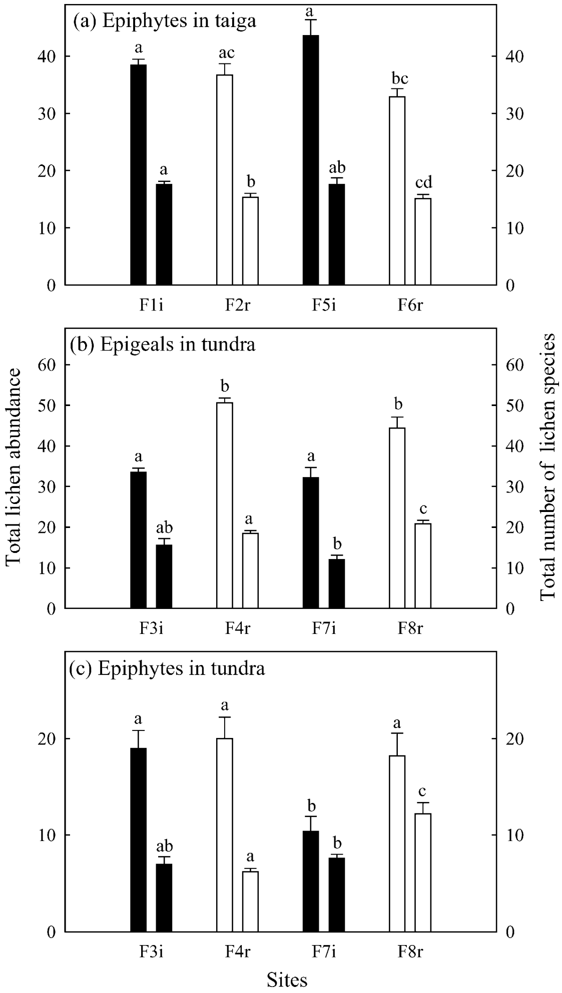

Fig. 8. Mean values of total abundance (left hand column) and total number of lichen species (right hand column) at each sampling site. Filled and open columns indicate 'industrial' and 'reference' sites respectively. Plotted values are mean ± 1 S.E. (n = 5–9). Significant differences were determined by one-way ANOVA followed by Tukey's test; within each measured attribute sites with the same letters were not significantly different and sites with different letters were significantly different at the $P < 0.05$ level.

tions of Ni were found in soil and moss samples taken close to oil-fired power plants. It is likely that soils from F7$_i$, and those at other sites in this study, have not been significantly modified and remain close to pristine, although previous studies have documented contamination of soils from oil spills in the area (Rusanova, 1995). It is doubtful that soils at these sites have been subject to an oil spill.

4.2. Lichen biodiversity

There was no evidence of deleterious industrial impacts on epiphytic lichens at taiga sites while values of lichen abundance and α-diversity were lower at industrial tundra sites, most notably at F7$_i$. However, interpretation of these data must take into consideration the potential confounding effects of other man-made disturbances such as forestry around the industrial sites producing younger, more open forests and reindeer husbandry. Since little data exist on the pollution sensitivity of epigeal lichens, and the tundra sites were subject to heavy reindeer grazing, greater emphasis was placed on data for epiphytic species for which more information exists on pollution sensitivity. In addition, it should be noted that the cover abundance scale used in this work was subjective and non linear, potentially over emphasising the importance of species with low abundance scores.

The abundance and α-diversity of lichens varied significantly between sites. Among taiga sites differences were small, with 'reference' sites having the lower values. Site F7$_i$, appeared to be less disturbed by reindeer. This might be because roads and pipelines restrict the movement of reindeer herds, or because reindeer herdsmen avoid industrial sites where the occurrence of discarded debris on the tundra can injure animals (O. Harbeck, personal communication, 2000). The epigeal lichens here formed much deeper mats, and hence greater biomass but comprised fewer species. By contrast, sites F3$_i$, F4$_r$ and F8$_r$ had more epigeal species, and hence greater diversity, but low biomass due to heavy grazing and trampling by reindeer. Crittenden (2000) gives a detailed review of lichen growth and the impacts of trampling and grazing by reindeer. In the shrub tundra east of Vorkuta mat-forming lichens were largely eliminated from the ground cover, as Walker et al. (2003a) found *C. arbuscula* at most sites but usually only in trace quantities, possibly due a combination of efficient grazing as well as trampling. Manseau et al. (1996) also reported damage to epigeal lichen communities in tundra used by migratory caribou in northern Quebec. The problem of lichen cover depletion due to reindeer is widespread in the Russian tundra (Ahti and Oksanen, 1990; Vilcheck and Tishkov, 1997; Klein, 2000).

