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# Impact of forest conversion to agriculture on carbon and nitrogen mineralization in subarctic Alaska

JOSÉ M. GRÜNZWEIG[1,3,*], STEPHEN D. SPARROW[2] and F. STUART CHAPIN III[1]

*[1]Institute of Arctic Biology, University of Alaska Fairbanks, Fairbanks, AK 99775-7000, USA; [2]Agricultural and Forestry Experiment Station, University of Alaska Fairbanks, Fairbanks, AK 99775-0080, USA; [3]Current address: Department of Environmental Sciences and Energy Research, Weizmann Institute of Science, Rehovot, 76100, Israel; *Author for correspondence (e-mail: jose.gruenzweig@weizmann.ac.il; phone: +972-8-9344227; fax: +972-8-9344124)*

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**Abstract.** Land-use change is likely to be a major component of global change at high latitudes, potentially causing significant alterations in soil C and N cycling. We addressed the biogeochemical impacts of land-use change in fully replicated black spruce forests and agricultural fields of different ages (following deforestation) and under different management regimes in interior Alaska. Change from forests to cultivated fields increased summer temperatures in surface soil layers by 4–5 °C, and lengthened the season of biological activity by two to three weeks. Decomposition of a common substrate (oat stubble) was enhanced by 25% in fields compared to forests after litter bags were buried for one year. In-situ net N mineralization rates in site-specific soil were similar in forests and fields during summer, but during winter, forests were the only sites where net N immobilization occurred. Field age and management had a significant impact on C and N mineralization. Rates of annual decomposition, soil respiration and summer net N mineralization tended to be lower in young than in old fields and higher in fallow than in planted young fields. To identify the major environmental factors controlling C and N mineralization, soil temperature, moisture and N availability were studied. Decomposition and net N mineralization seemed to be mainly driven by availability of inorganic N. Soil temperature played a role only when comparing forests and fields, but not in field-to-field differences. Results from soil respiration measurements in fields confirmed low sensitivity of heterotrophic respiration, and thus decomposition to temperature. In addition, both soil respiration and net N mineralization were limited by low soil water contents. Our study showed that (1) C and N mineralization are enhanced by forest clearing in subarctic soils, and (2) N availability is more important than soil temperature in controling C and N mineralization following forest clearing. Projecting the biogeochemical impacts of land-use change at high latitudes requires an improved understanding of its interactions with other factors of global change, such as changing climate and N deposition.

## Introduction

Land-use change is the dominant component of global change in terms of its impact on terrestrial ecosystems (Walker and Steffen 1999). Such a change, and particularly conversion of native land to agriculture, profoundly alters land cover, biota, and biogeochemical cycles. The potential effect of land-use change on C storage in the boreal region is large. Boreal forests are the second largest biome on Earth, occupying 8.5% of the total terrestrial area (Whittaker 1975). They contain 13–18% of the terrestrial C pool (Anderson 1992; Houghton 1995), due to their high soil C contents per unit area (Dixon et al. 1994; McGuire et al. 1995; Houghton et al. 1999). Much of this C is vulnerable to loss through combustion and enhanced decomposition after clearing of boreal forests.

Plant production and microbial activity are constrained at high latitudes by low temperatures and a short growing season (Van Cleve et al. 1983; Parry 1992). Temperature increases of about 0.3 °C per decade during summer and up to 2 °C per decade during winter have occurred during the last 30–40 years in Alaska (Maxwell 1997; ARAG-USGCRP 1999; Barber et al. 2000; Serreze et al. 2000). Even greater warming of 1.5–3.5 °C and 4–10 °C is predicted for Alaska by the years 2030 and 2100, respectively (ARAG-USGCRP 1999). This warming trend will not only lengthen the growing season, but also likely increase boreal disturbances, such as fire and insect outbreaks (Starfield and Chapin 1996). Warming might also make conditions more favorable for agriculture at high latitudes by lengthening the growing season (Myneni et al. 1997) and reducing frost probability. Together with increased competition for land (Parry 1992), disturbances associated with warming might accelerate the conversion of forest to other land-surface types.

Temperature is often viewed as the dominant control over soil decomposition (Schimel et al. 1994; Townsend et al. 1995; Trumbore et al. 1996), with soils in cold climates being particularly sensitive to warming (Kirschbaum 1995; Goulden et al. 1998). This notion has been challenged, recently (Liski et al. 1999; Giardina and Ryan 2000). Alternative hypotheses include control of decomposition by an interaction of temperature and soil nutrient status (Scowcroft et al. 2000), by an interaction of temperature and soil moisture (Wang et al. 2000) and by litter quality (Flanagan and Van Cleve 1983; Giardina and Ryan 2000; Hobbie et al. 2000).

Loss of soil organic C following land-use change is caused by enhanced decomposition of organic matter rather than by the slow climate-driven pedogenic processes (Anderson 1992). Decomposition of black-spruce forest-floor material was more rapid when buried in a cultivated field than in its original site (Sparrow et al. 1992), but the relative importance of environmental factors could not be determined. Management of fields might also influence decomposition and C storage. Decomposition of agricultural residues is often enhanced by N fertilization (Mary et al. 1996; Henriksen and Breland 1999) and inhibited by no-till practices, use of perennial vegetation, and reduced frequency of bare fallow (Paustian et al. 2000).

In the present study, we investigated the effects of land use, time after deforestation and field management on mineralization of C and N in subarctic soils. We also attempted to identify the major factors controlling these effects, particularly microclimate, N availability, and soil properties. Decomposition, soil respiration and N mineralization were studied in replicate forests and fields of various types in two regions of interior Alaska. We hypothesized that C and N mineralization rates would be lower in forests than in fields, and higher in new fields and cropland as compared to old fields and grassland, respectively. We further hypothesized that differences among land-use types would be driven primarily by soil temperature, soil moisture and/or N availability.

## Methods

### *Study sites*

Our study design allowed a replicated comparison of field and forest sites spread across two regions in a similar climate regime. Our two study regions were located in interior Alaska near Delta Junction (64°10′ N, 145°44′ W; elevation 350–370 m) and Fairbanks (64°51′ N, 147°43′ W; elevation 125 m; some sites near Eielson [64°40′ N, 147°10′ W; elevation 150 m] and Two Rivers [64°50′ N, 146°45′ W; elevation 200 m]) on old terraces and floodplains of the Tanana River and the Chena River. The two regions had similar mean annual air temperatures (−2.3 °C in Delta Junction [Big Delta station] and −3.1 °C in the Fairbanks area [average of Fairbanks and Eielson Airport stations]), and mean annual precipitation (296 and 297 mm, respectively) (NOAA 1999).

Across both regions, we selected sites that were similar with respect to soils, topography and original vegetation. In each region, sites consisted of one black spruce (*Picea mariana*)-dominated forest, the main climax community on lowlands (Viereck et al. 1986), and seven agricultural fields. Most of the fields had originated from black spruce forests, a few fields originated from white spruce forests (*P. glauca*). All fields and the Delta Junction forest were on Inceptisols (Table 1). The Fairbanks forest was on Gelisol, a soil order similar to Inceptisols but underlain by permafrost. Soils were similar in texture ranging from loamy sand to loam with clay contents between 4 and 17% (Table 1). Forest soils were characterized by a thick organic horizon (approximately 30 cm deep) covered with mosses and lichens. All soils developed on alluvium derived from glacially eroded materials from the Alaska Range and weathering products of Birch Creek Schist, which occupies a large portion of the Tanana-Yukon uplands.

