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Tami

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Hi Fred: Below is what is in the database for Julie's dissertation:

3940. Spears, Julie. 2002. The imprint of coarse woody debris on soil biological and chemical properties in the western Oregon Cascades. Corvallis, OR Oregon State University. 99 p. Ph.D. dissertation. 050627/HJA4/LTER5-2/DIRT/NC.

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Fred 1/12/06

Thanks Much! Tami

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The Imprint of Coarse Woody Debris on Biological and Chemical Properties of Soil

Julie D.H. Spears

Introduction:

The spatial heterogeneity of coarse woody debris (CWD) on the forest floor is a prominent feature of Pacific Northwest forest ecosystems. It has only recently become evident that the spatial variability of CWD inputs may affect forest soil processes. Previous research has focused on the importance of CWD as habitat for vertebrates and invertebrates (Marra and Edmonds 1998), and as a nursery for seedlings (Harmon et al.. 1986). Many studies have investigated the nutrient stores and the nutrient dynamics within CWD of various age classes (Holub et al. 2001, Krankina et al. 1999, Busse 1994, Means et al. 1992, Arthur and Fahey 1989, Sollins et al. 1987, Graham and Cromack 1982, Lambert et al. 1980, Grier 1978), while other studies have considered N- fixation in CWD (Roskoski 1980), N mineralization in highly decayed wood (Hart 1999) and the nutrient losses from CWD by fungal sporocarps (Harmon et al. 1994). The seasonal and long-term water balance of CWD in early decay stages has been investigated (Harmon and Sexton 1995), as well as wood mineralization by invertebrates (Ausmus 1977), and the chemistry of the interstitial water in a montane forest of western Wyoming (Yavitt and Fahey 1985). However, the effect of CWD on soil solution chemistry, nutrient cycling and availability, soil physical structure and formation of soil organic matter is less well known. Therefore, studies on the spatial and temporal imprint of CWD on

forest soils are timely. This information can fill critical gaps in our understanding of the role of CWD in Pacific Northwest forest ecosystems.

Current forest management practices that include whole tree harvesting and clear-cutting are dramatically decreasing the amount of CWD left on the forest floor in the Pacific Northwest (Sollins et al. 1987). Furthermore, a decrease in the amount of CWD may affect the amount of organic matter and nutrients in PNW soils. Soluble organic matter that leaches from CWD into the soil profile may be a long-term source of carbon (C) and other nutrients to the soil beneath decomposing CWD. Decaying boles may also act as a nutrient pool that is slowly released over hundreds of years (Arthur and Fahey 1990). However nutrient dynamics during CWD decomposition vary across species and ecosystems and therefore, no clear picture of the effect of CWD on soils has emerged.

Carbon in CWD has three fates: it can be respired as CO_2, fragemented into smaller chunks, or leached as DOM into the soil beneath it. DOM leaching from CWD may contain incompletely degraded soluble components of wood, which is not only an important source of soil organic carbon, but may also play an important role in humification in soils (Yavitt and Fahey 1985). DOM can determine the quantity and quality of soil organic matter, and hence soil fertility (Schoenau and Bettany 1987), as well as the balance of N and P in the soil (Qualls and Haines 1991). Organic compounds containing N and P are mobilized as dissolved organic nitrogen and dissolved organic

phosphorus through heterotrophic microbial activity and are then carried from CWD through the soil horizons by percolating water. Fahey and Yavitt (1988) found more than 95% of the total dissolved N and P leaching from decomposing boles to be organic in form.

Dissolved organic carbon (DOC) chemistry may affect its movement through the soil profile. The DOC in the interstitial water of decaying wood has been shown to have a higher percent hydrophobic acid content than water in lakes and rivers as well as soil solutions percolating through the O, A, and B horizons (Qualls and Haines 1991). Over 80% of the DOC in the log leachate was found to be acidic in nature, dominated by hydrophobic compounds, which Yavitt and Fahey (1985) believe to reflect the highly lignified nature of the wood. Many studies have shown the selective removal of the hydrophobic acid fraction as the soil solution percolates through the soil profile (Qualls and Haines 1991). Aromatic hydrophobic compounds are tightly retained in mineral soils, and can displace low molecular weight compounds, hydrophilic compounds or even anions such as phosphates and sulfates (Kaiser and Zech 1998, Kaiser et al. 1996, Guggenberger and Kaiser 1997, Kaiser and Zech 1996). This variation in sorption affinity may affect the distribution of nutrients and carbon throughout the soil profile below CWD.

Only one study has looked at the effect of leachates from CWD on the microbial community in soils under CWD. Busse (1994) found that microbial biomass carbon, and

the ratio of microbial carbon to total carbon were greater under decaying boles than in soils without CWD in a lodgepole pine ecosystem in central Oregon (Busse 1994). Busse (1994) concluded that nutrient accumulation and/or release from CWD contributed little to the O horizon or mineral soil, however he did state that several soil processes were indirectly influenced beneath the decaying boles, such as a large influx of carbon from the decaying bole into the mineral soil below it.

Clearly, more information is necessary on the effect of CWD on soil chemistry and biology. Therefore, I investigated the effect of CWD on soil chemistry and biology with a two-part study at the H.J. Andrews Experimental Forest log decomposition site (Harmon 1992). This study lasted two years during which time soils were sampled beneath CWD frequently for seasonal differences, and leachate and soil solution from under CWD was collected as often as possible from the fall of 1999 until the spring of 2001.

Hypotheses:

Chapter two explores the hypothesis that the chemistry of CWD leachate is more acidic, and contains more total DOC, polyphenols, and hydrophobic DOC than leachates from the forest floor. If this hypothesis holds, the acidic DOC-rich solution would potentially acidify the soil beneath the CWD, leach base cations from the surface soils under CWD. Acidification would also contribute to podzolization at deeper soil depths below CWD by releasing aluminum and iron from soil matrices. I also

hypothesized that hydrophobic rich, and nutrient poor DOC would dominate the chemistry of the leachates in the later stages of decomposition. To investigate CWD leachate chemistry, I sampled leachates from under CWD of different stages of decay with zero-tension lysimeters. I also compared bulk soils to a depth of 60 cm from under CWD of advanced decay stages with soils that have not had CWD on them. These soils were analyzed for differences in pH, exchangeable acidity and cations, and iron and aluminum chemistry to determine if the acidic CWD leachates did indeed change the chemistry of the underlying soil.

Chapter 3 focuses on the effect of CWD on soil biology and nutrient cycling. I hypothesized that decomposing CWD would contribute a large amount of carbon (C) as DOC to the underlying soil. I further hypothesized that this large C influx would change the structure of the soil microbial community. To investigate these hypotheses, I looked at several indicators of microbial activity including tests for enzyme activity and substrate specificity. To determine the effect of CWD on soil nutrients, I analyzed leachate collected from three depths under a chronosequence of CWD logs for C, N and P. I also analyzed soils for nitrogen cycling rates and nutrient pools.

All results are documented in the proceeding chapters and a final conclusion discussing these results is presented in chapter 4.

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The imprint of coarse woody debris on soil chemistry in the Oregon Cascades

J.D.H. Spears and K. Lajtha

Abstract:

Coarse woody debris (CWD) may affect soil processes through the release of acidic dissolved organic matter (DOM). Acidic DOM has been implicated in many soil processes such as podzolization, the displacement of anions from the soil matrix, the release of P from iron and aluminum hydroxides, and the dissolution of soil minerals. We investigated the potential imprint of CWD on soil chemistry at the H.J. Andrews Long-term Ecological Research Site in the Oregon Cascades. CWD leachates were sampled with zero-tension lysimeters under CWD at different stages of decay and under the forest floor without CWD as controls. Solutions were analyzed for total dissolved organic carbon (DOC) and polyphenol concentrations. DOC was further separated by chemical fractionation. We also sampled soil from underneath CWD and from soils without CWD to a depth of 60 cm. Samples were analyzed for pH; exchangeable acidity; exchangeable aluminum (Al) and iron (Fe) in $BaCl_2$; aluminum and iron bound inorganic phosphorus and organic phosphorus; and free, amorphous and organically bound iron and aluminum. CWD leachates were lower in pH and contained more polyphenols and DOC than leachates without CWD, although chemical DOC fractions from CWD and the forest floor were similar. Leachate from under class

3 CWD contained more hydrophobic DOC compounds than from soils under other decay classes. Surface mineral soils under CWD were lower in pH, had more exchangeable acidity and more exchangeable aluminum ($p=0.04$) and iron ($p=0.06$) than adjacent soils. There were no significant differences between soils under CWD and control soils for any of the Al or Fe fractions. At depth of greater than 5 cm, there were no differences between control soils and soils under CWD. Our results suggest that CWD acidifies the surface soil as it decomposes by decreasing exchangeable bases and increasing exchangeable acidity and aluminum. However, soils under the most highly decayed CWD, or in deeper soils were not affected by CWD leachates. Although we hypothesized that CWD may have a long-term influence on soil pedogenesis by creating a permanent mosaic of wood imprints in the soil, this study suggests that the effects of CWD on soil chemistry were evident only during the earlier stages of decay and only affected the top 5 cm of the soil.

Introduction:

The spatial heterogeneity of coarse woody debris (CWD) on the forest floor is a prominent feature of Pacific Northwest forest ecosystems. It has only recently become evident that the spatial variability of CWD inputs may affect forest soil processes. However, current forest management regimes differentially affect the amount of CWD contributed to the forest floor altering the spatial mosaic of the forest floor.

Spatial heterogeneity of litter inputs may have profound effects on ecosystem processes such as nutrient cycling and pedogenesis. In desert ecosystems, spatially discrete litterfall may have cascading effects on nutrient cycling, plant establishment, soil erosion, and ultimately on net primary productivity (e.g. Schlesinger et al. 1996). However, the spatial variation in nutrient cycling and pedogenesis has been less well studied in old-growth forests where CWD may provide 'islands of carbon' which may be likened to the concept of 'islands of fertility' in desert ecosystems (Schlesinger et al. 1996).

If CWD acts as well decayed, lignin-rich islands of carbon in a forest floor spatial mosaic, highly acidic organic compounds that leach from the CWD into the soil may significantly impact the underlying soil through chemical weathering processes. Long-term productivity depends, in part, on the ability of forest soils to re-supply inorganic nutrient cations through long-term chemical weathering (Arocena and Glowa 2000, Giesler et al. 2000). Removing CWD may reduce soil chemical weathering in these

ecosystems. Mismanagement of forest soil organic matter and nutrient supplies may have long-term consequences such as the decline in forest productivity (Perry 1994).

Chemical soil weathering is the dominant process that releases inorganic nutrients for plant uptake (McKeague et al. 1986). Chemical weathering occurs when acidic substances leach from decaying plant tissues, percolate through the soil profile in solution, and dissolve soil minerals by two methods: (1) Protons promote soil mineral dissolution by bonding to the oxide surfaces, promoting the detachment of ions into the soil solution and (2) Organic ligands can also bind with the surface oxides forming metal-ligand complexes, which then detach into solution removing the metal from the mineral (Sparks 1995). Therefore, the composition and concentration of leachates, in addition to the concentration of acidity, may be an important influence on the rates of soil mineral weathering. Lundstrom (1995) found that in surface soils, dissolved acidic organic matter increased aluminum complexation from primary minerals three and half times faster than occurred without the acidic DOM.