Table 4
An example of epigeal lichen species and their cover abundance recorded in plots at site $F8_r$ (Mareyu river)

Species	$F8_r$ Mareyu river					Mean (S.E.)
	1	2	3	4	5	
Epigeal						
Alectoria nigricans[a]	2	3	2	2	4	**2.6**
A. ochroleuca[a]	2	3	2	1	3	**2.2**
Bryocaulon divergens[a]	2	3	2	1	3	**2.2**
Flavocetraria nivalis	4	4	3	5	5	**4.2**
F. cucullata	2	3	2	2	2	**2.2**
Cetraria islandica	2	3	3	3	2	**2.6**
F. nigricans	0	1	0	0	0	**0.2**
Cladonia amaurocraea	2	2	2	2	2	**2.0**
C. arbuscula	3	4	3	4	4	**3.6**
C. bellidiflora	1	2	1	2	1	**1.4**
C. borealis	1	1	2	2	2	**1.6**
C. cervicornis	0	0	0	1	0	**0.2**
C. chlorophaea	0	2	1	1	2	**1.2**
C. crispate	0	0	0	0	2	**0.4**
C. gracilis	2	2	2	2	2	**2.0**
C. pleurota	0	2	0	0	0	**0.4**
C. rangiferina/stygia	2	2	2	2	2	**2.0**
C. stellaris	1	2	0	2	2	**1.4**
C. sulphurina	0	2	0	0	0	**0.4**
C. uncialis	2	2	4	3	2	**2.6**
Hypogymnia physodes	2	0	0	0	0	**0.4**
Ichmadophila ericetorum	1	0	0	0	0	**0.2**
Ochrolechia frigida	0	2	2	1	2	**1.4**
Peltigera neckeri	0	1	0	0	0	**0.2**
P. malaceae	0	0	3	0	0	**0.6**
Pertusaria coccodes	0	0	0	0	1	**0.2**
P. panygra	0	1	0	1	0	**0.4**
Sphaerophorus globosus	2	2	2	2	2	**2.0**
Stereocaulon alpinum[a]	1	0	1	0	0	**0.4**
S. paschale[a]	3	2	3	0	1	**1.8**
Thamnolia vermicularis	2	2	1	0	2	**1.4**
Sum of abundance	**39**	**53**	**43**	**39**	**48**	**44.4 (2.71)**
Number of species	**20**	**24**	**20**	**19**	**21**	**20.8 (0.86)**

Other sites recorded in this way were $F3_i$, $F4_r$ and $F7_i$.

[a] Epigeal species which are pollution sensitive according to McCune and Geiser (1997) and Gilbert (2000).

Table 5
An example of epiphytic lichen species and their cover abundance recorded growing on *Betula nana* in plots at site $F8_r$ (Mareyu river)

Species	$F8_r$ Mareyu river					Mean (S.E.)
	1	2	3	4	5	
Epiphytes						
Alectoria nigricans[a]	0	2	0	0	2	**0.8**
A. ochroleuca[a]	0	1	0	0	2	**0.6**
Cetraria nigricans	2	2	2	1	1	**1.6**
Bryocaulon divergens[a]	0	1	0	0	0	**0.2**
Bryoria fuscescens[a]	1	0	0	0	1	**0.4**
Hypogymnia physodes	3	3	0	0	2	**1.6**
H. tubulosa	1	0	0	0	0	**0.2**
Lecanora argentata	2	2	2	2	2	**2.0**
Melanelia olivacea	2	2	1	0	1	**1.2**
Ochrolechia frigida	2	2	2	3	2	**2.2**
Parmelia sulcata	2	2	0	1	1	**1.2**
Parmeliopsis ambigua	2	2	2	2	2	**2.0**
P. hyperopta	2	2	2	2	2	**2.0**
Pertusaria corallina	0	0	0	1	2	**0.6**
P. coccodes	0	2	1	0	0	**0.6**
P. dactylina	2	0	1	0	2	**1.0**
Tuckermanopsis sepincola	5	4	3	4	4	**4.0**
Vulpicida pinastri	2	2	2	2	2	**2.0**
Sum of abundance	**21**	**23**	**13**	**12**	**22**	**18.2 (2.35)**
Number of species	**13**	**14**	**10**	**9**	**15**	**12.2 (1.16)**

