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Impact of Agricultural Land-use Change on Carbon Storage in Boreal Alaska

JOSÉ M. GRÜNZWEIG*, STEPHEN D. SPARROW†, DAN YAKIR‡ and F. STUART CHAPIN III*

Institute of Arctic Biology, University of Alaska Fairbanks, Fairbanks, AK 99775-7000, USA, †Agricultural and Forestry Experiment Station, University of Alaska Fairbanks, Fairbanks, AK 99775-0080, USA, ‡Department of Environmental Sciences and Energy Research, Weizmann Institute of Science, Rehovot 76100, Israel

Abstract

Climate warming is most pronounced at high latitudes, which could result in the intensification of the extensively cultivated areas in the boreal zone and could further enhance rates of forest clearing in the coming decades. Using paired forest-field sampling and a chronosequence approach, we investigated the effect of conversion of boreal forest to agriculture on carbon (C) and nitrogen (N) dynamics in interior Alaska. Chronosequences showed large soil C losses during the first two decades following deforestation, with mean C stocks in agricultural soils being 44% or 8.3 kg m^{-2} lower than C stocks in original forest soils. This suggests that soil C losses from land-use change in the boreal region may be greater than those in other biomes. Analyses of changes in stable C isotopes and in quality of soil organic matter showed that organic C was lost from soils by combustion of cleared forest material, decomposition of organic matter and possibly erosion. Chronosequences indicated an increase in C storage during later decades after forest clearing, with 60-year-old grassland showing net ecosystem C gain of 2.1 kg m^{-2} over the original forest. This increase in C stock resulted probably from a combination of large C inputs from belowground biomass and low C losses due to a small original forest soil C stock and low tillage frequency. Reductions in soil N stocks caused by land-use change were smaller than reductions in C stocks (34% or 0.31 kg m^{-2}), resulting in lower C/N ratios in field compared with forest mineral soils, despite the occasional incorporation of high-C forest-floor material into field soils. Carbon mineralization per unit of mineralized N was considerably higher in forests than in fields, which could indicate that decomposition rates are more sensitive in forest soils than in field soils to inorganic N addition (e.g. by increased N deposition from the atmosphere). If forest conversion to agriculture becomes more widespread in the boreal region, the resulting C losses (51% or 11.2 kg m^{-2} at the ecosystem level in this study) will induce a positive feedback to climatic warming and additional land-use change. However, by selecting relatively C-poor soils and by implementing management practices that preserve C, losses of C from soils can be reduced.

Keywords: δ^{13}C analysis, chronosequence, cultivation, deforestation, nitrogen mineralization, soil organic matter quality

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Introduction

Among components of global change, land-use conversion has the greatest impact on terrestrial ecosystems, profoundly altering land cover, biota and biogeochemical cycles (Walker & Steffen, 1999). Historically, the most important change of land use was the expansion of agriculture (Houghton, 1999), with most of the

Correspondence: José M. Grünzweig, Department of Environmental Sciences and Energy Research, Weizmann Institute of Science, Rehovot 76100, Israel, fax + 972 8 9344124, e-mail: jose.gruenzweig@weizmann.ac.il

conversion to agriculture occurring in the tropics during the last 150 years (Houghton, 1995a, 1999). At high latitudes, rates of deforestation amounted to 0.7×10^6 ha yr^{-1} in the late 1980s (Dixon *et al*., 1994), with several regions losing relatively large areas of boreal forests. For example, the annual rate of deforestation in the boreal plains of Saskatchewan, Canada was 0.87% yr^{-1} during the mid-1960s to the mid-1990s (Hobson *et al*., 2002), more than twice the current mean annual global deforestation rate of 0.38% yr^{-1} (FAO, 2001). Land-use change affects 30–50% of the boreal forest in the Russian Plain (Alayev *et al*., 1990) and 73% of the formerly forested area in the boreal plains of Saskatchewan (Hobson *et al*., 2002).

Climatic warming is most pronounced at high latitudes, with Alaska experiencing a temperature increase of about 0.3 °C per decade during summer and up to 2 °C per decade during winter in the last 30–40 years (Maxwell, 1997; Weller *et al*., 1999; Barber *et al*., 2000; Serreze *et al*., 2000). While plant production is constrained at high latitudes by a short growing season and low temperatures, the warming trend lengthens the growing season and reduces temperature constraints on high-latitude agriculture (Van Cleve *et al*., 1983; Parry, 1992). The northward intensification of agricultural, forestry and mining activities, which already resulted in increased population in Canada's mid-north and even in arctic areas (McCarthy *et al*., 2001), is possibly related to the temperature trend. The potential for deforestation and expansion of agricultural activity in the boreal region is increasing due to the expected further rise in temperature (Cohen, 1997; Watson *et al*., 1997; Weller *et al*., 1999; McCarthy *et al*., 2001). Warming of 1.5–3.5 °C by the year 2030 and of 4–10 °C by the year 2100 is predicted for Alaska (Weller *et al*., 1999). Climate change is expected to further lengthen the frost-free season by 1–9 weeks (McCarthy *et al*., 2001), which might increase the competition for land in the region (Parry, 1992). Under such conditions, deforestation and agriculture can be increased by population growth or by better economic conditions independent of population size, such as improved infrastructure, profitability, technology and land holdings (Fitzsimmons, 2002). Additionally, an increase in disturbance by fire and insect outbreaks is expected in a warmer climate (Starfield & Chapin, 1996; Weller *et al*., 1999), possibly leading to additional losses of forests.

The potential effect of land-use change on the C cycle 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 stock (Anderson, 1992; Houghton, 1995b), due to their high soil C content per unit area (Dixon *et al*., 1994; McGuire *et al*., 1995; Houghton *et al*., 1999). Among all forests on Earth, high-latitude forests (50–70° latitude) account for 25% of vegetation and 60% of soil C stocks (Dixon *et al*., 1994). Some high-latitude ecosystems are already acting as a C source to the atmosphere as a consequence of climatic warming (Oechel *et al*., 2000), but land-use change could strongly enhance this trend (Dixon *et al*., 1994). In addition, disturbance increases the sensitivity of CO_2 exchange to temperature, with disturbed high-latitude ecosystems being a source of CO_2 during warm, dry years (Zimov *et al*., 1999).

Forest harvests and conversion of forests to agriculture not only reduce C stocks in vegetation but can also cause significant losses of soil organic C. Soil C stocks may decrease because the input of organic matter decreases and the decomposability of plant residues increases (Post & Kwon, 2000). In addition, decomposition of soil organic matter is enhanced as a result of a reduction in its physical protection following tillage and a rise in soil temperature after removal of vegetation and organic soil (Sharratt, 1998; Post & Kwon, 2000).

In the present study, we investigated the effect of land-use change on ecosystem C stocks and on soil C and N dynamics in interior Alaska. We used the chronosequence approach (Jenny, 1980) by comparing fields of different ages (time after forest clearing) to assess the time course of changes induced by forest conversion to cultivated land. Additionally, we used stable C isotopes and assessed soil-C quality to follow processes that resulted in changes in C stocks.

Materials and methods

Study areas

We selected chronosequences of forests and agricultural fields in interior Alaska. The sites were concentrated in four locations along the central part of the Tanana Valley, the region with the most extensive agricultural development in Alaska: Nenana (64°40′N, 148°56′W; elevation 110 m), Fairbanks (64°51′N, 147°43′W; elevation 125 m), Two Rivers (64°50′N, 146°45′W; elevation 200 m) and Delta Junction (64°10′N, 145°44′W; elevation 350–370 m; Fig. 1). The mean annual temperature and precipitation are similar among study areas (−4.1 °C and 274 mm for Nenana, −2.5 °C and 315 mm for Fairbanks, −2.5 and 345 mm for Two Rivers and −2.2 °C and 296 mm for Delta Junction). The mean summer temperature (June–August) in the respective areas is 13.8, 14.5, 14.5 and 14.1 °C. By sampling all major agricultural areas in interior Alaska, we could safely extrapolate our results to this broad region of Alaska.

Fig. 1 Location of sites in interior Alaska.

Sites were on old terraces and flood plains of either the Tanana River or one of its tributaries. Undisturbed vegetation included forests dominated by black spruce (*Picea mariana*), the main climax community on lowlands (Viereck *et al.*, 1986, 1993), and forests dominated by white spruce (*P. glauca*), another common late-successional community in this landscape. Stand characteristics for the main tree species and genera at the different sites are listed in Table 1. Understory species included shrubs, such as Labrador tea (*Ledum groenlandicum*), lingonberry (*Vaccinium vitis-idaea*), dwarf blueberry (*V. uliginosum*) and wild rose (*Rosa acicularis*) and grasses, such as bluejoint reedgrass (*Calamagrostis canadensis*). Soils in both forest types are characterized by a thick organic horizon covered with mosses (mainly *Pleurozium schreberi*, with some *Hylocomium splendens* and *Sphagnum* spp.) and lichens.

Forests were either on Gelisols (permafrost-dominated soils) or Inceptisols (soils with minimal soil development) that often had gravel within the top 2 m. Forests were typically cleared by bulldozers shoving trees, understory plants and parts of the forest soil into rows (berms) 30–50 m apart. After raking of large roots from the newly exposed areas, berms were burned several times during the following two to four winters. Fields were normally plowed, fertilized and seeded to crops in the first to third spring after clearing the forest. Forest-floor material remaining on the field after clearing was incorporated into the mineral A_p horizon in the field. We selected fields of different ages that were on Inceptisols, with the exception of the 1-year-old field at Nenana I that was on Gelisol. To the best of our knowledge, fields did not receive additions of organic fertilizers, such as manure. Fields were managed as croplands (annual crops, mainly barley, *Hordeum vulgare*, and oats, *Avena sativa*), grasslands (perennial crops for hay production; mainly seeded to smooth bromegrass, *Bromus inermis*, but often dominated by weedy grasses), or were fallowed (mostly during a few years after forest clearing preceding the first crop). The age of the fields, i.e. the time since clearing of the forests, was obtained from farmers and extension workers and was verified with ground and aerial photographs. The age of the two oldest fields (>20 years) is approximate. Field age was determined according to the number of summers that a field experienced after it was converted from the forest. We sampled an undisturbed forest adjacent to each field. At times, the same forest served as reference for more than one nearby field. The growing season lasts from mid-May to early September, with grain harvests starting in late August. Grasslands are normally cut in early July and, providing there is sufficient grass production, again in late August. Soils are frozen and covered by snow from October to April.