The acidic organic components of leachate can range from simple aliphatic acids to more complex aromatic and heterocyclic acids (Tan 1986) and are collectively termed dissolved organic matter (DOM). DOM contains many different compounds that have varying chemical reactivity towards soil exchange surfaces and as such are difficult to identify. Therefore, fractionation methods have been developed to determine the functional chemical categories of DOM, rather than specific compounds. Solution DOM

can be chemically fractionated into hydrophobic and hydrophilic compounds (Qualls and Haines 1992). Hydrophobic compounds include high molecular weight carboxylic acids and polyphenols and have been shown to displace loosely held hydrophilic compounds, as well as sulfates and some phosphates from soil exchange sites (Qualls 2000, Kasier and Zech 1996, Qualls and Haines 1991). Other components of DOM, such as low molecular weight organic acids, may comprise less than 10% of total dissolved organic carbon (DOC) in soils, but their ability to form strong complexes with metals can have a large influence on soil processes (Tani and Higashi 1999, Fox 1995).

Acidic organics have also been hypothesized to be involved in podzolization (Yavitt and Fahey 1985, Kayahara and Klinka 1996), which is the translocation of organic matter, aluminum, and iron to lower soil horizons (Browne et al. 1995, David et al. 1995). Because acidic organic functional groups have been shown to form complexes with metals, Yavitt and Fahey (1985) have suggested DOM leaching from CWD would complex metals in soils underlying CWD. Acidic ligand-metal complexes migrate downward in the soil profile until microbial activity releases the metals by decomposition of the organic ligand (Lundstrom 1993). This results in the redistribution of metals in the soil profile (Jones 1998) and thus CWD may affect the distribution of metals throughout the soil profile.

Many soils show evidence of podzolization without necessarily developing a spodic horizon (Birkeland 1994) and thus podzolization processes may occur in soils

underlying CWD but no spodic horizon may develop. Kayahara and Klinka (1996) investigated B horizon soils beneath CWD in British Columbia but found no evidence of podzolization or spodic horizon development. However, Crampton (1982) found podzolization of soils under individual tree canopies in the same ecosystem that he hypothesized were caused by the acidic stemflow of the trees. We expect that CWD leachates would have a similar affect on the underlying soil chemistry.

Acidic DOM may also affect the availability of nutrients such as phosphorus in soil by increasing the dissolution of aluminum and iron hydroxides (Yavitt and Fahey 1985). Free Al and Fe may complex P. This will increase the amount of metal-bound P in soils under CWD and limit P availability for plant or microbial uptake. Alternatively, organic acids in CWD leachates may increase the rate of P release from iron and aluminum hydroxides by complexing the metals, which will release the phosphate ion back into solution (Tan 1986, Jones 1998).

We investigated the influence of CWD on soil chemical weathering, podzolization, carbon, aluminum, iron, and phosphorus chemistry in soils in an old-growth Douglas-fir-western hemlock forest in the Oregon Cascades. We hypothesized that the concentrations of DOM leaching from CWD would be greater than concentrations of DOM leaching from the forest floor. The leachates from CWD should also be much more acidic and rich in hydrophobic and polyphenol compounds. These acidic inputs will bind to the soil exchange sites, complex metals, decrease soil pH and

increase the exchangeable acidity in soils beneath CWD. Base cations will be decreased as they are leached as base partners to maintain electrical neutrality with the organic ligands. Furthermore, we hypothesized that there will be a strong signal of podzolization as measured by the ratio of the organically bound Al to C. Podzolization processes will control Al and Fe chemistry, binding Al and Fe to carbon compounds and eluviating these complexes to lower soil horizons.

Site description

This study was conducted at the H.J. Andrews Experimental Research Forest in the Willamette National Forest in the western Oregon Cascades (44 13' 53" N, 122 13' 40" W). The study site within the forest supports a *Tsuga heterophylla/Rhododendron macrophyllum/ Berberis nervosa* plant association (Franklin and Dyrness 1988). Tree species include large *Pseudotsuga menziesii* (Mirb.) Franco (Douglas-fir) with smaller *Tsuga heterophylla* (Raf.) Sarg. (western hemlock) and *Thuja plicata* Donn ex D. Don (western red cedar) in the overstory. *Tsuga heterophylla, Taxus brevifolia* Nutt. (pacific yew), *Rhododendron macrophyllum* G. Don, and *Acer cercinatum* Pursh (vine maple) make up the majority of the understory vegetation. The forest floor is covered with a layer of mosses including *Eurhynchium oreganum* (Sull.) Jaeg. (Oregon beaked moss) and *Hylocomium splendens* (Hedw.) B.S.G. (step moss).

The climate is Mediterranean. Precipitation at the site averages 350 cm y^{-1} and the average annual temperature is 10° C. The elevation of the site is approximately 535

meters. Soils have been classified as Inceptisols; Humic Dystrodept. More recent chemical analysis, however, suggests that these soils may be classified as Andisols (Keys to Soil Taxonomy, 1998). Bulk density averages 0.7 g/cm^3, percent soil carbon is > 2% in surface samples, and oxalate extractable Al plus one half the oxalate extractable Fe totals more than 2 as well (see results below).

Experimental Design and Field Methods:

CWD was classified with the CWD decay classes according to the method of Triska and Cromack (1979). Class 1 is a freshly fallen log with sound wood and intact bark. A class 2 log has an initial rotting of the sapwood. The bark on class 3 log begins to slough off and the sapwood decays. The log is classified as class 4 after the sapwood has rotted, the bark has sloughed off, and the heartwood is thoroughly rotted. When the log has no structural integrity, has settled into the soil and is thoroughly rotted red mush, it is classified as class 5.

We used class 2 CWD from the log decomposition experiment, which is described more extensively in Harmon (1992). Class 3, 4, and 5 CWD were naturally occurring in the same site. We used class 2 CWD from the log decomposition experiment, which are described more extensively in Harmon (1992). Since class 2 CWD was placed for the log decomposition experiment, we chose CWD of later decay classes that were adjacent to the class 2 logs. Three sets of logs were selected, with each

set containing one CWD in class 2, 3, 4, and 5 and controls. Two additional sets were selected from the surrounding areas to increase the sample size to 5.

To sample CWD leachates, five zero-tension lysimeters were made from 15 cm long, 10.5 cm diameter PVC pipes cut in half lengthwise and installed directly under CWD of decay class 2, 3, and 4. Because class 5 logs are unstructured and settled into the soil, it was not possible to sample leachates with a zero-tension lysimeter. Five 10 cm square plastic sandwich boxes were used to sample solution from under the forest floor as a control (hereafter referred to as forest floor leachates).

Soil solution was sampled with 5 Prenart tension lysimeters at two depths (30 cm and 50 cm) beneath CWD of class 2, 3, and 4 and in control soils with no CWD (hereafter called control solutions). The first two sample of leachate and soil solution were discarded.

To determine if CWD decomposition affects mineral soil chemistry, bulk soils were sampled from under class 4 and class 5 logs where we expected the CWD imprint to be the strongest. Five of the class 4 CWD were the same as those used for the leachate study, although soils were not sampled near the lysimeter. An additional five class 4 CWD were selected from the site area to increase the sample size to 10. Class 5 CWD and control soils were chosen in the adjacent area of the class 4 CWD. Control mineral soils were sampled under the forest floor where no CWD could be detected throughout

the soil profile. Soils were sampled from 4 depths; 0-5 cm, 5-15 cm, 15-30 cm, and 30-60 cm, and homogenized thoroughly within each depth.

Lab Analysis:

Solutions were analyzed for pH with an Orion Scientific electrode Lysimeter prior to filtration. Solutions were filtered using an ashed GF/F filter and stored at 0° C until further analysis. Solutions were analyzed for DOC with a Shimadzu TOC analyzer. To determine the chemical behavior of the leachate, total DOC was further fractionated into hydrophilic and hydrophobic compounds using XAD-8 resin (Qualls and Haines 1991). Polyphenols were analyzed using the Folin-Denis method on a Shimadzu UV1201 spectrophotometer (Harmon and Lajtha 1999).

Field-moist samples were sieved through a 4 mm mesh and processed immediately for pH, percent soil moisture, exchangeable cations in barium chloride (Hendershot et al. 1993), and exchangeable acidity in potassium chloride (Robertson et al. 1999). Sub-samples of 0-5 cm soils were analyzed for bulk density.

Sub-samples were air-dried, ground to a 35 mesh and extracted with sodium hydroxide for amorphous, crystalline Fe and Al bound P (Spears et al. 2001, Lajtha 1999). This extract can contain both organic (P_o) and inorganically bound phosphorus (P_i). The organic fraction was determined by subtracting NaOH P_i from total NaOH P after digestion. Free Al and Fe were extracted with dithionite-citrate (Ross & Wang 1993).

Additional sub-samples were powdered in a Spex mixer-mill to pass through 100 mesh for analysis of total C. Amorphous Al and Fe were extracted with sodium pyrophosphate and organically bound Fe and Al were extracted with acid ammonium oxalate (Ross & Wang 1993). All Al and Fe extracts were analyzed on a Liberty 150 Inductively coupled plasma emission spectrometer.

Statistics:

All statistics were performed in SAS using Analysis of Variance (ANOVA) (SAS Institute, Cary NC, 2000). A pair-wise comparison of the means (Tukey) was performed only if the F-statistic was significant at alpha=0.05. Lysimeter data for pH, polyphenol concentration, and the percentage of hydrophobic and hydrophilic fractions were analyzed by repeated measures ANOVA to determine if there was an effect of date on concentration. If there was no significant effect of sampling date on the concentration, concentrations were averaged over the sampling period and the averages and standard errors were reported.

Leachate Results:

All results are summarized in table 2.1. Dissolved organic carbon concentrations in CWD leachate from zero tension lysimeters were much higher than those from the forest floor (figure 1a and 1b). DOC concentrations from zero tension lysimeter leachate under class 3 CWD were generally highest while forest floor DOC concentrations

remained lowest of all treatment. Class 2 CWD leachate showed considerable variation in DOC concentration with no apparent trends in relation to the other decay classes. table 2. 1 figure 1

Class 3 leachate contained significantly more hydrophobic compounds and correspondingly, less hydrophilic compounds than leachates from class 2 or class 4 CWD ($p \leq 0.015$, figure 2). However, the percent of total DOC that was hydrophobic in class 3 leachate was not different than forest floor leachate. Solutions collected at 30 and 50 cm were too low in DOC concentration to fractionate.

figure 2

Zero-tension leachate from Class 3 CWD had more polyphenols than class 2 CWD and class 4 CWD, and forest floor leachates ($p<0.0001$). Class 2 CWD and class 4 CWD leachates contained more polyphenols than forest floor leachates, however, the concentrations were not significantly different (figure 3). Concentrations of polyphenols increased from class 2 to class 3 and then decreased in decay class 4. At 30 cm, polyphenols in control solutions (3.29 ppm) were lower than in leachates under CWD. However there was no difference among concentrations collected under various decay classes. The mean of class 2 was 5.52 (se=1.36), the mean of class 3 was 5.97 (se=1.04), and the mean of class 4 was 5.03 (se=1.47). At 50 cm, there was no difference between control solutions (3.81, se=0.43) and leachates from under class 4 CWD (3.68, se=0.42).

figure 3

CWD leachate in zero tension lysimeters was more acidic than leachate from under the forest floor ($p>0.001$, figure 4). However, the pH of leachates from under CWD was not different among decay classes.

figure 4

Soil Results:

Mineral soils at 0-5 cm were more acidic under class 4 and 5 CWD than mineral soils under the forest floor ($p\leq0.005$, table 2.2). Soils under class 4 and 5 were not different from each other. Soils under class 5 CWD were not different than control soils. Soil pH did not differ among decay classes or controls at deeper soil depths. Soil pH increased with depth under all treatments.

table 2

There was no difference in bulk density among soils under CWD and control soils at 0-5 cm. Soil bulk density at 0-5 cm under CWD averaged 0.76 g/cm^3 (se= 0.03) and soil bulk density under the forest floor averaged 0.70 g/cm^3 (se = 0.02).

