Restoration of Highly Impacted Subalpine Campsites in the Eagle Cap Wilderness, Oregon

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Abstract

The effects of intensive recreation impacts and restoration amendments on soil parameters were assessed at four campsites in the Eagle Cap Wilderness, northeastern Oregon. Sites (2,215- to 2,300-m elevation) are characterized by shallow granitic soils, an *Abies lasiocarpa/Pinus albicaulis* overstory, and a *Vaccinium scoparium* understory. In fall 1995, plots were established at four campsites on three subalpine lakes in which soils were scarified, compost amended, and planted to native species. In summer 1998, we sampled surface soils (0–15 cm) on undisturbed sites (between and under vegetation) and unamended and compost-amended campsite soils. Samples were analyzed for total organic C, total N, potentially mineralizable N (PMN), NH_4, soil moisture, microbial biomass, basal 5-day respiration rates, and microbial community carbon utilization profiles. Unamended campsite soils had significantly lower levels of PMN, microbial biomass, basal respiration, and number of substrates metabolized in carbon utilization profiles. Compost addition elevated all these impacted parameters on campsite soils, although the increase in basal respiration rate was neither statistically significant nor sufficient to approach rates found underneath vegetation on undisturbed soils. Only the number of substrates metabolized in the carbon utilization profiles was significantly higher on compost-amended soils than on undisturbed soils. Levels of PMN indicate that campsite soils may lack sufficient N for rapid plant regeneration, whereas amended and undisturbed soils contained adequate quantities of available N. This work suggests that compost amendments can ameliorate impacts to soil chemistry and microbial populations caused by camping, without exceeding the N fertility found on undisturbed soils.

Key words: subalpine campsite restoration, recreation impacts, compost, soil chemistry, microbial communities.

Introduction

Wilderness areas are to be managed such that human impact is minimal. In many wilderness areas, however, recreation use causes substantial impact, particularly at popular campsites at accessible lakes in the subalpine zone. Impacts include loss of vegetation, reduction in depth of organic horizon, loss of surface organic matter, increased soil bulk density, and reduced infiltration rates (Marion & Merriam 1985; Marion & Cole 1996; Hammit & Cole 1998). These physical and chemical impacts directly and indirectly affect soil microbial communities and processes, nutrient status, and water-holding capacity (Lal & Stewart 1992; Zabinski & Gannon 1997). In many cases, highly impacted conditions on campsites are simply accepted; in other cases, managers seek to restore vegetation and soil on damaged campsites.

Restoration of wilderness sites generally involves closure of campsites. However, rates of recovery are slow, particularly at high elevation sites, where short growing seasons, shallow soils, and cool temperatures limit revegetation success (Cole & Ranz 1983; Cole & Hall 1992). Planting vegetation on disturbed subalpine campsites is often not highly successful (Moritsch & Muir 1993). Scarification and organic matter amendments of soil on closed campsites have enhanced the rate of vegetative recovery in some situations (Legg et al. 1980) but not in others (Zabinski & Cole 2000). Slow recovery rates on closed campsites are problematic because obvious impact invites continued use of sites that still are clearly campsites.

To date, there has been limited research performed on the restoration of campsites in the subalpine zone. Campsite restoration has focused on vegetative recovery and the restoration of soil physical properties (e.g., Marion & Cole 1996) with little research on microbial process recovery after campsite restoration attempts. Microbial communities in soils develop in response to

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plant root exudates and organic matter deposited in the form of senescent shoot and root material. The microbial community, in turn, contributes to ecosystem functioning through the mobilization of nutrients, transformation of soil organic matter, production of phytohormones, and contribution to soil food webs (Arshad & Frankenberger 1993; Bolton et al. 1993).

One means of restoring the chemical and microbial properties of disturbed soils is to amend them with organic treatments, such as composted sewage sludge (Sopper 1992). Although various organic soil amendments are widely used in more developed settings, the appropriateness of amendments in a wilderness setting has not been evaluated. Ideally in wilderness, native levels of microbial activity and soil nutrients should be restored without creating artificially fertile soils. Restoration treatments should move the state of the disturbed system toward, but not beyond, that of the undisturbed system.

