The Impact of Timber Harvest on Surface Soil Microbial Community Activity in Clearcut Missouri Ozark Forest Ecosystem Project Plots

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Abstract.—Transformations of carbon (C), sulfur (S), nitrogen (N), potassium (K), and magnesium (Mg) were determined for Oa+A- and EB-horizon soils collected from 12 permanent subplots located in MOFEP sites 1 and 3 from May 1995 to June 1999. Six of the subplots were clearcut in 1996, and six were left undisturbed. Mineralization of ^{14}C-lignocellulose (of *Quercus alba*) was used as a measure of microbial catabolic activity in the soils. *Quercus alba* is the predominant overstory tree species on MOFEP plots. Incorporation of $^{35}SO_4$ into organic sulfur compounds was used as a measure of microbial anabolic activity in the soils. Total C, N, and S (TC, TN, and TS) were determined by elemental analysis. Exchangeable K and Mg were determined using ammonium acetate extraction followed by atomic absorption spectrophotometry. After harvest, Oa+A-horizons from clearcut sites initially had higher rates of lignocellulose mineralization than controls, but these had declined by 2 years post-harvest. Organic S production rates were reduced to a large extent (by 80 to 90 percent, t-test, $p<0.05$) for soils from plots both high and low in the landscape. Concentrations of TC in Oa+A-horizons were lower in clearcut soils than controls on all dates beginning 1 year post-harvest (declining by as much as 35 percent compared with pre-treatment soils). Total S of both Oa+A-horizons decreased by nearly 40 percent after clearcutting, while EB-horizons tended to show an increase in TS following clearcutting (up nearly 73%). The TS content of litter falling into clearcut sites increased by nearly 50 percent 2 years post-harvest. Total N was lowest in clearcut plots. Exchangeable K concentrations in Oa+A-horizons from clearcut plots declined by nearly 75 percent 2 years post-harvest. Trends in exchangeable Mg concentrations in Oa+A-horizons were not easily determined for the clearcut plots 2 years post-harvest.

Microbial activity in forest surface soils is critical to the recycling of most elements used by plants and animals of the forest ecosystem. The predominant form of C found on the forest floor is lignocellulose. Soil microorganisms are responsible for the decomposition of lignocellulose, and they help to recycle the C, N, and S associated with this compound (Atlas and Bartha 1993, Stolp 1988). Thus, the degradation of lignocellulose in forest soils is dependent on the presence of an active community of bacteria and fungi possessing the requisite enzymes to degrade these compounds.

Microbial activity associated with degradation of lignocellulose in forest soils tends to be greater near the surface of the soil than at depth. Although soil type dependent, analyses of the distribution of saprophytic bacterial communities within forest soils suggest that both cell numbers and diversity of microbial communities decline with depth from the soil surface (Zvyagintsev *et al.* 1993). Thus, assessments of forest soil decomposition activities, and their relationships to nutrient cycling, must focus on analyses of surface soils.

Organic matter that enters forest surface soils is usually not completely degraded by microbial decomposers. Some of the more recalcitrant forms of lignin will accumulate in the soil's humus pool (Atlas and Bartha 1993). Microbial

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activity, therefore, plays an essential role in the ultimate production of soil humus.

Organic compounds present in forest soil humus may interact with other elements. The presence of ionizable carboxyl, sulfhydryl, or amino groups on soil organic matter enables this substance to serve as excellent ion exchange sites for cations and anions present in soil solution. In fact, numerous studies have documented interactions between forest soil organic sulfur and ionizable elements, such as N, Mg, K, and Ca (Homann and Harrison 1992, Mitchell *et al.* 1989, Rechcigl and Sparks 1985, Spratt 1998, Spratt 1997b, Watwood *et al.* 1993, Wiklander 1978). Thus, in addition to providing both carbon and energy sources for soil microbial communities, organic matter entering forest soils may be critical to maintaining the equilibrium of other critical potentially soluble elements found in those soils.

Disturbances that impact surface soils of forests may have major effects on soil microbial communities. In a study conducted in a clearcut pine forest, Lundgren (1982) demonstrated a 2-year increase in soil bacterial numbers in his cut plots immediately after timber harvest, followed by a reduction in bacterial numbers in comparisons with control plots. In another study where bacterial phospholipids were monitored 2 years post clearcut, soils from harvested sites demonstrated a 23 percent reduction in these indicators of bacterial biomass (Pietikainen and Fritze 1995). Pietikainen and Fritze (1995) also observed a significant reduction (by 21%) in microbial carbon 3 years post clearcut in the coniferous forest they studied. Lastly, Spratt (1997b) demonstrated that in surface soils of a Missouri forest that had been clearcut between 2 and 3 years prior to sampling, microbial incorporation of sulfate into organic matter was reduced by over 80 percent compared to control plots. These studies suggest an overall negative impact of clearcutting on soil microbial populations and processes they catalyze. So, too, might timber harvest be expected to have an impact on other critical features of the soil environment where microbes convert litter into soil humus.

Studies of surface soil C and S transformations and exchangeable bases in sample plots of MOFEP prior to experimental treatment (clearcutting) indicated dynamic seasonal changes in these elemental pools. Most notable changes were observed for total S (TS), organic S (OS), and the exchangeable bases K and Mg (Spratt 1997a). Observed relationships between OS and concentrations of exchangeable K and Mg in forest surface soils suggest that loss of OS from forest surface soils, due to natural variation in these compounds year to year or to some type of ecosystem disturbance, may influence the concentrations of exchangeable K and Mg available in those forest surface soils. Lost OS may reflect loss of an especially reactive component of the humus, which is critical to ion exchange reactions within the soil (Spratt 1998). We are currently studying how reduced concentrations of exchangeable K and Mg may affect organisms dependent on these nutrients in the surface soils.

This report will focus on post-treatment effects on surface soils of the Missouri Ozark Forest Ecosystem Project (MOFEP) in 12 subplots located within two watersheds of MOFEP sites 1 and 3 from May 1995 to June 1999. Six of these subplots were clearcut in 1996, and the other six were left undisturbed as controls.

OBJECTIVES

The major objectives of this study are:

1. To determine the short-term effects of even-aged and non-manipulative (no-harvest) forest management practices on soil carbon and sulfur constituents in MOFEP surface soils.
2. To assess any changes in indicators of surface soil microbial activity due to even-aged and no-harvest forest management practices in MOFEP surface soils.
3. To determine any relationships that may exist between soil microbial activity and concentrations of nutrient cations (e.g., K^+ or Mg^{2+}) in MOFEP surface soils.