Exchangeable acidity was higher in 0-5 cm soils under class 4 CWD and under class 5 CWD than control soils (table 2.3, $p\leq0.002$). Trends suggest that exchangeable acidity remained relatively high in soils under class 5 CWD to a depth of 60 cm. However, the large variability in soils under class 5 masked any significant differences (see standard errors in table 2.3).

table 3

Exchangeable Al was higher in soils under class 4 CWD than in control soils ($p<0.008$). Trends also suggest that exchangeable Al was higher in soils under class 4 and 5 CWD than control soils to a depth of 60 cm (figure 5) although differences were no statistically significant. There was a strong positive correlation between exchangeable Al and exchangeable acidity for 0-5 cm samples ($p=0.0001$, adjusted $R^2=0.68$).

figure 5

Free, organically bound, and amorphous Al and Fe concentrations were not significantly different between soils under CWD or soils under the forest floor. B-horizon soils under class 4 CWD did have a higher ratio of oxalate to dithionite extractable Al (figure 6). This trend was not evident for Fe extractions; the ratios of oxalate to dithionite extractable Fe decreased over depth (figure 7). The ratio of organically bound Al to total C also decreased with depth for all treatments (figure 8).

figure 6,7,8

Percent base saturation was much lower in 0-5 cm soils under class 4 and class 5 CWD than in control soils ($p<0.001$, table 2.4). In general, base saturation increased with depth up to 30 cm for all soils, although control soils maintained a higher percent base saturation than soils under CWD throughout the soils profile. At 5-15 and 15-30 cm, soils under class 4 CWD had a higher percent base saturation than control soils or

soils under class 5. However, these differences were not statistically significant and a large amount of variation was present.

table 4

The inorganic NaOH P fraction (NaOH P_i) was not significantly different among soils under class 4 and 5 CWD or from control soils to a depth of 0-30 cm (figure 9). At 30-60 cm, however, soils under class 4 CWD had more NaOH P_i than soils from under class 5 CWD (p-value$\leq$0.03). The organic NaOH P fraction (NaOH P_o) was higher in 0-5 cm soils under the forest floor and under class 4 CWD than soils under class 5 CWD. However, there was no difference in this fraction between soils under class 4 CWD or control soils. No differences in phosphorus chemistry were seen at depths greater than 5 cm (figure 10).

figure 9, 10

Discussion:

Although we had hypothesized that the most profound spatial effects of CWD on soils chemistry would be most evident under class 5 CWD after the log had thoroughly decomposed, most of the chemical analyses in this study suggested that CWD had a larger effect on soil when the heartwood was rotting (class 4) than when the heartwood was red mush (class 5). Soils under class 5 CWD were rarely different than control soils, which have not been under the influence of CWD decomposition. This suggests that the earlier decay classes may have the largest effect on soil chemistry

when the sapwood and inner bark are actively decaying and that CWD does not have a profound long-term effect on soil chemistry at this site, but rather a more subtle effect that changes over time.

We had hypothesized that the DOC in CWD leachates would be much more acidic, and contain more hydrophobic and polyphenol compounds than DOC in the forest floor leachates. DOC concentrations in CWD leachate were generally much higher than those from the forest floor. However, we found small differences in the proportions of hydrophobic or hydrophilic compounds in leachates from various decay classes, or among CWD leachates and forest floor leachates. Our ratio of hydrophobic to hydrophilic compounds was greater than 2:1 across all treatments and was not different among leachates from the forest floor and leachates from CWD. Yavitt and Fahey (1985) found a 1.5:1 ratio of hydrophobic to hydrophilic DOC in class 4 leachates; a ratio, which was did not differ from forest floor leachates in their study.

Because certain chemical components of DOM, such as polyphenols, can be both hydrophobic and hydrophilic, our chemical fractionation method may have been inadequate to determine quality differences between CWD and forest floor leachates. Polyphenols concentrations in CWD and forest floor leachates were more accurate in describing the differences in DOM quality. Polyphenols are components of the lignocellulose in CWD and are also the by-products of decomposition. Thus we suspected that there would be a much larger concentration of polyphenols in solutions

under CWD in the middle stages of decay. Indeed, polyphenol concentrations were highest in class 3 CWD leachates whose sapwood is most actively decaying (figure 3). Although we did not measure class 5 leachates, we suspect that polyphenol concentrations in class 5 leachates would be lower than class 4. The remaining substrate in class 5 is thoroughly degraded, lignin rich, and mostly non-soluble and therefore may not leach as readily from the CWD (Prescott et al. 1990).

Concentrations of soluble polyphenols were generally similar to Yavitt and Fahey (1985). There was a particularly large fall flush in 1999 when polyphenol concentrations exceeded 25 mg/L, and this trend was also evident for total DOC concentrations leaching from class 3 logs (figure 1). However, this spike did not return in the fall of 2000, which may suggest an initial disturbance effect of zero-tension lysimeter installation. Additionally, the 2000-2001 water year was a severe drought for the PNW, which might have prevented strong leaching of the class 3 CWD.

We had hypothesized that the acidic leachate would have a strong impact on the pH of the soil and indeed, soil pH was much lower under CWD of all decay classes. However, it was only lower in the surface soils and not in deeper soils, as we had expected. Several sources of acidity could be found in soils under CWD, including hydrolysis of Al^{3+}, adsorption of OH^- to complexes, organic matter, and H^+ ions. The release of Al^{3+} from the soil matrix, as it is broken down during chemical weathering, creates H+ ions as the Al is rapidly hydrolyzed. Consistent with this, exchangeable Al

was found to be much higher in soils under CWD than in soils under the forest floor, particularly in surface soils (figure 5). However, exchangeable Al did not account for the total exchangeable acidity, and the remainder must be from the acidic groups on soil organic matter.

Soils were either high in exchangeable Al reflecting an increased amount of Al dissolution from soil matrices, or were high in exchangeable Ca reflecting that little leaching had occurred. Correspondingly, we found that shallow soils under CWD consistently had a lower percent base saturation than soils under the forest floor (table 3). Soils under class 4 CWD were particularly low in base saturation in the surface soils. A correspondingly higher base saturation at 15-30 cm under class 4 CWD may perhaps reflect a redistribution of cations to lower soil horizons. Al and acidic DOC from the CWD leachates probably leach the base cations to lower soil horizons as partners to maintain electrical neutrality. Thus we see that percent base saturation tends to increase with depth as the higher soil horizons are being actively leached of base cations.

Kaiser et al. (2000) suggest that hydrophobic DOM would replace Al and Fe organic compounds on soil exchange sites, making Al and Fe leach in solution to the lower horizons. Therefore, hydrophobic DOM may be responsible for the transport of Al and Fe compounds to deeper soils forming the Bs horizons of podzol soils. Although we found more than twice as many hydrophobic as hydrophilic compounds

in CWD leachates, there was only a suggestive increase in the Fe and Al compounds in lower soil horizons.

The larger pool of exchangeable Al that we observed under CWD would suggest that the soil P should be tied up as unavailable forms of Al hydroxide P. However, there was little difference in P chemistry in soils under logs compared to control soils. As mentioned previously, CWD leachate may also be rich in organic acids, which have been shown to solubilize P from Fe and Al hydroxides (Fox et al. 1990). Therefore, although there was more exchangeable Al in soils under CWD, there may also have been more organic acids to chelate metals and reduce P-fixation by soils under CWD.

Conclusion

Soils under the most highly decayed CWD, or in deeper soils did not reflect a CWD imprint. Although we had hypothesized that CWD may have a long-term influence on soil pedogenesis, chemical weathering of the soil beneath CWD was limited. Therefore, this study suggests that the spatial effects of CWD on soil processes were limited to surface soils beneath CWD. Surface mineral soils under CWD were lower in pH, had more exchangeable acidity and more exchangeable aluminum than adjacent soils. Acidic leachates were probably responsible for the increase in acidity in the mineral soil below CWD. This would act to release more Al from soil minerals, which would act as a positive feedback increasing soil acidity. Previous studies have suggested that there were more organic acids, namely citric acid, under CWD of class 4

than in control soils (Krzyszowska-Waitkus, personal com.). However, no study has directly investigated the effect of LMWOA on soil chemistry beneath CWD and therefore, further study is necessary to elucidate the mechanisms of soil acidification under CWD.

Although we had hypothesized that wood would spatially imprint on soils for the long-term (thousands of years), the imprint was temporary (hundreds of years) and was not evident under the most highly decayed CWD. Therefore, CWD may not create the long-term spatial mosaic that we had hypothesized, but rather a surficial spatial pattern that changes on a more moderate time span than previously believed. However, the scope of this study is limited to Douglas-fir CWD decaying at the H.J. Andrews experimental sites. More extensive research in different ecosystems is needed to determine whether the patterns that we observed are generalizable.

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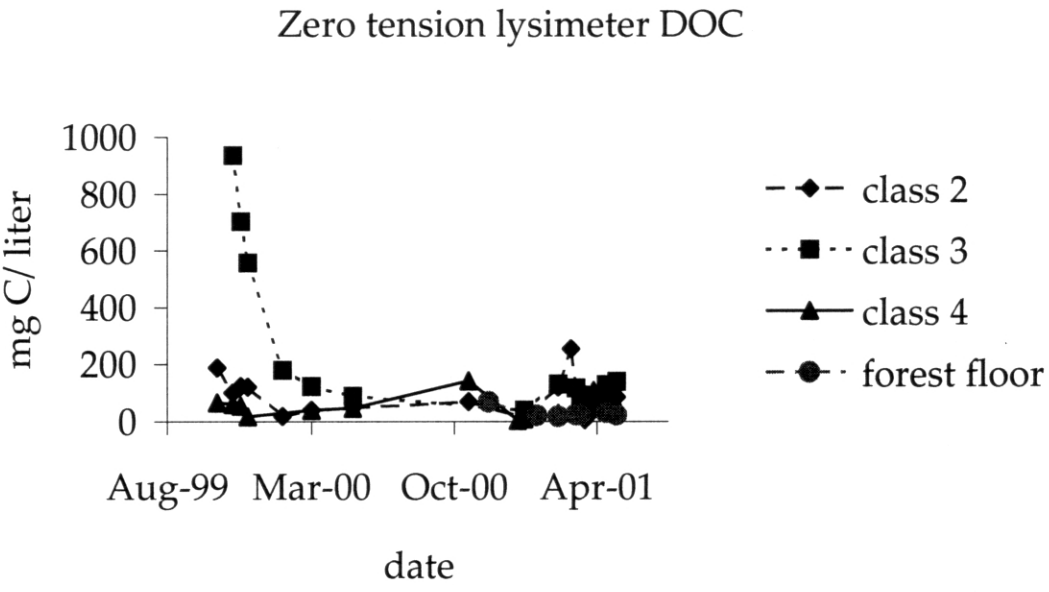

Figure 2.1a. Mean DOC concentrations collected from zero tension lysimeter under CWD of decay class 2,3, and 4 and from under the forest floor. Forest floor samples were not collected until the fall of 2000. Area under the bar is shown in close up in figure 2.1b.