Fields were established using the conventional method: removing and burning native forest vegetation and part (or all) of the organic horizon, and subsequent raking to remove roots. In each region, we included three newly established fields (0–2 years after forest clearing), two old croplands and two old grasslands (at least 10 years under their respective management; age of fields was obtained from owners, and was verified with aerial photographs and records of extension services). New fields were either fallow (one in Delta Junction, two in Fairbanks) or planted to annual crops (’new croplands’; two in Delta Junction, one in Fairbanks). Some of the fallow new fields had been under production in the previous one or two years (ploughed, fertilized and planted to annual crops), others had only been deforested and had never been ploughed or fertilized. All of these fields were fallow in 1999, and were neither ploughed and nor fertilized during that year. Some of the old croplands had previously been in a grain-fallow or grain-potato rotation, but in 1999

*Table 1.* Region, time after forest clearing and soil properties for all sites. Soil texture and particle size distribution were determined for the top 20 cm of the mineral soil, pH and bulk density for the top 10 cm below the soil surface (i.e. $O_{ie}$ horizon in forests). C and N concentrations are for the whole organic layer in forests and for the top 20 cm in fields. Results for particle size distribution and pH are from composites of six subsamples per site, results for bulk density ($n$ = 6) and C and N concentrations ($n$ = 5) are means (standard error)

| Land use, age, management | Region | Time after clearing (years) | Soil classification (USDA system) | Soil texture | Clay content (%) | Silt content (%) | Sand content (%) | Bulk density (g $cm^{-3}$) | pH carbon | Organic nitrogen concentration (%) | Total nitrogen concentration (%) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Forest | Delta Jct. | – | Aquic Eutrocryept | Sandy loam | 6 | 36 | 58 | 0.07 (0.01) | 4.7 | 24.9 (0.4) | 0.69 (0.05) |
| | Fairbanks | – | Typic Haploturbel | Sandy loam | 12 | 22 | 66 | 0.04 (0.00) | 4.0 | 37.5 (1.5) | 0.86 (0.04) |
| New fallow field | Delta Jct. | 2 | Aquic Eutrocryept | Sandy loam | 6 | 48 | 46 | 0.72 (0.03) | 5.6 | 7.0 (0.3) | 0.31 (0.01) |
| | Fairbanks | 0 | Aquic Eutrocryept | Sandy loam | 7 | 24 | 69 | 0.77 (0.03) | 5.6 | | |
| | | 1 | Typic Eutrocryept | Sandy loam | 11 | 22 | 67 | 0.93 (0.05) | 5.2 | 3.0 (0.4) | 0.15 (0.02) |
| New cropland | Delta Jct. | 1 | Aquic Eutrocryept | Loamy sand | 5 | 17 | 78 | 0.87 (0.03) | 5.8 | 5.6 (0.9) | 0.30 (0.05) |
| | | 2 | | Sandy loam | 9 | 19 | 72 | 0.83 (0.02) | 6.0 | | |
| | Fairbanks | 1 | Typic Eutrocryept | Loam | 17 | 35 | 48 | 1.03 (0.02) | 5.1 | | |
| Old cropland | Delta Jct. | 15 | Aquic Eutrocryept | Sandy loam | 6 | 28 | 66 | 0.70 (0.02) | 5.8 | 7.1 (0.4) | 0.38 (0.02) |
| | | 20 | Aquic Eutrocryept | Silt loam | 8 | 66 | 26 | 0.90 (0.02) | 5.6 | 4.7 (0.1) | 0.25 (0.01) |
| | Fairbanks | > 10 | Aquic Eutrocryept | Sandy loam | 11 | 20 | 69 | 1.16 (0.06) | 6.6 | | |
| | | > 40 | Typic Eutrocryept | Loam | 15 | 32 | 53 | 1.03 (0.02) | 6.9 | | |
| Old grassland | Delta Jct. | 17 | Aquic Eutrocryept | Sandy loam | 4 | 40 | 56 | 0.79 (0.02) | 5.9 | | |
| | | 40 | Aquic Eutrocryept | Sandy loam | 5 | 26 | 69 | 0.90 (0.03) | 5.7 | 5.6 (0.5) | 0.31 (0.02) |
| | Fairbanks | 21 | | Loamy sand | 7 | 10 | 83 | 1.02 (0.03) | 5.1 | | |
| | | > 60 | Aquic Eutrocryept | Loam | 15 | 32 | 53 | 0.82 (0.03) | 6.8 | 6.3 (0.4) | 0.39 (0.02) |

all of these fields were planted to annual grain or fodder crops, using conventional farming practices. Grasslands (mainly smooth bromegrass, *Bromus inermis*, for hay production) were not ploughed, and plants resprouted from rhizomes in spring. In May 1999, new and old croplands were ploughed and fertilized prior to sowing either barley (*Hordeum vulgare*) or oat (*Avena sativa*). These fields and the grasslands were fertilized with 50–80 kg N ha$^{-1}$. Two of the four grasslands received an additional 25–40 kg N ha$^{-1}$ after the first harvest (re-fertilized grasslands). All grasslands were harvested between early and mid July 1999, but only the two re-fertilized fields were cut again in September. The number of hay harvests per season depends on weather conditions and farming priorities. Croplands were harvested between mid-August and mid-September 1999. All fields were commercially managed by farmers. This wide range of management regimes was selected to encompass the range of agricultural practices used in interior Alaska.

The current approach substitutes differences in sites for differences in time. Since most fields originated from black-spruce forests as a stable climax community, conversion of forests to agricultural land can be simulated by simultaneously comparing forests and fields. Similarly, fields of different ages were compared to simulate field development from newly cleared land to established older fields.

### *Field and laboratory methodology*

For the decomposition experiment, we used a common substrate, in order to reveal site-specific environmental factors controlling decomposition rates. We collected stubble that over-wintered on the soil surface (the normal situation in the region) from an oat field in Delta Junction and from a bromegrass field in Fairbanks in May 1999 (see Table 2 for initial litter quality). Litter was air dried, cut into 5 cm lengths, mixed, weighed (3 g), and placed into 0.1-mm polyester mesh bags (10.5 × 10.5 cm). Litter bags with oat stubble were buried at 10-cm depth in all fields and forests (10 cm beneath the bottom of the green moss), whereas bags with bromegrass stubble were placed on the soil surface of grasslands only. A soil depth of 10 cm was used for most experiments in this study, as this depth was within the most active horizon in forests ($O_{ie}$ layer) and fields (mineral top soil; Flanagan and Van Cleve (1983) and Sparrow et al. (1992)). We also collected $O_a$ horizon material (15% organic C, which is considered an organic horizon [Boone et al. (1999)]) of a black spruce forest, sieved it through a 2 mm sieve, passed the material > 2 mm through a Wiley mill equipped with 2-mm mesh screen and mixed it with the rest of the $O_a$ soil, without allowing any part of the material to dry out. Forty grams of the moist $O_a$ soil were weighed into a 0.1-mm mesh bag (13 × 13 cm), and the bags were buried at 10 cm depth in new fields and at the original depth in forests (20 cm, measured from the bottom of the green moss layer). Subsamples of the litter and forest floor were oven-dried to determine the original dry mass of each litter bag.

To estimate the in-situ net N mineralization and nitrification rates, we sampled a 10 cm deep core of 5-cm diameter at each site. Half of the core was retained as a control for initial $NH_4^+$ and $NO_3^-$ concentrations, and was frozen for further analy-

*Table 2.* Initial chemical characteristics of litter used in decomposition trials. C fractions were reported on an ash-free basis. Mean (standard error), $n = 3$

| Litter type | Carbon (%) | Nitrogen (%) | Ash (%) | Carbon fractions(%)[a] | | | |
|---|---|---|---|---|---|---|---|
| | | | | Nonpolar extractable | Water soluble | Acid soluble | Acid insoluble |
| Oat stubble | 43.3 (0.1) | 1.05 (0.04) | 4.70 (0.20) | 3.04 (0.35) | 13.17 (0.35) | 65.9 (0.2) | 17.94 (0.25) |
| Bromegrass stubble | 41.9 (0.1) | 1.21 (0.01) | 6.72 (0.05) | 4.35 (0.35) | 21.31 (0.05) | 62.9 (0.4) | 11.48 (0.07) |

[a] The nonpolar extractable fraction contains fats, waxes and chlorophylls; the water soluble fraction contains sugars, phenols and amino acids; the acid soluble fraction contains cellulose, hemicellulose and starch; the acid insoluble fraction contains lignin.

sis. The other half of the core was reburied in polyethylene bags that allowed $CO_2$ exchange with the soil atmosphere (Gordon et al. 1987). The study was initiated within 2 weeks after fertilization in most fields, and within 4 weeks in a few fields. During this time span most of the inorganic N remained in the top soil layer (Knight and Sparrow 1993). In forests, where the organic horizon had a lower bulk density than the mineral soils in the fields, the entire core was reburied for mineralization, and another core was collected 20 cm away as the control.