MATERIALS AND METHODS

Sample Sites and Collection

Sample site selection for this study has been described in detail by Spratt (1997a). Descriptions of the overall MOFEP design and the harvest techniques used may be found in Sheriff, this proceedings. This post-harvest study was established to observe potential

changes in surface soil nutrients (e.g., C, N, and S) and indicators of microbial activity immediately following harvest, and for a period of nearly 3 years post-harvest. The plots studied here were located in the four paired watersheds described in the MOFEP pre-treatment document (see Spratt 1997a). These watersheds are located in MOFEP sites 1 and 3 (no-harvest and even-aged treatments, respectively, fig. 1). Of the watershed plots described (Spratt 1997a),

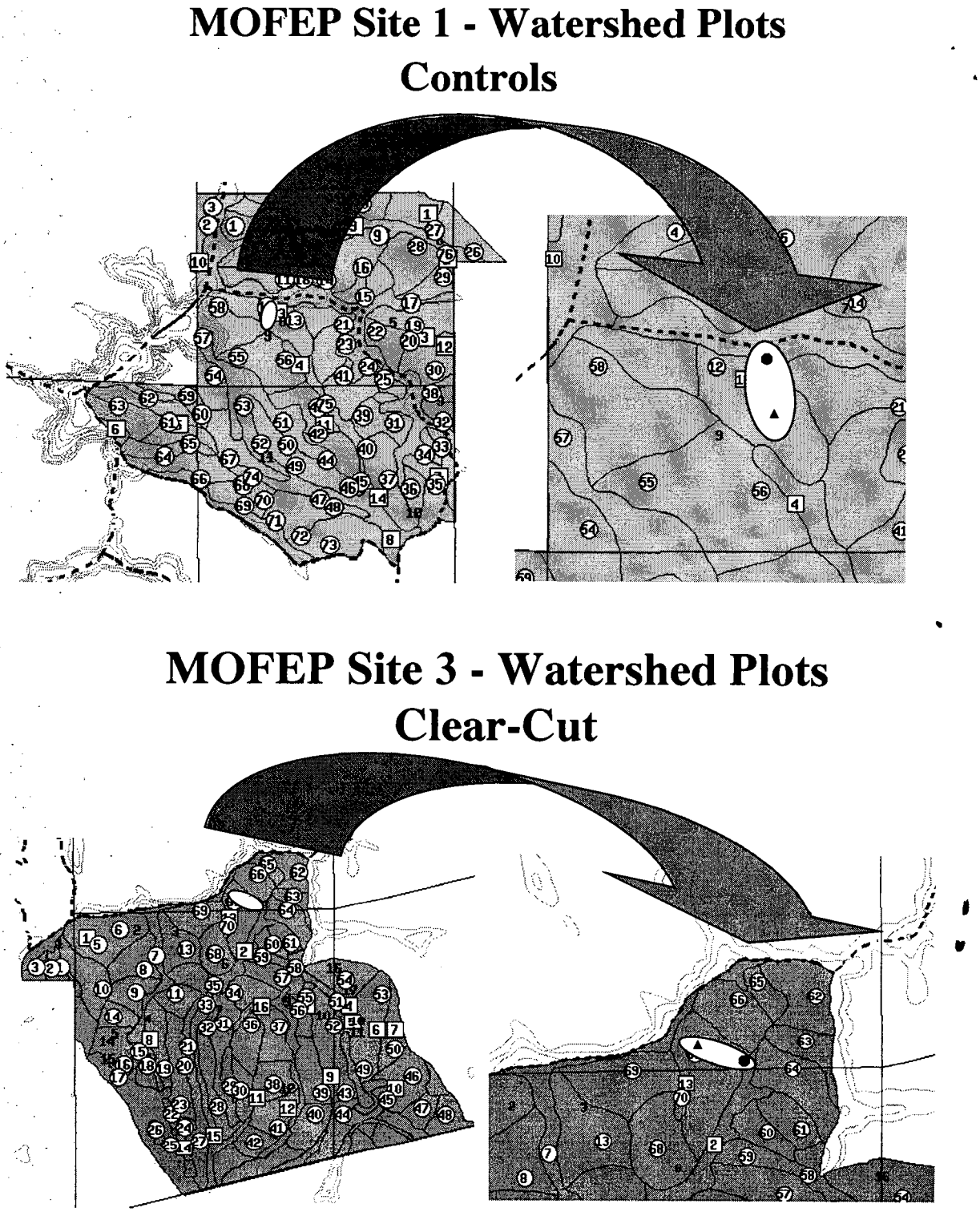

Figure 1.—Map of the MOFEP plots showing watershed plots in MOFEP site 1 (controls); watershed plots in MOFEP site 3 (clearcut). Numbers in circles indicate vegetation plots, numbers in squares indicate mast plots, remaining numbers indicate herpetofaunal arrays.

only those with north and east aspect (ELT 18) were actually clearcut. Thus, this report includes only results from clearcut and control subplots with this aspect. Additionally, within each of the two watersheds sampled, three subplots were established near the top of the slope (HI plots), and three subplots were established near the bottom of the slope (LO plots, see figure 2). Samples from sites with the uneven-aged treatment are not discussed here. Soil samples were collected from these 12 watershed subplots on the following sample dates: May 1995 (20°C), September 1995 (17°C), March 1996 (3°C), May 1996 (18°C), December 1996 (8°C), March 1997 (10°C), May 1997 (20°C), October 1997 (21°C), December 1997 (6°C), March 1998 (7°C), May 1998 (22°C) , December 1998 (6°C), February 1999 (8°C), and June 1999 (23°C) [field Oa+A-horizon soil temperatures indicated in parentheses]. For the locations of the watershed plots reported here, please refer to figure 1 or the detailed maps presented in Brookshire *et al.* (1997).

Sampling in the watershed plots included collection of litter, Oa+A-horizons, and EB-horizons from all subplots. On each plot the litter was removed from the forest floor in an area of ca. 100 cm^2 and placed in sample bags. The Oa+A-horizons to a maximal depth of approximately 2 cm from just below the litter were then carefully cut with a sharp spatula and placed in a sample bag. Finally, EB-horizons were collected down to a total depth of approximately 15 cm using a small trowel, carefully avoiding contamination of these soils with litter or Oa+A-horizons, and placed in a sample bag. All samples were stored in coolers on ice and transported to laboratories at the University of Tennessee at Chattanooga for processing.

Once at the laboratory, the soils and other samples were stored at 4°C and, within 3 days of collection, processed according to the chart in figure 3. To remove unwanted root material, rocks, and any other recognizable litter soils were passed through a 2-mm polyethylene sieve. The sieved samples were then subdivided into four fractions: one to measure extractable sulfate; a second percent moisture determination, elemental analysis, and determination of the exchangeable bases (K and Mg); a third to measure ^{35}S-sulfate incorporation into organic matter; and a fourth to measure ^{14}C-lignocellulose mineralization. The exchangeable sulfate extracts were placed in sealed vials and frozen at -20°C until further processing (see below); the samples for percent moisture were weighed and then dried at 60°C until a constant weight was obtained to determine the weight of moisture lost. After the percent moisture was determined,