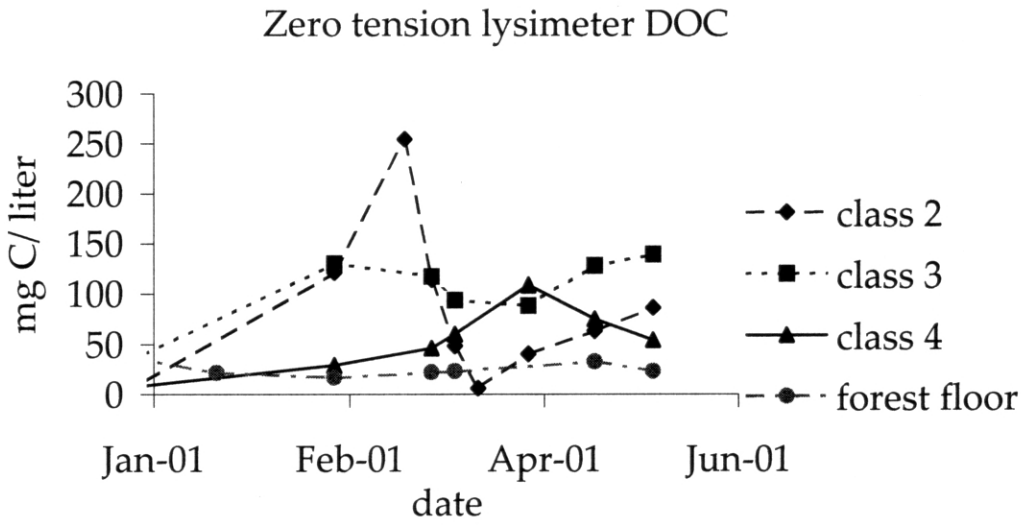

Figure 1b is a close-up of figure 1a

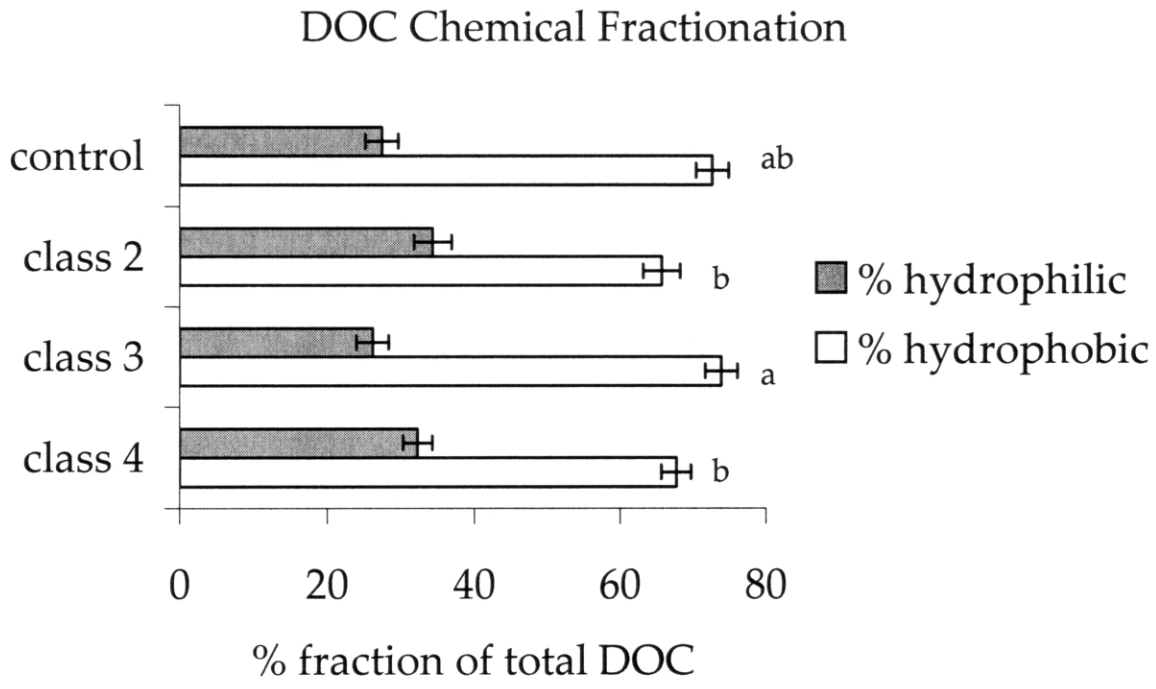

Figure 2.2. Mean percent hydrophobic DOC of total DOC across all collection dates. Samples were collected with zero-tension lysimeters from under CWD of decay class 2,3, and 4 and from under control soils. Bars represent the standard errors. Decay class 3 leachates had slightly higher percent hydrophobic compounds than class 2 and class 4 leachates ($p=0.015$), but were not different from forest floor leachate.

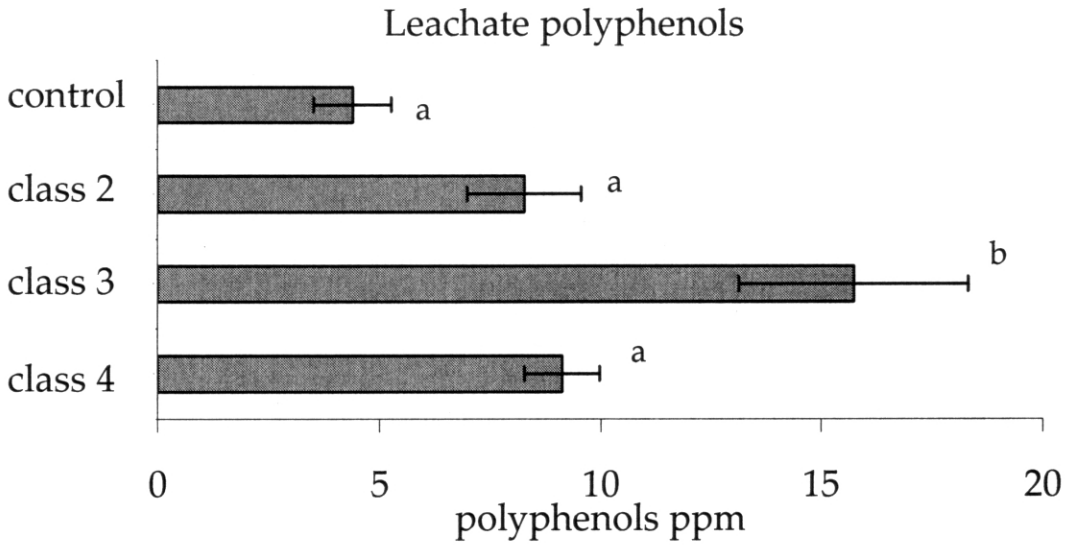

Figure 2.3. Mean polyphenol concentrations across all collection dates. Samples were collected with zero-tension lysimeter under CWD class 2, 3, and 4 and control leachates from under the forest floor. Bars represent the standard errors and letters represent statistical difference using Tukey's multiple comparison test. Treatments with different letters are significantly different at alpha=0.05.

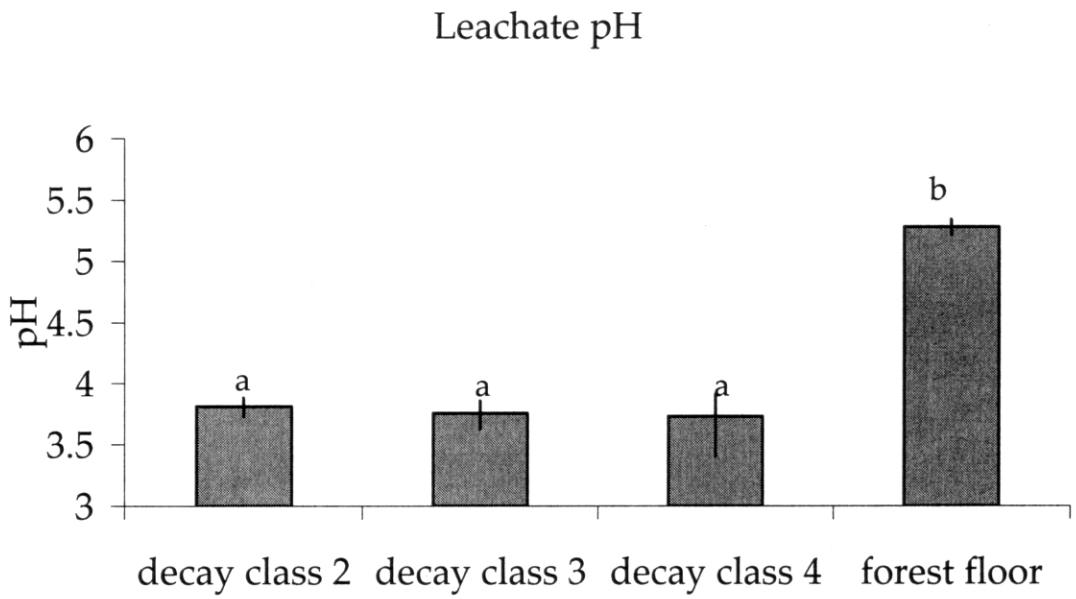

Figure 2.4. Mean pH of leachates across all dates from under CWD of decay class 2, 3, and 4, and from under the forest floor. Bars represent the standard errors and letters represent statistical difference using Tukey's multiple comparison test. Treatments with different letters are significantly different. Forest floor leachates are significantly higher than CWD leachates, $p<0.001$.