Other sites recorded in this way were $F3_i$, $F4_r$ and $F7_i$.

[a] Epiphytic species which are pollution sensitive according to Hawksworth and Rose (1970).

It is not known why there were more epiphytic species at $F1_i$ than at the 'reference' site $F2_r$, or why the sum of abundance of epiphytes was significantly higher at $F5_i$ compared with $F6_r$. All are lowland taiga sites, but $F1_i$ is situated at a distance of approximately 10 km from petrochemical operations, principally oil processing, around Ukhta and the nearby gas processing operations in Sosnogorsk. Site $F1_i$ is also affected by forestry, which has left the taiga in this area fragmented, with forest stands of different ages. Svetly Vuktyl ($F5_i$) is almost exclusively characterised by its gas extraction industry which accounts for more than 95% of all industrial production in the area, and a small timber industry (Gimadi, 2002). However, lichens in the genera *Usnea* and *Bryoria* (see below) were present at this site, and at $F1_i$, presumably currently unaffected by emissions from the local oil and gas industries. If potential sources of pollution exist here, there may be a time lag before even the most sensitive species begin to decrease, possibly over several decades. These species are known to be pollution sensitive (Helle et al., 1990; Oksanen et al., 1990; Aamlid and Venn, 1993; Pirintsos et al., 1993). For example, Kauppi and Halonen (1992) found that the most SO_2 responsive epiphytic lichens around Oulu, northern Finland were, *Plastimatia glauca* and *Usnea filipendula*, together with *Bryoria fuscescens* and *B. capillaris*. Kuusinen et al. (1990) and Hyvärinen et al. (1992) found that these species also tended to favour older forest stands.

Factors other than air pollution affecting epiphytic lichen abundance include climate and stand age (Kuusinen et al., 1990; Hyvärinen et al., 1992). Site $F5_i$ comprises fragmented lowland taiga, whereas $F6_r$ is pristine old growth taiga, protected in the Yugyd Va national park in the foothills of the pre-Polar Ural mountains. Furthermore, the annual precipitation in the Urals is higher than in lowland taiga (Christensen and Kuhry, 2000). The stand age of the forest at $F6_r$ is greater than parts of the fragmented forest of $F5_i$ (T. Virtanen, personal communication, 2001). Trees of similar size were selected to restrict sampling to lichen communities at broadly comparable successional stages (Stone, 1989). Hyvärinen et al. (1992) found that *Bryoria fuscescens* was common in all types of stand, whilst species such as, *B. capillaris* prefer older or denser stands. Both *B. capillaris* and *B. fuscescens* were abundant at $F5_i$. Accordingly, differences in lichen abundance between the two sites could be due to regional variation in precipitation and impacts of forestry.

Epigeal lichen species which were present at $F8_r$ and $F4_r$, but were absent or poorly represented at $F7_i$, included species that are known to be pollution-sensitive according to McCune and Geiser (1997) viz. *Alectoria nigricans*, *Bryocaulon divergens* and *Sphaerophorus globosus*.

5. Conclusions

Data presented here for lichen and soil chemistry, and for lichen diversity suggest that the effects of terrestrial pollution at the industrial sites examined were generally below detection limits. An exception is $F7_i$ where a suite of minor shifts in environmental chemistry and lichen abundance might be an early indicator of industrial activity.