Two sets of litter bags and one set of N-mineralization bags were buried in early June 1999 in each field and forest. Each set consisted of six replicated bags that were randomly located within a 30 × 50 m area at each site. In late September, the N-mineralization bags and one set of litter bags were retrieved, transported to the laboratory on ice, and frozen until analysis. At the same time, a new set of bags for net N mineralization was buried using fresh soil cores. Again, half of each core was used as a control for initial $NH_4^+$ and $NO_3^-$ concentrations. At the end of the experiment in May 2000, the remaining litter bags and the N-mineralization bags were retrieved. In the re-fertilized grasslands, the N-mineralization bags were removed after the second fertilization in mid July, and a new set of bags was buried using fresh soil cores and new controls; these fields therefore had two sets of N-mineralization bags until the end of the summer. The summer period lasted 102–105 days, the winter period 247–250 days. Bags were not in place during the two weeks when farmers ploughed, fertilized and planted the fields.

We measured soil respiration at selected sites and dates between June and September 1999 using a closed-system Licor 6200 portable infrared gas analyzer (LI-COR, Lincoln, Nebraska, USA; Grogan and Chapin (2000)). For field soils, we used a 270 $cm^3$ cylindrical steel chamber, whose 4-cm diameter base was driven 1 cm into the soil. In forests, measurements were performed only during late summer and fall 1999. For these measurements, 28-cm diameter rings were driven 10 cm into the forest floor, and a light-impermeable 8000 $cm^3$ cylindrical chamber was mounted for respiration measurements. In a preliminary study conducted in a cropland, both chambers yielded similar results (data not shown). Chambers remained in place during the performance of the measurement, which lasted approximately 5 min. Forest soil respiration included a live moss component. Root severing by the rings was assumed to be minimal. Biomass in fields was sampled at near-maximal vegetative development in mid August.

Biomass of the early-July harvest from grasslands was estimated from farmers' records. Tree biomass in forests was estimated from diameter and height measurements using allometric equations, understory biomass was evaluated from 20-cm diameter cores, since understory plants were usually around 0.05–0.5 m high (Grünzweig et al., unpublished data). Root biomass for crops was determined from cores at 0–40 cm (5-cm diameter) and 40–100 cm (2-cm diameter) depths, and included only live roots.

We monitored soil temperature continuously at 10 cm depth in every forest and field using two Hobo H8 loggers (Onset, Bourne, Massachusetts, USA). Soil temperature measurements covered the entire experimental period of one year and included data from the previous winter (1998–1999) for some sites. Temperature was recorded every half hour during summer and every hour during winter. Sum of soil degree-days (DD) on 0 °C basis was calculated as follows:

$$DD = \sum_{i=1}^{n} T_i \qquad (1)$$

where $T_i$ is the average daily temperature above 0 °C on the $i^{th}$ of a total of n days.

For determination of soil moisture, we took six soil cores per site to 10-cm depth every 2–3 weeks during the summer, and determined water contents gravimetrically. Water contents were expressed as volume percent (v/v), and were averaged across all sampling points per site. Snow depth was measured in mid March 2000.

Soil texture was analyzed by the hydrometer method after dispersion with sodium hexametaphosphate (Sheldrick and Wang 1993). Soil pH was measured by a glass electrode in supernatant above a 1:1 soil/water suspension. Oat and bromegrass stubble was chemically characterized by a C fraction analysis (proximate analysis; Hobbie (1996)) conducted at the Natural Resources Research Institute, University of Minnesota, Duluth, Minnesota, USA. For determination of total C and N contents, samples were ball milled and combusted in a Carlo-Erba NA 1500 analyzer (Carlo-Erba, Milano, Italy). Organic materials were dried at 60 °C, and mineral soils were dried at 105 °C. For determination of net N mineralization and nitrification, soils were thawed, and immediately 5 g of soil were extracted in 50 ml 2*N* KCl shaken for one hour at 180 rpm. Filtered extracts were analyzed colorimetrically for $NH_4^+$ and $NO_3^-$ (phenol-hypochlorite method – Berthelot reaction and Cu-Cd reduction – modified Griess-Illosvay procedure, respectively; Keeney and Nelson (1982)) by an automated rapid-flow analyzer (RFA-300, Alpkem, Clackamas, OR). Freezing soil samples prior to analysis is a common procedure for soil preservation (Keeney and Nelson 1982), and was also used for N mineralization studies in forest and fields of interior Alaska (Klingensmith and Van Cleve 1993; Sparrow et al. 1993; Van Cleve et al. 1993). Freezing and thawing of annually frozen soils has no significant effect on inorganic N concentration, if thawed out samples were kept at low temperatures and were immediately extracted (Esala 1995). Preserving samples at around −20 °C halts all microbial processes, as minimal temperatures for microbial activity at high latitudes are typically around −5 °C (Flanagan 1978; Persson et al. 2000).

### *Statistical analyses*

Soil temperature, oat-stubble decomposition, net N mineralization and net nitrification were analyzed to test for the effects of land-use type and season or duration of the experiment using model I two-way ANOVA. Contrast analysis was used to test statistical significance of specific differences in particular parts of the dataset (planned comparisons of subsets of land-use types) (Statistica '99, StatSoft Inc., Tulsa, Oklahoma, USA). Because of no significant region effect for most variables, values obtained in Delta Junction and Fairbanks were averaged. Soil respiration and soil moisture data were analyzed with multivariate and univariate tests of repeated-

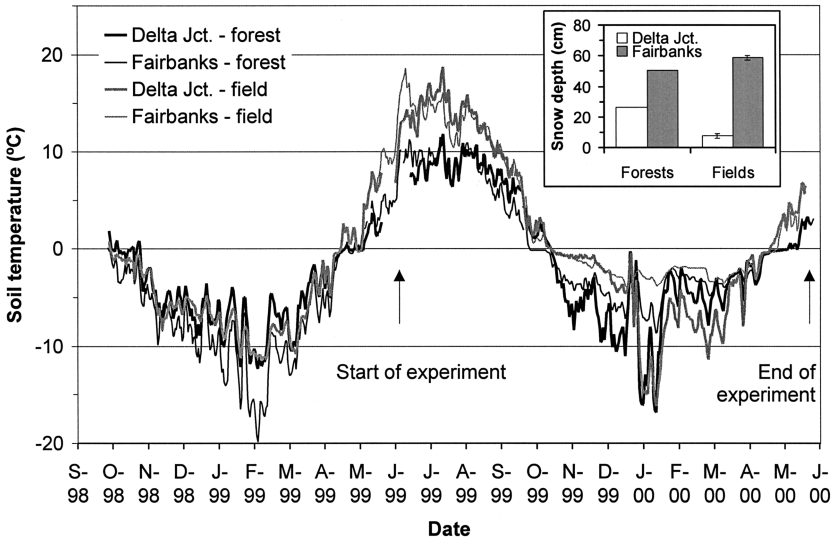

*Figure 1.* Daily soil temperature at 10 cm depth in a forest and a nearby old cropland in Delta Junction and Fairbanks during the experimental period of June 1999–May 2000 and the preceding winter 1998–1999 (no data was available for the Fairbanks field between October 1998 and April 1999). The inset shows snow depth separately for Delta Junction and Fairbanks in one forest and six to seven fields (mean ± one standard error) in mid March 2000 ($P < 0.001$ for the regional effect).

measures ANOVA(excluding mid-July measurements, where data were missing for some land-use types). Net N mineralization and nitrification data of the re-fertilized grasslands were excluded from the analyses. Differences were considered statistically significant, if $P \leqslant 0.05$, and marginally significant, if $0.05 < P \leqslant 0.1$.

## Results

### *Soil physical and chemical properties*

Mean 'summer' soil temperature (early June–late September 1999) at 10 cm depth (organic horizon in the forests, top mineral soil in the fields) was 4–5 °C or 400–500 degree-days higher in fields than in forests (Figures 1 and 2, Table 3). Maximum daily soil temperature in summer was also greater in fields (19 °C) than in forests (12 °C; Figure 1). In 'winter' (late September 1999–late May 2000), forests experienced 100–150 degree-days less than fields, and minimum daily temperatures reached −17 °C in the Delta Junction forest. Soils thawed 2–3 weeks earlier (late April) in fields than in forests (early May; Figure 1), thus lengthening the season of biological activity. Field age and management did not affect soil temperature, except for slightly lower summer temperatures in old grassland vs. old cropland (marginally significant; Figure 2, Table 3).