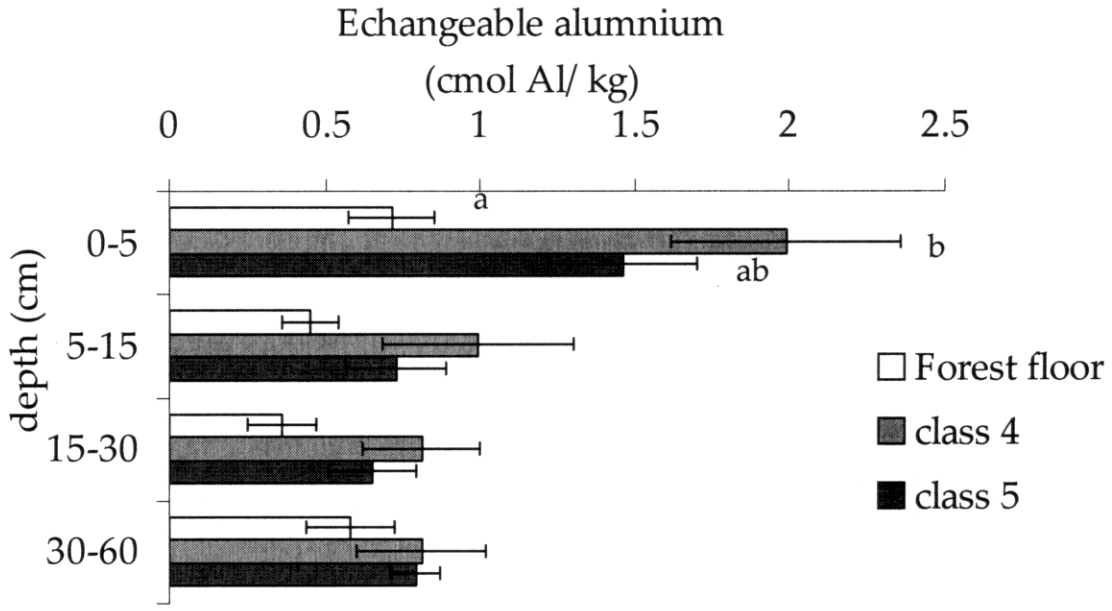

Figure 2.5. Mean exchangeable aluminum concentrations and standard errors from 4 soil depths below CWD of decay class 4 and 5 and control soils. Letters indicate that there is a significant difference between treatments using Tukey's multiple comparison test.

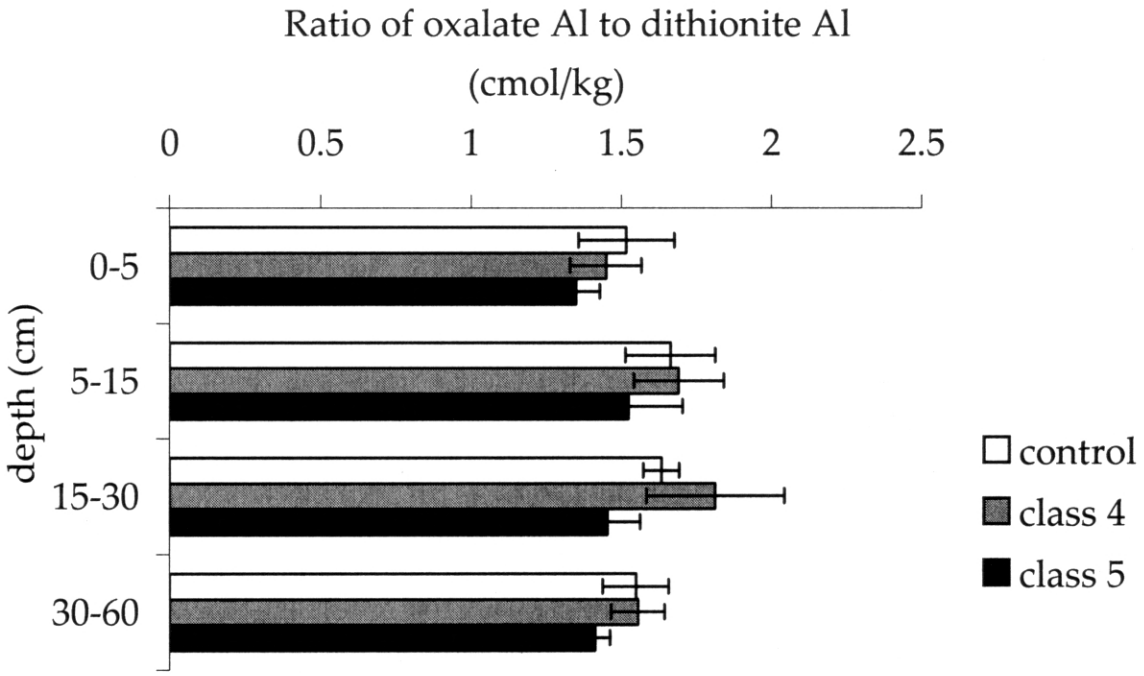

Figure 2.6. Ratio of oxalate extractable aluminum to dithionite extractable aluminum from soils under class 4 and 5 CWD and from under the forest floor. Bars represent the standard error.

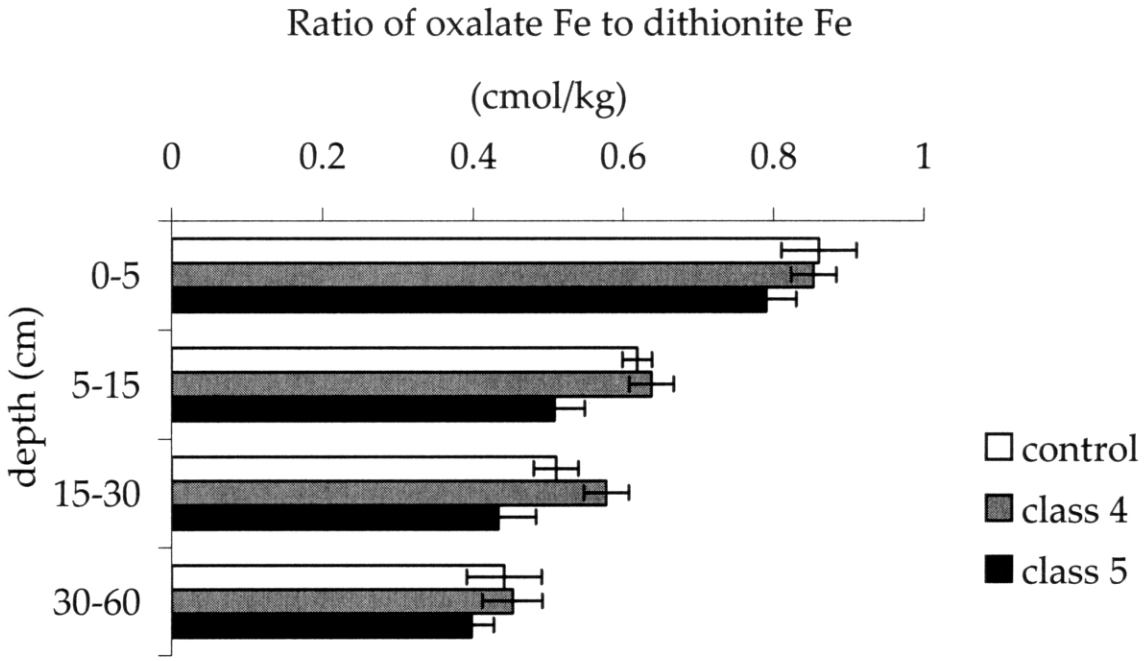

Figure 2.7. Ratio of oxalate extractable iron to dithionite extractable iron in soils under class 4 and 5 CWD and from under the forest floor by depth. Bars represent the standard error.

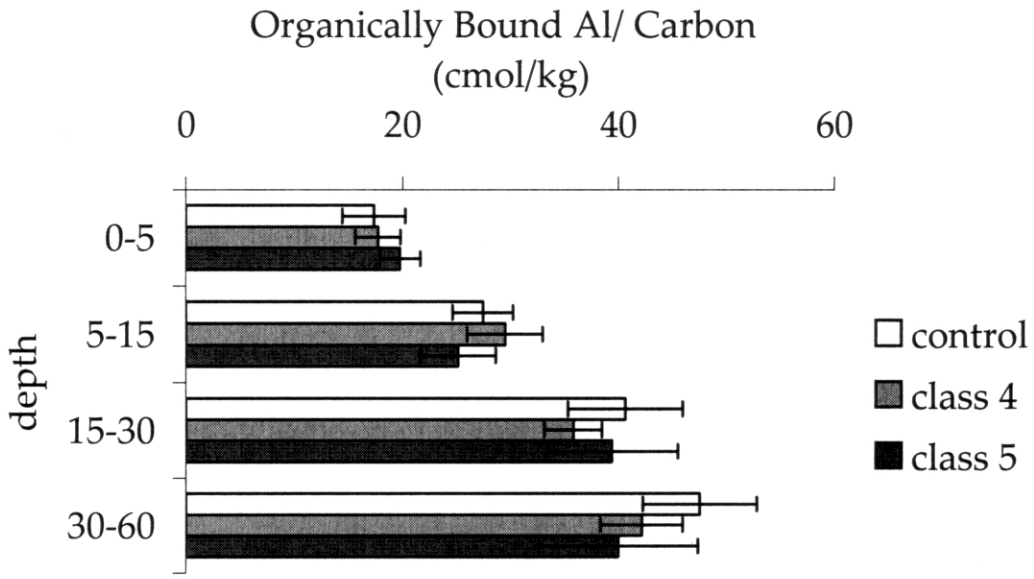

Figure 2.8. Ratio of organic aluminum to soil carbon from soils under class 4 and 5 CWD and from under the forest floor by depth. Bars represent one standard error.

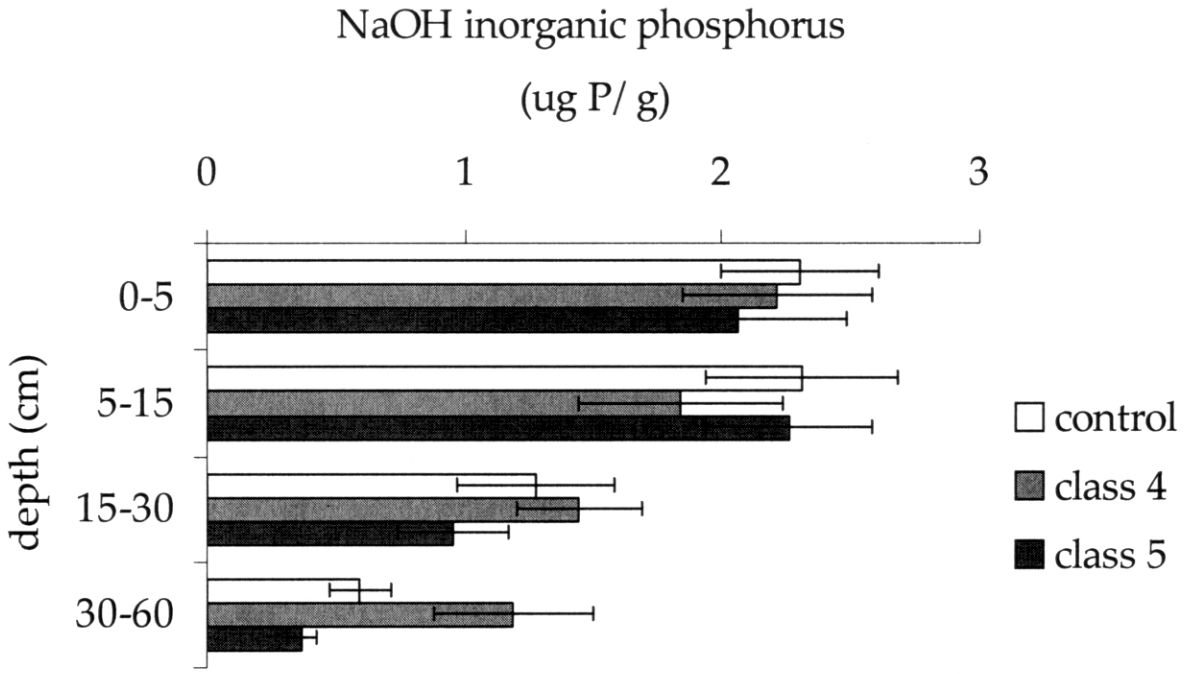

Figure 2.9. Mean inorganic phosphorus extracted with NaOH and standard errors for soils under decay class 4 and 5 and for soils under forest floor.