Field and laboratory methodology

Height and diameter at breast height (DBH, at 1.3 m) of trees higher than 1.3 m was measured in four to five randomly selected circles of 2–3 m radius per site (diameter at stem base for $\geq$1 m high dwarf birch, *Betula glandulosa*). The aboveground tree mass was calculated using allometric equations for black spruce, trembling aspen and white spruce with DBH <6 cm (Manning *et al.*, 1984), for white spruce with DBH> 6 cm (Yarie & Van Cleve, 1983), and for willows (Clark *et al.*, 1986; general equation for soft hardwoods). We developed the following equation for dwarf birch at the Nenana I site:

$$\text{biomass (kg)} = -0.01165 + 0.02613 D_b{}^2 + 0.01713 D_b{}^2 H, \quad (1)$$

where D_b is the diameter at stem base (cm), and H the height (m); $n = 20$, $r^2 = 0.94$. The dry weight of dead trees was calculated as live-tree mass without foliage (Yarie & Van Cleve, 1983; Manning *et al.*, 1984; Clark *et al.*, 1986; Roth & Davis, 1991). To convert tree mass to C stocks, a C content of 50% (w w^{-1}) was assumed. Aboveground biomass of the understory (<1.3 m; including green moss) was sampled together with the organic layer of the soil using a 20 cm diameter corer (see below). Vegetation in fields was sampled in late August 1998 before harvest, and again as stubble after harvest. Dry weight of the first grassland harvest was estimated from farmers' records. Root biomass for crops was determined from cores at 0–40 cm (5 cm diameter) and 40–100 cm (2 cm diameter) depths, and data included only live roots (visual distinction of live from dead roots was possible). For total ecosystem C

Table 1 Stand characteristics of forests

Site (forest type)*	Mass of trees ($kg\,m^{-2}$)	Mass of understory and moss ($kg\,m^{-2}$)	Main tree species or genera*	Tree density (ha^{-1})		Basal area ($m^2\,ha^{-1}$)		Mean tree height (m)	Mean DBH (cm)
				Live trees	Dead trees	Live trees	Dead trees		
Nenana I (BS)	0.85 (0.25)	0.51 (0.17)	BS	3980 (1040)	480 (320)	29.2 (8.6)	0.3 (0.2)	4.2 (0.8)	4.7 (0.9)
			W	2860 (740)	5250 (2540)	0.7 (0.2)	4.3 (1.6)	1.5 (0.2)	0.5 (0.2)
			Al	320 (190)	–	0.2 (0.1)	–	2.3 (0.9)	1.4 (0.6)
Nenana II (BS)	2.53 (0.79)	0.69 (0.06)	BS	1910 (600)	–	8.1 (2.5)	–	3.1 (0.1)	4.8 (0.4)
			DB	14640 (3700)	–	1.3 (0.3)	–	1.3 (0.1)	0.3 (0.0)
			W	2230 (730)	5570 (2060)	1.1 (0.4)	3.2 (1.2)	1.8 (0.4)	1.1 (0.3)
Fairbanks (BS)	9.42 (1.53)	0.59 (0.09)	BS	24830 (1450)	1430 (390)	56.6 (11.4)	1.5 (0.6)	3.6 (0.3)	3.3 (0.3)
Two Rivers (BS)	4.44 (1.61)	0.57 (0.29)	BS	8950 (2630)	–	10.4 (4.0)	–	2.8 (0.4)	2.9 (0.5)
			As	4380 (1790)	1190 (1190)	14.0 (7.0)	1.6 (1.6)	5.2 (0.5)	4.8 (0.4)
Delta Junction I (BS)	3.95 (1.62)	0.33 (0.06)	BS	4770 (3660)	–	16.5 (6.8)	–	5.6 (2.3)	7.6 (3.3)
			WS	1430 (640)	480 (320)	6.8 (4.0)	1.2 (1.1)	4.0 (1.7)	5.3 (2.4)
			W	1590 (670)	320 (190)	1.1 (0.5)	0.1 (0.1)	2.7 (0.7)	1.9 (1.0)
Delta Junction II (BS)	5.88 (1.55)	0.78 (0.07)	BS	9750 (2110)	1190 (760)	39.7 (9.8)	1.2 (0.8)	3.8 (0.2)	5.0 (0.3)
Delta Junction III (WS)	13.97 (3.32)	0.30 (0.06)	WS	1980 (430)	–	56.3 (6.2)	–	11.4 (1.4)	13.3 (2.7)
			As	230 (160)	230 (160)	12.6 (10.3)	7.8 (5.1)	13.9 (1.1)	18.4 (4.7)
			W	280 (130)	350 (160)	2.2 (0.9)	1.9 (0.9)	4.8 (0.7)	6.9 (1.2)
Delta Junction IV (WS)	6.64 (1.32)	0.28 (0.08)	WS	6100 (1040)	710 (320)	36.0 (9.4)	0.5 (0.2)	5.3 (0.6)	5.1 (0.9)
			W	2210 (640)	620 (270)	11.7 (4.0)	1.4 (0.8)	4.1 (0.7)	3.8 (0.7)

*BS, black spruce (*Picea mariana*); W, willow (*Salix* spp.); Al, alder (*Alnus* spp.); DB, dwarf birch (*Betula glandulosa*); As, trembling aspen (*Populus tremuloides*); WS, white spruce (*Picea glauca*).

Mass (dry weight) was recorded for all aboveground parts, including dead trees. For number of trees, basal area, mean height and mean diameter at breast height (DBH), live trees of the main species and genera were used. Number of dead trees and their basal area were also listed. Mean (SE), $n = 4$–5 plots per site.

stocks in planted fields, aboveground mass remaining after crop harvest was used, which amounted to less than 1% of the total ecosystem C.

We collected soil cores from all forests and fields in September 1998. Five replications were randomly selected within a 30×50 m area at each site. The forest floor was sampled with a 20 cm diameter corer down to the interface of the organic and the mineral soil. Mineral soil was sampled at 0–20 and 20–50 cm depths with a 5-cm diameter corer, and at 50–100 cm depth with a 2 cm diameter corer. Field soils were sampled like forest mineral soils. A few fields were included only for estimations of total C and N stocks, and they were sampled with a 2 cm diameter corer throughout. Results were shown for the organic horizon in forests and for mineral soil at 0–20 and 20–100 cm depth (pooled from 20–50 and 50–100 cm) in forests and fields. The mineral soil of the Nenana I forest could not be separated into different layers because it was waterlogged. A nearby forest with similar vegetation was sampled for changes in soil properties with depth.

Mineral soils including roots were passed through a 2 mm mesh sieve (larger roots were cut or coarsely ground by a coffee grinder to pass the sieve). Samples were then mixed and air-dried. Roots were included as part of the belowground C stock because it was impossible to exclude all roots as a separate pool. In addition, most roots become part of the dead organic C pool after forest clearing or crop harvest. Soil texture was analyzed by the hydrometer method after dispersion with sodium hexametaphosphate (Sheldrick & Wang, 1993). Soil pH was measured by a glass electrode in the supernatant of a 1 : 10 soil/water suspension. For determination of C and N concentrations, samples were dried at 105 °C, ball milled and combusted in a CN analyzer (NA 1500, Carlo-Erba, Milano, Italy; Sollins *et al.*, 1999); all samples were analyzed in duplicate or triplicate. Carbonate was only detected in the deeper layer of a few fields (not exceeding 0.05% of dry soil mass), and was determined by gas chromatography on samples treated with HCl. Carbon concentrations and stocks presented exclude carbonate-C.

Stable carbon isotope analyses were conducted on CO_2 samples in a ratio mass spectrometer (Optima, Micromass, Manchester, UK or PDZ Europa GEOS 20-20, Cheshire, UK) after quantitative combustion in an elemental analyzer (EA 1108, Carlo-Erba, Milan, Italy or ANCA GSL, Cheshire, UK). Stable carbon isotope ratios were expressed as

$$\delta^{13}\mathrm{C}\,(‰) = (R_{\mathrm{sample}}/R_{\mathrm{standard}} - 1) \times 1000, \qquad (2)$$

where $R = {}^{13}\mathrm{C}/{}^{12}\mathrm{C}$ and the standard was V-PDB. The precision of the ^{13}C analysis was $\pm$ 0.1‰.

Laboratory incubations of mineral soils were used to separate soil organic C into pools of different quality (decomposability; Townsend *et al.*, 1995). Fresh samples (stored at 4 °C for less than 10 days between field sampling and initiation of incubations) were sieved and mixed as described above. Subsamples of 40 g soil at field capacity (Sparrow & Cochran, 1988) were incubated in 500 mL Mason jars in the dark at 20 °C for 1 year. CO_2 concentration in the headspace was measured periodically (once a week during the first 12 weeks of incubation, every 2 weeks during the next 20 weeks and every 4–6 weeks thereafter) by gas chromatography (Shimadzu, Kyoto, Japan). After measurements, jars were aerated and their weight readjusted by adding distilled water. In addition to incubations, subsamples of 2 g dry soil were analyzed for recalcitrant C (Leavitt *et al.*, 1996; Sollins *et al.*, 1999). Briefly, coarse plant remains were removed by floating in concentrated NaCl solution (0.35 g mL^{-1}) and subsequent hand picking under a microscope. After sieving through a 150 μm mesh, soils were treated with 6 N HCl, and the mixture was heated to boiling in a block digester for 16 h. Recalcitrant C was then analyzed in the CN analyzer. This method consistently yields a fraction of resistant organic matter that is old in age and has a very long turnover time (Paul *et al.*, 1997). The pool size of labile and intermediate C was estimated by fitting the following equation to the results obtained from the incubations (Wieder & Lang, 1982; Robertson *et al.*, 1999) using non-linear regression statistics:

$$C_{\mathrm{min}} = k_{\mathrm{l}}C_{\mathrm{l}}\mathrm{e}^{-k_{\mathrm{l}}t} + k_{\mathrm{i}}C_{\mathrm{i}}\mathrm{e}^{-k_{\mathrm{i}}t}, \qquad (3)$$

where C_{min} is the rate of CO_2-C emission during incubations (mg g^{-1} soil day^{-1}), t the time (days), k_x the decay constant (day^{-1}) and C_x the concentration (mg g^{-1} soil) of labile (x = l) and intermediate C (x = i). For line fitting, C_{i} in Eqn (3) was substituted by $C_{\mathrm{t}} - C_{\mathrm{r}} - C_{\mathrm{l}}$, where C_{t} is the concentration of total organic C and C_{r} the recalcitrant C (both measured as above). Line fits had an r^2 of 0.93 or higher. We assumed that CO_2 emitted during incubations originated from the labile and intermediate, but not from the recalcitrant pool (Townsend *et al.*, 1995). The turnover time was calculated as $1/k$ for labile and intermediate pools.

Potential net N mineralization and net nitrification rates were obtained from laboratory incubations of 5 g fresh mineral soil at field capacity and 20 °C for 12 weeks. For determination of ammonium and nitrate concentrations, soils were extracted in 50 mL 2 N KCl after shaking for 1 h at 180 rpm. Filtered extracts were analyzed colorimetrically for ammonium and nitrate (phenol-hypochlorite method – Berthelot reaction and Cu–Cd reduction – modified Griess–Illosvay procedure, respectively; Keeney & Nelson, 1982) by an

automated rapid-flow analyzer (RFA-300, Alpkem, Clackamas, OR, USA; Sparrow & Cochran, 1988). Microbial biomass was determined in fresh soil using chloroform-fumigation extraction and dichromate-digestion analysis (Vance *et al*., 1987; Paul *et al*., 1999). Estimates of dissolved organic C (DOC) were obtained from controls for determination of microbial biomass (non-fumigated extracts). Moist soil was extracted in 0.5 M KCl, filtered through a 0.4 μm polycarbonate filter, and analyzed by dichromate digestion. The results for DOC were expressed per liter of soil solution using actual field soil moisture.