Other aspects of environmental chemistry, and components of biodiversity were examined at the same eight locations by collaborating research groups and data on environmental chemistry, provides additional evidence that emissions at $F7_i$ can be detected in the environment. These include polycyclic aromatic hydrocarbons, Hg and As in lake sediments (Solovieva et al., 2002; E. Patova, personal communication, 2002).

Acknowledgements

Help with logistics and fieldwork was provided by our Russian colleagues at the Institute of Biology—Komi Science Centre, Syktyvkar, especially Vasily Ponomarev. Olga Lavrinenko for help with lichen identification. Andy Tye and John Corrie for help with F-AAS and GF-AAS analysis. Dr. Peter Kuhry, for co-ordinating the SPICE project. This investigation was a component of the SPICE project which was supported by the INCO-COPERNICUS 2 Programme of the 5th Framework of the European Commission (Contract number: ICA2-CT-2000-10018).

References

Aamlid, D., Venn, K., 1993. Methods of monitoring the effects of pollution on forest and vegetation of eastern Finnmark, Norway. Nor. J. Agric. Sci. 7, 71–87.

Ahti, T., Oksanen, J., 1990. Epigeic lichen communities of taiga and tundra regions. Vegetatio 86, 39–70.

Alexeeva-Popova, N.V., Igoshina, T.I., Drosdova, I.V., 1995. Metal distribution in the Arctic ecosystems of the Chukotka Peninsula. Russia. Sci. Total Environ. 160–161, 643–652.

Bargagli, R., Mikhailova, I., 2002. Accumulation of inorganic contaminants. In: Nimis, P.L., Scheidegger, C., Wolseley, P.A. (Eds.), Monitoring with Lichens-Monitoring Lichens, NATO Science Series. Kluwer Academic Publishers, Netherlands, pp. 65–84.

Boyd, R., Niskavaara, H., Chekushin, V.A., Pavlov, V., Often, M., Reimann, C., 1997. Anthropogenic noble-metal enrichment of topsoil in the Monchegorsk area, Kola Peninsula, Northwest Russia. J. Geochem. Explor. 58, 283–289.

Bremmer, J.M., Breitenbeck, G.A., 1983. A simple method for determination of ammonium in semimicro-kjeldhahl analysis of soils and plant materials using a block digester. Commun. Soil Sci. Plant Anal. 14 (10), 905–913.

Briffa, K.R., Jones, P.D., Schweingruber, F.H., Shiyatov, S.G., Cook, E.R., 1995. Unusual twentieth-century summer warmth in a 1000-year temperature record from Siberia. Nature 376, 156–159.

Chapin, D.M., Bledsoe, C.S., 1992. Nitrogen fixation in Arctic plant communities. In: Chapin, III, F.S., Jeffries, R.L., Reynolds, J.F., Shaver, G.R., Svoboda, J. (Eds.), Arctic Ecosystems in a Changing Climate. Academic Press, London, pp. 301–319.

Christensen, J.H., Kuhry, P., 2000. High resolution regional climate validation and permafrost simulation for the East-European Russian Arctic. J. Geophys. Res. 105 (D24), 29647–29658.

Conti, M.E., Cecchettib, G., 2001. Biological monitoring: lichens as bioindicators of air pollution assessment; a review. Environ. Pollut. 114 (3), 471–492.

Crittenden, P.D., 1998. Nutrient exchange in an Antarctic macrolichen during summer snowfall-snow melt events. New Phytol. 139, 697–707.

Crittenden, P.D., 2000. Aspects of the ecology of mat-forming lichens. Rangifer 20 (2–3), 127–139.

Dobson, F.S., 1979. Lichens, an Illustrated Guide to the British and Irish Species. Cambrian Printers, Aberystwyth, Wales.

Ellis, C.J., Crittenden, P.D., Scrimgeour, C.M., Ashcroft, C.J., 2003. The natural abundance of ^{15}N in mat-forming lichens. Oecologia 136 (1), 115–123.

Farmer, A.M., Bates, J.W., Bell, J.N.B., 1991. Seasonal variations in acidic pollutant inputs and their effects on the chemistry of stemflow, bark and epiphyte tissues in three oak woodlands in N.W. Britain. New Phytol. 118, 441–451.