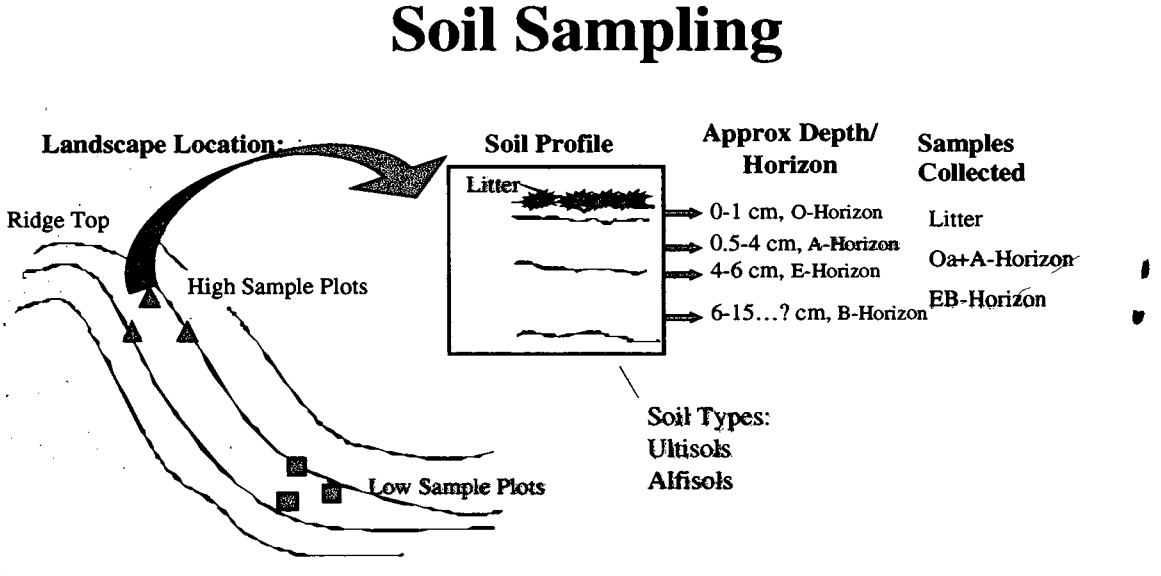

Figure 2.—*Soil sampling, showing location of subplots high and low in the landscape, and approximate location of soil samples from the soil horizons found in these ultisols and alfisols.*

Sample Processing

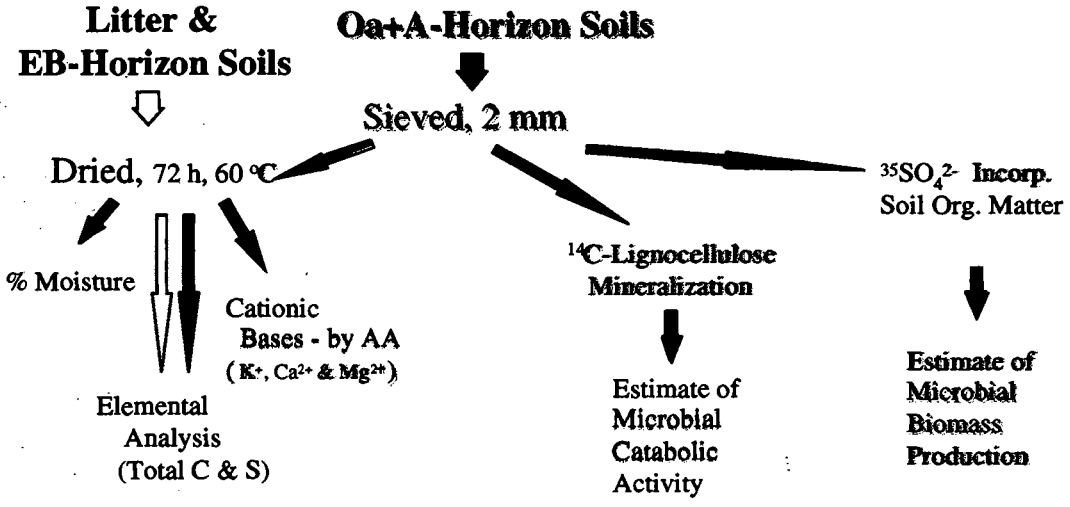

Figure 3.—*Sample processing for surface soil elemental analyses and measures of microbial activity.*

the dried soils were used to determine the soil TC, TN, and TS content and extractable base content (see below). Note: all data are presented on a g dry weight basis to negate changes due to different moisture content throughout the sampling period.

^{14}C-Labeled Lignocellulose Mineralization

Published techniques to specifically label the lignin or cellulose moiety of woody plant tissue were followed (Benner *et al.* 1984, 1985; Crawford and Crawford 1976; Crawford *et al.* 1977; Hackett *et al.* 1977). White oak (*Quercus alba*) was chosen as the species to be radiolabeled, based on its distribution throughout the MOFEP sites. The process of generating the ^{14}C-labeled lignocellulose using tree cuttings from MOFEP site 8 is described elsewhere (Spratt 1997a). Mineralization of white oak ^{14}C-lignin and ^{14}C-cellulose was determined using 200-ml microcosms. Suspended below stoppers inserted in the openings of the microcosm bottles was a test tube (3-ml capacity) into which 2 ml of 0.1 N NaOH was added. This NaOH served as a trap for any $^{14}CO_2$ generated during incubations. To set up the microcosms, 1 g field moist Oa+A-horizon soil was added to the bottom of each bottle. Time course experiments were initiated by addition of radiolabeled lignocellulose to soil in the bottoms of the microcosms. At specified times the NaOH in the trapping tube was removed and replaced with fresh NaOH. Radiolabel present in the NaOH removed from the traps was quantified using liquid scintillation counting. Maximal rates of lignin or cellulose mineralization were determined by calculating the maximal change in DPM (backgrounds subtracted) recovered for different times over the time course of the incubation. Lignocellulose mineralization is used in this study as an indicator of microbial catabolic activity (fig. 4).

^{35}S-Sulfate Incorporation Experiments

Incorporation of ^{35}S-sulfate into different Oa+A-horizon soil S pools was monitored using a modification of the technique of Watwood and Fitzgerald (1988), and which is described by Spratt (1997a). Approximately 1 g sieved soil was added to 12-ml conical centrifuge tubes. ^{35}S-sulfate, as $Na_2{}^{35}SO_4$, was added (0.2 ml, ca. 1 µCi containing a total of 8 pmols sulfate) to the soil samples to initiate the incubations. The soils were incubated at field temperature, aerobically, for 48 hours in the dark. To terminate incubations, the soils were placed in a -20°C freezer. The fate of ^{35}S-sulfate added to the soils was determined by sequential extraction of the soils to quantify the radiolabel present in the water soluble and adsorbed

Measures of Soil Microbial Activity and Associated Elements

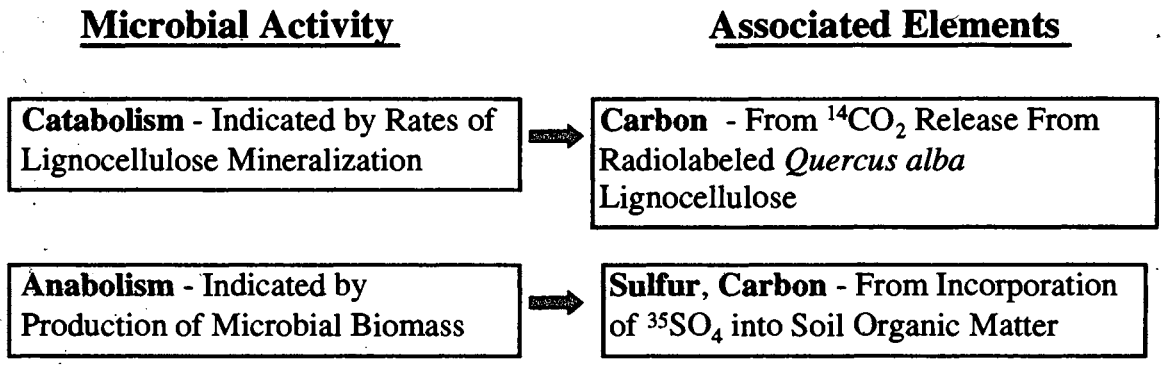

Figure 4.—*Definitions of microbial activity and elements involved in microbial metabolism.*

sulfate pools, and the OS fraction (Spratt 1997a, Watwood and Fitzgerald 1988). Sequential extraction with salts (1M Na_2SO_4, 1M NaH_2PO_4, and 1M NH_4Cl) was used to determine the amount of ^{35}S-sulfate adsorbed onto soil surfaces during the incubation. Radiolabel present in the OS fraction of the soil was determined using a strong acid/high temperature hydrolysis followed by a strong base extraction. Short-term production of OS in these soils represents microbial biomass production, and is used here as a proxy for microbial anabolic processes (fig. 4). The ^{35}S present in these fractions was determined using liquid scintillation counting.