In situ soil temperature was monitored at the organic–mineral interface in two forests and at 10 cm depth in 14 fields during summer 1999 (early June–early September). Temperature was recorded every half hour using two Hobo H8 loggers (Onset, Bourne, MA, USA) per forest or field.

Calculations and statistical analyses

To track the fate of forest soil C and N after agricultural land-use change, we followed the convention used by many ecosystem scientists (e.g. Post *et al*., 1982) of focusing on the upper 100 cm of the soil profile. This is the layer with the highest concentrations of C and N, and although low amounts of C and N are likely to be present below 100 cm, the largest changes in stocks resulting from deforestation are expected above 100 cm.

All aboveground biomass C was assumed to be lost by combustion because there is no commercial use for wood or fiber in these unproductive lowland forests. Part or the entire forest floor and a shallow layer of the top mineral soil (both layers contain belowground biomass and dead organic C) could have been mechanically removed during forest clearing. The extent of soil removal depended on the season and type of equipment used for clearing, and on preferences of the farmer. If only a part of the forest floor was removed, the remaining deeper organic horizon was ploughed into the A_p horizon of the new field.

Soil C and N stocks in forests were compared with stocks in corresponding fields. However, field soils were normally compacted during forest clearing and cultivation, decomposition of organic matter and/or soil erosion. Thus, the original 100 cm of forest mineral soil covered a shallower soil layer in the corresponding field, and sampling to 100 cm depth included non-target soil from a deeper layer that was not sampled in the forest. To track changes of C and N in the predefined target layer with its fixed soil mass, corrections were necessary to account for the extra soil sampled in fields (Crépin & Johnson, 1993; Davidson & Ackerman, 1993; Veldkamp, 1994; Fearnside & Barbosa, 1998). An additional condition that made corrections necessary was the presence of permafrost and gravel at some sites. These situations prevented sampling to the intended maximal depth of 100 cm, and again resulted in different masses of soil being sampled in forests and corresponding fields.

Corrections were performed according to the principle of 'equivalent soil mass,' i.e., by calculating soil C and N stocks for the same mass of mineral soil in both forests and fields (Davidson & Ackerman, 1993; Veldkamp, 1994; Ellert & Bettany, 1995; Neill *et al*., 1997; Fearnside & Barbosa, 1998). According to this principle, field soil was mathematically adjusted to the mass of the uncompacted mineral soil in the forest by subtracting the extra deep soil collected in the field. Since C and N concentrations decreased with depth, the above subtracted soil mass was multiplied by C or N concentration from the deepest soil layer to yield the following corrected C and N stocks ($kg\,m^{-2}$):

$$M_{fi}' = M_{fi} - (m_{fi} - m_{fom})c_{deep}, \quad (4)$$

where M_{fi} is the uncorrected C or N stock ($kg\,m^{-2}$) in the field, m the total sampled soil mass ($kg\,m^{-2}$) of the field (fi) or forest mineral horizon (fom) and c_{deep} is the concentration of C or N in the deepest layer sampled ($kg\,kg^{-1}$ soil). Corrected field stocks M_{fi}' were compared with corresponding forest mineral stocks (M_{fom}; Table 4, columns 5 and 11) and forest total-soil stocks (mineral plus organic (foo) stocks, $M_{fom} + M_{foo}$; Table 4, columns 7 and 13). This first correction did not use forest-floor material plowed into field soil, and assumed that the entire forest floor was mechanically removed during forest clearing. Yet, forest floor C and N were included as part of the forest-field comparison ('total soil' in Table 4).

For the second correction, we assumed that the entire forest floor was included in the newly established field. In this case, the profile of the field was corrected to the mineral plus organic soil of the forest, as follows:

$$M_{fi}' = M_{fi} - (m_{fi} - m_{fom} - m_{foo})c_{deep}. \quad (5)$$

Corrected field stocks M_{fi}' were compared with $M_{fom} + M_{foo}$ (Table 4, columns 8 and 14). Field stocks as corrected by Eqn (4) (Table 4, columns 7 and 13) were very close to those corrected by Eqn (5) (Table 4, columns 8 and 14) because addition or disregard of forest organic layers (m_{foo}) with their low masses (9–35 $kg\,m^{-2}$) did not make a large difference to corrections of field soil masses (m_{fi}; 800–1400 $kg\,m^{-2}$). The only exception was Nenana I with some slight differences between Eqns (4) and (5) caused by the presence of a thick organic horizon.

To distinguish between changes occurring to soil organic C due to physical removal (forest clearing, soil

erosion) and changes due to other mechanisms, forests and fields were compared for differences in the intermediate + recalcitrant C pool of the top 20 cm of the mineral soil. This approach assumed that no large changes occurred in intermediate and recalcitrant C within the initial years after land-use change, and that changes in C stocks reflected primarily changes in C input and decomposition of labile C. The turnover of labile C typically occurs over one to a few years, that of intermediate C over decades and that of recalcitrant C over centuries to millennia (Parton *et al.*, 1987; Leavitt *et al.*, 1996; Paul *et al.*, 1997). This approach could not be applied to the oldest fields (>20 years) because decomposition can significantly affect the intermediate pool and new crop biomass can add new intermediate and recalcitrant C over these time scales (Follett *et al.*, 1997; Sollins *et al.*, 1999). Carbon stocks in the top 20 cm of field soils were corrected to those in forests according to Eqn (5). Deviation of field intermediate + recalcitrant C stock from that of the corresponding forest was interpreted either as an addition of forest-floor material to field soil (more intermediate + recalcitrant C in fields than in forests) or complete removal of forest floor and removal of a shallow mineral layer during clearing or erosion (less intermediate + recalcitrant C in fields than in forests). The amount of forest-floor C added to fields was calculated from the mass and percentage of intermediate + recalcitrant C in the forest floor. The latter was not analyzed in this study, but was determined by fitting Eqn (3) (C_i was assumed to be intermediate + recalcitrant C) to laboratory incubations of O_{ie} and O_a horizons in a black-spruce forest as performed by Sparrow & Cochran (1988).

Differences in C and N stocks between forests and corresponding fields were tested at each site by one-way ANOVA. Multiple comparisons were performed with the Tukey–Kramer 'honestly significant difference' *post hoc* test. In order to average percent change in C and N stocks (differences between forest and corresponding field) across all or parts of the dataset, we first log-transformed the ratios of fields to forests because ratios and percentages are not normally distributed (Sokal & Rohlf, 1995). The mean log-transformed ratio was backtransformed and the mean percent change was calculated. Changes in soil C stocks and stable isotopes with time after clearing were presented on a logarithmic scale, similar to other studies (Murty *et al.*, 2002), as the rate of these changes is high in young fields and slows with increasing age (Davidson & Ackerman, 1993; Neill & Davidson, 2000). Statistical analyses and line fitting were conducted with Statistica '99 (StatSoft Inc., Tulsa, OK), SYSTAT 7 (SPSS, Chicago, IL, USA), and Origin 7.0 (OriginLab Corporation, Northampton, MA, USA).

Results

Soil properties

Forests in Nenana and Fairbanks were on Gelisols, i.e. soils underlain by shallow permafrost (Table 2). After forest clearing, permafrost melted rapidly during the first few years, which led to a change in soil order to Inceptisol. No such change occurred in the permafrost-free sites in Two Rivers and Delta Junction, where both forests and fields were on Inceptisols.

Soil textural classes ranged from loamy sand to silt loam, with most soils being classified as sandy loams (Table 2). Clay content averaged 8.7%, and did not exceed 20%, which is typical for interior Alaska (Knight & Sparrow, 1993; Van Cleve *et al.*, 1993). A weakly acidic to neutral pH was measured in the top 20 cm of the mineral soils, with no significant differences among sites.

The organic soil layer in forests on permafrost (Nenana, Fairbanks) had considerably higher water content (2.9 ± 0.4 g $H_2O\,g^{-1}$ soil, mean ± SE) than the organic layer on non-permafrost soils (Two Rivers, Delta Junction; $0.9 \pm 0.1\,g\,H_2O\,g^{-1}$ soil; $P < 0.001$; Table 3). No such differences between permafrost and non-permafrost soils were found in the mineral soil. Forest mineral soils, particularly the top mineral soils under black spruce, were wetter than most field soils, whereas soils under white spruce tended to be drier than soils in the corresponding fields. Soil temperature at the organic–mineral interface in forests during summer was low, and averaged $3.8 \pm 1.5\,^{\circ}C$ (mean ± SE, $n = 2$ forests). In field soils at 10 cm depth, temperature was considerably higher, averaging $12.8 \pm 0.2\,^{\circ}C$ ($n = 14$ fields).

Bulk density was 10–30% lower in the top mineral soils of forests compared with the same layer in fields, except for the fields in Nenana I and Fairbanks, where bulk density was higher in forests than in fields (Table 3). Bulk density was normally higher and differences between forests and fields were smaller at depth compared with the top layer. In chronosequences with fields of more than one age group, bulk density tended to increase with time after clearing. Similar trends with land-use change and time since clearing have been observed in other studies. (Davidson & Ackerman, 1993; Keller *et al.*, 1993)

Plant biomass and production

Aboveground tree mass in black- and white-spruce forests was mostly within the range of 1–11 and 6–25 $kg\,m^{-2}$, respectively, found across boreal North America (Viereck *et al.*, 1983; Gower *et al.*, 1997; Harden

Table 2 Soil classification (Soil Survey Staff, 1998) and soil properties in forests and fields