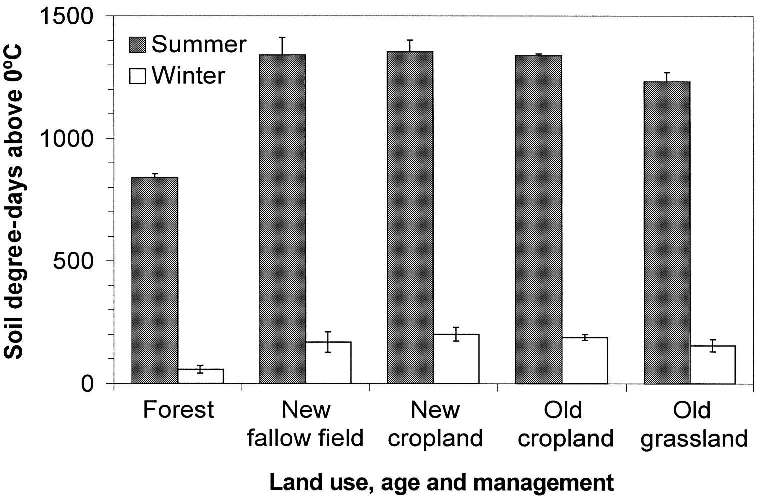

*Figure 2.* Mean summer (June–September) and winter (October–May) temperatures in soils under different land use (mean ± one standard error, $n$ = 2–4 forests or fields). Temperatures were expressed as soil degree-days above 0 °C, and are equivalent to mean temperatures in the summer (no below-0 °C temperatures), but different from mean temperatures in the winter.

During summer 1999, soil temperatures among fields and among forests were similar in the two regions (Delta Junction, Fairbanks; Figure 1). The same trend continued after freezing in early winter, with fields having higher temperatures than forests. However, soil temperatures differed by region rather than land use after late December 1999, with values in Fairbanks being higher than those in Delta Junction. The reason for this change were two thaw events in December 1999, which melted most of the (insulating) snow layer in Delta Junction, but not in Fairbanks (Figure 1 inset). As a consequence, mean winter temperatures in the soils of Delta Junction were lower by 1.2 °C in forests and by 2.5 °C in fields compared to those in Fairbanks. Despite different mean winter soil temperatures, both locations had similar soil degree-days above 0 °C, reflecting concurrent thawing of soils in the two regions.

Volumetric soil water content in summer 1999 was lower in forests than in fields (Figure 3), although the forest soil had a higher mass-based soil moisture than fields (data not shown) because of high organic content and low bulk density of the forest floor (Table 1). Fallow fields were always wetter than planted fields, whether values were expressed per soil volume or weight. Soil moisture remained relatively constant in all fields throughout the growing season, except for a decline in soil moisture in planted fields during a drought period in the first half of July. A significant log-normal relationship was found between mean summer soil moisture (cm$^3$ $H_2O$ cm$^{-3}$ soil) and late-summer aboveground biomass (g m$^{-2}$) across all land-use types including forests [vol. water content = –0.0524 log (biomass) + 0.5955], with biomass explaining 80% of the variation in soil water content. This indicates the importance of transpiration in influencing soil moisture in the dry climate of inte-

*Table 3.* *P*-values for two-way analyses of variance and contrast analyses

| Effect/ comparison | *df* temperature[1] | Soil mass loss | Oat stubble mineralization | Net N nitrification | Net |
|---|---|---|---|---|---|
| *Analysis of variance* | | | | | |
| Land use | 4 | < 0.001 | < 0.001 | 0.001 | 0.003 |
| Season/duration[2] | 1 | < 0.001 | 0.004 | < 0.001 | < 0.001 |
| Interaction | 4 | < 0.001 | 0.962 | 0.005 | 0.008 |
| *Contrast analysis* | | | | | |
| Land use (forests vs. fields) | | | | | |
| summer | 1 | < 0.001 | 0.009 | 0.842 | 0.041 |
| winter/annual[2] | 1 | 0.006 | 0.013 | < 0.001 | 0.121 |
| Field age (new vs. old fields) | | | | | |
| summer | 1 | 0.178 | 0.134 | 0.012 | 0.052 |
| winter/annual[2] | 1 | 0.594 | 0.084 | 0.172 | 0.985 |
| Cropland age (new vs. old cropl.) | | | | | |
| summer | 1 | 0.784 | 0.008 | 0.001 | 0.043 |
| winter/annual[2] | 1 | 0.724 | 0.006 | 0.061 | 0.388 |
| New field manag. (fallow vs. crop) | | | | | |
| summer | 1 | 0.837 | 0.173 | 0.064 | 0.470 |
| winter/annual[2] | 1 | 0.427 | 0.303 | 0.922 | 0.145 |
| Old field manag. (crop vs. grass) | | | | | |
| summer | 1 | 0.075 | 0.041 | 0.115 | 0.050 |
| winter/annual[2] | 1 | 0.342 | 0.021 | 0.069 | 0.003 |

[1] Expressed as degree-days above 0 °C.

[2] Summer – whole year for oat stubble decomposition, summer – winter for the other variables.

rior Alaska. No consistent trend was found between soil water content and temperature. Soil moisture did not differ between regions.

Soil pH was 1–3 units lower in forests than in fields (Table 1; $P = 0.038$ for one-way ANOVA with the factor land use), which was caused by the typically low-pH forest floor as compared to field mineral soils. Organic C and total N concentrations in fields were 10–23% and 20–50% of the concentrations in forests, reflecting the lower organic content of the surface soil in fields compared to forests. There were no significant differences in pH, bulk density, or C and N concentrations among fields of different ages or management (Table 1).

### *Decomposition, soil respiration and nitrogen mineralization*

Decomposition rate of the oat-stubble substrate buried in the surface soil was significantly higher in fields than in forests (Figure 4, Table 3). The 37% mean mass loss of this crop residue during the summer in fields was 25% greater than the average mass loss in forests (30%). An additional, statistically significant 3.5–6% mass loss occurred during the subsequent winter. The ratio of summer to annual decomposition was similar among all land-use types (no land-use × duration inter-

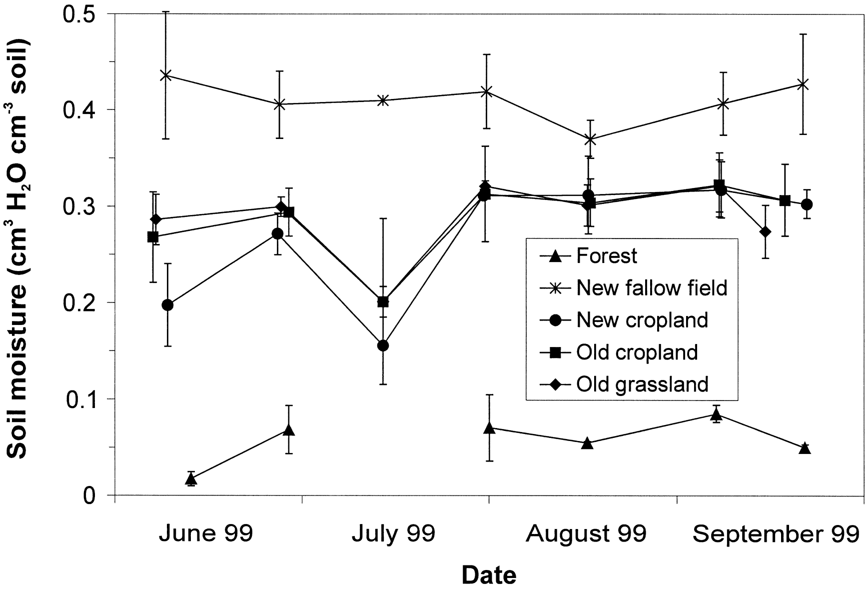

*Figure 3.* Volumetric soil water content under different types of land use during summer 1999 (mean ± one standard error, $n$ = 2–4 forests or fields; only one fallow field in mid July). Repeated-measures ANOVA showed marginally significant differences among sites ($P$ = 0.073). All univariate tests were statistically significant ($P$ = 0.003 to < 0.001).

action, Table 3). Field age and management also influenced decomposition. Mass loss was 27% more rapid in old than in young croplands, and 18% more rapid in old croplands than in old grasslands. Bromegrass stubble left in litter bags on the soil surface of grasslands lost 42% of the initial mass during summer, and thus tended to decompose faster than oat stubble buried in grassland soil (marginally significant, $P = 0.067$; Figure 4). During winter, however, surface bromegrass stubble only lost an additional 2.5% as compared to the 3.5–6% mass loss of oat stubble over the same period. No change in total C and N concentration of $O_a$ horizon material was found at any of the sites over the summer (data not shown), so we assume that no detectable C loss occurred in this material.