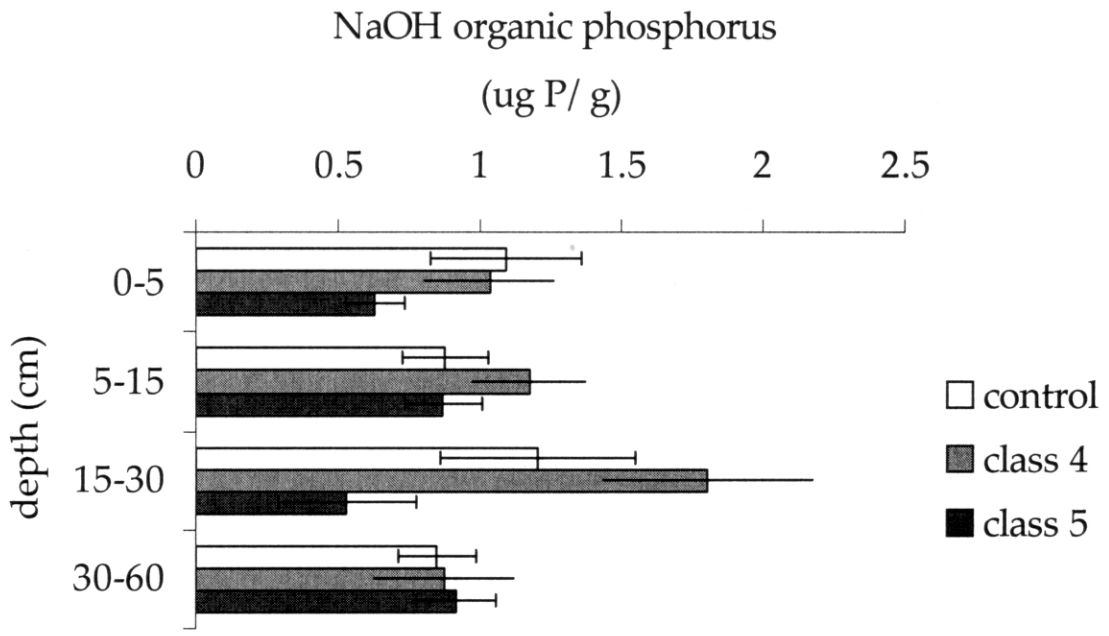

Figure 2.10. Mean organic phosphorus extracted with NaOH for soils under decay class 4 and 5 and control soils. Standard errors are represented by bars.

Chemical Analysis:	**Result:**
DOC in leachates	higher under CWD
Polyphenols in leachates	higher at under class 3 CWD
Leachate pH	lower under CWD
Chemical fractionation	higher % hydrophobic under class 3, than class 2 or 4
Soil pH	lower under CWD at 0-5 cm
Exchangeable aluminum	higher under class 4 CWD at 0-5 cm
Exchangeable acidity	higher under CWD at 0-5 cm
Percent base saturation	lower under CWD at 0-5 cm
Aluminum and iron chemistry	no difference
NaOH phosphorus	no difference

Table 2.1. Summary of the results found in this study. 'No difference' means that there was no significant difference found among classes 4 or 5 CWD or control soils, alpha=0.05.

depth	control	class 4 CWD	class 5 CWD
0-5 cm	5.5 (0.09)[a]	5.01 (0.14)[b]	5.19 (0.08)[ab]
5-15 cm	5.79 (0.09)	5.52 (0.10)	5.64 (0.08)
15-30 cm	5.98 (0.06)	5.87 (0.11)	5.91 (0.07)
30-60 cm	5.93 (0.11)	5.97 (0.09)	6.07 (0.06)

Table 2.2. Mean pH values and standard errors in parentheses for soils under decay class 4 and 5 CWD and control soils at 4 depths. Letters indicate values are statistically significant using tukey's comparison of means. Control soils and soils under class 4 CWD are significantly different at 0-5 cm, p=0.004.

depth	control	class 4 CWD	class 5 CWD
0-5 cm	3.93 (0.84)[a]	11.86 (2.11)[b]	10.01 (1.52)[b]
5-15 cm	3.01 (1.02)	5.07 (1.61)	11.22 (6.03)
15-30 cm	3.76 (0.97)	5.78 (1.81)	6.98 (1.70)
30-60 cm	8.09 (1.37)	6.58 (0.74)	8.74 (1.42)

Table 2.3. Mean exchangeable acidity in cmol kg^{-1} and standard errors in parentheses for soils under decay class 4 and 5 CWD and control soils at 4 depths. Letters indicate values are statistically significant using tukey's comparison of means. Control soils and soils under class 4 and 5 CWD are significantly different at 0-5 cm, p=0.05.

depth	control	class 4 CWD	class 5 CWD
0-5 cm	56.20 (6.96)a	14.87 (5.19)b	22.52 (6.94)b
5-15 cm	63.20 (8.01)	50.61 (10.20)	34.35 (9.92)
15-30 cm	57.93 (12.35)	60.28 (8.66)	44.06 (5.88)
30-60 cm	47.09 (7.00)	37.82 (5.40)	43.23 (5.54)

Table 2.4. Mean percent base saturation values and standard errors in parenthesis for soils under decay class 4 and 5 CWD and control soils at 4 depths.
Control soils are significantly different than soils under CWD of class 4 and 5 at 0-5 cm, $p<0.001$.

Life under Logs; the influence of coarse woody debris on soil biology and nutrient cycling

J.D.H. Spears, S.M. Holub, M.E. Harmon and K. Lajtha

Abstract

Although many studies have quantified the nutrient stores and dynamics in decomposing logs, few studies have investigated the effect of CWD on soil nutrient pools and microbial communities. We hypothesized that CWD may contribute carbon and nutrients to the underlying soil as dissolved organic matter (DOM) over long-term decomposition, and that these inputs may have profound influences on the soil microbial community. We investigated the effect of CWD on soil biology and nutrient cycling by examining nutrient inputs from CWD leachates over a log decay chronosequence. We measured CWD leachates for carbon, nitrogen, and phosphorus. We measured soil from under CWD for moisture, temperature, carbon (C), nitrogen (N) , gross N mineralizations, labile phosphorus (P), and water-soluble organic carbon (WSOC). We measured microbial biomass and enzyme activity as measures of the microbial community. We used Biolog plates to test which substrate the microbial community would degrade. We also investigated the effect of CWD species on leachate chemistry by comparing 4 species of CWD in the early stage of decay. We found no difference in C, N, and P, microbial biomass and enzyme activity between soils under CWD under the forest floor. We found no differences in dissolved inorganic and

organic nitrogen, and dissolved inorganic phosphorus in CWD leachate than forest floor leachate. However, results indicate that the mineral soil under CWD was warmer, but not wetter than mineral soil under the forest floor. Water-soluble organic carbon (WSOC) was higher under CWD than under the forest floor at 5-15 cm ($p<0.02$). Gross N mineralization was significantly faster in mineral soils without CWD. Biolog © SFP2 plates showed that microbial communities varied seasonally but were not different between soils under CWD and control soils. Differences between CWD species were limited as well. Although we had hypothesized that the mineral soil under CWD of later decay classes would be different than the mineral soil under the forest floor, we found few differences suggesting that the effect of CWD on mineral soil biology and nutrient cycling is small. The most profound differences between CWD and forest floor leachates were evident only in the early stages of CWD decay.

Introduction

Coarse Woody Debris (CWD) has been considered by some researchers as stockpiles of carbon on the forest floor that may increase soil organic matter (SOM) over the long term (Sollins et al. 1987, Harmon et al. 1986), while others have considered grinding up CWD and mixing the residue into the soil to abate the deleterious affects of nitrogen (N) saturation (Waring and Running 1998, Waring and Pitman 1985). Indeed, CWD could have very large effects on forest soils. However the mechanisms by which

CWD affects the underlying soil remain relatively unknown and few studies have addressed the processes by which CWD can affect the soil.

Previous studies have investigated nutrient stores and dynamics in logs of various age classes that may contribute to soil organic matter (Holub et al. 2001, Krankina et al. 1999, Busse 1994, Means et al. 1992, Arthur and Fahey 1989, Sollins et al. 1987, Graham and Cromack 1982, Lambert et al. 1980, Grier 1978). . Additionally pathways of soil nutrient enrichment may include: 1) nutrient losses from CWD by fungal sporocarps (Harmon et al. 1994) 2) nitrogen fixation in CWD may contribute large amounts of N to the soil ecosystem through leaching (Roskoski 1980, Hicks 2000) and 3) leaching losses of carbon (C) and cations in the interstitial water of CWD and soil (Yavitt and Fahey 1985).

Although there have been many studies on the role of CWD in the ecosystem, few studies, until recently however, have directly tested the effect of CWD on soils (Krzyszowska-Waitkus et al. in review, Kayahara et al. 1996, and Busse 1994). The results of these studies have been conflicting and therefore, the effect of CWD on soil biology and nutrient cycling deserves more attention.

CWD may affect soil ecosystems through its physical presence on the forest floor. CWD has long been known to be a source of water for plant roots because it acts as a sponge, retaining water late into the dry summer seasons of the Pacific Northwest (PNW). It may also moderate the temperature of the underlying soil. This may prevent

freeze-thaw cycles, which disrupt biological activity and cause nutrient leaching losses. Because temperature and moisture are key driving variables of biological activity, CWD may increase activity in the summer by providing moisture, while also increasing activity in fall and spring by moderating temperature extremes. These influences may have cascading effects on soil microbial community composition and nutrient cycling.

CWD may contribute C and nutrients as dissolved organic matter (DOM) inputs to the underlying soil over long-term decomposition. Yavitt and Fahey (1985) hypothesized that because decaying-wood is only partially oxidized, soluble organic matter could be transferred to the mineral soil. This may contribute to the formation of soil organic matter. Indeed Krzyszowska-Waitkus et al. (in review) found that mineral soils under CWD of later decay stages contained significantly more C than control soils. The difference between mineral soil C under CWD and control mineral soils in this study, however, was small and limited to the surface horizons.

Additionally, Yavitt and Fahey (1985) found only a small increase in mineral soil C under CWD although reporting that CWD may contributes approximately 12 g C m^{-2} to the soil per year in a Montane lodgepole pine stands. Fine litter inputs were estimated at 11g C m^{-2} y^{-1} (Yavitt and Fahey 1985). In this case, CWD contributed about the same amount of C as fine litter. Conversely in an eastern Oregon lodgepole pine forest, Busse (1994) reported a decrease in total organic C content in soils beneath CWD. The contrasting results of Busse (1994) versus Yavitt and Fahey (1985) and

Krzyszowska-Waitkus et al. (in review) suggest that the effect of CWD on soil C may be variable across ecosystems. Results from Lodgepole pine ecosystems suggest that C input to soils underlying CWD is only slightly greater compared to soils underlying the forest floor. Lodgepole pine ecosystems are more fire-prone than the old-growth Douglas-fir ecosystems in the PNW. This, in part, allows the Douglas-fir forests to have a much larger biomass than the lodgepole ecosystems. Thus the size of CWD may be larger and CWD may persist longer on the forest floor in PNW ecosystems.