The objectives of this study were to assess the degree to which campsite activity altered soil chemical and microbial properties relative to undisturbed soils and to assess the degree of recovery after the application of composted sewage sludge on impacted campsites. Specifically, we tested the following hypotheses: (1) Heavily used campsite soils have lower levels of carbon, nitrogen, and microbial activity than soil from adjacent undisturbed sites; (2) application of compost to campsite soils results in higher levels of carbon, nitrogen, and microbial activity than in unamended campsite soils; and (3) after the application of compost, soil parameters on amended campsite soils are similar to those on adjacent undisturbed sites.

Methods

Study Site

We conducted this study on four well established (>50 years) highly impacted campsites at subalpine lakes in the Eagle Cap Wilderness, Wallowa Mountains (45° 24′ N, 117° 44′ W), northeastern Oregon. Sites were located at an elevation of 2,215 to 2,300 m in forests with an overstory of *Abies lasiocarpa* (subalpine fir), *Pinus contorta* (lodgepole pine), and *Pinus albicaulis* (whitebark pine) and an understory dominated by *Vaccinium scoparium* (grouse whortleberry). Soils are shallow Cryochrepts and Cryorthents derived from granitic substrates, typically sandy and acidic (pH between 4.2 and 4.8).

Campsite restoration began in the summer of 1995. Sites were closed to recreation use, and 1-m^2 experimental plots were established on each of the campsites to test the effect of soil amendments, mulching, and planting on revegetation success (Cole & Spildie 2000). Treatments were randomly assigned to the experimental plots. In this study, we evaluated the effects of one treatment, the application of a 2.5-cm layer of compost raked into the top 10 cm of soil in the fall of 1995, relative to untreated campsite soils, and undisturbed soils adjacent to the campsites. The compost (Eko Kompost, Missoula, MT, U.S.A.) was a mixture of log yard waste composted with sewage sludge, 34% C, 1.5% N, C:N 20:1 (DeLuca & Lynch 1997). Two 40-pound bags of compost were added to 6 m^2 of experimental plots on each campsite. In July 1998 we sampled soils from untreated portions of the campsite (no soil or plant amendments), campsite plots that had been treated with compost, and adjacent undisturbed sites. Undisturbed sites were located as close as possible to campsites, in places with comparable topography and understory vegetation (which appeared to be present on the campsite before use). Vegetation cover on undisturbed sites is about 60%, with vegetation growing in clumps surrounded by areas without plants. Because vegetation cover is likely to affect underlying soil properties, we took separate samples from beneath and between established clumps of vegetation.

Plot Sampling

Soil microbial activity and nutrient availability were analyzed from samples collected from 0- to 15-cm depths (including surface organic matter) using a 2.5-cm diameter soil probe. In each of the four treatments at each campsite, we created three composite samples from eight subsamples. The carbon utilization profile of soil microbes was analyzed from three composite samples created from three subsamples (2.2 cm dia × 15 cm depth). The soil probe was flame sterilized between sample collections. Soil samples were placed in sterile plastic bags and stored on snow or ice for approximately 36 hr before processing in the laboratory.

Laboratory Analyses

For nutrient analyses and measurements of respiration, samples were shaken through a 2-mm sieve to remove coarse fragments. Ammonium (NH_4) was assessed from a 10-g subsample of soil, extracted in 50 mL of 2M KCl, filtered, and analyzed for NH_4^+-N using the Berthelot reaction (Willis et al. 1993). Potentially mineralizable N (PMN) was determined using 14-day anaerobic incubation (Hart et al. 1994). Five grams (oven dry equivalent) of moist soil was placed in a centrifuge tube, covered with 12.5 mL of water, head space air was displaced with N_2 (gas), and the tube sealed and incubated for 14 days at 25°C. Ammonium was then extracted from the anaerobic incubation soil samples by adding 12.5 mL of 4M KCl to each tube, shaking for 30 minutes, and filtrating through filter paper (Whatman no. 2).

PMN was calculated as anaerobic incubation NH_4 – natural soil NH_4. Microbial biomass C was determined by the fumigation/extraction-ninhydrin method as described by Joergensen and Brookes (1990) and modified by DeLuca and Keeney (1993).