Soil respiration (i.e. the respiration of roots and microorganisms) was significantly influenced by land use (Figure 5). Soil respiration was higher in fields than in forests in late summer, with rates being twice as great on new fallow soils and four to five times as great in old grasslands than in forests. Field age and management regime also had a large effect on soil respiration. Young planted soils had lower soil respiration than young fallow soils over most of the season despite the fact that roots contributed to soil respiration in the planted fields. This indicates that decomposition of soil organic matter was more rapid in the fallow than in the young planted fields. Among planted croplands, higher soil respiration in the old than in the new croplands might be partly related to the larger aboveground biomass (430–600 g $m^{-2}$ vs. 250–310 g $m^{-2}$) and presumably larger belowground bio-

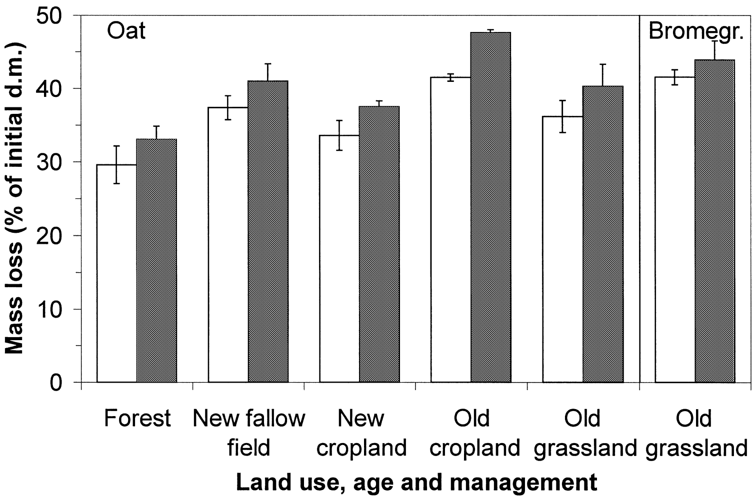

*Figure 4.* Mass loss of oat stubble (buried at 10 cm depth) and bromegrass stubble (at soil surface in grasslands only) in forest and various field types over summer 1999 (open bars) and over the whole year (shaded bars) (mean ± one standard error, *n* = 2–4 forests or fields).

mass in the former. Greater soil respiration in old grasslands than in old croplands probably also resulted from differences in belowground biomass, which was 1300 ± 270 g $m^{-2}$ in grasslands and 180 ± 30 g $m^{-2}$ in croplands (means ± one standard error, *n* = 4 fields) in late summer 1998 (Grünzweig et al., unpublished data).

Land use had a large effect on in-situ net N mineralization and net nitrification. In summer, net N mineralization rate was similar in forests, newly cleared fallow fields and old grasslands (Figure 6), despite large differences in soil properties and soil environment. Young croplands displayed an 80% reduction in N mineralization rate compared to fallow fields, and a 90% reduction compared to old croplands (Table 3). Winter rates of net N mineralization were about an order of magnitude less than in summer when calculated on a daily basis (Figure 6). Because of the length of the winter period, the total amount of N mineralized in winter (calculated over the entire period of 250 days) was substantial: 17–20% of the amount mineralized during summer in old croplands, grasslands and fallow new fields, and 100% of the amount mineralized during summer in new croplands. The pattern of site differences in net N mineralization also differed between summer and winter. Whereas forests mineralized substantial amounts of N in summer, they immobilized N in winter. Fallow and planted croplands were similar to one another in winter net N mineralization, in contrast to their differences in summer (marginally significant; Table 3, Figure 6). Net nitrification was close to zero in forests in both seasons (Figure 6). In contrast, fields had substantial net nitrification rates, except for grasslands with very low rates during winter. Winter net nitrification rates were an order of magnitude lower than summer rates, and were highest in old croplands in both seasons.

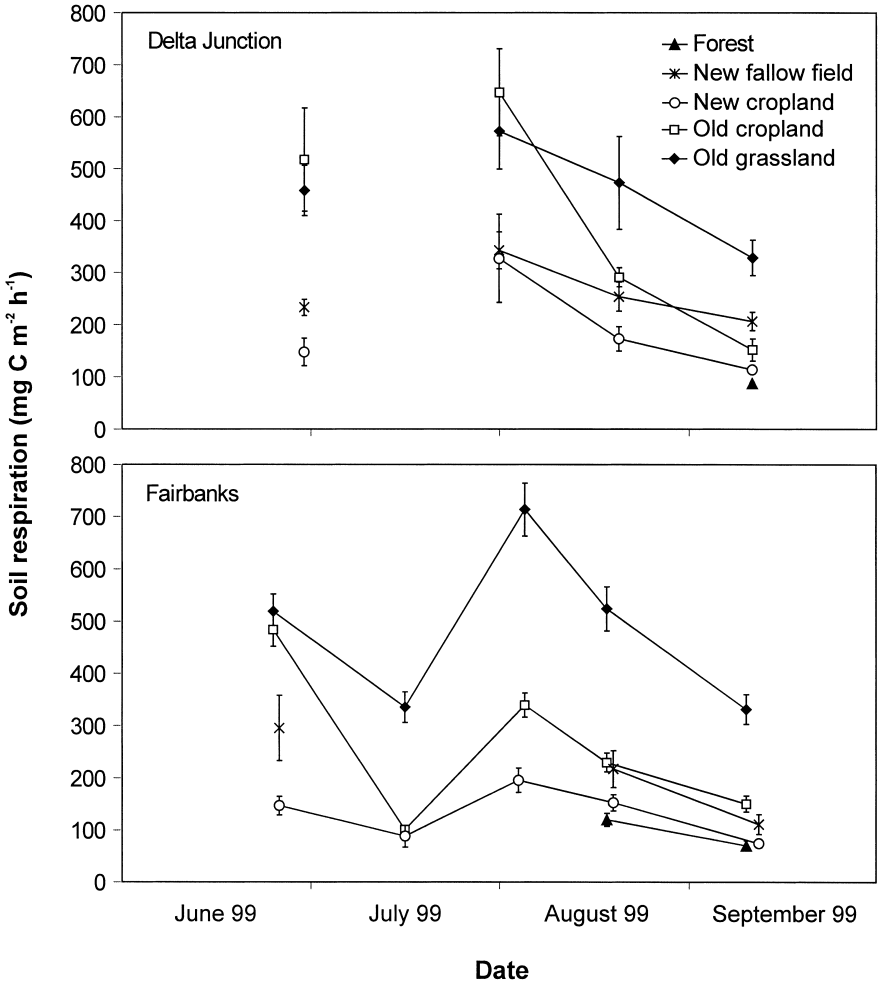

*Figure 5.* Soil respiration at selected sites during summer 1999. Mean ± one standard error, *n* = 6 replications within forest/field. Repeated-measures ANOVA for all fields except the fallow field in Fairbanks showed significant differences among sites for both Delta Junction ($P < 0.001$) and Fairbanks ($P < 0.001$). For each location, all univariate tests were statistically significant ($P = 0.004$ to $< 0.001$).

Net N mineralization correlated with mass loss of oat stubble in summer 1999 ($P = 0.006$; Figure 7). This indicates that the effect of land use on net N mineralization of site-specific soil was similar to its effect on decomposition of the common substrate (lowest in young croplands and highest in old croplands). The variation in decomposition explained by net N mineralization increased from 48% to 63% when only fields were analyzed.