Garty, J., Cohen, Y., Kloog, N., 1998. Airborne elements, cell membranes, and chlorophyll in transplanted lichens. J. Environ. Qual. 27, 973–979.

Genoni, P., Parco, V., Santagostino, A., 2000. Metal biomonitoring with mosses in the surroundings of an oil-fired power plant in Italy. Chemosphere 41, 729–733.

Gilbert, O., 2000. Lichens. Harper Collins Publishers, London.

Gimadi, I., 2002. The Preliminary variant of the social-economical development scenarios. Unpublished report of the Institute of Economy, Ural Division. Russian Academy of Sciences, Yekaterinburg.

Gonzalez, C.M., Pignata, M.L., 1997. Chemical response of the lichen Punctelia subrudecta (Nyl.) Krog transplanted close to a power station in an urban-industrial environment. Environ. Pollut. 97 (3), 195–203.

Goward, T., McCune, B., Meidinger, D., 1994. The Lichens of British Colombia, Illustrated keys. Part 1. Foliose and squamulose species Ministry of Forests, British Colombia.

Haapala, H., Goltsova, N., Lodenius, M., 2001. Heavy metal solubility in podzolic soils exposed to the alkalizing effect of air pollutants. Environ. Pollut. 115, 33–41.

Hawksworth, D.L., Rose, F., 1970. Qualitative scale for estimating sulphur dioxide air pollution in England and Wales using epiphytic lichens. Nature 227, 145–148.

Helle, T., Norokorpi, Y., Saastamoinen, O., 1990. Reduction of arboreal lichens in two spruce stands in northern Finland between 1976 and 1988. In: Kinnunen, K., Varmola, M. (Eds.). Effects of Air Pollutants and Acidification in Combination with Climate Factors on Forests, Soils and Waters in Northern Fennoscandia, pp. 57–65. Nordic Council of Ministers.

Hyvärinen, M., Crittenden, P.D., 1996. Cation ratios in *Cladonia portentosa* as indices of precipitation acidity in the British Isles. New Phytol. 132, 521–532.

Hyvärinen, M., Crittenden, P.D., 1998a. Relationships between atmospheric nitrogen inputs and the vertical nitrogen and phosphorous concentration gradients in the lichen *Cladonia portentosa*. New Phytol. 140, 519–530.

Hyvärinen, M., Crittenden, P.D., 1998b. Growth of the cushion-forming lichen, *Cladonia portenosa*, at nitrogen-polluted and unpolluted heathland sites. Environ. Exp. Bot. 40, 67–76.

Hyvärinen, M., Halonen, P., Kauppi, M., 1992. Influence of stand age and structure on the epiphytic lichen vegetation in the middle-boreal forests of Finland. Lichenologist 24 (2), 165–180.

IPCC, 2001. Climate Change 2001. The scientific basis. In: Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., Van der Linden, P.J., Xiaosu, D. (Eds.), Third Assessment Report IPCC Working Group I. Cambridge University Press, UK.

IUCN, 1993. Oil and Gas Exploration and Production in Arctic and Subarctic Onshore Regions. IUCN Gland, Switzerland and Cambridge, UK, with E and P Forum, London.

Jaffe, D.A., Hornrath, R.E., Furness, D., Conway, T.J., Dlugokencky, E., Steele, L.P., 1995. A determination of the CH_4, NO_x and CO_2 emissions from the Prudhoe Bay, Alaska oil development. J. Atmos. Chem. 20, 213–227.

Jónsdóttir, I.S., Callaghan, T.V., Lee, J.A., 1995. Fate of added nitrogen in a moss-sedge Arctic community and effects of increased nitrogen deposition. Sci. Total Environ. 160–161, 677–685.

Kauppi, M., Halonen, P., 1992. Lichens as indicators of air pollution in Oulu, northern Finland. Annal. Botanici Fenni. 29, 1–9.

Klein, D.R., 2000. Arctic grazing systems and industrial development: can we minimize conflicts? Polar Res. 19 (1), 91–98.

Krankina, O.N., Dixon, R.K., Kirilenko, A.P., Kobak, K.I., 1997. Global climate change adaptation: examples from Russian Boreal forests. Climatic Change 36, 197–215.