In addition to C, leaching of decomposing CWD may transfer nutrients, such as N, to the underlying soil. Nitrogen fixation in CWD in PNW forests has been estimated at approximately 1 kg ha^{-1} yr^{-1} (Harmon et al. 1986). This is approximately 1/4 of the total annual input of N from all other N sources to these old-growth PNW ecosystems and is available to roots and mycorrhizae that colonize decomposing trees. However, N in CWD may also be leached as dissolved organic nitrogen (DON) to the underlying soil. Yavitt and Fahey (1985) have shown that concentrations of DON leaching from well-decayed CWD were twice as large as concentrations of DON in throughfall and were two to five times larger than concentrations of DON in soil solution (Fahey and Yavitt 1988).

It is well known that N is limiting to plant growth in most terrestrial ecosystems. Therefore root and microbial uptake of N in CWD may exceed leaching losses and may therefore decrease the concentration of N leaching from CWD. More N may leach from

the forest floor where litter components are higher in N than CWD. In British Columbia, lower amounts of mineralizable N were found in B-horizon soils beneath well-decayed CWD than in control soils (Kayahara et al. 1996). In eastern Oregon, Busse (1994) also found lower concentrations of inorganic N under CWD. Thus, nitrogen concentrations in mineral soils underlying CWD may be lower than in mineral soils underlying the forest floor.

In addition to N, CWD leachates may affect soil P pools. Using a log chronosequence, a net efflux of P from CWD has been shown as CWD decomposes (Grier 1978, Lambert et al. 1980, Sollins et al. 1987). Because CWD leachates has a high percentage of hydrophobic compounds (Yavitt and Fahey 1985), and hydrophobic DOC compounds have a stronger affinity for soil exchange sites than phosphorus (Kaiser and Zech 1996), hydrophobic DOC may displace phosphates into the soil solution flushing P from the soil exchange sites and leaching labile P from the system. As a result, labile soil P pools may be lower in mineral soils underlying CWD than in soils underlying the forest floor but P may be more available in soil solution for plant and microbial uptake.

These differential inputs of nutrients (lower N and P, and higher C) to soils under CWD may change soil community structure by favoring for those organisms that can degrade recalcitrant, N-poor DOM. Conversely, the DOM may provide more labile carbon (DOC) and nitrogen (DON) sources as DOM to the microbial community, which may prime the degradation of the more recalcitrant ligninitic soil organic matter and

change the composition of the microbial community. Microbial biomass can be higher under CWD than under the forest floor (Busse 1994), suggesting that the inputs of DOM increased microbial biomass under CWD compared to the microbial biomass in the mineral soil beneath the forest floor.

In this paper, our objectives were to investigate the effect of CWD on soil biology and nutrient cycling by examining: 1) the nutrient inputs in CWD leachates using a chronosequence of decayed CWD and 2) nutrient pools, soil microbial biomass and microbial community structure in soils beneath CWD. We hypothesized that 1) DOC concentrations of CWD leachate would be largest in the later stages of decay and 2) that soils underlying well-decomposed CWD would have more C and lower N and P than soils under the forest floor. Changes in nutrient pools and N fluxes that occur as CWD ages would be likely to change the microbial community structure by favoring organisms that are able to degrade more recalcitrant, lignin rich substances.

In addition to studying the effects of time on the quality of one species of CWD leachate, we also investigated potential differences among four species of CWD. Major constituents of wood chemistry are similar among coniferous species, but secondary compounds may differentially influence the decomposition of CWD and thus influence the quality of CWD leachates. Species that are less resistant to decay will show a faster release of C and nutrients, whereas species that are more resistant will show a slower release.

Methods and Materials

Site description:

This study was conducted in one site at the H.J. Andrews Experimental Research Forest in the Willamette National Forest in the in the Western Cascade Province of the Oregon Cascade Range (44 13' 53" N, 122 13' 40" W). The site is classified as a *Tsuga heterophylla/Rhododendron macrophyllum/ Berberis nervosa* habitat (Franklin and Dyrness 1988). Dominant tree species include large *Pseudotsuga menziesii* (Mirb.) Franco (Douglas-fir) with smaller *Tsuga heterophylla* (Raf.) Sarg. (western hemlock), and *Thuja plicata* ex D. Don (western red cedar) in the overstory. *Tsuga heterophylla, Taxus brevifolia* Nutt. (pacific yew), *Rhododendron macrophyllum* G. Don, and *Acer cercinatum* Pursh (vine maple) make up the majority of the understory vegetation. The forest floor is covered with a layer of mosses including *Eurhynchium oreganum* (Sull.) Jaeg. (Oregon beaked moss) and *Hylocomium splendens* (Hedw.) B.S.G. (step moss).

The climate is Mediterranean. Precipitation at the site averages 350 cm per year and the average temperature is 10° C. Soils have been classified as Inceptisols; Humic Dystrodept. More recent chemical analysis, however, suggests that these soils may be classified as Andisols (Keys to Soil Taxonomy, 1998). Evidence for this includes: bulk density averages 0.7 g/cm^3, percent soil carbon is > 2% in surface samples, and the percent of oxalate extracted-Al plus ½ oxalate extracted Fe totals more than 2 (Spears and Lajtha, in review).

Experimental Design and Field Methods:

CWD was classified with the CWD decay classes described by Triska and Cromack (1979). Class 1 is a freshly fallen log with sound wood and intact bark. Class 2 log has an initial rotting of the sapwood. Class 3 logs have sloughing bark and very decayed sapwood. Logs are classified as class 4 after the sapwood has completely rotted, the bark has completely sloughed off, and the heartwood is thoroughly rotted. When the log has no structural integrity, has settled into the soil and is thoroughly rotted red "mush", it is classified as class 5.

Our experimental design included Douglas-fir CWD of four decay classes. We used class 2 CWD from the log decomposition experiment, which are described more extensively in Harmon (1992). Class 3,4, and 5 CWD were naturally occurring in the same site. Since class 2 CWD had been selected and placed at the site for the log decomposition experiment, we chose CWD of later decay classes that were adjacent to the class 2 logs. Three sets of logs and associated controls were selected, with each set containing one CWD in each of class 2, 3, 4, and 5 and controls. We used areas under the forest floor where there was no CWD as controls. Two additional sets were selected from the surrounding areas to increase the sample size to five sets.

Leachate Sampling

Leachates were collected with zero-tension lysimeters and soils solutions were collected with tension lysimeters. Zero tension lysimeters were installed under decay

class 2,3, and 4 logs. Control zero-tension lysimeters were placed beneath the forest floor in areas where there was no CWD evident (hereafter referred to as control leachates). Prenart tension lysimeters were installed under class 2, 3, and 4 at 30 cm and under class 4 at 50 cm and control tension lysimeters were placed in areas with no CWD at 30 and 50 cm (hereafter referred to as control solutions). Lysimeters were collected from fall of 1999 until the spring of 2002 after discarding the first two samples.

Species differences

To determine if log leachate chemistry differs between species, zero-tension lysimeters were installed under four species of CWD and replicated at six Log Decomposition Sites (Harmon 1992). Log species included *Thuja Plicata* (THPL), *Psuedotsuga menziesii* (PSME), *Abies amabilis* (ABAM) and *Tsuga heterophilia* (TSHE).

Soil Sampling

In the spring of 2000, soils were sampled from under 10 PSME logs of decay class 4 and 5, where we expected to find the most pronounced effect of CWD. We sampled under class 4 CWD that previously had lysimeters installed under them. A large distance between the lysimeters and the soil sampling was included to prevent edge effects on the leachates. Five additional samples were chosen from surrounding areas. All class 5 logs and control soils under the forest floor (hereafter referred to as control soils) were chosen to be as close to the class 4 CWD as possible. Control soils were sampled where there was no evidence of CWD.

Soils were sampled to a depth of 60 cm and homogenized over four depths: 0-5, 5-15, 15-30 and 30-60 cm. In the fall of 2000, a second set of soil samples were collected from 0-15 cm under the same CWD and controls to test for seasonal differences in biologic activity under CWD. These collections were made far enough away from the previously sampled areas to prevent a disturbance or edge effects on lysimeters. In the fall of 2001, soils were re-sampled for gross N mineralizations.

Stow Away Tidbit temperature loggers were placed under the class 3 CWD and control areas for a short period in the spring of 2000 to determine if mineral soil temperatures are affected by CWD. Percent soil moisture was determined gravimetrically in each season.

Chemical Analysis:

Lysimeter solutions were analyzed for DOC with a Shimadzu TOC analyzer, and dissolved inorganic nitrogen (NH_4^+, NO_3^- or collectively, DIN) and total dissolved nitrogen (TDN) after digestion in alkaline persulfate (Cabrera and Beare 1993) with an Orion Scientific AC 100 Autoanalyzer. DON was calculated as the difference between TDN and DIN. Dissolved inorganic phosphorus (DIP) was analyzed on an Orion Scientific AC 100 autoanalyzer by the acetic acid method (Lajtha 1999). DIP in 30 cm and 50 lysimeter leachates were below detection limits.

Soils were sieved through a 4 mm mesh and extracted with K_2SO_4 for microbial biomass C using the method of Voroney et al. (1999). KCl extractable-NH_4^+ and NO_3^- in

soils were measured on a Orion Scientific AC100 autoanalyzer. Phosphatase and Beta-glucosidase enzyme assays were analyzed by the method of Caldwell (1999) as an index of general microbial activity.

Soil microbial community differences were assessed with Biolog ® plates. Biolog ® SFP2 plates contained ninety-six wells with each well containing a different substrate for microbial growth and one control well with water. One gram of field-moist soil was added to 10 mmol PO_4 buffer (pH=6.8), shaken for 20 min., and allowed to settle. One hundred and fifty ul of the buffer solution was pipetted into each well and allowed to incubate for five days. If the extracted microbes could degrade to the substrate in each well, the well would turn turbid. We scored the results as positive when the well became cloudy and negative when there was no change. We realize that we do not specify community type or functional groups with Biolog ® plates, but rather indicate differences in microbial preferences. This may include only those microbes that are able to reproduce during the incubation time.

Soils to be analyzed for total C and N were first powdered with a spex-mixer mill. Total C&N was analyzed on a Carlo Erba NA1500 CHN analyzer at the Stable Isotope Biology Lab at the University of Georgia. To check for differences in C chemistry, water-soluble organic carbon (WSOC) was extracted using 2 parts water for every part of soil. Slurries were filtered through a GF/F filter and analyzed on a Shimadzu TOC analyzer. Labile P was extracted with resin and sodium bicarbonate

using the first two steps of the Hedley fractionation as described in Lajtha (1999) and analyzed on an Orion AC100 Autoanalyzer.