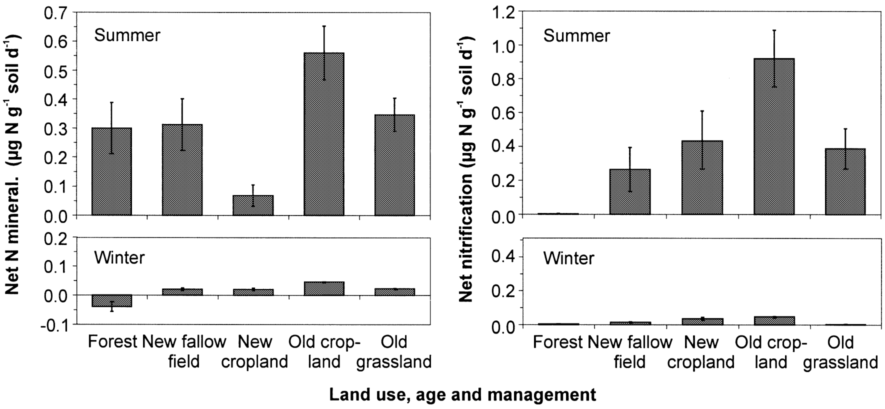

*Figure 6.* In-situ net N mineralization and net nitrification rates in forest and various field types over summer 1999 and over winter 1999/2000 (mean ± one standard error, *n* = 2–4 forests or fields). In the two re-fertilized grasslands (data not included in the figure), summer net N mineralization rates were 0.70 ± 0.10 and 0.72 ± 0.14 g N $g^{-1}$ soil $d^{-1}$ (mean ± one standard) for the periods June–July and July–September, respectively. The respective net nitrification rates in the two fields were 0.86 ± 0.16 and 0.65 ± 0.10 g N $g^{-1}$ soil $d^{-1}$.

### *Effect of environmental factors on carbon and nitrogen mineralization*

Decomposition of oat stubble correlated weakly with soil temperature (expressed as the sum of soil degree-days), resulting mainly from differences between forests and fields. Temperature accounted for 27% of the variation in mass loss of the oat stubble over the summer ($P = 0.038$; data not shown) and for 26% over the whole year ($P = 0.044$; Figure 7). Field temperatures differed by up to 400 degree-days annually (statistically significant at $P = 0.027$), but temperature could not explain differences in stubble decomposition once forests were disregarded.

Soil respiration did not significantly correlate with temperature in fallow fields (Figure 8). However, in planted fields (new and old croplands, grasslands), soil respiration increased exponentially with increasing temperature. A single exponential model explained 90–97% of the variation in soil respiration after removing measurements that were taken early in the season (before there was much root biomass) or during a drought in July (29–57% reduction in soil respiration compared to other summer rates).

In-situ net N mineralization and net nitrification showed no correlation with soil temperature in summer (data not shown), but increased with degree-days in winter (Figure 9; $R^2 = 0.36$, $P = 0.014$ for the linear relationship between net nitrification and soil degree-days). No significant correlation was found between winter net N mineralization and winter temperature among fields.

Summer net N mineralization and net nitrification rates increased with increasing concentration of initial inorganic N, i.e. $NH_4$-N + $NO_3$-N concentrations at the beginning of the experiment in spring 1999 (Figure 9). This relation was valid for forests and for those fields that had an initial soil water content above 0.20 $cm^3$ $H_2O$

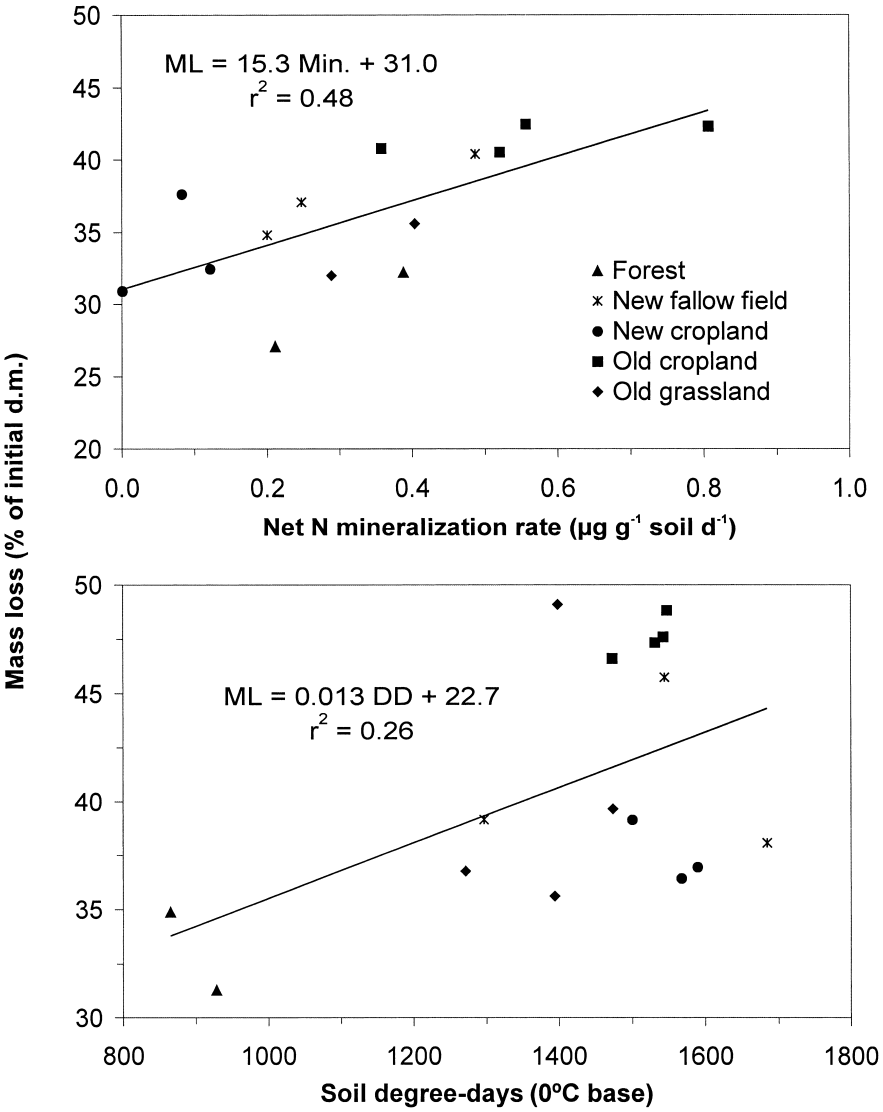

*Figure 7.* Relationship between oat stubble mass-loss (ML) and net N mineralization (Min.) in the summer (top), and relationship between mass loss and effective soil temperature (degree-days, DD) over the whole year (bottom). Data points are means of 6 replications per forest or field.

cm$^{-3}$ soil for net N mineralization and 0.15 cm$^3$ $H_2O$ cm$^{-3}$ soil for net nitrification, but not for fields with drier soils. Initial N concentration was not a significant factor for explaining variation among sites in winter net N mineralization and net nitrification (data not shown).