Liquid Scintillation Counting

Quantification of the ^{14}C and ^{35}S used in all of the above experiments was made using a Wallac 1409 liquid scintillation counter. A biodegradable scintillation cocktail was used (Packard - Ultima Gold XR) for both radionuclides on all dates. Quenching of the samples was accounted for using external quench monitoring techniques (Wallac quench correction). For the ^{35}S-extraction samples specific quench curves were prepared using soils with no added ^{35}S, but extracted exactly as the radiolabeled soils. This was necessary because of the humic acids extracted from the soils along with OS compounds, which caused significant color quenching.

Determination of C, N, and S Pools

Soil TC and TS were determined for all Oa+A-horizons sampled. For some samples TN was also determined. Beginning with samples collected in March 1996, elemental analysis was conducted for Oa+A- and EB-horizons, as well as litter. A Leco CNS 2000 elemental analyzer, standardized with sulfamethazine and drift corrected using a NIS-traceable soil standard, was used for these analyses.

Exchangeable Bases

The exchangeable bases K and Mg were determined for all samples, using an ammonium acetate extraction procedure (Simard 1993, Spratt 1997a). Extracts from Oa+A- and EB-horizons were analyzed for K and Mg, using a Varian Spectr AA10 atomic adsorption spectrophotometer.

Statistical Analyses

Data presented here represent the results of a detailed study of two watersheds located within MOFEP. Due to limitations associated with potential pseudoreplication, multivariate analyses were not performed on these data. However, comparisons of the two watersheds were made using t-tests. To help demonstrate changes over

time for the treatment, compared with the controls, most data presented are normalized to control data (e.g., [experimental plot data] / [control plot data]). In this way, values greater than 1.0 represent increases in the parameter, while values less than 1.0 represent reductions in the parameter in comparison with the control plots.

RESULTS

Total C concentrations in watershed plots were either minimally affected by harvest or exhibited substantial differences from controls. For litter, minimal changes (t-test, $p<0.05$) were observed in clearcut plots compared with controls. In clearcut plots, the total C in Oa+A-horizons from 38 to 43 mmol/g dry weight for all sample dates from March 1996 to June 1999, while in control plots, total C in Oa+A-horizons ranged from 20 to 34.5 mmol/g dwt. Changes in TC for Oa+A-horizons were most notable for clearcut plots (t-test, $p<0.001$). A general trend towards lower TC in Oa+A-horizons from clearcut plots both high and low in the landscape began to show up in May 1997 (table 1). Total C in Oa+A-horizons of clearcut plots from low in the landscape differed from that of controls on each date sampled post-harvest. By June 1999, TC in Oa+A-horizons of clearcut plots low in the landscape was nearly 70 percent lower than in control plots and nearly 40 percent lower than clearcut plots pre-treatment.

Total C in EB-horizons was considerably lower than that found in Oa+A-horizons, averaging 2.7 and 2.8 mmol/g dwt for EB-horizons vs. 26.0 and 27.3 mmol/g dwt for Oa+A-horizons, from control plots high and low in the landscape, respectively. Treatment effects were observed for TC in comparison of clearcut plots with control plots (t-test, $p<0.05$). For plots high in the landscape, EB-horizons lost approximately 35 percent of the TC compared with control plots in June 1999 (table 1). Clearcut plots low in the landscape exhibited a loss of nearly 60 percent of TC in their EB-horizons compared with control plots in June 1999.

Total S from the plots studied here also varied considerably over the period sampled. Litter and Oa+A-horizons had 7 and 10 times higher TS than was found in EB-horizons (averaging 43.2 and 42.5 for litter, 43.4 and 49.0 for Oa+A-horizons, and 6.1 and 7.7 µmol/g dwt for EB-horizons; control plots high and low in the landscape, respectively, averaged over the entire study). Litter inputs to surface soils exhibited increases in TS in the clearcut plots compared

Table 1.—*Total carbon in clearcut plots (HI – high, and LO – low in the landscape), normalized to controls, May 1995 to June 1999. Mean values presented, (+/- 1 SD), n=3.*

Sample date	Clearcut HI			Clearcut LO		
	Litter	Oa+A-horizon	EB-horizon	Litter	Oa+A-horizon	EB-horizon
May 95	—	0.71 (0.23)	—	—	0.51 (0.08)	—
Sep 95	—	0.84 (0.25)	—	—	0.59 (0.05)	—
Mar 96	0.98 (0.02)	0.92 (0.27)	0.99 (0.26)	1.00 (0.01)	0.81 (0.24)	0.76 (0.26)
May 96	1.00 (0.03)	0.76 (0.10)	0.60 (0.15)	1.00 (0.01)	0.65 (0.29)	0.89 (0.17)
Dec 96	1.04 (0.02)	0.66 (0.04)	0.77 (0.10)	0.93 (0.03)	0.75 (0.09)	0.92 (0.18)
Mar 97	0.97 (0.01)	0.72 (0.05)	1.02 (0.16)	0.96 (0.02)	0.88 (0.26)	0.82 (0.19)
May 97	0.96 (0.01)	0.47 (0.37)	0.71 (0.32)	0.96 (0.02)	0.38 (0.02)	0.62 (0.20)
Oct 97	0.84 (0.16)	0.80 (0.37)	0.55 (0.15)	0.98 (0.03)	0.31 (0.11)	1.46 (0.94)
Dec 97	1.04 (0.02)	0.57 (0.25)	0.96 (0.23)	0.93 (0.03)	0.53 (0.02)	0.70 (0.09)
Mar 98	0.95 (0.06)	0.92 (0.10)	—	0.97 (0.08)	0.54 (0.14)	0.74 (0.07)
May 98	0.94 (0.06)	0.53 (0.10)	0.93 (0.13)	0.97 (0.01)	0.31 (0.14)	0.64 (0.06)
Dec 98	—	0.68 (0.28)	—	—	0.34 (0.07)	—
Feb 99	0.99 (0.03)	0.41 (0.16)	0.73 (0.16)	0.93 (0.01)	0.23 (0.04)	0.40 (0.19)
Jun 99	0.94 (0.03)	0.62 (0.22)	0.67 (0.07)	0.93 (0.02)	0.30 (0.09)	0.39 (0.02)

with controls over the period sampled (t-test, $p<0.001$). For Oa+A-horizons clearcutting resulted in marked loss of TS from plots both low and high in the landscape (table 2). TS in control plot Oa+A-horizons exhibited seasonal and annual variations that ranged from 25 to 72 µmol/g dwt for plots high in the landscape and from 27 to 103 µmol/g dwt for plots low in the landscape.

EB-horizons also exhibited changes in TS over the period sampled here. For samples from both control and clearcut plots, the general trend in TS was for reduced levels in EB-horizons in the clearcut plots (table 2, t-test, $p<0.05$). For soils from plots low in the landscape, the greatest loss of TS from the EB-horizons was approximately 40 percent for June 1999 samples.