Respiration rates were measured by static incubation using alkaline traps (Zibilske 1994). Twenty-five grams (oven dry equivalent) of mineral soil was placed in a glass jar, and water was added to bring soils to approximately 60% water-holding capacity. An open scintillation vial containing 30 mL of 1.0N NaOH was placed in the jar, and the sealed jar was incubated at 25°C for 3 days. NaOH was quantitatively transferred to 200-mL flasks, 30 mL of 2N $BaCl_2$ and 5 drops of phenylthaline indicator solution added, and the remaining NaOH titrated to a white endpoint with 1.0N HCl.

Air-dried soils were ground with a mortar and pestle and sieved to 76 μm (200 mesh). The fine ground samples were then analyzed for total C and N by dry combustion using a Fissions EA 1100 CNSHO analyzer (Fission Instruments, Milano, Italy). Additional subsamples of soil were air dried and analyzed for pH (2:1, 0.01M $CaCl_2$-to-soil), water-holding capacity at –30 kPa (Cassel & Neilsen 1986), and particle size distribution by hydrometer (Gee & Bauder 1986).

The carbon utilization capabilities of the microbial community were estimated using GN Biolog (Biolog, Inc., Hayward, CA, U.S.A.) microplates (Garland & Mills 1991; Grayston et al. 1994; Zak et al. 1994; Bossio & Scow 1995; Haack et al. 1995; Knight 1997; Zabinski & Gannon 1997). Microplates hold 95 unique sole carbon sources and a control well with no carbon, along with minimal salts for growth and tetrazolium redox indicator dye. Soil for carbon utilization profiles were homogenized, diluted 1:10 in 0.1% sodium pyrophosphate, and sonicated for 10 minutes. Heterotrophic plate counts (72 hr, R2A agar, Difco Laboratories, Detroit, MI, U.S.A.) were used to adjust the samples to a standard cell density for microplate inoculation. A second soil slurry was prepared (1:10 soil in sodium pyrophosphate, sonicated for 10 minutes) and further diluted in phosphate-buffered saline to a standard density of 10^4 CFU/mL. One hundred fifty microliters of each sample were inoculated into Biolog GN microplates and incubated at 25°C for 115 hr. The optical density of the wells was measured approximately every 24 hr with an ELISA microplate reader at 570 nm (Molecular Devices, Menlo Park, CA, U.S.A.).

From these we derived two indicators of microbial carbon substrate use—average well color development (AWCD) across time (Garland & Mills 1991) and number of substrates used. AWCD was assessed by averaging the absorbance readings for each well in a single sample plate after first subtracting the time zero reading. The number of substrates used was determined by counting the number of substrates for which absorbance readings exceeded 0.25. Carbon utilization profiles are an approximation of relative activity and functional diversity of culturable soil microbes to a range of substrates. There is no direct extrapolation of these results to the ecological function of the soil; as with any method that relies on culturing soil microorganisms in the lab, only a subset of microbes survives in lab cultures. Carbon utilization profiles, however, can still serve as a relative measure of microbial activity in disturbed soils (Zabinski & Gannon 1997).

Statistical Analysis

The three composite samples from each treatment at each site were averaged, for a total of four replicates (one per campsite) of each treatment. The resulting measures were analyzed with paired *t* tests. One-tailed *t* tests were used to test our a priori hypotheses, that soil nutrients and microbial populations would be reduced on campsites relative to undisturbed sites and that the addition of compost would elevate nutrient and microbial levels. Because we had no directional hypothesis, two-tailed tests were used to assess whether treated campsite soils differed from those on undisturbed sites. Differences in total C and N were tested with a Mann-Whitney rank sum test, because data for those parameters were not normally distributed.