Site	Land use	Time after clearing (years)	Soil classification	pH	Clay content (%)	Textural class
Nenana I	Black spruce forest	0	Typic Histoturbel	6.6 (0.2)	10.0 (1.1)	Sandy loam
	Fallow field	1	Typic Aquiturbel	5.9 (0.1)	6.2 (2.4)	Sandy loam
	Fallow field	3	Aquic Eutrocryept	6.4 (0.1)	9.8 (1.0)	Sandy loam
	Grassland	5	Aquic Eutrocryept	–	–	–
	Grassland	6	Aquic Eutrocryept	6.5 (0.1)	7.0 (0.3)	Sandy loam
Nenana II	Black spruce forest	0	Typic Historthel	6.3 (0.1)	10.0 (0.7)	Sandy loam
	Cropland	6	Typic Eutrocryept	–	–	–
	Grassland	9	Typic Eutrocryept	6.6 (0.2)	18.7 (0.4)	Silt loam
	Cropland	14	Typic Eutrocryept	6.0 (0.2)	10.3 (0.5)	Sandy loam
Fairbanks	Black spruce forest	0	Typic Haploturbel	5.7 (0.2)	11.6 (0.7)	Sandy loam
	Grassland	60	Aquic Eutrocryept	6.8 (0.05)	15.3 (0.5)	Loam
Two Rivers	Black spruce forest	0	Typic Eutrocryept	5.1 (0.1)	14.8 (1.3)	Sandy loam
	Fallow field	1	Typic Eutrocryept	5.2 (0.1)	10.6 (0.4)	Sandy loam
Delta Junction I	Black spruce forest	0	Aquic Eutrocryept	5.9 (0.1)	4.1 (0.5)	Sandy loam
	Fallow field	1	Aquic Eutrocryept	5.8 (0.1)	4.8 (1.3)	Loamy sand
Delta Junction II	Black spruce forest	0	Aquic Eutrocryept	5.5 (0.1)	5.6 (0.5)	Sandy loam
	Cropland	2	Aquic Eutrocryept	5.6 (0.1)	6.4 (0.6)	Sandy loam
	Cropland	15	Aquic Eutrocryept	5.6 (0.04)	5.6 (0.2)	Sandy loam
Delta Junction III	White spruce forest	0	Typic Eutrocryept	5.6 (0.1)	6.7 (0.6)	Sandy loam
	Cropland	3	Typic Eutrocryept	5.7 (0.04)	7.8 (0.8)	Sandy loam
	Cropland	3	Typic Eutrocryept	–	–	–
Delta Junction IV	White spruce forest	0	Aquic Eutrocryept	5.7 (0.1)	4.0 (0.5)	Sandy loam
	Cropland	20	Aquic Eutrocryept	5.8	8.0	Silt loam
	Grassland	40	Aquic Eutrocryept	5.7 (0.2)	5.4 (0.6)	Sandy loam

Soil properties were measured in the top 20 cm of the mineral soil. Mean (SE), $n = 4$–5. Where standard error is lacking, a composite of five subsamples was tested.

Table 3 Changes in moisture, bulk density and C and N concentrations of forest and field soils with depth

Site	Land use	Time after clearing (years)	Layer/ depth (cm)*	Gravimetric soil moisture ($g\,H_2O\,g^{-1}$ soil)	Bulk density ($g\,cm^{-3}$)	Organic C concentration ($mg\,g^{-1}$ soil)	Total N concentration ($mg\,g^{-1}$ soil)	C/N ratio
Nenana I	Black spruce forest	0	O layer	3.27 (0.04)	–	417.7 (12.9)	16.60 (0.21)	25.2 (1.0)
			0–20	0.63 (0.15)	1.05 (0.31)	49.6 (19.5)	2.92 (1.08)	16.4 (0.8)
			20–59	0.34 (0.04)	1.47 (0.12)	12.3 (5.0)	0.86 (0.36)	14.3 (0.6)
	Fallow field	1	0–20	2.50 (0.63)	0.38 (0.14)	220.6 (48.8)	11.34 (2.42)	19.4 (1.5)
			20–93	0.52 (0.07)	1.36 (0.10)	32.1 (8.8)	2.02 (0.55)	15.9 (0.4)
	Fallow field	3	0–20	0.63 (0.12)	0.72 (0.15)	91.8 (26.4)	4.95 (1.32)	17.8 (0.8)
			20–100	0.30 (0.01)	1.42 (0.03)	17.7 (2.6)	1.13 (0.14)	15.5 (0.4)
	Grassland	5	0–20	0.40 (0.03)	0.98 (0.06)	37.4 (6.9)	2.31 (0.40)	16.0 (0.4)
			20–100	0.29 (0.01)	1.19 (0.06)	10.5 (1.1)	0.69 (0.06)	15.3 (0.5)
	Grassland	6	0–20	0.35 (0.06)	1.01 (0.09)	32.1 (10.5)	1.95 (0.62)	16.3 (0.2)
			20–100	0.25 (0.01)	1.34 (0.02)	9.0 (0.8)	0.55 (0.04)	16.2 (0.4)
Nenana II	Black spruce forest	0	O layer	3.32 (0.29)	–	457.7 (5.7)	12.66 (0.51)	36.4 (1.4)
			0–20	0.79 (0.08)	0.80 (0.08)	73.5 (14.0)	4.35 (0.90)	17.2 (0.7)
			20–61	0.38 (0.02)	0.94 (0.13)	18.6 (4.4)	1.15 (0.25)	15.9 (0.3)
	Cropland	6	0–20	0.33 (0.01)	0.98 (0.05)	25.8 (2.1)	1.25 (0.09)	20.7 (0.2)
			20–100	0.25 (0.01)	1.22 (0.05)	3.4 (0.5)	0.26 (0.02)	13.0 (1.1)

(*Continued*)

Table 3 (Continued)

Site	Land use	Time after clearing (years)	Layer/ depth (cm)*	Gravimetric soil moisture ($g H_2O g^{-1}$ soil)	Bulk density ($g cm^{-3}$)	Organic C concentration ($mg g^{-1}$ soil)	Total N concentration ($mg g^{-1}$ soil)	C/N ratio
	Grassland	9	0–20	0.50 (0.05)	0.88 (0.13)	52.2 (9.6)	3.02 (0.51)	16.9 (0.7)
			20–100	0.28 (0.01)	1.48 (0.06)	5.6 (1.0)	0.41 (0.07)	13.6 (0.3)
	Cropland	14	0–20	0.26 (0.01)	1.20 (0.04)	16.4 (1.4)	0.89 (0.05)	18.3 (0.5)
			20–100	0.23 (0.01)	1.44 (0.02)	5.4 (0.5)	0.47 (0.04)	11.6 (0.2)
Fairbanks	Black	0	O layer	2.16 (0.19)	–	375.0 (15.1)	8.58 (0.37)	43.7 (0.7)
	spruce		0–20	0.44 (0.04)	1.51 (0.10)	19.3 (4.8)	0.92 (0.16)	19.9 (1.6)
	forest		20–46	0.33 (0.02)	1.34 (0.26)	9.2 (2.3)	0.44 (0.07)	19.5 (2.2)
	Grassland	60	0–20	0.41 (0.01)	0.79 (0.05)	62.8 (3.8)	3.92 (0.25)	16.1 (0.2)
			20–100	0.25 (0.01)	1.43 (0.06)	20.6 (5.5)	1.25 (0.26)	15.8 (1.1)
Two Rivers	Black	0	O layer	1.19 (0.18)	–	313.9 (30.5)	10.34 (0.86)	30.3 (1.4)
	spruce		0–20	0.45 (0.03)	0.89 (0.08)	35.8 (5.4)	1.73 (0.24)	20.6 (0.4)
	forest		20–100	0.23 (0.02)	1.32 (0.07)	6.4 (0.9)	0.50 (0.05)	12.7 (0.8)
	Fallow	1	0–20	0.39 (0.02)	1.01 (0.03)	29.9 (4.2)	1.51 (0.16)	19.3 (1.1)
	field		20–100	0.29 (0.01)	1.32 (0.04)	7.8 (1.3)	0.65 (0.06)	11.7 (1.0)
Delta Junction I	Black	0	O layer	1.06 (0.18)	–	291.3 (17.3)	7.50 (0.49)	39.5 (3.8)
	spruce		0–20	0.58 (0.03)	0.57 (0.04)	71.9 (8.7)	3.30 (0.35)	21.8 (1.3)
	forest		20–89	0.29 (0.01)	1.32 (0.02)	8.8 (0.6)	0.56 (0.02)	15.5 (0.9)
	Fallow	1	0–20	0.45 (0.05)	0.66 (0.09)	55.8 (8.6	2.96 (0.52)	19.1 (0.6)
	field		20–91	0.27 (0.01)	1.41 (0.03)	12.8 (2.1)	0.78 (0.08)	16.1 (1.3)
Delta Junction II	Black	0	O layer	0.57 (0.11)	–	249.3 (4.1)	6.88 (0.47)	36.8 (2.0)
	spruce		0–20	0.62 (0.03)	0.53 (0.05)	91.0 (8.3)	4.00 (0.37)	22.8 (0.6)
	forest		20–100	0.24 (0.01)	1.16 (0.06)	11.1 (1.2)	0.57 (0.06)	19.4 (1.0)
	Cropland	2	0–20	0.45 (0.01)	0.71 (0.02)	70.0 (3.5)	3.07 (0.15)	22.8 (0.3)
			20–100	0.23 (0.01)	1.21 (0.04)	13.5 (1.9)	0.70 (0.08)	19.0 (0.7)
	Cropland	15	0–20	0.37 (0.01)	0.74 (0.03)	71.1 (3.9)	3.80 (0.25)	18.7 (0.2)
			20–100	0.19 (0.01)	1.38 (0.01)	13.1 (1.1)	0.71 (0.04)	18.3 (0.6)
Delta Junction III	White	0	O layer	0.93 (0.06)	–	311.1 (12.1)	9.85 (0.40)	31.9 (2.2)
	spruce		0–20	0.16 (0.01)	0.72 (0.02)	43.0 (1.3)	1.95 (0.03)	22.1 (0.4)
	forest		20–86	0.08 (0.004)	1.38 (0.04)	5.9 (0.8)	0.42 (0.03)	13.8 (0.9)
	Cropland	3	0–20	0.25 (0.02)	0.81 (0.05)	24.1 (3.5)	1.19 (0.12)	20.0 (0.9)
			20–65	0.19 (0.01)	1.35 (0.04)	3.5 (0.2)	0.31 (0.01)	11.1 (0.6)
	Cropland	3	0–20	0.18 (0.01)	0.83 (0.04)	34.1 (5.0)	1.57 (0.17)	21.4 (0.9)
			20–86	0.12 (0.01)	1.18 (0.07)	4.1 (0.3)	0.33 (0.03)	12.6 (0.4)
Delta Junction IV	White	0	O layer	0.76 (0.1)	–	260.0 (18.5)	7.78 (0.31)	33.8 (3.6)
	spruce		0–20	0.24 (0.03)	0.51 (0.06)	80.6 (14.8)	3.36 (0.47)	23.4 (1.4)
	forest		20–100	0.12 (0.01)	1.08 (0.03)	18.0 (1.4)	0.96 (0.07)	18.9 (0.5)
	Cropland	20	0–20	0.26 (0.01)	0.87 (0.02)	47.2 (1.3)	2.55 (0.05)	18.5 (0.4)
			20–100	0.20 (0.01)	1.01 (0.02)	16.0 (0.4)	0.91 (0.02)	17.6 (0.3)
	Grassland	40	0–20	0.33 (0.03)	0.84 (0.04)	55.6 (4.8)	3.14 (0.17)	17.6 (0.7)
			20–100	0.19 (0.03)	1.22 (0.02)	14.2 (0.7)	0.80 (0.04)	17.7 (0.5)

*Organic layer and depth into mineral soil (measured from organic-mineral interface) down to either 100 cm, permafrost or gravel. A maximal depth of less than 100 cm indicates the top of permafrost (Nenana, Fairbanks) or gravel (Two Rivers, Delta Junction). Mean (SE), $n = 3$–5.

et al., 1997; Table 1). The annual aboveground production in forests was not measured in this study. However, another study in interior Alaska reported a mean aboveground production of $0.11 kg m^{-2} yr^{-1}$ for black-spruce stands and $0.37 kg m^{-2} yr^{-1}$ for white-spruce stands (Viereck *et al.*, 1983). Aboveground production in planted fields as measured in the current study averaged $0.46 \pm 0.05 kg m^{-2} yr^{-1}$ ($n = 12$ fields, ranging 0.16–$0.75 kg m^{-2} yr^{-1}$). Management (cropland, grassland) had no statistically significant effect on production (data not shown). Belowground biomass in grasslands ($1.26 \pm 0.21 kg m^{-2}$ to a depth of 100 cm, $n = 5$ fields; 55% roots and 45% rhizomes) was considerably larger than aboveground biomass, and

seven-fold greater than belowground biomass in croplands ($0.18 \pm 0.03\,kg\,m^{-2}$ to a depth of 100 cm or gravel, $n = 4$ fields; $P = 0.003$).