Surface soil TN data were not available for samples collected earlier than May 1997 due to a malfunction in the elemental analyzer. Therefore, the data presented in table 3 are representative of only approximately two annual cycles

Table 2.—*Total sulfur in clearcut plots (HI – high, and LO – low in the landscape), normalized to controls, May 1995 to June 1999. Mean values presented, (+/- 1 SD), n=3.*

Sample date	Clearcut HI			Clearcut LO		
	Litter	Oa+A-horizon	EB-horizon	Litter	Oa+A-horizon	EB-horizon
May 95	—	0.60 (0.15)	—	—	0.46 (0.10)	—
Sep 95	—	0.69 (0.16)	—	—	0.57 (0.09)	—
Mar 96	0.81 (0.16)	0.91 (0.30)	0.91 (0.22)	0.86 (0.02)	0.78 (0.23)	0.76 (0.23)
May 96	0.79 (0.14)	0.72 (0.12)	0.64 (0.40)	0.64 (0.03)	0.55 (0.27)	0.77 (0.48)
Dec 96	1.15 (0.05)	0.63 (0.07)	1.15 (0.25)	1.79 (0.21)	0.58 (0.09)	1.22 (0.50)
Mar 97	1.41 (0.29)	0.82 (0.07)	0.82 (0.33)	1.16 (0.07)	1.09 (0.31)	0.69 (0.49)
May 97	1.10 (0.18)	0.46 (0.35)	0.65 (0.09)	1.10 (0.14)	0.36 (0.04)	0.64 (0.10)
Oct 97	0.74 (0.28)	0.86 (0.32)	0.60 (0.27)	1.19 (0.06)	0.32 (0.12)	1.16 (0.39)
Dec 97	1.15 (0.05)	0.60 (0.18)	1.09 (0.15)	1.80 (0.21)	0.49 (0.07)	0.55 (0.18)
Mar 98	1.31 (0.25)	0.81 (0.07)	—	1.40 (0.29)	0.42 (0.07)	0.71 (0.24)
May 98	1.20 (0.17)	0.33 (0.05)	0.84 (0.09)	1.05 (0.09)	0.36 (0.07)	0.66 (0.10)
Dec 98	—	0.74 (0.25)	—	—	0.34 (0.02)	—
Feb 99	0.94 (0.10)	0.50 (0.13)	0.94 (0.17)	1.15 (0.19)	0.26 (0.05)	0.60 (0.23)
Jun 99	1.31 (0.24)	0.66 (0.12)	0.98 (0.18)	1.17 (0.24)	0.30 (0.09)	0.59 (0.05)

Table 3.—*Total nitrogen in clearcut plots (HI – high, and LO – low in the landscape), normalized to controls, May 1997 to June 1999. Mean values presented, (+/- 1 SD), n=3.*

Sample date	Clearcut HI			Clearcut LO		
	Litter	Oa+A-horizon	EB-horizon	Litter	Oa+A-horizon	EB-horizon
May 97	1.22 (0.21)	—	0.92 (0.27)	1.06 (0.18)	—	0.78 (0.22)
Oct 97	0.90 (0.31)	0.84 (0.28)	0.77 (0.14)	1.23 (0.07)	0.37 (0.17)	1.68 (1.05)
Dec 97	—	0.83 (0.38)	1.33 (0.25)	—	0.62 (0.04)	0.84 (0.08)
Mar 98	1.47 (0.35)	0.91 (0.01)	—	1.40 (0.38)	0.59 (0.14)	0.71 (0.24)
May 98	1.24 (0.22)	0.63 (0.10)	1.33 (0.17)	1.14 (0.14)	0.41 (0.15)	1.10 (0.35)
Dec 98	—	1.03 (0.44)	—	—	0.38 (0.07)	—
Feb 99	0.94 (0.15)	0.64 (0.13)	1.13 (0.19)	0.93 (0.17)	0.47 (0.12)	2.15 (0.78)
Jun 99	0.65 (0.11)	0.77 (0.25)	0.72 (0.13)	1.46 (0.16)	0.36 (0.07)	0.49 (0.04)

post-harvest (from May 1997 to June 1999), with no pre-treatment data. The TN concentrations in litter and Oa+A-horizons were 8 to 10 times greater than the concentrations found in EB-horizons (averaging 938.3 and 843.4 µmol/g dwt for litter, 806.7 and 899.0 µmol/g dwt for Oa+A-horizons, and 93.6 and 111.9 µmol/g dwt for EB-horizons; control plots high and low in the landscape, respectively, averaged over the entire study). Total N concentrations in litter ranged from about 700 to 1,200 µmol/g dwt for all sample dates. For Oa+A-horizons, the control plots had greater TN concentrations than did the clearcut plots. The greatest change in TN was observed in comparisons between control and clearcut plots located low in the landscape. On each of the sample dates, TN in Oa+A-horizons was lower than that found in control soils (table 3, differing by as much as nearly 60% in December 1998). For EB-horizons, TN in clearcut soils differed little from that in control soils. However, in February 1999, one sample from clearcut plots low in the landscape had the highest TN value observed for all EB-horizons sampled over all dates (table 3). On that date, TN in this plot was more than two times greater than in the controls.

The concentration of exchangeable K was from four to five times greater in Oa+A-horizons than in EB-horizons of the plots studied, averaging 26.1 and 9.5 µmol/g dwt, respectively, for the sampling period for soils from the two different areas. Exchangeable K in Oa+A-horizons of the plots studied here also exhibited notable change as a result of the experimental treatment (t-test, $p<0.05$). Variation of exchangeable K in Oa+A-horizons by up to 1.9-fold was observed for the control plots over all dates sampled (table 4, ranging from a high of 42 to a low of 16.5 µmol/g dwt). However, comparisons of exchangeable K in Oa+A-horizons of clearcut plots high in the landscape with controls showed an initial increase immediately after harvest, followed by a decrease back to levels close to those originally found (table 4). Within surface soils from clearcut plots, variation in exchangeable K was more dramatic, dropping by as much as four-fold following treatment (from a high of 28 to a low of 7 µmol/g dwt). In comparisons of clearcut plots located low in the landscape with controls, the same pattern as seen for soil from plots located high in the landscape was observed. Exchangeable K levels increased initially following harvest, only to drop back to near initial conditions by 2 years after harvest (table 4). Exchangeable K values in surface soils of the low plots of the clearcut treatment also exhibited a reduction in exchangeable K concentrations, dropping by 2.8-fold (22 vs. 8 µmol/g dwt). For EB-horizons, exchangeable K from plots high in the landscape changed only a small amount compared with controls (table 4). Plots located low in the landscape generally had higher levels of exchangeable K over the period observed post-harvest.

