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The CLIMEX soil-heating experiment: soil response after 2 years of treatment

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Abstract Most model predictions concerning the response of boreal forest ecosystems to climate change are inferred from small-scale experiments on artificial, simplified systems. Whole-ecosystem experiments designed to validate these models are scarce. We experimentally manipulated a small forested catchment in southern Norway by increasing soil temperature (+3 °C in summer to +5 °C in winter) using heating cables installed at 1 cm depth in the litter layer. Especially nitrification in the 0 to 10-cm soil layer increased as a result of the climate manipulation. *Betula* litter, produced after exposing trees for 2 years to ambient and elevated CO_2 in greenhouses, was incubated for 1 year in the manipulated catchment. Exposure to elevated CO_2 did not affect the C/N ratio or decomposition of the *Betula* litter, but lignin content decreased by 10%. We found no effect of elevated temperature on litter decomposition, probably due to desiccation of the litter. The heating cables caused a permanent increase in soil temperature in this soil layer, but when soils were dry, the temperature difference between control and heated plots decreased with increasing distance from the cables. When soils were wet, no gradients in temperature increase occurred.

Key words Climate warming · Decomposition · Nitrogen mineralization · Whole catchment manipulation · Soil heating

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Introduction

Elevated concentrations of greenhouse gases due to combustion of fossil fuels and changes in land use may cause an increase in global temperatures (Houghton et al. 1995). Estimated temperature increases during the last decades at northern latitudes range from 0.7 °C (Möberg and Alexandersson 1997) to 2–4 °C (Lachenbruch and Marshall 1986). Elevated temperatures may stimulate decomposition of soil organic matter (Kirschbaum 1995) and mineralization of N (Stanford et al. 1973). Higher N availability may enhance net primary production in addition to the effect of CO_2 fertilization. If N mineralization exceeds immobilization by the living biomass, excess N can leach to the groundwater and surface waters, which may result in increased acidification and eutrophication of aquatic ecosystems (Wright and Schindler 1995). The effect of elevated temperature on decomposition may be buffered if litter produced under elevated CO_2 has a higher C/N ratio and decomposes more slowly (Norby et al. 1986; Lambers 1993; Cotrufo et al. 1994).

To date, predictions regarding whole ecosystem responses are largely based on modeling, in which results from small-scale greenhouse studies and laboratory incubations have been extrapolated to the ecosystem level (Overpeck et al. 1990; Rastetter et al. 1992; Schimel et al. 1997). There are only few data from large-scale experiments available that can be used to validate these models.

CLIMEX (Climate Change Experiment) is an international multidisciplinary project in which forested catchments are subjected to increased air temperature and CO_2 or to increased soil temperature alone (Jenkins and Wright 1995). In this paper we report on the effects of soil heating on N mineralization and litter decomposition. In addition, we studied effects of elevated CO_2 on litter quality and decomposition by incubating birch litter produced under ambient and elevated CO_2.

Materials and methods

Site description

The CLIMEX site is located at Risdalsheia (58°23'N, 8°19'E) near Grimstad, southernmost Norway. The site is located 300 m above sea level on a large biotite granite plateau and is representative for large areas of upland southern Norway. Mean annual precipitation is 1400 mm and mean annual temperature is 5 °C (–3 °C in January and +16 °C in July). Depressions in the granite surface are filled with post-glacial soil material in which acid, peaty podsolic soils have developed. In the studied EGIL catchment (400 m^2), maximum soil depth is 40 cm. Vegetation is dominated by heather (*Calluna vulgaris* (L.) Hull) with few scattered Scots pines (*Pinus sylvestris* L.) and birches (*Betula pubescens* Ehrh.). In 1983, the catchment was covered by a transparent roof as part of the RAIN (Reversing Acidification In Norway) project (Wright et al. 1993).

Soil heating

In spring 1994, in the lower 80% of the catchment (EGIL-t), heating cables were placed at a depth of 1 cm in the litter layer with a spacing of 10–15 cm. The upper 20% of the catchment (EGIL-c) acted as an untreated control. Temperatures in EGIL-c and EGIL-t were monitored hourly at 20 locations at 6 levels (25 and 10 cm above the soil; 0, 5, 15, and 30 cm in the soil). Starting in June 1994, the temperature in the (EGIL-t) part was increased by 5 °C in January and 3 °C in July, with intermediate temperature increases in the intervening months (Fig. 1). An uncovered catchment (METTE) served as outside control.