Results

As expected, many parameters of undisturbed soil differed, depending on whether samples came from under or between clumps of vegetation. Soils beneath vegetation had substantially higher PMN ($t = 2.28$, $p = 0.05$), microbial biomass C ($t = 3.74$, $p = 0.02$), basal respiration rates ($t = 3.15$, $p = 0.03$), total organic carbon ($t = 2.17$, $p = 0.06$), number of substrates used ($t = 2.76$, $p = 0.04$), and AWCD ($t = 2.22$, $p = 0.06$) (Tables 1 & 2). For these parameters, comparisons with undisturbed sites were made separately for soil samples from beneath and between vegetation. Differences were not pronounced for total N ($t = 1.58$, $p = 0.11$) or ammonium (NH_4; $t = 1.03$; $p = 0.19$). For these parameters, data from both under and between vegetation were averaged for a single measure of undisturbed conditions.

Our first hypothesis, that heavily used campsites have lower levels of C, N, and microbial activity, was generally supported. We found that PMN, biomass C, basal respiration, and number of substrates used were all substantially lower in disturbed campsite soils than in the soils of undisturbed sites, particularly those directly beneath vegetation (Tables 1 & 2). PMN in campsite soil was barely over 50% of that found under vegetation on undisturbed sites. Microbial biomass C and

Table 1. Soil parameters across treatment types.

Campsite and Treatment	Organic C(%)	Total N (%)	NH_4 (mg/kg)	PMN (mg/kg)
Undisturbed soil (avg.)	5.16 ± 0.77	0.18 ± 0.02	1.54 ± 0.12	23.88 ± 6.33
Under vegetation	5.56 ± 0.74 ab	0.19 ± 0.03 ab	1.47 ± 0.10 a	25.41 ± 6.90 a
Between vegetation	4.76 ± 0.85 ab	0.17 ± 0.02 ab	1.61 ± 0.16 a	22.35 ± 5.78 b
Campsite soil	3.57 ± 0.48 a	0.17 ± 0.02 a	1.33 ± 0.22 a	14.34 ± 5.17 c
Amended campsite soil	5.46 ± 0.35 b	0.20 ± 0.02 b	2.23 ± 0.39 a	26.86 ± 6.79 ab

Values are mean ± SE. Letters in a column signify statistically different means ($p < 0.05$), as measured by paired t tests or Wilcoxon sum rank tests, as explained in text.

respiration on campsites, indicators of the magnitude of microbial populations, were less than 50% of their levels under vegetation on undisturbed sites. Differences between campsite soils and soils from between vegetation on undisturbed sites were less but still substantial.

For PMN, differences were statistically significant when campsite soils were compared with undisturbed soils, both under ($t = 5.89$, $p = 0.005$) and between vegetation ($t = 12.76$, $p < 0.001$). For the other parameters, tests of statistical significance were as follows: (1) microbial biomass C on campsite soils versus undisturbed soils under vegetation ($t = 3.73$, $p = 0.02$) and between vegetation ($t = 1.49$, $p = 0.12$); (2) basal respiration rates on campsites soils versus undisturbed soils under vegetation ($t = 3.88$, $p = 0.02$) and between vegetation ($t = 1.67$, $p = 0.10$); and (3) the number of substrates metabolized on campsite soils versus undisturbed soils under vegetation ($t = 2.43$, $p = 0.05$) and between vegetation ($t = 2.09$, $p = 0.06$). Disturbed campsite soils also had lower levels of organic C ($t = 1.49$, $p = 0.12$), NH_4 ($t = 0.64$, $p = 0.28$), and AWCD (under, $t = 1.49$, $p = 0.12$; between, $t = 0.88$, $p = 0.22$), but differences were not statistically significant. Total N did not differ between campsite and undisturbed sites.

Our second hypothesis, that the addition of compost to disturbed campsite soils would result in higher concentrations of soil nutrients and larger microbial populations, was also generally supported. Three years after they were applied, compost amendments resulted in significantly greater levels in three of the four soil parameters that were reduced on disturbed campsites (Tables 1 & 2). Amended campsite soils, when compared with unamended campsite soils, had significantly higher levels of PMN ($t = 3.11$, $p = 0.027$), microbial biomass C ($t = 2.98$, $p = 0.03$), and number of substrates used ($t = 6.31$, $p = 0.004$). Higher levels of microbial respiration on amended campsite soils were not statistically significant ($t = 1.23$, $p = 0.15$). Amended soils also had significantly higher levels of total C ($t = 6.01$, $p = 0.005$), total N ($t = 2.65$, $p = 0.04$), and AWCD ($t = 10.77$, $p = 0.001$), as well as more NH_4 ($t = 1.69$, $p = 0.09$).