Exchangeable Mg was as much as eight times more concentrated in Oa+A-horizons than in EB-horizons of the plots studied, averaging 41.8

Table 4.—*Exchangeable potassium in clearcut plots (HI – high, and LO – low in the landscape), normalized to controls, May 1995 to May 1998. Mean values presented, (+/- 1 SD), n=3.*

Sample date	Clearcut HI				Clearcut LO			
	Oa+A-horizon		EB-horizon		Oa+A-horizon		EB-horizon	
May 95	0.56	(0.08)	—		0.45	(0.01)	—	
Sep 95	0.95	(0.28)	—		0.55	(0.06)	—	
Mar 96	0.96	(0.12)	—		0.92	(0.21)	—	
May 96	1.21	(0.19)	—		0.64	(0.08)	—	
Dec 96	1.01	(0.22)	1.09	(0.14)	0.79	(0.04)	1.27	(0.18)
Mar 97	0.58	(0.13)	1.03	(0.03)	0.79	(0.12)	0.95	(0.05)
May 97	1.87	(1.19)	0.87	(0.04)	0.62	(0.03)	1.05	(0.24)
Oct 97	0.66	(0.08)	0.84	(0.17)	0.45	(0.04)	1.24	(0.51)
Dec 97	0.65	(0.22)	0.68	(0.01)	0.50	(0.06)	0.78	(0.19)
Mar 98	—		—		—		1.14	(0.51)
May 98	0.46	(0.12)	1.49	(0.42)	0.46	(0.27)	0.97	(0.33)

and 5.5 µmol/g dwt, respectively, for the sampling period for soils from the two different areas. Comparing exchangeable Mg from clearcut plots with controls, a slight increase in this element for Oa+A-horizons post-harvest is evident. For EB-horizons, exchangeable Mg was generally higher in all samples tested post-harvest (table 5).

Microbial catabolic activity in Oa+A-horizons from the plots studied, as evidenced by rates of lignocellulose mineralization, exhibited a marked increase in activity within 2 years post-harvest, followed by a notable reduction in activity between 2 and 3 years post-harvest. This trend was evident for samples collected from plots high and low in the landscape. Rates of cellulose mineralization peaked by December 1998 (increasing by as much as fourfold compared with controls for plots from low in the landscape, fig. 5). However, data for both February and June 1999 are not consistent with the higher rates observed in December 1998.

Lignin mineralization in surface soils of the watersheds studied followed a pattern similar to that described above for cellulose mineralization. There was a notable increase in lignin mineralization in May 1998 for Oa+A-horizons from high in the landscape (fig. 6, an increase from the controls of over sevenfold). No similar result was observed for soils from low in the landscape, however.

Microbial anabolic metabolism, as evidenced by rates of incorporation of SO_4 into soil organic matter, indicated substantial changes in the microbial communities present in clearcut Oa+A-horizons, compared with pre-treatment conditions. With variations between all plots pre-treatment ranging by no more than about 45 percent, the reductions of 82 percent and nearly 90 percent in clearcut plot soil OS production for soils from both high and low in the landscape, respectively, were significant (t-test, $p<0.05$, fig. 7).

DISCUSSION

The data presented in this report represent a study of the relatively short-term effects of clearcutting on surface soil microbial communities and certain elemental pools in selected plots of the MOFEP study. Although the initial scope of this study was scaled down after it was determined that some pre-treatment soil sample plots were not harvested, the data presented for the north and east aspect plots in these watersheds provide some useful insights into the short-term effects of clearcutting in surface soils of these Missouri forests. This is especially true when considered in light of the earlier study conducted in south and west aspect plots in Deer Run State Forest (Spratt 1997b). Direct comparisons may not be made with results of the Deer Run study (the clearcut sites in that

Table 5.—*Exchangeable magnesium in clearcut plots (HI – high, and LO – low in the landscape), normalized to controls, May 1995 to May 1998. Mean values presented, (+/- 1 SD), n=3.*

Sample date	Clearcut HI				Clearcut LO			
	Oa+A-horizon		EB-horizon		Oa+A-horizon		EB-horizon	
May 95	0.79	(0.26)	—		1.49	(0.12)	—	
Sep 95	1.05	(0.29)	—		1.43	(0.30)	—	
Mar 96	0.99	(0.18)	—		2.18	(0.05)	—	
May 96	1.23	(0.34)	—		1.23	(0.23)	—	
Dec 96	0.62	(0.23)	1.34	(0.52)	1.94	(0.30)	3.93	(1.97)
Mar 97	1.09	(0.10)	1.72	(0.29)	1.93	(0.38)	0.92	(0.42)
May 97	1.23	(0.21)	1.94	(0.62)	1.40	(0.14)	1.26	(0.37)
Oct 97	0.83	(0.07)	1.28	(0.23)	0.76	(0.31)	2.11	(0.86)
Dec 97	1.20	(0.09)	1.38	(0.22)	1.68	(0.22)	1.92	(0.44)
Mar 98	—		—		—		0.89	(0.32)
May 98	0.89	(0.31)	2.07	(0.82)	0.95	(0.47)	0.82	(0.12)

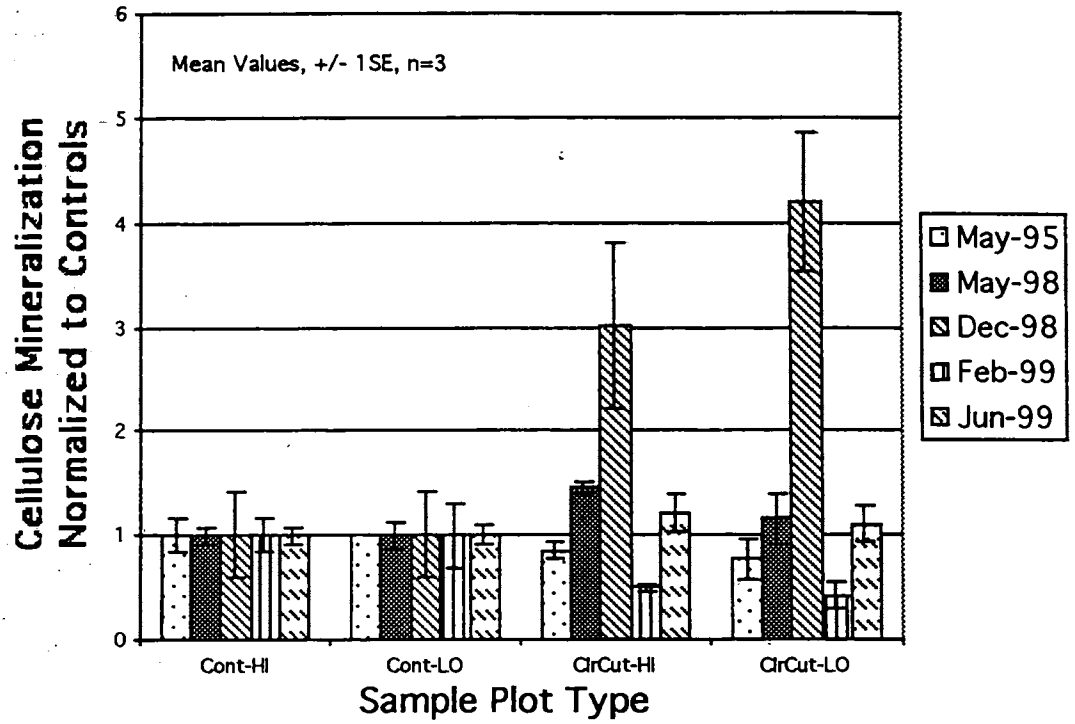

Figure 5.—*Cellulose mineralization normalized to mean control cellulose mineralization for Oa+A-horizons from watershed subplots located both high (HI) and low (LO) in the landscape. Cont indicates control plots and ClrCut indicates clearcut plots.*