At selected locations, additional thermistors were installed at 1 cm depth at 1, 3 and 7.5 cm from the cable and at 5 cm depth, at 1 and 7.5 cm from the cable. We measured thermal conductivity (λ) of the LF, H, Ah and Bw layer at different moisture levels in the laboratory according to Van Loon et al. (1989). Gravimetric moisture content (g H_2O g dry $soil^{-1}$) was determined by drying a subsample for 24 h at 105 °C. Volumetric heat capacity (C_h) of the soil was calculated as:

$$C_h = \Sigma\, \theta_i\, C_i \quad (1)$$

where θ_i is the volume fraction of soil constituent i and C_i the volumetric heat capacity of material i, assuming a volumetric heat capacity of 2.0 MJ m^{-3} K^{-1} for minerals, 2.5 MJ m^{-3} K^{-1} for organic matter and 4.2 MJ $^{-3}$ K^{-1} for water (Koorevaar et al. 1983). Organic matter content of the soil material was measured as the loss on ignition at 600 °C after 4 h of heating. Thermal diffusivity (D_h) was calculated as λ/C_h. We also calculated D_h using field data with the equation given by Van Wijk (1966):

$$D_h = 0.5 d^2 \omega \varphi^{-2} \quad (2)$$

where φ is the phase shift in the diurnal heat cycle between two depths (–), d the difference between depths (mm), ω the angular frequency [$=2\pi/(24\times3600)$ radians per s], and D_h diffusivity (mm^2 s^{-1}). To determine φ, we fitted a sine function through the temperatures measured at 1, 5 and 15 cm depth under dry and wet conditions.

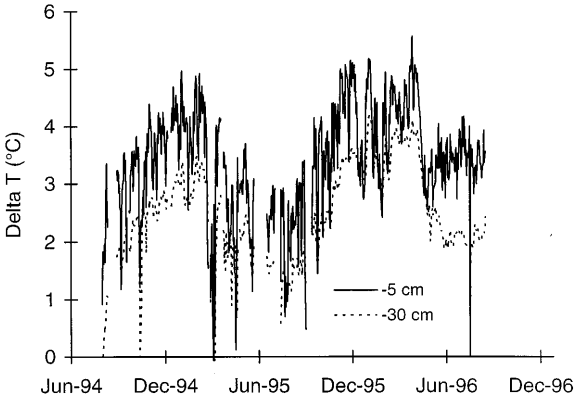

Fig. 1 Average temperature difference (*Delta T*) between heated-soil and control plots at 5 and 30 cm depth

Soil N mineralization and nitrification

Net N mineralization was measured in EGIL-c, EGIL-t and METTE during the pre-treatment (June 1993–June 1994) and treatment period until August 1996 using the sequential core incubation method (Berendse et al. 1987, 1989; Raison et al. 1987; Berendse 1990). We measured N mineralization in ten plots of 0.1 m^2. Each year was split into four incubation periods: April–June, June–August, August–October, and October–April. At the start of each incubation period, two samples were taken 5–10 cm apart using pre-weighed PVC tubes (length 15 cm, diameter 2.8 cm, wall thickness 2 mm). The soil was sampled to a depth of 10 cm, unless the soils were shallower. One sample was taken to the laboratory whereas the second sample was covered on both ends with soft polyethylene caps and put back into the soil. The incubated cores had four 4-mm holes in the cylinder walls to allow for gas exchange. At the end of each incubation period, the samples were collected from the field and stored overnight at 4 °C. A subsample of 20 g field-moist soil was extracted with 50 ml 1 M KCl. The KCl extracts were analysed for NH_4^+-N and NO_3^--N by colorimetry in an autoanalyser. Net N mineralization was calculated as the difference in NO_3^--N + NH_4^+-N, and net nitrification as the difference in NO_3^--N between paired incubated and reference samples.

Soil moisture content was measured after drying the samples at 105 °C for 24 h. Gravimetric moisture content in samples collected at any one time showed a large variation, reflecting differences in soil organic C content (6.7–53.0%). We normalized water content to total pore space rather than to dry soil, and expressed moisture as relative water content (RWC = volume fraction water/total pore space; Skopp et al. 1990) according to Verburg (1998).