Soil carbon and nitrogen concentrations

Organic C concentrations in forest soils declined from $337 \pm 24\,mg\,g^{-1}$ soil (mean ± SE; $n = 9$ forests) in the forest floor to $58 \pm 9\,mg\,g^{-1}$ in the top mineral soil and to $11 \pm 2\,mg\,g^{-1}$ in the deep layer (Table 3). Organic C in fields decreased similarly from top to deep layers. Carbon concentrations in the organic and the top mineral soil of the forests correlated positively where permafrost was present (Nenana, Fairbanks; $r^2 = 0.84$, $P = 0.085$, $n = 4$) and negatively where no permafrost was present (Two Rivers, Delta Junction; $r^2 = 0.91$, $P = 0.011$, $n = 5$). Nitrogen concentrations also declined with depth, but to a lesser extent than C. Consequently, C/N ratios decreased with depth, probably reflecting the loss of C from the more highly decomposed organic matter at depth. C/N ratios of top and deep soils were wider in most forests compared with the same layers in the corresponding fields, except for the sites in Nenana where no such trend was observed. Nitrogen concentrations across all soil layers in forests and fields correlated highly and positively with C concentrations [$\log(N) = 0.8161 \log(C) - 2.2969$, $r^2 = 0.98$, $P < 0.001$, $n = 62$], thus reflecting variation in soil organic matter among soil layers. Variation in C concentration of the top mineral soil was unrelated to site variation in clay content or pH (data not shown).

Carbon and nitrogen stocks

Carbon inventories of soils under black-spruce forests (organic and mineral soil) was in the range 15.1–$24.4\,kg\,m^{-2}$, similar to C stocks in other black-spruce forests in Alaska and Manitoba (Viereck *et al*., 1983; Rapalee *et al*., 1998; Table 4). Changes in total soil C stocks (as corrected by Eqn (5)) caused by deforestation ranged from 75% loss to 44% gain, or in absolute terms $17.0\,kg\,m^{-2}$ loss to $6.9\,kg\,m^{-2}$ gain, with the Fairbanks grassland being the only field that showed net C accumulation (marginally significant, $P = 0.060$; Table 4). These changes averaged −44% or $-8.3\,kg\,m^{-2}$ across fields 3 years or more after clearing. Reductions in C stocks in the mineral and the total soil were significant at least in some fields in Nenana I and II and Delta Junction III and IV. Plotting the relative change in soil C stocks against time after clearing yielded a statistically significant trend of decrease in C stocks during the first 10–15 years followed by an increase over the subsequent decades (Fig. 2a). This trend was strongly influenced by the oldest field, but even when this field was omitted from the analysis, the same model was still statistically significant ($r^2 = 0.50$, $P = 0.046$).

Aboveground C stocks comprised 2–6% of total ecosystem C in forests at the Nenana sites and 10–35% at the other sites. All of the aboveground forest mass was assumed to be burnt when clearing forests, which resulted in total changes in ecosystem C stocks of -18.4 to $+2.1\,kg\,m^{-2}$ (Table 4). Changes in total ecosystem C across all fields that were 3 years or older averaged −51% or $-11.2\,kg\,m^{-2}$ of the initial ecosystem C stock.

Changes in N stocks were proportional to changes in C stocks ($r^2 = 0.91$, $P < 0.001$, $n = 16$), with N losses being smaller than C losses and N gains being larger than C gains (Table 4). Some field soils in Nenana I and II and Delta Junction III lost considerable amounts of N, while the field in Fairbanks gained N by almost 140% as compared with the entire forest soil profile. Changes in N stocks averaged across fields 3 years or more after clearing amounted to −34% or $-0.31\,kg\,m^{-2}$. C/N ratios in the entire mineral soil (calculated as C stocks over N stocks) tended to be wider in most forests (18.4 ± 0.7 mean ± SE) than in corresponding fields (17.1 ± 0.4), but differences were statistically significant only in some fields (Table 4).

Stable carbon isotope analysis

Black-spruce tissues became progressively enriched (i.e. less negative) in ^{13}C from needles (-27.4 ± 0.3‰) to twigs (-26.6 ± 0.1‰) and trunk (−24.7‰). Annual (oats, barley) and perennial crops (bromegrass) had a slightly lower $\delta^{13}C$ than spruce trees, and weeds (quackgrass) and mosses had a noticeably lower $\delta^{13}C$ than spruce trees. Interannual variation of $\delta^{13}C$ in barley was low and amounted to a difference of 0.3‰ between 2 successive years.

Forest floors had a $\delta^{13}C$ between −26.3‰ and −27.6‰ (Table 5). The top mineral soils in forests were higher in ^{13}C than the forest floor by 1.7‰ on average ($P < 0.001$) and showed a narrow range of −24.8‰ to −25.4‰, similar to forests in Saskatchewan (Flanagan *et al*., 1999). Plotting differences in $\delta^{13}C$ between forest and field mineral soils against field age showed a decrease in field compared with forest $\delta^{13}C$ during the first decade after deforestation and an increase during later decades (Fig. 2b). This trend was statistically significant even when the oldest field was omitted from the analysis ($r^2 = 0.73$, $P = 0.007$).

Soil-carbon quality and physical removal of carbon

The labile pool of soil organic matter in the top mineral soil was a relatively small portion of total soil C (4.6%

Table 4 Effect of land-use change on C and N inventories

Site	Land-use type	Time after clearing (years)	Soil depth* (cm)	Carbon stock: Mineral soil Eqn (4) (kg m^{-2})	% change	Carbon stock: Total soil Eqn (4) (kg m^{-2})	Eqn (5) (kg m^{-2})	% change	Carbon stock: Total ecosystem Eqn (5) (kg m^{-2})	% change	Nitrogen stock: Mineral soil Eqn (4) (kg m^{-2})	% change	Nitrogen stock: Total soil Eqn. (4) (kg m^{-2})	Eqn (5) (kg m^{-2})	% change	C/N ratio: Mineral soil Eqn (4)
Nenana I	Black spruce forest	0	46	12.2^{ab}		24.4^{ab}	24.4^{ab}		25.0 (1.8)		0.73^{ab}		1.37^{ab}	1.37^{ab}		17.0^{a}
	Fallow field	1		25.6^{a}	+110	25.6^{a}	26.7^{a}	+10	26.7 (4.8)	+7	1.50^{a}	+107	1.50^{a}	1.57^{a}	+15	17.3^{a}
	Fallow field	3		19.6^{ab}	+60	19.6^{abc}	20.1^{abc}	−18	20.1 (4.7)	−20	1.14^{ab}	+57	1.14^{abc}	1.17^{abc}	−14	16.8^{a}
	Grassland	5		10.7^{b}	−12	10.7^{bc}	11.1^{bc}	−55	11.1 (1.9)	−56	0.67^{b}	−7	0.67^{bc}	0.70^{bc}	−49	15.9^{a}
	Grassland	6		8.2^{b}	−33	8.2^{c}	8.5^{c}	−65	8.6 (1.7)	−66	0.51^{b}	−30	0.51^{c}	0.53^{c}	−61	16.0^{a}
Nenana II	Black spruce forest	0	61	17.5^{a}		22.6^{a}	22.6^{a}		24.1 (1.8)		1.06^{a}		1.20^{a}	1.20^{a}		16.7^{b}
	Cropland	6		6.5^{bc}	−63	6.5^{bc}	6.6^{bc}	−71	6.6 (0.6)	−73	0.35^{b}	−67	0.35^{b}	0.35^{b}	−71	18.8^{a}
	Grassland	9		10.4^{b}	−41	10.4^{b}	10.4^{b}	−54	10.5 (1.2)	−56	0.63^{b}	−40	0.63^{b}	0.64^{b}	−47	16.2^{b}
	Cropland	14		5.6^{c}	−68	5.6^{c}	5.7^{c}	−75	5.7 (0.4)	−76	0.35^{b}	−67	0.35^{b}	0.35^{b}	−71	16.1^{b}
Fairbanks	Black spruce forest	0	46	8.4^{b}		15.1^{a}	15.1^{a}		20.0 (1.6)		0.41^{b}		0.56^{b}	0.56 ^{b}		19.8^{a}
	Grassland	60		21.8^{a}	+158	21.8^{a}	22.1^{a}	+44	22.1 (2.4)	+11	1.32^{a}	+220	1.32^{a}	1.33 ^{a}	+136	16.5^{a}
Two Rivers	Black spruce forest	0	100	12.7^{a}		17.2^{a}	17.2^{a}		19.6 (1.6)		0.82^{a}		0.97^{a}	0.97^{a}		15.5^{a}
	Fallow field	1		13.8^{a}	+8	13.8^{a}	14.0^{a}	−19	14.0 (1.9)	−29	0.96^{a}	+18	0.96^{a}	0.98^{a}	+1	14.0^{a}
Delta Junction I	Black spruce forest	0	89	16.1^{a}		19.5^{a}	19.5^{a}		21.6 (1.0)		0.89^{a}		0.98^{a}	0.98^{a}		18.2^{a}
	Fallow field	1		18.5^{a}	+15	18.5^{a}	18.7^{a}	−4	18.7 (1.8)	−13	1.07^{a}	+20	1.07^{a}	1.07^{a}	+10	17.4^{a}
Delta Junction II	Black spruce forest	0	100	19.5^{a}		23.6^{a}	23.6^{a}		26.8 (2.0)		0.94^{b}		1.05^{a}	1.05^{a}		20.8^{a}
	Cropland	2		22.2^{a}	+14	22.2^{a}	22.3^{a}	−5	22.4 (1.7)	−16	1.07^{ab}	+14	1.07^{a}	1.08^{a}	+3	20.7^{a}
	Cropland	15		23.4^{a}	+20	23.4^{a}	23.5^{a}	0	23.6 (1.1)	−12	1.25^{a}	+33	1.25^{a}	1.25^{a}	+19	18.8^{b}
Delta Junction III	White spruce forest	0	65	10.2^{a}		13.3^{a}	13.3^{a}		20.3 (1.9)		0.56^{a}		0.65^{a}	0.65^{a}		18.3^{a}
	Cropland	3		5.9^{b}	−43	5.9^{b}	5.9^{b}	−56	5.9 (0.5)	−71	0.38^{b}	−32	0.38^{b}	0.38^{b}	−42	15.6^{b}
	Cropland	3		8.2^{ab}	−20	8.2^{b}	8.2^{b}	−38	8.2 (0.7)	−59	0.46^{b}	−17	0.46^{b}	0.46^{b}	−28	17.6^{ab}
Delta Junction IV	White spruce forest	0	100	25.7^{a}		29.2^{a}	29.2^{a}		32.7 (1.6)		1.26^{a}		1.36^{a}	1.36^{a}		20.5^{a}
	Cropland	20		21.2^{b}	−18	21.2^{b}	21.3^{b}	−27	21.3 (0.5)	−34	1.18^{a}	−6	1.18^{a}	1.19^{a}	−13	17.9^{b}
	Grassland	40		21.5^{b}	−16	21.5^{b}	21.7^{b}	−26	21.7 (0.9)	−34	1.21^{a}	−4	1.21^{a}	1.22^{a}	−11	17.8^{b}

*Common soil depth used to calculate C and N stocks for all land-use types within a site.
Stock size was corrected for differences in bulk density and depth. Mean (SE), $n = 4$–5; errors for total ecosystem C stocks were pooled from soil and vegetation C stock errors. Relative changes are the percentage of change from forest to field stocks. Values within sites followed by unidentical letters are significantly different at $P \leq 0.05$.