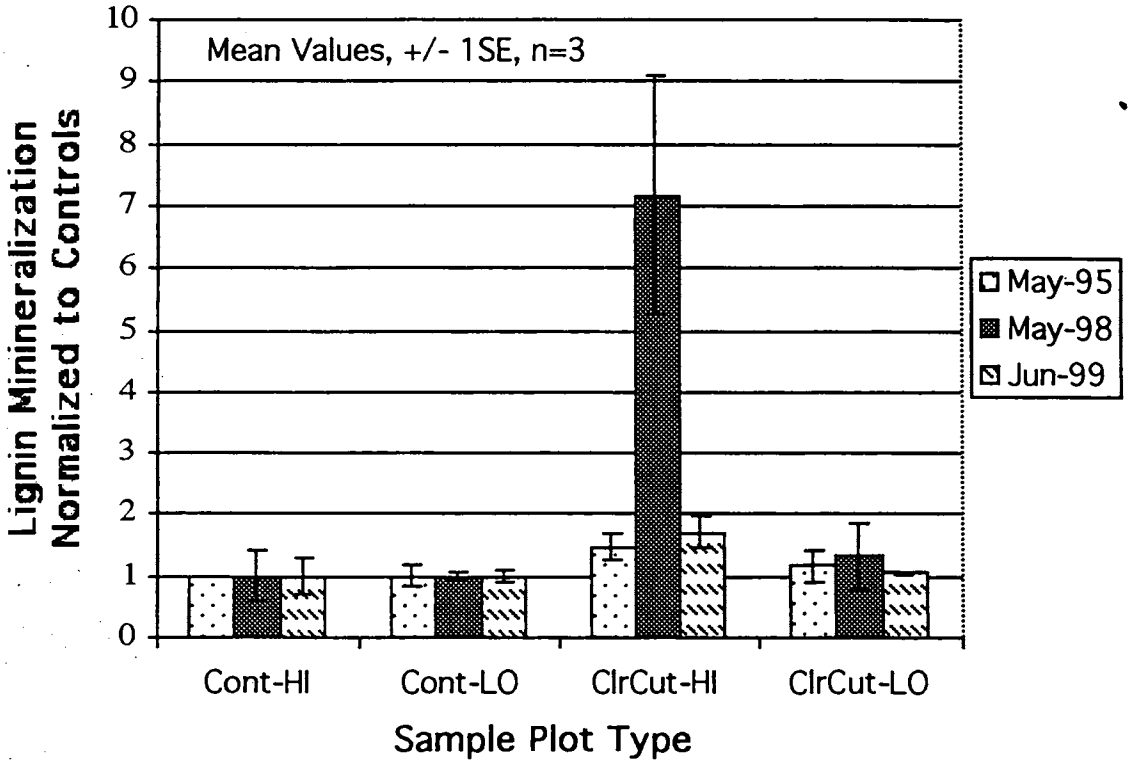

Figure 6.—*Lignin mineralization normalized to mean control cellulose mineralization for Oa+A-horizons from watershed subplots located both high (HI) and low (LO) in the landscape. Cont indicates central plots and ClrCut indicates clearcut plots.*

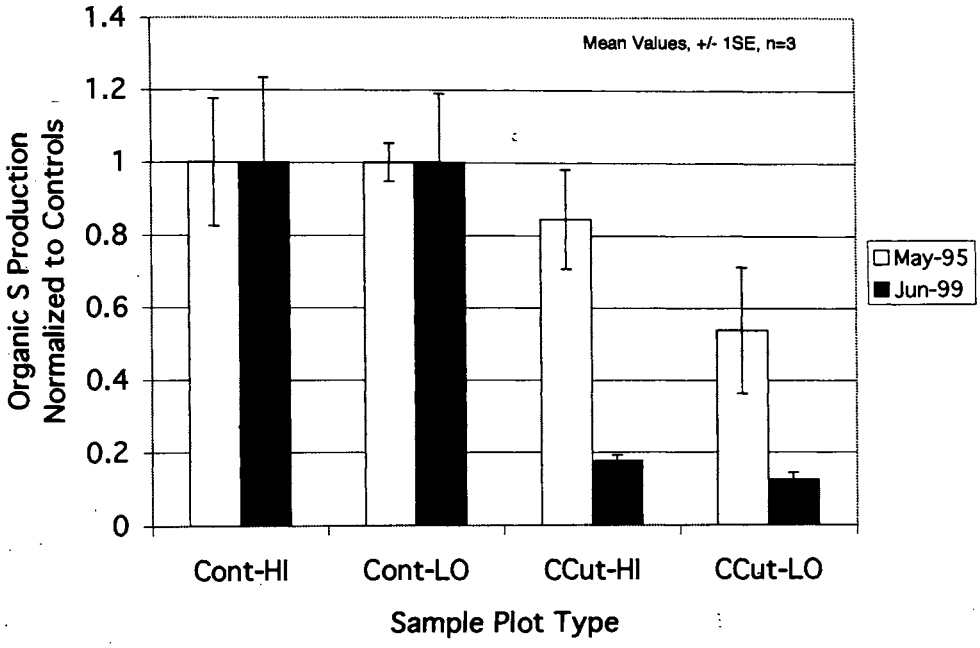

Figure 7:—*Microbial OS production as an indicator of microbial anabolism. Data are normalized to mean control OS production for Oa+A-horizons from watershed subplots located both high (HI) and low (LO) in the landscape. Cont indicates control plots and ClrCut indicates clearcut plots.*

pilot study were not part of MOFEP) detailing changes in TS and OS in Oa+A-horizons (actually called A-horizon soils in Spratt 1997b, but subsequently determined to be Oa+A-horizon soils), as well as changes in exchangeable K and Mg, since all of the plots studied in the Deer Run study were located in south and west aspect habitat. However, the Deer Run study did suggest potential changes in surface soil nutrient status that might be expected within 3 years post-clearcutting in soils from other aspect plots. The basic findings of the pilot study have been supported for the north and east aspect plots studied here.

Changes in TC in surface soils from the clearcut plots 2 to 3 years post-harvest may represent initial changes in the soil habitat similar to those observed in other forest ecosystems post-harvest. Two indicators of microbial activity in soils, cell numbers and microbial biomass, were significantly reduced within 2 years post-clearcut in European forest ecosystems (Lundgren 1982, Pietikäinen and Fritze 1995). In the pilot study conducted in Deer Run State Forest, similar reductions in indicators of soil microbial activity were observed 2 to 3 years post-harvest (Spratt 1997b). In the current study, TC reductions in Oa+A-horizons from clearcut plots may indicate loss of potential microbial carbon and energy sources. The observed reduction in TC from Oa+A-horizons (of approximately 25 percent for plots high in the landscape), if directly linked with microbial biomass, would lead to losses of microbial biomass of a similar magnitude to those observed in the European forests mentioned above (Lundgren 1982, Pietikäinen and Fritze 1995). The greater than 60 percent loss of TC for Oa+A-horizons from plots located low in the landscape may represent an even greater impact on the microbial communities in those soils.

Direct measurements of microbial activity using lignocellulose mineralization as a proxy for microbial catabolism in these soils support findings of the European studies to some degree (Lundgren 1982, Pietikäinen and Fritze 1995). For the soils studied here, mineralization activity appears to have peaked between 1 and 2 years post-harvest, and then declined. There was also, however, a large increase in microbial catabolic activity in control plot soils, presumably due to climatic effects of a seasonal nature, making this assertion more difficult to support.