Kuusinen, M., Mikkola, K., Jukola-Sulonen, E-L., 1990. Epiphytic lichens on conifers in the 1960's to 1980's in Finland. In: Kauppi, P., Anttila, P., Kenttämies, K. (Eds.), Acidification in Finland. Springer-Verlag, Berlin, pp. 397–420.

Larssen, T., Carmichael, G.R., 2000. Acid rain and acidification in China: the importance of base cation deposition. Environ. Pollut. 110, 89–102.

Lausala, T., Valkonen, L., 1999. Economic geography and structure of the Russian territories of the Barents region. Arctic Centre Report No. 33, University of Lapland, Rovaniemi.

Locatelli, C., 1999. The Russian oil industry restructuration: towards the emergence of western type enterprises? Energ. Policy 27, 435–449.

Manseau, M., Huot, J., Crête, M., 1996. Effects of summer grazing by caribou on composition and productivity of vegetation: community and landscape level. J. Ecol. 84, 503–513.

McCune, B., Geiser, L., 1997. Macrolichens of the Pacific Northwest. Oregon State University Press, Corvallis.

Moberg, R., Holmåsen, I., 1982. Lavar. Interpublishing, Stockholm (in Swedish).

Nash III, T.H., 1989. Metal tolerance in lichens. In: Shaw, A.J. (Ed.), Heavy Metal Tolerance in Plants: Evolutionary Aspects. CRC Press, Boca Raton, FL, pp. 119–131.

Nash III, T.H., Gries, C., 1995. The use of lichens in atmospheric deposition studies with an emphasis on the Arctic. Sci. Total Environ. 160, 729–736.

Nimis, P.L., Castellello, M., Perotti, M., 1990. Lichens as bioindicators of heavy metal pollution: a case study at La Spezia (N. Italy) In: Markert, B. (Ed.), Plants as Biomonitors. VCH, Weinheim, New York, pp. 265–284.

Niskavaara, H., Reimann, C., Chekushin, V.A., Kashulina, G., 1997. Seasonal variability of total and easily leachable element contents in topsoils (0–5 cm) from eight catchments in the European Arctic (Finland, Norway and Russia). Environ. Pollut. 96 (2), 261–274.

Oksanen, J., Tynnyrinen, S., Kärenlampi, L., 1990. Testing for increased abundance of epiphytic lichens on a local pollution gradient. Annal. Botani. Fenni. 27, 301–307.

Pakarinen, P., Kaistila, M., Hasanen, E., 1983. Regional concentration levels of vanadium, aluminium and bromine in mosses and lichens. Chemosphere 12, 1477–1485.

Pelley, J., 2001. Will drilling for oil disrupt the Arctic National Wildlife Refuge? Environ. Sci. Technol. 35 (11), 240A–247A.

Pirintsos, S.A., Vokou, D., Diamantopoulos, J., Galloway, D.J., 1993. An assessment of the sampling procedure for estimating air pollution using epiphytic lichens as indicators. Lichenologist 25 (2), 165–173.

Press, M.C., Callaghan, T.V., Lee, J.A., 1998. How will European Arctic ecosystems respond to projected global environmental change? Ambio 27 (4), 306–311.

Pryde, P.R., 1991. Environmental Management in the Soviet Union. Cambridge University Press, Cambridge.

Reimann, C., Boyd, R., Caritat, P. de, Halleraker, J.H., Kashulina, G., Niskavaara, H., Bogatyrev, I., 1997. Topsoil (0–5 cm) composition in eight Arctic catchments in northern Europe (Finland, Norway and Russia). Environ. Pollut. 95 (1), 45–56.

Reimann, C., Halleraker, J.H., Kashulina, G., Bogatyrev, I., 1999. Comparison of plant and precipitation chemistry in catchments with different levels of pollution on the Kola Peninsula, Russia. Sci. Total Environ. 243–244, 169–191.

Reimann, C., Banks, D., Caritat, P. de, 2000. Impacts of airborne contamination on regional soil and water quality: the Kola Peninsula, Russia. Environ. Sci. Technol. 34, 2727–2732.