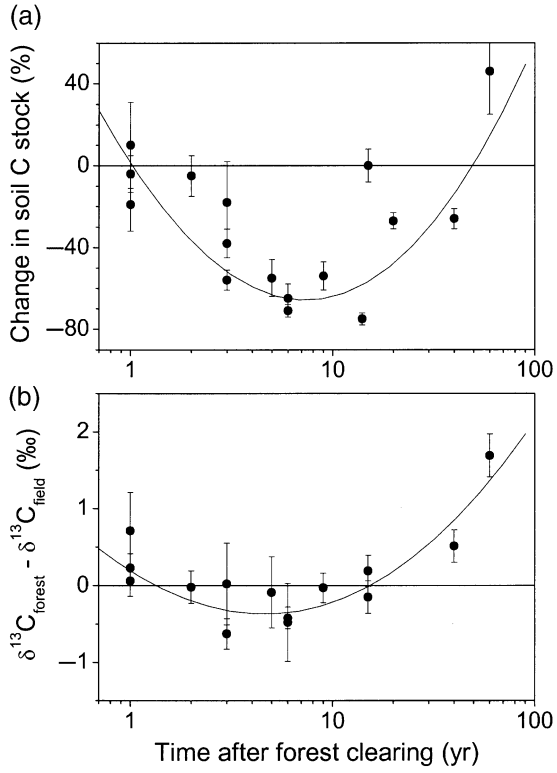

Fig. 2 Soil C stocks and $\delta^{13}C$ as a function of time after forest clearing. (a) Relative change in field C stocks (adjusted by Eqn (5)) compared with total forest soil C stocks. (b) Change in $\delta^{13}C$ in field topsoil (calculated as forest top mineral soil $\delta^{13}C$ minus field topsoil $\delta^{13}C$). Data points are means ± SE, $n = 4$–5. Note: logarithmic scale on x-axes. The following polynomial regressions were performed on log-transformed time (t) data and were weighted by the standard errors: (a) % change in soil C = $1.39 - 151.8 \log t + 82.1(\log t)^2 + 4.14(\log t)^3$, with $n = 16$, $r^2 = 0.53$, $P = 0.025$; (b) $\delta^{13}C_{forest} - \delta^{13}C_{field} = 0.19 - 1.65 \log t + 1.17(\log t)^2 + 0.07(\log t)^3$, with $n = 14$, $r^2 = 0.81$, $P < 0.001$.

on average), with a rather wide range of about 0.5–11.9% (Table 6). Forests on permafrost soils (Nenana, Fairbanks) had low labile C concentrations of less than $2\,mg\,C\,g^{-1}$ soil, whereas forests on non-permafrost soils (Two Rivers, Delta Junction) had 2.3–5.5 $mg\,C\,g^{-1}$ soil ($P = 0.041$). Soils under grasslands had a higher concentration of labile C than soils under croplands, which probably resulted from the larger root mass that was included as soil organic C. A majority of the organic C was in the intermediate and recalcitrant pools. In black-spruce forests and fields derived from them, half to almost all C was recalcitrant, whereas in white-spruce forests and their fields, most C was in the intermediate pool. Forests on non-permafrost soils had larger labile and intermediate pools than fields. The turnover time for labile and intermediate C was higher for forests than for their corresponding fields, except for Nenana II.

The fate of the forest floor and the top mineral soil during and after clearing was estimated by comparing intermediate + recalcitrant C stocks in forests and fields. Most permafrost soils lost an amount of intermediate + recalcitrant C that was equivalent to the entire forest floor plus a smaller or larger part of the top mineral soil (Table 7). This resulted from mechanical removal during the clearing process and possibly also from erosion after establishment of fields. In contrast, a high percentage of the forest floor was incorporated into the newer fields at the Nenana I site. Non-permafrost soils lost a quantity of intermediate + recalcitrant C equivalent to the forest floor, and normally lost only a small portion of their mineral soil C. On average, 4.6 $kg\,C\,m^{-2}$ were lost from the forest floor by physical removal (mean across all fields in Table 7).

Nitrogen mineralization, microbial biomass and DOC

Nitrate concentrations were low in the top mineral soil of forests and considerably higher in most corresponding fields (Table 8). Ammonium concentrations were also lower in most forests compared with fields. Laboratory incubations revealed potential net N mineralization rates of $0.09 \pm 0.02\,\mu g\,N\,g^{-1}\,soil\,day^{-1}$ in mineral soils of forests (mean ± SE) and $0.49 \pm 0.12\,\mu g\,N\,g^{-1}\,soil\,day^{-1}$ in fields. Net N mineralization (net ammonium + nitrate production) and net nitrification (net nitrate production) had similar rates at each site, and the two processes correlated strongly ($r^2 = 0.99$, $P < 0.001$, $n = 20$), indicating that nearly all of the mineralized N was nitrified during the incubation. CO_2 emission during the first 12 weeks of incubation correlated positively with net N mineralization, but the slope of C vs. N mineralization was considerably steeper in forest soils than in field soils (Fig. 3).

Microbial biomass (determined at the onset of soil incubations) was low in the top mineral soil of the Fairbanks forest, but high in the corresponding field (Table 8). Intermediate values were recorded for forests and fields at Two Rivers and Delta Junction II. Cumulative CO_2 emissions during 1 year of soil incubations correlated positively with initial microbial biomass ($r^2 = 0.76$, $P = 0.010$, $n = 7$ sites). The concentration of DOC was low in the forest in Fairbanks, reaching values similar to those in an upland black-spruce forest (MacLean *et al.*, 1999; Table 8). Higher DOC levels were measured in the forests at Two Rivers and Delta Junction II, and in the fields at all three sites.

Discussion

The comparisons between forests and fields suggested large losses of C from ecosystems following land-use

Table 5 $\delta^{13}C$ of plant material, organic layer and mineral soil (top 20 cm) collected in 1998 and 1999

Site	Land use, main crop	Time after clearing (years)	Plant/soil material	$\delta^{13}C$ (‰)
Nenana I	Black spruce forest	0	Organic layer	−26.96 (0.09)
			Mineral soil	−25.22 (0.46)
	Fallow field	1	Mineral soil	−25.94 (0.20)
	Fallow field	3	Mineral soil	−25.25 (0.27)
	Grassland, bromegrass	5	Mineral soil	−25.14 (0.06)
	Grassland, bromegrass	6	Mineral soil	−24.74 (0.22)
Nenana II	Black spruce forest	0	Organic layer	−27.59 (0.36)
			Mineral soil	−25.17 (0.11)
	Grassland, bromegrass	9	Mineral soil	−25.14 (0.15)
	Cropland, oats	14	Mineral soil	−25.02 (0.18)
Fairbanks	Black spruce forest	0	Black spruce, needles	−27.76 (0.44)
			Black spruce, twigs	−26.64 (0.20)
			Black spruce, trunk	−24.71 (0.14)
			Green moss (mainly *Pleurozium*)	−29.67 (0.51)
			Organic layer	−26.30 (0.16)
			Mineral soil	−24.77 (0.29)
	Grassland, bromegrass/quackgrass	60	Quackgrass, shoot	−29.61 (0.30)
			Mineral soil	−26.46 (0.10)
Two Rivers	Black spruce forest	0	Organic layer	−26.64 (0.13)
			Mineral soil	−24.92 (0.04)
	Fallow field	1	Mineral soil	−24.98 (0.20)
Delta Junction I	Black spruce forest	0	Organic layer	−26.92 (0.35)
			Mineral soil	−25.41 (0.17)
	Fallow field	1	Mineral soil	−25.64 (0.06)
Delta Junction II	Black spruce forest	0	Black spruce, needles	−27.07 (0.33)
			Black spruce, twigs	−26.46 (0.27)
			Green moss (mainly *Pleurozium*)	−30.32 (0.35)
			Organic layer	−26.87 (0.38)
			Mineral soil	−25.37 (0.19)
	Cropland, oats	2	Oats, shoot	−28.71 (0.14)
			Mineral soil	−25.35 (0.09)
	Cropland, barley	15	Barley, shoot (1998)	−27.82 (0.14)
			Barley, shoot (1999)	−28.13 (0.17)
			Mineral soil	−25.55 (0.07)
Delta Junction III	White spruce forest	0	Organic layer	−27.20 (0.26)
			Mineral soil	−25.43 (0.12)
	Cropland, barley	3	Mineral soil	−24.81 (0.16)
Delta Junction IV	White spruce forest	0	Organic layer	−26.62 (0.14)
			Mineral soil	−25.10 (0.16)
	Grassland, brome	40	Mineral soil	−25.61 (0.13)

Most grasslands were sown to bromegrass ($\delta^{13}C = -28.95$‰, sampled from a field in Fairbanks, spring 1999). For croplands, the main crop since the beginning of cultivation was indicated, for grasslands, current and past dominant species were specified. Mean (SE), $n = 4$–5.

change in boreal Alaska. Assuming that our chronosequences represent a valid time course, some of these losses must occur over a relatively short period of time (see below *Processes involved in soil carbon loss*). In contrast, the oldest field showed that considerable net C storage following forest clearing is possible under certain conditions, as discussed later (see below *Processes involved in soil carbon gain*).