Soil water content showed no significant correlation with soil respiration (Figure 8). However, soil moisture might control soil respiration and N mineralization,

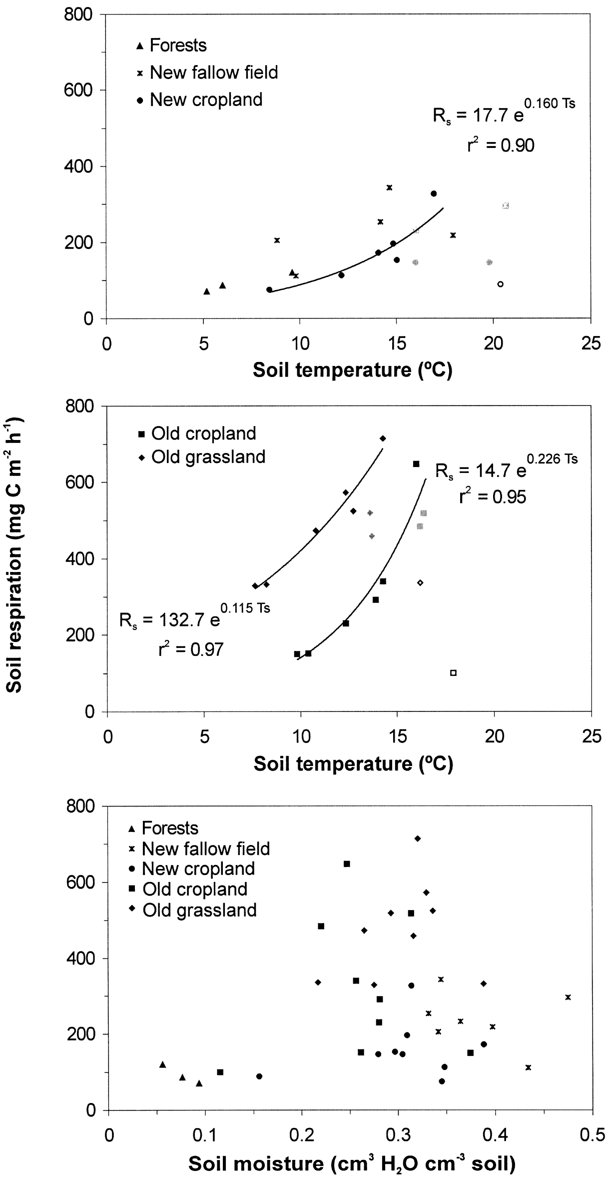

*Figure 8.* Relationship between soil temperature ($T_s$; top and center) or soil moisture (bottom) and soil respiration ($R_s$) under various land-use types in summer 1999. Gray symbols indicate measurements performed prior to July, and open symbols indicate measurements taken during a drought period in mid July. Exponential regressions were fitted to data points indicated by black symbols (respiration measurements performed after mid July) for new croplands (top) and old croplands and grasslands (center). Regression for new fallow fields (top) was not statistically significant. Data points are means of 6 replications per forest or field per date.

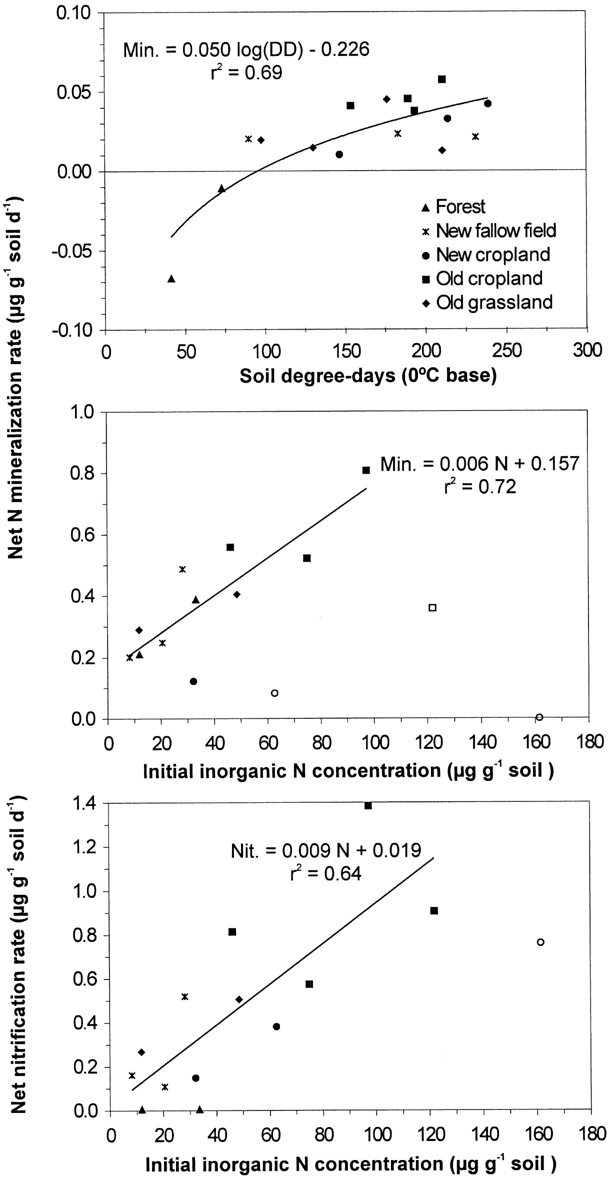

*Figure 9.* Relationship between effective soil temperature (degree-days, DD) and net N mineralization (Min.) in winter 1999–2000 (top chart), and relationships between initial mineral N concentration (N) and net N mineralization (center chart) or net nitrification (Nit.; bottom chart) under various land-use types in summer 1999. Open symbols indicate fields with *in situ* incubations that had an initial volumetric soil water content below 0.20 $cm^3$ $H_2O$ $cm^{-3}$ soil (center) and below 0.15 $cm^3$ $H_2O$ $cm^{-3}$ soil (bottom). These fields were not included in linear regressions. For winter incubations in fields, initial soil water contents were always above 0.20 $cm^3$ $H_2O$ $cm^{-3}$ soil. Data points are means of 6 replications per forest or field.

once values dropped below a certain threshold (see above). No significant correlation was found between stubble decomposition and pH, clay content (for fields only) or late-summer aboveground biomass. Furthermore, neither net N mineralization nor net nitrification correlated with pH, clay content (for fields only) or initial soil moisture (data not shown).

## Discussion

### *Consequences of land use for microclimate and carbon and nitrogen mineralization*

Our results from fully replicated forests and fields showed that land-use type in interior Alaska had a significant effect on all microclimatic and biogeochemical variables tested in this study. Particularly, temperature and moisture in the most active soil layer were increased and biogeochemical processes, including decomposition, soil respiration, in-situ net N mineralization (winter only) and net nitrification were enhanced in fields compared to forests.

Higher temperatures in field than in forest soils during summer probably are a consequence of higher surface heat flux as found by Sharratt (1998) in a field in Delta Junction. Field soil temperatures were also higher in winter, despite a much higher albedo above snow-covered fields (Sharratt 1998). The mid-winter snowmelt events and subsequent drop in soil temperatures in Delta Junction emphasizes the role of snow for insulation of the soil (Figure 1). Climatic warming, which at high latitudes is particularly pronounced in winter (ARAG-USGCRP 1999), might lead to warm spells and mid-winter snowmelt. Paradoxically, this effect could result in a decrease in winter soil-temperatures, particularly in fields.

The more rapid C turnover in field soils resulted in lower C storage compared to the original forests (Grünzweig et al., unpublished data). High-latitude soils contain 40% of the world's reactive soil C (McGuire et al. 1995), so widespread land-cover change at high latitudes could act as a significant source of $CO_2$ to the atmosphere and a positive feedback to climate warming. Substituting differences in sites for differences in time, our results show that enhanced C emissions that accompany forest conversion to agriculture in temperate and tropical areas (Haynes 1986) also occur at high latitudes.

In addition to forest clearing, field age and management significantly affected C and N dynamics. Analyzed over all fields, rates of soil respiration and summer net N mineralization and net nitrification were lower in newly established fields than in old fields. Among croplands of different age, stubble decomposition was also slower in new fields. The slow rates of carbon and nitrogen cycling in young fields could be caused by several factors, such as low root biomass, small amount of labile organic matter and low concentration of inorganic N. Lower rates of soil respiration in new vs. old fields were consistent with lower shoot and presumably lower root biomass in the former. Moreover, young fallow and planted fields normally had smaller amounts of labile organic matter than old crop- and grasslands (Grünzweig et al., unpublished data), which suggests less microbial biomass and lower microbial activity. New fields had little labile organic matter because forest clearing removed most of the easily decomposable C in the forest floor and surface mineral soil, whereas old fields had accumulated considerable quantities of new organic matter. The rate of net N mineralization correlated positively with the size of the labile C pool across different forest types in interior Alaska (Vance and Chapin 2001) and across different North American ecosystems (Zak et al. 1994). Lower rates of net N mineralization in all new fields and of stubble decomposition in new croplands might result from low concentrations of inorganic N. Low rates of oat-stubble decomposition in fields were related to low N mineralization rates, and low net N mineralization rates were related to low initial soil inorganic N concentrations (see also next section).