Litter decomposition

To study the effect of elevated CO_2 and temperature on decomposition, we incubated foliar litter produced at 365 and 700 ppm CO_2 in EGIL-t, EGIL-c and METTE. We grew 1-year-old birches (*Betula pubescens* Ehrh.) in 10-l pots in a sandy soil (C = 2.16%, N = 0.10%, pH-H_2O = 6.59) at either 365 or 700 ppm CO_2 in greenhouses in The Netherlands. Temperature, light and humidity followed ambient outside conditions. The litter was collected after 2 years exposure to elevated CO_2. Total C and N of the litter was measured using an EA 1108 CHN element analyser. Klason lignin was determined according to Van Vuuren and Van der Eerden (1992).

In April 1995, 30 litterbags with litter from either the low- or high-CO_2 treatments were incubated in EGIL-c, EGIL-t and METTE. Each litterbag (15 × 15 cm; mesh size 1.5 mm) contained 4 g litter. At $t = 0$, 0.5, and 1 year, ten litterbags of high- and low-CO_2 litter were collected from the three locations. After sampling, litterbags were cleaned, dried at 70 °C and weighed.

Statistical methods

Time trends in N mineralization and nitrification data were analysed by analysis of variance (ANOVA) for each location (METTE, EGIL-c, EGIL-t) separately. For ANOVA on RWC, we added 'RWC prior to and after incubation' and 'season' as additional factors. Residual mass data of the birch litter at $t = 0.5$ and 1 year were analysed by two-way ANOVA using 'location' (METTE, EGIL-c and EGIL-t) and 'CO_2 treatment' as factors at each sampling time. Tests for significance were carried out using Student's *t*-tests. Effects or differences were considered significant when $P < 0.05$. For the statistical analyses we used SPSS version 6.1.

Results and discussion

Soil N mineralization and nitrification

In METTE, EGIL-c and EGIL-t, N mineralization in the 0 to 10-cm soil layer did not significantly change in treatment years compared to the pre-treatment period (Figs. 2, 3). The change in mineralization (%) in EGIL-t between the second treatment year and the pre-treatment year tended to be larger ($P = 0.061$) than in METTE. In EGIL-t, nitrification was significantly higher ($P = 0.044$) in the second treatment year than in the pre-treatment year, whereas in EGIL-c and METTE nitrification tended to decrease (EGIL-c: $P = 0.089$; METTE: $P = 0.06$). These data suggest that the heating treatment predominantly affected nitrification.

During the pre-treatment year, both mineralization and nitrification were lower in EGIL-c and METTE than in EGIL-t. The main cause for this difference is, however, difficult to assess. During this period, in the EGIL catchment, moisture content was lower in EGIL-c than in EGIL-t (Fig. 4). In EGIL-c, the soils are shallower than in EGIL-t and are thus more likely to dry out. During the two treatment years, moisture content in the tubes was similar in EGIL-c and EGIL-t. Whether differences in moisture alone explain differences in pre-treatment mineralization/nitrification between METTE and EGIL-t is unlikely since both locations vary not only in soil depth. Other factors, such as presence of a roof or variation in organic matter quality due to differences in litter produced by overstorey vegetation to the soil organic matter pool, may have caused differences in N transformations between or even within the locations.

The in situ incubation method for measuring N mineralization is widely used but may underestimate mineralization (Tietema et al. 1992). Covering the cores at both ends prevents throughflow of water, whereas water movement through the soil may stimulate microbial

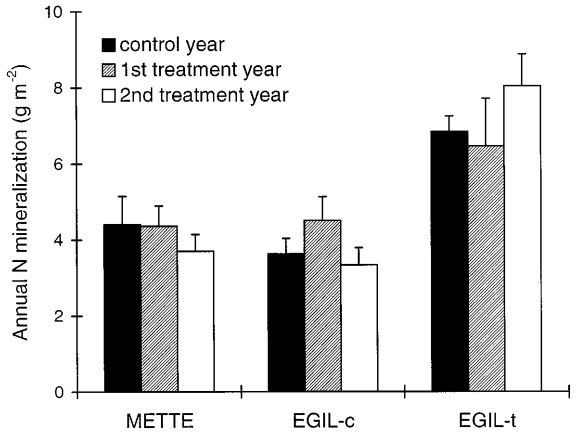

Fig. 2 Annual net N mineralization in the study sites METTE, EGIL-c and EGIL-t during pre-treatment and treatment years. *Error bars* represent standard errors of the mean ($n = 10$)