To compare the effects of land-use change on C stocks in boreal Alaska with those from other biomes, we averaged changes in C stocks across fields that passed the initial stage of agricultural establishment (assumed 3 years after forest clearing). The mean relative soil-C loss of 44% observed in this study is at the upper limit of 22-42% mean C losses following conversion of forests to cultivated land in temperate and tropical regions

Table 6 Carbon-quality pools (labile, intermediate and recalcitrant C) in the top 20 cm of the mineral soil of forests and corresponding fields as obtained from lab incubations

Site	Land use	Time after clearing (years)	Concentration (mg C g^{-1} soil)			% of total organic C			Stock (kg C m^{-2})			Turnover time (years)	
			Labile	Intermediate	Recalcitrant	Labile	Intermediate	Recalcitrant	Labile	Intermediate	Recalcitrant	Labile	Intermediate
Nenana I	Black spruce forest	0	0.12*	0.7	32.7	0.4	2.0	97.7	0.04	0.21	10.34	–	–
	Fallow field	1	9.45	109.3	101.8	4.3	49.6	46.1	0.89	10.28	9.57	0.6	148
	Fallow field	3	3.14	33.7	54.9	3.4	36.7	59.9	0.65	7.01	11.43	0.4	44
	Grassland	6	1.92	9.9	20.2	6.0	31.0	63.1	0.42	2.18	4.45	0.3	24
Nenana II	Black spruce forest	0	1.43	18.3	53.8	1.9	24.9	73.2	0.23	2.91	8.57	0.4	17
	Grassland	9	2.37	23.8	26.0	4.5	45.6	49.9	0.38	3.82	4.17	0.3	30
	Cropland	14	1.95	6.6	7.9	11.9	40.1	48.0	0.36	1.23	1.46	0.5	126
Fairbanks	Black spruce forest	0	1.80	6.9	10.6	9.3	35.8	54.9	0.50	1.93	2.95	1.0	>300
	Grassland	60	3.35	28.6	30.9	5.3	45.4	49.2	0.75	6.38	6.91	0.3	34
Two Rivers	Black spruce forest	0	2.26	14.2	19.3	6.3	39.7	53.9	0.39	2.43	3.29	0.6	>300
	Fallow field	1	0.38	12.4	17.1	1.3	41.4	57.3	0.07	2.26	3.12	0.2	31
Delta Junction I	Black spruce forest	0	2.79	34.0	35.1	3.9	47.3	48.8	0.31	3.80	3.91	0.6	62
	Fallow field	1	0.66	18.5	36.6	1.2	33.2	65.6	0.08	2.16	4.28	0.2	22
Delta Junction II	Black spruce forest	0	5.46	43.2	42.4	6.0	47.4	46.6	0.56	4.46	4.37	0.7	>300
	Cropland	2	1.59	24.9	43.5	2.3	35.6	62.2	0.19	2.93	5.13	0.4	27
	Cropland	15	1.18	35.1	34.8	1.7	49.4	49.0	0.14	4.10	4.06	0.2	35
Delta Junction III	White spruce forest	0	2.66	23.9	16.4	6.2	55.6	38.2	0.38	3.45	2.37	0.4	48
	Cropland	3	0.94	12.5	10.7	3.9	51.7	44.4	0.14	1.87	1.60	0.2	15
Delta Junction IV	White spruce forest	0	3.28	56.6	20.7	4.1	70.2	25.7	0.34	5.91	2.16	0.5	277
	Grassland	40	2.60	31.2	21.8	4.7	56.1	39.2	0.34	4.06	2.84	0.3	38

*No curve fitting was possible because of low and relatively constant CO_2 emissions during incubations; the concentration of labile C was directly estimated from CO_2 emissions. C stocks were corrected according to Eqn (5). Turnover time is a function of conditions during incubations, and was calculated as $1/k$ (Eqn (3)). Note that turnover time in the field will be longer than the one that was obtained from incubations under optimal experimental conditions.

Table 7 C-quality analysis for estimation of physical soil loss

Chronosequence	Land use	Time after clearing (years)	Absolute change ($kg\,m^{-2}$)	Relative change (%)*
Nenana I	Fallow field	1	+9.30	+83
	Fallow field	3	+7.88	+70
	Grassland	6	−3.92	−37
Nenana II	Grassland	9	−3.49	−30
	Cropland	14	−8.79	−77
Two Rivers	Fallow field	1	−0.34	−6
Delta Junction I	Fallow field	1	−1.27	−16
Delta Junction II	Cropland	2	−0.77	−9
	Cropland	15	−0.67	−8
Delta Junction III	Cropland	3	−2.34	−40

*Negative values were interpreted as the relative amount of original top mineral layer removed, positive values as relative amount of forest floor added to fields.
Change in the intermediate + recalcitrant organic C pool in field top soils was compared with change in top mineral layers of corresponding forests. Not applicable to fields older than 20 years (see Materials and methods, *Calculations and statistical analyses*).

Table 8 Inorganic N concentrations, net N mineralization, net nitrification, microbial biomass and dissolved organic C (DOC) in the top 20 cm of the mineral soil of forests and corresponding fields

Site	Land use	Time after clearing (years)	NH_4-N ($\mu g\,g^{-1}$ soil)	NO_3-N ($\mu g\,g^{-1}$ soil)	Net N mineralization ($\mu g\,g^{-1}$ soil day^{-1})	Net nitrification ($\mu g\,N\,g^{-1}$ soil day^{-1})	Microbial biomass ($mg\,C\,g^{-1}$ soil)	DOC ($mg\,L^{-1}$)
Nenana I	Black spruce forest	0	0.58 (0.31)	0.18 (0.06)	0.022 (0.017)	0.021 (0.012)		
	Fallow field	1	3.63 (1.18)	0.00 (0.00)	0.218 (0.134)	0.199 (0.105)		
	Fallow field	3	3.45 (1.37)	3.48 (1.20)	0.792 (0.341)	0.795 (0.344)		
	Grassland	6	0.82 (0.27)	0.01 (0.01)	0.060 (0.036)	0.046 (0.035)		
Nenana II	Black spruce forest	0	1.77 (0.35)	0.01 (0.01)	0.100 (0.039)	0.114 (0.042)		
	Grassland	9	2.29 (0.52)	0.50 (0.22)	0.822 (0.304)	0.829 (0.305)		
	Cropland	14	1.22 (0.20)	1.85 (0.65)	0.129 (0.048)	0.132 (0.048)		
Fairbanks	Black spruce forest	0	0.21 (0.08)	0.08 (0.06)	0.019 (0.011)	0.016 (0.012)	0.12 (0.02)	14.3 (4.9)
	Grassland	60	2.69 (0.45)	0.72 (0.13)	1.330 (0.127)	1.331 (0.129)	1.25 (0.12)	46.2 (5.3)
Two Rivers	Black spruce forest	0	0.37 (0.19)	0.07 (0.02)	0.037 (0.010)	0.019 (0.008)	0.56 (0.05)	49.8 (5.2)
	Fallow field	1	0.46 (0.09)	0.12 (0.03)	0.045 (0.027)	0.046 (0.026)	0.40 (0.03)	32.5 (3.0)
Delta Junction I	Black spruce forest	0	1.31 (0.32)	0.14 (0.08)	0.099 (0.033)	0.095 (0.038)		
	Fallow field	1	3.74 (1.16)	6.82 (1.95)	0.262 (0.075)	0.298 (0.078)		
Delta Junction II	Black spruce forest	0	2.75 (0.47)	0.66 (0.23)	0.147 (0.064)	0.112 (0.060)	0.81 (0.07)	56.7 (5.2)
	Cropland	2	2.06 (0.31)	1.76 (0.42)	0.393 (0.089)	0.408 (0.088)	0.58 (0.04)	40.0 (5.0)
	Cropland	15	3.11 (0.33)	5.75 (1.63)	0.916 (0.114)	0.944 (0.116)	0.59 (0.03)	40.0 (4.0)
Delta Junction III	White spruce forest	0	1.79 (0.38)	0.00 (0.00)	0.124 (0.049)	0.016 (0.007)		
	Cropland	3	2.23 (0.45)	0.58 (0.20)	0.251 (0.049)	0.270 (0.053)		
Delta Junction IV	White spruce forest	0	2.46 (0.67)	0.23 (0.13)	0.131 (0.049)	0.122 (0.048)		
	Grassland	40	6.43 (1.96)	2.87 (1.84)	0.686 (0.127)	0.735 (0.150)		

Net mineralization and nitrification are the result of 12-week laboratory incubations of soils. Mean (SE), $n = 4$–5.

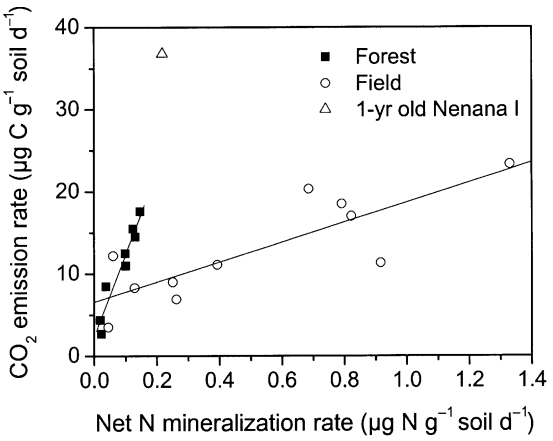

Fig. 3 Relationship between C and N mineralization in lab incubations. Data points are means, $n = 4$–5. Equations for linear regression were as follows: $CO_2 = 2.387 + 99.566N$, with $n = 8$, $r^2 = 0.94$, $P < 0.001$ for forests; $CO_2 = 6.598 + 12.104N$, with $n = 11$, $r^2 = 0.68$, $P = 0.002$ for fields (excluding 1-year-old Nenana I field).

(Davidson & Ackerman, 1993; Guo & Gifford, 2002; Murty *et al*., 2002). Differences in soil C losses between boreal and other regions become particularly pronounced when they are expressed in absolute terms. For example, losses of soil C from forest clearing in the Amazon region were estimated at 0.8 kg m^{-2} (Fearnside & Barbosa, 1998; mainly C-rich pastures), and losses from native grassland and forests in temperate regions averaged 3.3 kg C m^{-2} (Davidson & Ackerman, 1993; all soils sampled by generic horizons or corrected for changes in bulk density). With mean soil C losses of 8.3 kg m^{-2} in the boreal region, enlargement of the agricultural area in this region might result in considerably greater absolute reductions in soil organic C stocks than in a comparable area of land-use change in other regions of the world. The actual reduction in soil C stocks at the permafrost sites could be larger than the ones we measured. Large amounts of soil organic C are locked up in permafrost and were not included in our estimates of initial C stocks. This organic C will be available for mineralization following thaw (Goulden *et al*., 1998).

Currently, an area of almost 300×10^6 ha could potentially be cultivated in the boreal region (agricultural climate envelop), a value that might increase to up to 700×10^6 ha under a future warmer climate (Solomon & Leemans, 1997). Houghton (1999) used ecosystem C losses of 13.6 kg m^{-2} in land-use change models for the boreal biome, which puts the total theoretical reduction in boreal C at 40–95 Pg (10^{15} g; assuming the current or future agricultural potential will be realized). Our average ecosystem C loss of 11.2 kg m^{-2} (from vegetation and soil) for interior Alaska is slightly lower than the above estimate for the boreal biome, but in both cases the amount of C at stake is 1.5–4% of the total terrestrial organic C stock (Schimel, 1995).