Finally, we tested the null hypothesis that compost-amended soils would not differ from those found on undisturbed sites. This hypothesis was generally supported as well (Tables 1 & 2). Compost-amended campsite soils did not differ significantly from undisturbed soil in levels of organic C (under, $t = 0.06$, $p = 0.96$; between, $t = 0.70$, $p = 0.54$), total N ($t = 1.06$, $p = 0.37$), NH_4 ($t = 2.01$, $p = 0.14$), PMN (under, $t = 0.32$, $p = 0.77$; between, $t = 1.08$, $p = 0.36$), AWCD ($t = 0.99$, $p = 0.40$), or microbial biomass C (under, $t = 2.62$, $p = 0.08$; between, $t = 0.40$, $p = 0.72$). Although compost amendments resulted in significantly higher microbial biomass C, mean biomass on amended campsite soils was still only half that found under vegetation on undisturbed sites (Table 2).

Basal respiration rates on compost-amended sites were similar to those between vegetation on undisturbed sites ($t = 1.06$, $p = 0.37$). However, respiration rates remained significantly lower on compost-amended sites than under vegetation on undisturbed sites ($t = 5.90$, $p = 0.01$), suggesting that this is one soil parameter (per-

Table 2. Soil microbial parameters across treatment types

Campsite and Treatment	Biomass C (mg/kg)	Respiration (g/kg day)	AWCD	No. of Cs Used
Undisturbed soil (avg.)	181.96 ± 23.85	0.30 ± 0.02	0.53 ± 0.05	44.17 ± 2.31
Under vegetation	232.68 ± 27.55 a	0.37 ± 0.04 a	0.61 ± 0.08 a	49.42 ± 3.37 a
Between vegetation	131.24 ± 27.33 bc	0.23 ± 0.02 b	0.45 ± 0.04 a	38.92 ± 2.57 b
Campsite soil	71.79 ± 26.61 b	0.15 ± 0.03 b	0.37 ± 0.09 a	30.75 ± 5.80 b
Amended campsite soil	116.47 ± 34.92 ac	0.18 ± 0.03 b	0.65 ± 0.09 b	55.75 ± 2.17 ac

Values are mean ± SE. Letters in a column signify statistically different means ($p < 0.05$), as measured by paired t tests. AWCD, average well color development; No. of Cs, number of substrates metabolized in the carbon utilization profiles.

haps along with microbial biomass C) not adequately ameliorated by the compost. The higher number of substrates used on amended soils was statistically significant when compared with undisturbed soil between vegetation ($t = 7.19$, $p = 0.006$) but not when compared with undisturbed soil under vegetation ($t = 1.23$, $p = 0.31$). For this parameter, compost amendment exceeds the target suggested by undisturbed conditions, particularly when considering soils between vegetation clumps.

Discussion

Our results show that soils on heavily impacted campsites had lower soil nitrogen availability and soil microbial populations. Campsite disturbance of soils resulted in loss of the litter layer and degradation of soil physical structure. Loss of surface detritus (Oi horizon) and fermentation and humus layers (Oe and Oa horizons) results in reduced water-holding capacity, reduced nutrient reserves, and increased soil surface evaporation rates. This disturbance creates increased demand on organic matter reserves within the mineral soil and ultimately leads to a decline in microbial activity and diversity, as well as a decline in organic matter in the mineral soil (Jurgensen et al. 1997). The role of soil microorganisms is diverse, including nutrient cycling, contribution to soil aggregate stability, and interactions ranging from mutualistic to pathogenic with establishing plants (Lynch & Bragg 1985; Bever 1994; Haselwandter 1997; Watkinson 1998). Therefore, the decline in microbial biomass, basal respiration, and carbon utilization profiles suggests that important microbial processes could be greatly slowed in such disturbed soils. The reduced function of soil microorganisms can, in turn, impact revegetation success both immediately and over the long term.