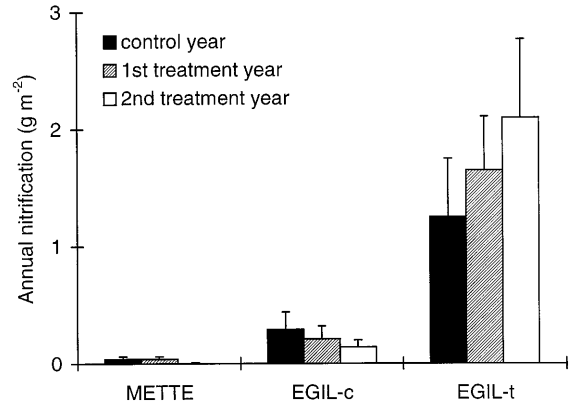

Fig. 3 Annual nitrification in the study sites METTE, EGIL-c and EGIL-t during pre-treatment and treatment years. *Error bars* represent standard errors of the mean ($n = 10$)

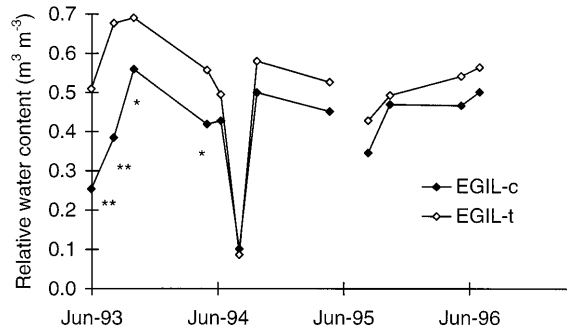

Fig. 4 Relative water content (= volume water divided by total pore space) of reference samples during the treatment and pre-treatment periods in the treated (*EGIL-t*) and control (*EGIL-c*) parts of the heated-soil catchment. The soil-heating started in June 1994. ** Significant differences at $P < 0.01$ and * at $P < 0.05$

processes (Koopmans et al. 1995). In addition, repeated wetting and drying of soil stimulates N mineralization (Nordmeyer and Richter 1985), which is more likely to have happened in the unconfined soil. Still, the observed increase in nitrification and possibly in total mineralization at elevated temperatures are consistent with the measured increase in NH_4^+ and NO_3^- efflux in runoff during the first 2 years of treatment (Lükewille and Wright 1997). By contrast, Van Cleve et al. (1990) found that, after 2 years of soil heating, NO_3^- concentrations in the soil solution were lower in soil-heated plots, which was ascribed to an increase in denitrification potential. However, these authors measured increased NH_4^+ and total N concentrations in heated plots. Peterjohn et al. (1993) found lower N concentrations in surface horizons in soil-heated plots, which was ascribed to faster mineralization.

Litter decomposition

Although soil temperatures in EGIL-t increased, mass loss of birch litter did not increase after 1 year of incubation (Table 1). In fact, after 6 months, mass loss was significantly lower in EGIL-t than in EGIL-c ($P=0.02$). After 6 months of incubation, litter from EGIL-t was significantly drier than from EGIL-c, most likely caused by the heat treatment. It appears that elevated temperature indirectly affected decomposition by decreasing moisture availability, which appeared to be more important than the direct (positive) effect of elevated temperature on (micro-) biological activity.

Two years of exposing birch trees to elevated CO_2 did not significantly alter the C/N ratio of the litter. Most studies report an increase in C/N ratio of tree leaves at elevated CO_2 (e.g. Norby et al. 1986; Coûteaux et al. 1991; Cotrufo et al. 1994). Ball (1997) reported that C/N ratios at elevated CO_2 decreased for several C_4 species, but increased for C_3 species. Fewer data are available on the effects of elevated CO_2 on secondary metabolites such as lignin. In our study, lignin content was significantly lower in the high-CO_2 litter (16.1%) than in the low-CO_2 litter (17.8%). Norby et al. (1986) found a decrease in lignin content for white oak, which contrasts with the results of Cotrufo et al. (1994), who observed an increase in lignin content for four tree species under elevated CO_2. Lambers (1993) also suggested that the concentrations of secondary metabolites may increase due to nutrient limitations induced by elevated CO_2.