New fallow fields tended to have higher rates of decomposition, soil respiration, and net N mineralization than young planted croplands of the same age. This is consistent with previous studies in the boreal region that showed more rapid straw decomposition rate in fallow fields than in wheat crops (Shields and Paul 1973). The higher rates of soil respiration in fallow fields than in new croplands were a further indication that fallow fields supported more rapid decomposition than did young croplands. Since roots could not contribute much to soil respiration in the fallow fields (only a few weeds were present), larger $CO_2$ emissions from these soils must have been caused by more rapid decomposition. Enhanced C mineralization in the absence of a crop might, in part, be a consequence of higher soil moisture. Fallow fields depleted soil moisture to a lesser degree, probably because they had low levels of transpiration from the few plants present, and because they were not ploughed, thereby not exposing subsurface soil water to evaporative loss. Higher water content will have its largest effect during dry periods, such as occurred during the first half of July 1999. Rochette et al. (1992) reported soil respiration rates that were 25% higher in fallow fields than in fields under barley, and attributed this result mainly to the higher soil water content. In addition, competition between microorganisms and plant roots for inorganic N could reduce decomposition rates in planted fields (Hu et al. 2001). Under such conditions, less inorganic N would be available for microbial activity.

Among old fields, croplands had higher rates of decomposition and net nitrification than grasslands. The higher soil respiration in grasslands than in croplands must therefore reflect greater root than microbial respiration. This conclusion is supported by the seven-fold greater root and rhizome biomass in grasslands compared to croplands. Low net nitrification in grasslands during winter suggests that N loss by nitrate leaching during snowmelt would be minimal. Nitrogen loss through leaching of nitrate is negligible in summer, when evapotranspiration exceeds precipitation in interior Alaska (Knight and Sparrow 1993). However, during snowmelt in early spring, large amounts of water pass through the soil profile eventually draining nitrate below the rooting zone. In croplands with relatively high nitrification rates, N loss will likely be more substantial than in grasslands (and forests) with their low rates of net nitrification.

The results from the fields suggest that management has a significant effect on C storage in subarctic Alaska. Planting young fields rather than fallowing them could reduce rates of C loss from these newly converted lands on one hand, and would add new C to soils on the other hand. Over the long term, it seems that cultivating perennial vs. annual crops would lead to higher soil organic C pools.

### *Environmental factors controlling carbon and nitrogen mineralization*

Soil temperature correlates strongly with rates of nutrient cycling and productivity in Alaskan boreal forests (Van Cleve et al. 1983). Artificial warming of soils in the coldest sites increased forest floor decomposition and N content (Van Cleve et al. 1990), indicating the key role soil temperature plays in boreal ecosystems. In our study, however, soil temperature had a surprisingly small effect on decomposition and explained less than 30% of the variation among all sites. Statistically significant relationships between soil degree-days and C and N dynamics were always driven by low soil temperature in forests, suggesting that the increase in soil temperature following forest clearing is one cause for enhanced C and N mineralization rates in fields compared to forests. Temperature was relatively unimportant, however, in explaining differences among fields. For net N mineralization and net nitrification, temperature explained more of the variation among sites in winter, when rates were low, than in summer. In boreal forests, microbial activity can be observed down to about −5 °C (Flanagan 1978; Persson et al. 2000). We therefore tested relationships between soil degree-days based on a minimum of −5 °C and annual stubble-mass loss, winter net N mineralization and winter net nitrification. We found that these relationships were always weaker than the corresponding ones with degree-days at the 0 °C-base, and that they were never statistically significant.

Temperature could not explain differences among fields. Stubble-mass loss, net N mineralization, and net nitrification did not correlate with soil temperature, whether it was tested over summer, winter, or annual datasets. The lack of a significant correlation of temperature with decomposition was confirmed by the respiration data. Soil respiration (a combination of root and microbial respiration) increased exponentially with temperature only in fields that were well covered with plants, but not early in the season (when root systems were still small) and not in nearly plantless fallow fields. This indicates that soil respiration is considerably more sensitive to temperature in the presence of a developed root system than when roots are absent (Boone et al. 1998). Microbial respiration, and thus decomposition, seem to be largely insensitive to temperature in cultivated fields of interior Alaska. Fields differed by 280 degree-days or 2.4 °C during summer, which is probably at the upper limit of temperature changes expected in Alaska during the next decades (ARAG-USGCRP 1999). It is therefore unlikely that climatic warming will enhance decomposition and N mineralization following cultivation.

Net N mineralization rates during the summer closely correlated with initial soil inorganic N concentrations, suggesting that net N mineralization was limited by the availability of inorganic N. This would logically occur, if microbes absorbed (immobilized) most N under conditions of high C availability. This condition is likely considering the high concentrations of total organic C (Table 1) and the relatively large amounts of labile C (0.2% of total soil mass on average; Grünzweig et al., unpublished results). The correlation of oat-stubble decomposition with N mineralization rate suggests that decomposition may also have been N-limited in these subarctic soils. Decomposition is often N-limited in soils of croplands (Mary et al. 1996; Henriksen and Breland 1999) and grasslands (Hunt et al. 1988), where herbaceous plant residues provide a source of labile C. In black-spruce forest floor, decomposition was constrained both by low inorganic N and by low quality of soil organic C (Vance and Chapin 2001).

External sources of N, such as fertilization and atmospheric N deposition might increase soil N availability, and might enhance decomposition and C loss to the atmosphere. N deposition was low during the past 20 years in interior Alaska (in the order of 0.2 kg $ha^{-1}$ $yr^{-1}$ for wet deposition; NADP 2002), but was considerably higher in more polluted boreal regions (e.g. 14 kg $ha^{-1}$ $yr^{-1}$ of total N deposition in Canada, 12 kg $ha^{-1}$ $yr^{-1}$ in Finland; Mäkipää et al. 1999; Köchy and Wilson 2001). These values are expected to rise during the next decades (Galloway et al. 1995). If boreal forests would be submitted to N deposition in the order of magnitude that some temperate regions are (around 40 kg $ha^{-1}$ $yr^{-1}$), an increase of available N in the forest floor by 15–20 μg $g^{-1}$ could be expected, eventually resulting in more than doubling of annual net N mineralization (Lamontagne and Schiff 2000). Such changes will have a pronounced effect on decomposition in forests and fields of the boreal region (see Figure 7).

Soil moisture appears to have a regulating effect on heterotrophic processes, once it is lower than a critical value. Low soil moisture limited net N mineralization and net nitrification below thresholds of 0.20 and 0.15 $cm^3$ $H_2O$ $cm^{-3}$ soil, respectively. Soil respiration was significantly reduced during the mid-July drought period, in which soil moisture dropped below 0.16 $cm^3$ $H_2O$ $cm^{-3}$ soil in croplands and below 0.22 $cm^3$ $H_2O$ $cm^{-3}$ soil in an old grassland. In temperate forests and in agricultural fields, dry conditions prevent temperature-dependent processes, such as soil respiration, from responding to changing temperatures (Davidson et al. 1998; Rochette et al. 1999). High-latitude warming appears to have reduced forest growth in interior Alaska as a result of increased water stress (Barber et al. 2000), and could have a similar effect on biogeochemical processes following clearing of boreal forest. If soil moisture dropped below a critical value for microbial activity, organic matter decomposition might slow down (Fox and Van Cleve 1983). With no change in precipitation, climatic warming could inhibit rather than enhance heterotrophic processes with low sensitivity to temperature and a threshold for soil moisture.

## *Conclusions*

Change in land use from black spruce forests to cultivated fields enhanced C and N mineralization. Moreover, increasing field age and management intensity enhanced rates of these biogeochemical processes, particularly in annually cropped old fields. Increased N availability played a key role in these biogeochemical changes through its effect on rates of N mineralization (a positive feedback to enhanced N supply), which in turn correlated with decomposition. We suggest that any source of N, such as fertilization and atmospheric N deposition, will enhance C loss from these ecosystems, except when they are limited by low soil moisture. Temperature increase after forest clearing also contributed to enhanced mineralization, an effect that might in part be caused by lengthening of the season. However, temperature did not affect mineralization among different field types. Based on these findings, understanding and projecting the impacts of land-use change at high latitudes requires an improved understanding of its interactions with a changing climate and N deposition.

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