The loss of organic N from these campsites may directly influence revegetation success. PMN as measured by anaerobic incubation has been identified as an excellent measure of the N available to plants during the course of a growing season (Powers 1980) and as a valuable index of biologically active N (Drinkwater et al. 1996). Levels of PMN in campsite soils are low enough to be considered potentially deficient for tree growth (Powers 1980), whereas levels under vegetation and in amended campsites are well within sufficiency levels.

Much of this impoverishment of soil nutrients and microbial populations may simply result from the removal of vegetation on campsites. Soil samples from sites between vegetation clumps consistently showed lower levels of microbial activity and PMN than soil from the rhizosphere directly underneath plants. This result, from the subalpine zone, is consistent with studies of desert and semi-arid shrublands that have documented islands of biological activity related to established patches of vegetation (Schlesinger et al. 1996; Kieft et al. 1998; Ayarbe & Kieft 2000). These results illustrate how plant litter and root exudates directly influence soil nutrient status and microbial activity (Lynch & Whipps 1991; Busse et al. 1996) and underscore the importance of plant reestablishment to soil restoration. The problem facing restorationists is how to restore soil conditions that are conducive to vegetation establishment and growth when those conditions result, to a great extent, from the existence of overlying vegetation.

Our results suggest that compost amendments could partially compensate for the lack of vegetation on disturbed campsites. The success of campsite restoration might be measured in terms of the degree to which the amended soils depart from the non-amended campsite and the degree of similarity between native soils and the amended soils. Three years after compost amendments were applied, levels of total carbon, PMN, and microbial carbon utilization profiles on campsites were equivalent to those under vegetation on undisturbed sites. Microbial biomass C and basal respiration also increased, but levels remained below those found on undisturbed sites, particularly those sites located under vegetation.

Addition of compost to campsite soils resulted in higher levels of microbial metabolic diversity, as measured by the number of substrates metabolized by soil microbes in carbon utilization profiles, than was found on undisturbed sites. This was the only parameter for which compost amendment exceeded the "target." We were particularly concerned that increases in available N, in excess of natural conditions, might cause shifts in plant species composition, even favoring exotic plant invasion (e.g., Wedin & Tilman 1996). However, PMN, an indicator of the amount of N available over a growing season, was not significantly higher on compost-amended soils than undisturbed soils, suggesting that amended campsite soils are within the natural range of N availability for these sites after three years of equilibration.

Application of composted sewage sludge to enhance revegetation success provides more than a nutrient source (Sopper 1992) to these disturbed soils. Compost as a soil amendment supplies a slow release of macro- and micronutrients, improved water-holding capacity, reduced albedo, and increased heat absorption in spring. All these factors will aid in the restoration of microbial processes and ultimately in the recovery of these sites to pre-disturbance conditions.

Results from the larger restoration experiment conducted on these campsites reinforce our conclusions that (1) the chemical and microbial impoverishment of

campsite soils is a significant barrier to campsite restoration in the subalpine zone and (2) compost amendments can be effective in ameliorating these conditions. Three years after closure, unamended and unplanted portions of the closed campsites still had virtually no vegetation cover. Density of seedling establishment was about twice as great on amended plots (187 vs. 103 seedlings per m^2), as were seedling growth rates. The mean height of seedlings on amended plots was 7.9 versus 4.0 cm on unamended plots (Cole & Spildie 2000). Seedlings growing in amended soils had three times the root biomass (0.12 g) of those in unamended soils (0.04 g) (Cole et al. 1999), and perhaps as a result seedling mortality rates during summer drought were only about one third as much (135 vs. 33%).

On heavily impacted granitic soils in the Eagle Cap wilderness, compost amendments mitigated the effect of camping disturbance on soil properties and enhanced revegetation success; however, such effects might differ elsewhere. Additional studies, on other soil types and at other elevations, would provide a better sense of the generality of our findings. Compost amendment proved feasible for small-scale backcountry restoration, as compost could be transported with horses and mules to the site. Large-scale wilderness restoration projects involving compost amendment might prove less feasible. Moreover, such treatments raise concerns about short-term nutrient releases, which were not monitored in this study. The potential for substantive releases and their likely effects should be evaluated before large-scale restorations are attempted.

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