The 10% lower lignin content of the high-CO_2 litter did not result in significant effects of elevated CO_2 on decomposition, according to ANOVA (Table 1). However, in the control catchment METTE, the high-CO_2 litter decomposed significantly faster ($P<0.001$). Cotrufo et al. (1994) found a significantly slower decomposition of high-CO_2 litter for several tree species, in agreement with higher C/N ratios.

Table 1 Residual mass (%) of *Betula* litter after 1 year of incubation. ANOVA results are given for residual mass data after 6 and 12 months of incubation

	6 months		12 months	
	365	700	365	700
EGIL-c	65 (2)	65 (8)	57 (4)	61 (9)
EGIL-t	70 (7)	73 (11)	62 (8)	65 (8)
METTE	71 (4)	64 (4)	61 (6)	53 (2)
Location[a]		*[b]		*
CO_2		ns		ns
Location × CO_2		**		**

[a] Location = EGIL-c, EGIL-t, METTE, CO_2 = 363 versus 700 ppm CO_2
[b] * $P<0.05$; ** $P<0.01$; ns = not significant

Heat distribution

Under dry conditions, the temperature difference between treatment and control decreased with increasing distance from the cable (Fig. 5). Under these conditions, thermal conductivity (λ) was very low, but increased with depth from 0.025 W m^{-1} K^{-1} in the LF layer to 0.116 W m^{-1} K^{-1} in the mineral soil (Bw horizon; Fig. 6), in agreement with a higher λ of minerals than of organic matter. Diffusivity (D_h), however, was highest in the organic surface layers (Fig. 7). When soils were wet, ΔT was similar at all points. In all layers, λ increased with increasing moisture content, but D_h only increased in the mineral soil (Bw). The diurnal pattern in ΔT under dry conditions may be caused by rewetting of the soils at night due to condensation. Water films around soil particles will increase λ, so heat from the surface is redistributed.

Although the field observations can be explained qualitatively using the laboratory measurements, differ-

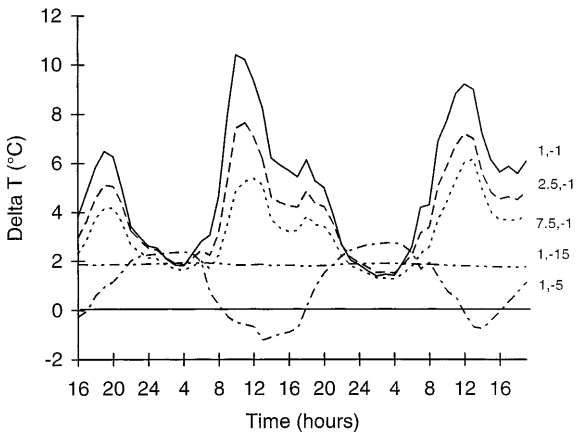

Fig. 5 Temperature difference (Delta T) between heated soil and control plots under dry conditions, 2 weeks after a rain event in July, as a function of vertical and horizontal distance from the cable. The first number of the coordinate gives horizontal distance (cm) from the cable; the second gives the depth (cm) in the soil

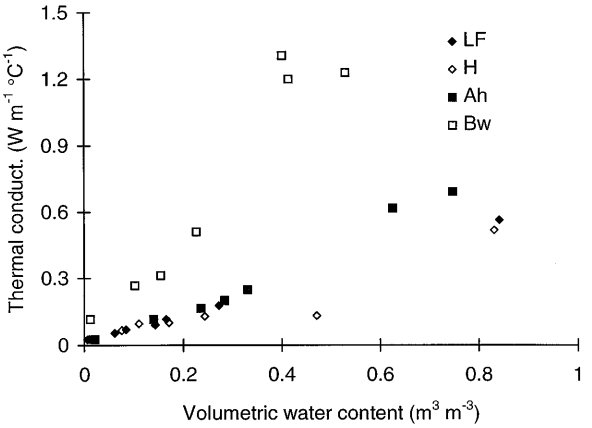

Fig. 6 Thermal conductivity of organic surface soil (*LF*, *H* and *Ah*) and mineral subsoil (*Bw*) as a function of moisture