Processes involved in soil carbon losses

The reductions in soil organic C stocks during the first 10–15 years after forest clearing (Fig. 2a) may result from relatively short-term physical removal and combustion of forest soil material during deforestation and from long-term processes, such as soil erosion, decomposition of organic matter and leaching of DOC following field establishment.

According to the C-quality analysis (Table 7), forest floor and some mineral soil were removed at most sites by physical means (either mechanically during forest clearing or by erosion). Forest clearing down into mineral soil was indeed visible from unplowed sections in the 1-year-old fields at Two Rivers and Delta Junction I (J. M. Grünzweig, unpublished observation). The increase in $\delta^{13}C$ of field compared with forest mineral soil during the first years after clearing (Fig. 2b) may also be interpreted as physical removal of a shallow layer of mineral soil (decomposition would be an alternative interpretation, see below). Some soil organic C is always lost by combustion following forest clearing, and this may include live moss and at least part of the upper O_{ie} horizon. In addition, while soil erosion by water flows is limited, since all fields in our study were on flat topography, strong surface winds, such as in Delta Junction (Sharratt, 1998) can potentially cause significant erosion on bare soils during spring and fall. Notably, fields that lost a sizeable part of the top 20 cm (e.g. 6-year-old grassland at Nenana I, cropland at Delta Junction III) also showed a relatively large increase in $\delta^{13}C$ compared with the isotopic signature of their corresponding forest mineral soil (Table 5). This increase in $\delta^{13}C$ is the result of the exposure of a deeper soil layer. The enrichment in ^{13}C with depth is due to a growing percentage of old decomposed organic matter (Nadelhoffer & Fry, 1988; Melillo *et al*., 1989; Balesdent *et al*., 1993; Flanagan *et al*., 1999), resulting from the preferential loss of the lighter isotope during the transformation of plant detritus to soil organic matter (Torn *et al*., 1997), and, to some extent, from the historic decrease in $\delta^{13}C$ of atmospheric CO_2 by fossil fuel combustion over the past 200 years (Boutton, 1996). Fields with an almost intact mineral horizon (Two Rivers, Delta Junction II) had $\delta^{13}C$ values that were very similar to forest values. The fields at Nenana II were the only exception of this pattern. At this site, part of the relatively large loss from the intermediate + recalcitrant C pools, particularly in

the 14-year-old field, might result from partial decomposition of intermediate C and not from physical removal of soil. This C pool was assumed to be mostly resistant to mineralization over a short time period of a few years to a decade. However, the short turnover time of intermediate C in the original Nenana II forest (Table 6) suggests higher decomposability of this pool compared with all other sites, even if *in situ* turnover time is longer than the one observed in lab incubations.

Decomposition was an important avenue of C loss in all fields as suggested by reduced labile C stocks (in the fields of Two Rivers and Delta Junction; Table 6), increase in $\delta^{13}C$ in fields without marked soil erosion and the narrowing of C/N ratios (Table 4). The initial increase in $\delta^{13}C$ of field compared with forest mineral soil (Fig. 2b) may be related to decomposition of organic matter left over from the forest, as decomposition typically results in higher $\delta^{13}C$ (Nadelhoffer & Fry, 1988; Melillo *et al.*, 1989; Balesdent & Mariotti, 1996; Boutton, 1996; Torn *et al.*, 2002). Assuming no major changes in N, narrowing of C/N ratios after forest clearing indicates C loss by decomposition. Narrower C/N ratios could also be caused by erosion and exposure of deeper soil; however, no relationship was found between loss of top mineral soil (Table 7) and changes in C/N ratio. A significant rise in field compared with forest soil-temperature during the growing season (Van Cleve *et al.*, 1990; Sharratt, 1998) and a reduction in physical protection of soil organic matter, when soils are plowed, are factors that promote decomposition of soil organic matter in the boreal region. Furthermore, higher potential net N mineralization and higher inorganic N content after forest clearing correlated with high decomposition rates in these soils (Grünzweig *et al.*, 2003b). If enhanced forest conversion to agriculture in the boreal region is expected with climatic warming, then increased temperatures themselves might affect C storage. However, a related study indicated that *after forest conversion*, a moderate temperature change in the range of predicted climatic warming had no effect on decomposition of soil organic matter and presumably also on C storage (Grünzweig *et al.*, 2003b).

DOC can be lost from soils by leaching from the soil profile. Fluxes of DOC below the forest floor in a boreal forest in Finland averaged only $0.017\,kg\,C\,m^{-2}\,yr^{-1}$ over a period of 3 years after land-use change [DOC concentrations were similar in Alaska (Table 8) and Finland; Piirainen *et al.*, 2002]. However, most of the leached DOC was retained in the mineral soil and did not leave the soil profile. Studies in boreal and temperate forests showed that hydrophilic DOC was the preferably leachable species of DOC (Raastad & Mulder, 1999; Kaiser *et al.*, 2001), and that it was higher in $\delta^{13}C$ than bulk organic matter (Kaiser *et al.*, 2001). Consequently, massive DOC leaching from the topsoil should have left the organic matter of that layer poorer in ^{13}C. However, $\delta^{13}C$ of a majority of field topsoils was higher than $\delta^{13}C$ of corresponding mineral soils in forests (Table 5). Additionally, $\delta^{13}C$ of field topsoils tended to increase rather than decrease compared with forest soils during the first years after clearing (Fig. 2b). Thus, both DOC budgets and $\delta^{13}C$ analysis suggest that DOC leaching played a minor role in C losses following boreal deforestation.

Processes involved in soil carbon gain

Chronosequences indicated *net accumulation of C* one decade or more after forest clearing. Losses in field C stocks relative to original forests tended to be lower in later years (Fig. 2a), and, at the same time, differences in soil $\delta^{13}C$ between forests and fields tended to increase, i.e. field $\delta^{13}C$ tended to decrease (Fig. 2b). These findings suggest accumulation of new crop-derived organic matter low in ^{13}C and lower decomposition rates after breakdown of the more labile forest-derived organic matter within the soil profile.

The trend for long-term net accumulation of C was strongly affected by the oldest field tested (Fairbanks), but was significant even without this field. The Fairbanks grassland experienced net soil C storage of about $7\,kg\,m^{-2}$ or $0.1\,kg\,m^{-2}\,yr^{-1}$(Table 4). Net C storage following land-use change has been shown in other studies, not only in forest plantations or after forest regrowth (Grünzweig *et al.*, 2003a) but also where perennial grasslands replaced native vegetation. For example, converting tropical moist forests to pastures can result in an increase of up to $0.1\,kg\,C\,m^{-2}\,yr^{-1}$ in the top 30 cm after 20 years (Neill *et al.*, 1997), and replacing native tropical savanna grasses with more productive exotic grasses might add a total of $0.4–1.7\,kg\,C\,m^{-2}\,yr^{-1}$ to a depth of 80–100 cm over periods of 3–6 years (Fisher *et al.*, 1994). The net increase in soil C stock in our grassland can be explained by some combination of high biomass production and low decomposition rates. Grassland soils experienced a relatively high input of biomass by perennial plants, most of it belowground, a fact that is also obvious from the stable isotope data. $\delta^{13}C$ of field soil in Fairbanks was 1.7‰ lower than the top mineral soil of the forest, which seems to reflect addition of new ^{13}C-poor biomass (bromegrass, quackgrass; Table 5). Decomposition of organic matter might be relatively slow in perennial grassland as compared with cropland due to physical protection of organic matter caused by low tillage frequency (Paustian *et al.*, 2000; Post & Kwon, 2000). Additionally, the large net increase in soil C stock in the Fairbanks grassland was

facilitated by a relatively small loss of original forest C (the forest in Fairbanks had the lowest C concentration in the mineral soil among all forests and fields in the present study; Table 2). In tropical ecosystems, net gains in soil C following deforestation and pasture establishment were associated with small original C stocks, whereas net C losses were associated with large original C stocks (Neill & Davidson, 2000). The mean summer soil temperatures were slightly lower in grasslands than in croplands, but these differences could not explain differences in decomposition (Grünzweig *et al*., 2003b).

Changes in soil nitrogen stocks

Nitrogen stocks in fields are the net result of addition (N deposition, fertilization) and loss of N (biomass harvest, leaching of dissolved N, gaseous N loss, physical removal during clearing and soil erosion). With N deposition from the atmosphere being low in interior Alaska (in the order of $0.2\,kg\,ha^{-1}\,yr^{-1}$ for wet deposition; National Atmospheric Deposition Program, http://nadp.sws.uiuc.edu), N is mainly added to fields by fertilization. Net fluxes of N into fields (fertilization minus biomass harvest when other processes were disregarded) probably do not exceed $+0.01\,kg\,m^{-2}\,yr^{-1}$ (Knight & Sparrow, 1993). As shown by C-quality analysis (Table 7), part of the soil N stock was removed together with organic matter during forest clearing and maybe also by soil erosion. In addition, significant amounts of N can be lost by denitrification, e.g. 15% of N added in fertilizers might have been lost by denitrification in an Alaskan barley field (Knight & Sparrow, 1993). Nitrate leaching was not significant during the growing season (Knight & Sparrow, 1993), but might be important during recharge after snowmelt (Sharratt, 1998), as some fields showed a relatively high nitrification potential (Table 8). Nitrogen might also be leached as dissolved organic N, an important source of N in high-latitude ecosystems (Chapin *et al*., 1993; Näsholm *et al*., 1998). In contrast, N loss by ammonia volatilization was negligible in Alaskan fields that are typically fertilized by urea or calcium nitrate (Knight & Sparrow, 1993). This finding is supported by our and other observations of low ammonium accumulation because of rapid nitrification once organic N was broken down to ammonium (Sparrow & Cochran, 1988; Table 8). Reductions in soil N stocks in interior Alaska appear to result from the removal of organic matter during forest clearing, and from erosion, leaching and/or denitrification.

Although N deposition is currently low in interior Alaska, it is considerably higher in some other boreal regions (up to $14\,kg\,ha^{-1}\,yr^{-1}$; Mäkipää *et al*., 1999; Köchy & Wilson, 2001), and is expected to increase globally during the next decades (Galloway *et al*., 1995). Lab incubations showed that C mineralization per unit of mineralized N is considerably higher in forests than in fields, which might be caused by the higher labile C concentration in most forests (Table 6). Because field N mineralization rates correlated positively with inorganic N availability (Grünzweig *et al*., 2003b), decomposition rates might be more sensitive to inorganic N addition (e.g. by increased N deposition) in forest than in field soils.

Conclusions

Forest conversion to agricultural land is occurring at relatively high rates in some boreal regions and could become more frequent in the coming decades because of climatic warming. Such land-use changes would cause large losses of C from newly established fields within a relatively short period after deforestation (a few years to a decade). Carbon stocks, stable C isotopes and C-quality analysis indicated that C was lost from ecosystems by combustion, decomposition and possibly soil erosion. Part of this C will be directly lost to the atmosphere, which will positively feedback to climatic warming and additional land-use change. However, by selecting relatively C-poor soils for land-use change and by implementing management of C preservation (e.g. perennial crops, low tillage) C losses from soils can be reduced and even net C gains are possible.

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