Indications of microbial production in the soils studied here suggest a similar effect of clearcutting on microbial anabolism in the Missouri

soils as has been observed for European soils post-clearcut. The OS production data presented here do provide compelling evidence for significant (t-test, $p<0.05$) changes in microbial activity leading to the production of microbial biomass. In the 1992 Deer Run pilot study, Oa+A-horizons from south and west aspect clearcut plots 2 to 3 years post-harvest (Spratt 1997b) supported rates of microbial OS production 70 to 80 percent lower than control plots. The reductions in rates of OS production presented here, ranging from approximately 80 to 90 percent for Oa+A-horizons from north and east aspect plots, suggest that microbial communities in soils from both habitats are affected in similar ways by the disturbance of clearcutting.

Other evidence that microbial communities of surface soils from clearcut plots may be indirectly affected by changes brought about by clearcutting may be seen in changes of nutrient concentrations in surface soils post-harvest. The TS found in Oa+A-horizons of MOFEP plots is known to be composed principally of OS (up to 98 percent of the TS, Spratt 1997a, 1997b). This is in keeping with studies of other forested ecosystems in the U.S., Canada, and Europe where OS has been found to dominate the TS of these soils (Johnson *et al.* 1986, Mitchell and Zhang 1992, Van Loon *et al.* 1987, and Zucker and Zech 1985). If the TS found in Oa+A-horizons is principally OS, and if that S represents a relatively reactive form of organic matter in the soil (e.g., either lignin or humic substances, or microbial biomass with abundant sulfhydryl groups), then the losses of TS described here for clearcut plots could indicate a potential for lost ion exchange sites within the organic fraction of these surface soils.

The TS or OS within surface soils of the plots studied here is dynamic. Interestingly, the TS data collected during March 1998 from all plots studied showed large increases in this element compared with previous sample dates (and for all later sample dates, for that matter). It is not known whether unusual weather patterns associated with the intense El Nino weather phenomenon that occurred during part of this study were responsible for these increases in TS, via increased S input in precipitation for the plots studied. Sulfur inputs to forested ecosystems via precipitation, although declining in recent years, have been important sources of this element in many eastern U.S. forests over the past 30 years (Johnson *et al.* 1986). It is possible that such a boost in S via precipitation input might overshadow any changes that could have been occurring as a result of the MOFEP treatments during that winter. However, in subsequent samplings, the levels of TS dropped for all plots studied, indicating that the added S during March 1998 was particularly liable.

Another question of note regarding the potential indirect impact of changing elemental pools on surface soil microbial communities has to do with the origin of OS to MOFEP surface soils. Organic S from plant origin was not directly measured in either the MOFEP pre-treatment study or here, but, it may be inferred from litter TS data in this study that significant changes in the quality and quantity of plant-derived OS may have occurred on the clearcut plots post-harvest. A very large increase (20% greater than observed in control plots) in TS in litter 2 years post-harvest may represent significant changes in the sources of OS in these plots. We know that in clearcut plots of other Missouri Ozark forests shrub species tend to dominate several years post-harvest (Annand and Thompson 1997). Spratt (1998) suggested that changes in the quality of OS found in litter in clearcut areas might change post-harvest, potentially leading to the observed loss of OS in the Deer Run pilot study plots. Further study of the type of S present in litter of MOFEP and watershed plots is currently underway and may help shed light on whether the changes in TS observed here for clearcut plots might affect surface soil S pools. Other changes in surface soil nutrients observed in the Deer Run pilot study that correlated with changes in OS of Oa+A-horizons may have occurred in the plots described here.

Exchangeable K and Mg concentrations for the surface soils from north and east aspect plots studied here have changed post-harvest. While control plot exchangeable K in Oa+A-horizons exhibited variation of some 1.9 times over the period of the study, exchangeable K from clearcut plots decreased nearly fourfold post-harvest, compared with pre-treatment data. In the pilot study of Deer Run State Forest Oa+A-horizons 2 to 3 years post clearcut, Spratt (1997b) found a nearly 40 percent reduction of exchangeable K. The loss of exchangeable K from Oa+A-horizons of the clearcut plots in this study approaches 75 percent. Data for exchangeable Mg from the Oa+A-horizons of the plots studied here do not indicate a loss of the order observed for Oa+A-horizons in the pilot study (a loss of from 40 to 70%). There does

appear to have been some increased input of Mg to all soils studied here (controls and experimental) during late winter 1997. This makes the analyses more difficult to interpret. Dolomitic outcrops located throughout the MOFEP study sites (Meinert *et al.* 1997), might provide added Mg via weathering to some surface soils. However, for either exchangeable K or Mg, the potential exists for a loss of organic cation exchange sites (either purely C-based or OS-based) from surface soils of clearcut plots. The correlation between reduced exchangeable K and Mg and OS concentrations in clearcut Oa+A-horizon soils of the Deer Run State Forest pilot study was discussed previously (Spratt 1997b). Other evidence of a role for OS in the retention of exchangeable bases in forested ecosystems has been documented. A study by Watwood *et al.* (1993) suggested that ecosystem leaching of Ca, Mg, and K was positively correlated with the loss of soil organic S from the A-horizons of a wide range of forest soils. In the Missouri soils studied here, nutrient cations may also have been leached out of the surface soils, either into soil horizons deeper than 15 cm, or potentially downslope and lost to the local system. Loss of nutrient cations from forest ecosystems might have a negative effect on the productivity of those ecosystems.

The potential for loss of exchangeable bases, especially K, from the watershed surface soils studied here may have special implications for these forested ecosystems. In all cases studied in this watershed study, as well as in the MOFEP plots pre-treatment, soils sampled were classified as either alfisols or ultisols. One characteristic of alfisols and ultisols is their limited K-supplying power (Hausenbuiller 1978). In these soils, K that is available to primary producers comes primarily from exchangeable and soluble forms. As a result of the limited K-supplying power of the soils of the MOFEP plots, the predominant source of this base to the forest ecosystem must be atmospheric deposition, a noted source of K to eastern U.S. forests (Ragsdale *et al.* 1992). As the vegetation uses base cations, deciduous trees tend to accumulate exchangeable bases in surface soils (Johnson 1992). Since the soils sampled in this study were well drained and mostly lack clays, any changes that might lead to loss of ion exchange sites, especially in the surface soils, for exchangeable bases might lead to increased leaching of these nutrients. Although such deficits might not prove significant to deep-rooted overstory trees, organisms dependent on surface soils for their nutrients might be stressed due to low levels of these bases in surface soils of clearcut plots. The potential that microbial populations responsible for nutrient cycling in the surface soils might become K limited several years post-clearcut was discussed by Spratt (1998).

Is there a minimal level of organic matter, including OS, that will retain adequate levels of K from precipitation to help keep the Missouri Ozark forest ecosystem adequately supplied with this nutrient? The Deer Run pilot study included samples from plots clearcut 8 to 10 years earlier, and exhibited levels of exchangeable K, Mg, TC, and OS much lower than those found in controls. The need for further study of relationships between forest disturbance and soil microbial processes, related to nutrient status of the ecosystem, should be evident. Further comparisons of post-treatment surface soil OS and nutrient cation data with pre-treatment baseline data may help answer this question.

FUTURE RESEARCH

To fully follow up on the results of the Deer Run State Forest pilot study will require at least one more season of sampling from the 27 MOFEP vegetation plots that were extensively studied during the pre-treatment study (see Spratt 1997a). These plots studied pre-treatment are all south and west aspect, as were the Deer Run pilot study plots. Such a followup study was recently initiated and should be completed during summer 2001. Studies of the effects of clearcutting on surface soil microbial community structure are also under consideration.

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