Revich, B.A., 1995. Public health and ambient air pollution in Arctic and Subarctic cities of Russia. Sci. Total Environ. 160–161, 585–592.

Riget, F., Asmund, G., Aastrup, P., 2000. The use of lichen (*Cetraria nivalis*) and moss (*Rhacomitrium languginosum*) as monitors for atmospheric deposition in Greenland. Sci. Total Environ. 245, 137–148.

Rovinsky, F., Pastukhov, B., Bouyvolov, Y., Burtseva, L., 1995. Present day state of background pollution of the natural environment in the Russian Arctic in the region of Ust-Lena Reserve. Sci. Total Environ. 160–161, 193–199.

Rusanova, G.V., 1995. Impacts of industrial activity on East European arctic soils. In: Global change and Arctic terrestrial ecosystems: Symposium Proceedings, Oppdal, Norway, pp. 307–313.

Ryaboshapko, A., Gallardo, L., Kjellström, E., Gromov, S., Paramonov, S., Afinogenova, O., Rodhe, H., 1998. Balances of oxidized sulfur and nitrogen over the Former Soviet Union Territory. Atmos. Environ. 32, 647–658.

Søchting, U., 1990. Reindeer lichens injured in Denmark. Brit. Lich. Soc. Bull. 67, 1–4.

Søchting, U., 1995. Lichens as monitors of nitrogen deposition. Crypto. Bot. 5, 264–269.

Solovieva, N., Jones, V.J., Appleby, P.G., Kondratenok, B.M., 2002. Extent, environmental impact and long-term trends in atmospheric contamination in the Usa basin of East-European Russian Arctic. Water Air Soil Poll. 139, 237–260.

State of the Environment of the Komi Republic, 1992–1998. Reports of the Ministry of Environmental Resources and Conservation. Komi Republic, Syktyvkar.

Stone, D.F., 1989. Epiphytic succession on *Quercus garryana* branches in the Willamette valley of western Oregon. Byrologist 92 (1), 81–94.

Thor, G., 1998. Red-listed lichens in Sweden: habitats, threats, protection, and indicator value in boreal coniferous forests. Biodivers. Conserv. 7, 59–72.

Tuovinen, J.-P., Laurila, T., Lättilä, H., Ryaboshapko, A., Brukhanov, P., Korolev, S., 1993. Impact of the sulphur dioxide sources in the Kola Peninsula on air quality in northernmost Europe. Atmos. Environ. 27A (9), 1379–1395.

Vilcheck, G.E., Tishkov, A.A., 1997. Usinsk oil spill. In: Crawford, R.M.M. (Ed.), Disturbance and Recovery in Arctic Lands. Kluwer Academic, Dordrecht, pp. 411–420.

Vinogradova, A.A., 2000. Anthropogenic pollutants in the Russian Arctic atmosphere: sources and sinks in spring and summer. Atmos. Environ. 34, 5151–5160.

Virtanen, T., Mikkola, K., Patova, E., Nikula, A., 2002. Satellite image analysis of human caused changes in the tundra vegetation around the city of Vorkuta, north-European Russia. Environ. Pollut. 120 (3), 647–658.

Walker, T.R., Crittenden, P.D., Young, S.D., 2003a. Regional variation in the chemical composition of winter snowpack and terricolous lichens in relation to sources of acid emissions in the Usa river basin, northeast European Russia. Environ. Pollut. 125, 401–412.

Walker, T.R., Young, S.D., Crittenden, P.D., Zhang, H., 2003b. Anthropogenic metal enrichment of snow and soil in northeastern European Russia. Environ. Pollut. 121, 11–21.

Will-Wolf, S., Esseen, P.-A., Neitlich, P., 2002. Monitoring biodiversity and ecosystem function: forests. In: Nimis, P.L., Scheidegger, C., Wolseley, P.A. (Eds.), Monitoring with Lichens-Monitoring Lichens, NATO Science Series. Kluwer Academic Publishers, Netherlands, pp. 203–222.

Wilson, E.O., 1988. Biodiversity. National Academy Press, Washington, DC.

Ziegler, C.E., 1987. Environmental Policy in the USSR. The University of Massachusetts Press, Amherst.