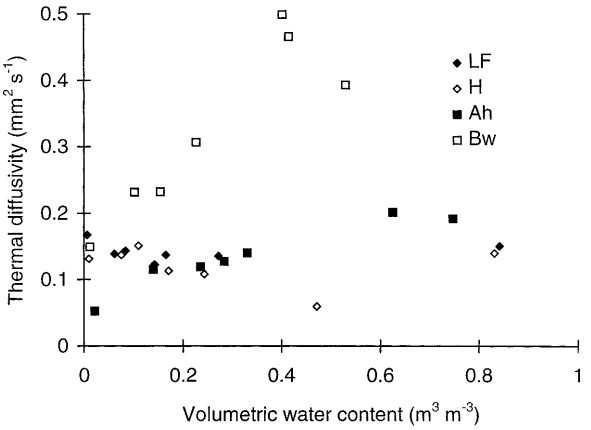

Fig. 7 Thermal diffusivity of organic surface soil (*LF*, *H* and *Ah*) and mineral subsoil (*Bw*) as a function of moisture

ences in λ and C_h between field and the laboratory may have occurred due to differences in arrangement of soil particles. Bulk density calculated by weighing moist soil and correcting for moisture content was lower than in the field (Verburg 1998), so contact between individual soil particles was likely to be less than in the field, resulting in a lower λ. We could not directly measure λ in the field, but we compared D_h determined in the laboratory and in the field using Eq. 2. Under dry conditions, the phase shift in diurnal temperature cycle (φ) was 3 h between 1 and 5 cm and 12 h between 5 and 15 cm. The resulting D_h was 0.15 $mm^2\ s^{-1}$ in the organic surface soil and 0.23 $mm^2\ s^{-1}$ in the mineral soil, which agrees well with the laboratory measurements (Fig. 7). Under wet conditions, φ between 1 and 5 cm remained 3 h, whereas between 5 and 15 cm it was 7 h. In the organic soil, D_h remained 0.15 $mm^2\ s^{-1}$, whereas in the subsoil D_h increased to 0.4 $mm^2\ s^{-1}$. The calculations do not resolve whether λ was measured correctly, because soil compaction causes both C_h and λ to increase, which could leave D_h ($=\lambda/C_h$) unchanged. However, under dry and wet conditions, laboratory estimates of D_h were consistent with field data.

In our experiment, we wove the cables through the litter layer so we did not have to cut through the forest floor where most fine roots are present. McHale and Mitchell (1996) observed increased N concentrations in the soil solution, which lasted for up to 1 year after cable installation at 15 cm depth. Although we did not have a disturbance control, neither mineralization nor nitrification showed sudden changes during the first year of treatment, suggesting no clear disturbance effects due to cable installation. Burying heating cables will, however, reduce risks of desiccation of the litter layer and provide a more even heat distribution near the soil surface. Peterjohn et al. (1993) showed that, when heating cables are buried at 10 cm depth, heat is distributed more evenly at 5 cm depth than when cables are buried at 5 cm depth. Our measurements suggest that, if soils are rich in organic matter, strong heating gradients can still occur at the depth at which cables are installed, which will affect decomposition of organic matter deeper in the soil. Heating patterns will also depend on the location of the control sensors. If control sensors are located far from the cables, especially under dry conditions the majority of the soil will be heated above the target temperature, causing large risks of desiccation. Installing control sensors close to the cables will reduce this risk of desiccation, but will cause the bulk of the soil to be heated below the targets when soils are dry. However, at low soil moisture, effects of elevated temperature are likely to be small (Moore 1986). Under wet conditions, most of the soil will be heated to the target temperatures due to the high conductivity of the soil. Therefore, in wet soils, location of control sensors is less critical.

Conclusions

We measured an increase in nitrification and possibly net N mineralization in the 0 to 10-cm soil layer after 2 years of elevated soil temperature, which is consistent with the observed increase in N export from the catchment in runoff (Lükewille and Wright 1997).

Although soil temperatures increased in the heated area, litter decomposition rates did not. The litter in the heated-soil area appeared to be drier, which may have reduced a direct effect of elevated temperature on litter decomposition rates (Moore 1986). We found no evidence for a decrease in decomposability of litter produced under elevated CO_2. In fact, the incubations in the control catchment METTE suggested the opposite.

The heating cables caused a permanent increase in soil temperature but, especially when soils were dry, heating was not uniform. Burying heating cables will cause more even heating of the soil surface and reduce desiccation of the litter layer but may cause variable heating patterns at greater depth and disturbance of the

soil. Provided these limitations and problems are recognized, soil heating can be a suitable approach. It is a robust and relatively cheap method for long-term and large-scale manipulations of soil temperature, which may help to reduce uncertainty in predictions concerning soil response to elevated temperatures.

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