Results of a preliminary net N mineralization study using the buried bag technique suggested no difference in mineralization rates (Spears, unpublished data). There were also no differences between KCl extractable-NH_4^+ and NO_3^- between soils under the logs and soils without logs in the spring and summer of 1999 (Spears, unpublished data). Therefore ^{15}N gross mineralization assesments were made using the isotope-pool-dilution technique (Hart et al. 1994) in the fall of 2001 on 5 soils 0-5 cm under class 4 and 5 CWD, and control soils. To determine potential N mineralization, soils were incubated in the lab at 25°C after a thorough mixing of the soil. We realize this does not represent *in situ* mineralization. Microbial biomass N and C were also analyzed on sub-samples of these samples (Voroney et al. 1999).

Lysimeter solutions from various tree species were composited over two seasons and analyzed for cations, total kjeldahl nitrogen (TKN) and DOC in September of 1986 and February of 1987. Nitrate and ammonium were analyzed from November of 1990 until April of 1992.

Statistics:

Statistical analyses were analyzed using SAS (SAS Institute, Cary NC, 2000). Because most lysimeters did not produce samples at all dates, it was not possible to use repeated measures analysis of variance (ANOVA) to analyze all lysimeter samples for

at all dates. Solution concentrations were graphed over time for visual inspection and if no effect of date on solution concentration was seen, samples were averaged across all sample date. The mean concentrations and standard errors are presented in the results. Lysimeter solutions for the different species of CWD were compared for each composite sample to determine if there was a difference among species using ANOVA. The means were compared with Tukey's multiple comparison test (alpha=0.05).

Chemical analyses of mineral soils were compared at each depth using ANOVA and the means compared using Tukey's test (alpha=0.05). The Nonmetric Multidimensional Scaling (NMS) program using the Sorenson Distance measure in PC-ORD (version 4.19, MjM Software Design 1999) was used to analyze microbial communities as indicated by Biolog® plates for both seasons.

Results

Overall results are summarized in table 3.1. In the spring, mean percent soil moisture soils from 0-5 cm depths was higher in soils under CWD (mean 55.32 %, se=5.09) than in soils under the forest floor (mean 46.22, se=6.62), although it is only suggestively higher. In the fall, there was no difference between soil moisture under CWD (mean 37.26 %, se=2.33) and under the forest floor (mean 39.26 %, se=2.36). Soil temperatures were more moderate under class 3 CWD as compared to soils under the forest floor in the spring (figure 3.2). Soil bulk density at 0-5 cm was not different under logs (mean= 0.76 g cm^{-3}, se= 0.03) than in control areas (mean=0.70 g cm^{-3}, se= 0.02).

table 1, figure 1

Dissolved organic carbon concentrations in zero-tension lysimeters were often highest under class 3 CWD, concentrations however varied over time (figure 3.2 a and b). Peak concentrations for lysimeters under class 2 CWD were 250 ppm, while class 3 CWD had a one-time peak of 935 ppm that decreased with time. Control lysimeter DOC concentrations were lowest of all treatments throughout the year, averaging 20 ppm.

figure 2

Dissolved organic carbon lysimeter solution was often highest in 30 cm soils under class 3 CWD and lowest in control solutions (figure 3.3a and b). Concentrations of DOC in class 2 and 4 lysimeters varied but were intermediate between those from class 3 CWD and controls. At 50 cm, solutions collected under class 4 had higher concentrations of DOC than controls for several dates, but concentrations were variable (figure 3.4). DOC concentrations in solutions collected under class 4 CWD and controls decreased with soil depth from approximately 100 ppm in zero-tension leachates, to approximately 5 ppm in 50 cm solution.

fig. 3 and 4

Dissolved organic nitrogen concentrations from zero-tension lysimeters and in tension lysimeters at 30 and 50 cm showed no clear pattern and variations between treatments was large (table 3.2). Class 2 CWD had the highest concentration of DON in

zero-tension leachate, but at 30 cm, solutions from under class 3 were highest. Ammonium and nitrate in zero-tension leachates were near detection limits at all depths. Again, leachate from class 2 CWD had suggestively higher NH_4^+ and NO_3^- concentrations than did leachates from other decay classes and controls. Ammonium and nitrate concentrations at 30 cm and 50 cm showed no consistent differences with regard to CWD decay classes, although concentrations in control solutions were always higher than those from class 4 at 50 cm. Zero-tension lysimeter DIP was similar among treatments and was extremely low for solutions at depths.

table 2

Total soil C and N did not differ between soils under decay classes 4 and 5, and neither differed from control soils at any depth (table 3.3). Percentages of total C and N decreased with depth across all treatments. Soil C:N ratios were higher under class 4 than under class 5 or in control soils throughout the profile. However soils underlying class 4 were significantly higher than in control soils only at 30-60 cm ($p<0.05$).

table 3

There were no statistically significant differences in WSOC among soils at 0-5 cm, but soils under class 4 CWD had more WSOC than soils under class 5 CWD at 5-15 cm ($p<0.02$, figure 3.5). Soils under class 4 CWD had suggestively more WSOC across all depths than control soils under the forest floor and soils under class 5 CWD. WSOC decreased with depth across all treatments.

figure 5

There were no differences in labile phosphorus concentrations between decay classes, nor among soils underlying CWD or control soils. Although soils under class 4 CWD often had more resin P, differences were not statistically significant because of the large amount of variation within treatments (figure 3.6). Resin P concentrations increased in 5-15 cm control soils but decreased with depth thereafter. Concentrations of resin P in soils under CWD also decreased across all depths. Labile phosphorus extracted with sodium bicarbonate was not different among treatments and decreased with depth for all treatments as well (figure 3.7).

fig. 3.6 and 3.7

Gross $^{15}NH_4^+$ mineralizations were significantly higher in mineral soils under the forest floor than in mineral soils under CWD ($p<0.015$, figure 3.8). Gross $^{15}NH_4$ production was 2 mg kg^{-1} d^{-1} lower for soils under CWD, while gross $^{15}NH_4$ consumption was also 2 mg kg^{-1} d^{-1}higher for soils under the CWD. Neither microbial biomass C nor microbial biomass N differed significantly between soils from under class 5 CWD and control soils (table 3.4).

figure 3.8, table 3.4

Beta-glucosidase, phosphatase activity and microbial biomass C were suggestively higher in 0-5 cm soils under the forest floor than in soils under CWD, however there was too much variation within treatments to detect significant

differences (table 3.5). Soils under class 4 and class 5 CWD had relatively similar amounts of enzyme activity and microbial biomass C. Microbial biomass C decreased with depth.

table 3.5

There were no clear differences in substrate specificity between soils under class 5 CWD and control soils from under the forest floor (figure 3.9). An NMS ordination graph produced a two-dimensional result with a Monte Carlo Test result $p \leq 0.02$ and the proportion of variance explained by the two axes was 84%. There was a clear separation of the soils by season along the two axes, suggesting that much of the variance explained the seasonal rather than the treatments differences. (figure 3.9

Results for different tree species

Dissolved organic carbon concentrations and TKN concentrations increased between fall and winter for all species of class 1 CWD (table 3.6). However there were no significant differences among species for either TKN or DOC. *Abies amabilis* (ABAM) had the highest leachate concentrations for DOC, TKN, and NH_4 of all species in the February of 1987. Calcium concentrations in THPL leachates were higher than other species in September of 1986 ($p<0.005$), but there was no difference in concentrations among species in February of 1987. THPL had higher concentrations of Mg in leachate than PSME ($p<0.02$), but it was not statistically different than ABAM or TSHE in the September of 1986. In February of 1987 there were no differences in Mg concentrations

between species. In September of 1986, concentrations of Mn in THPL leachate were significantly lower than ABAM and TSHE, but not PSME ($p<0.03$). In February of 1987, concentrations of Mn in THPL leachate were lower than TSHE ($p<0.05$) but were similar to concentrations of MN in PSME and ABAM leachates.

table 6

ABAM also showed a decrease in concentration for Al, Fe and Mn between sampling dates while cations leachates from the other three species increased in concentrations between September of 1986 and February of 1987. Leachate for all species showed an increase in Ca concentrations from September of 1986 to February of 1987.

In September of 1986, DOC in class 1 CWD leachate was weakly correlated with Mn concentrations ($p\leq0.001$, $r^2=0.42$) and with K concentrations ($p\leq0.04$, $r^2=0.19$). These correlations were not significant in February of 1987. There were small but significant correlations between DOC and Mg ($p<0.001$, $r^2=0.29$) and Mn ($p<0.045$, $r^2=0.08$) across all dates and among all species.

Ammonium concentrations showed no trends among species and varied enormously by season (table6). Concentrations increased in the fall months and decreased by the late summer into fall. Nitrate showed an initial decrease in concentration over time, followed by no noticeable trends (table 7).

table 7

Discussion

There were suggestive, but not significant, variations in the seasonal concentrations of DOC among decay classes, while DOC from the forest floor remained lower than DOC from CWD throughout the water year. Seasonal trends indicated the flushing of solutes associated with the beginning of the rainy season in the PNW. Dry summers contributed to a build up of solutes that were flushed in the beginning of the fall and therefore, DOC concentrations decreased over the water year. A seasonal flush in DOC from the interstitial zone of class 4 CWD and soil was also found by Yavitt and Fahey (1985). However, their flush began in the spring as the snowpack melted after solutes had accumulated through fall and winter, while the flush in PNW ecosystems begins in the fall after solutes accumulate through the dry summer. Thus, the influence of CWD on soil C varies temporally with ecosystem hydrology.

As expected, zero-tension leachates from CWD were approximately three times higher in DOC concentration than leachates from the forest floor indicating a very large input of C to the underlying soils. Even at 30 cm, solution from class 3 CWD had three times the concentrations of DOC as control soil solutions did. However, the concentration of DOC in solution had decreased by tenfold. Sorption of DOC to the mineral soil as the solution percolates through the profile could account for the decrease in DOC concentration with depth and could increase the amount of C in soil under CWD dramatically. This decrease in concentration of DOC with depth has been found

in many soils and can be accounted for by either C sorption by the mineral soil, or bio-degradation by the microbial pool (Zech and Guggenberger 1996). However, we must also take into account that this decrease may be from differential fluxes of DOC. If there was more water percolating through the soil under the forest floor, a lower DOC concentration would really be just an artifact of dilution. Thus a decrease in concentration may not completely represent what is happening in the soils under CWD.

We would expect that a threefold increase in DOC leaching from CWD would greatly increase mineral soil C, especially after the CWD has been decomposing *in situ* for greater than 300 years. Although we had hypothesized that a large amount of C would be sorbed into the mineral soil under well-decayed CWD, we found that there was no increase in soil C under well-decayed CWD. Krzyszowska-Waitkus et al. (in review) found more C under CWD in the surface soil, but neither their study, nor ours found differences in deeper soils horizons. Similarly Kayahara et al. (1996), found no difference in the C content of B-horizon soils under CWD and in control soils. The possible fates of carbon from CWD include leaching from the system as DOC or respiration as CO_2. Respiration from class 4/5 CWD has been reported at 5.52 kg C ha^{-1} y^{-1} by Rose and Li (1997). We measured the flux of C in class 4 CWD leachate as 7.3 kg C ha^{-1} y^{-1}. Thus leaching losses of C from CWD are similar to those lost from C respiration.

Busse (1994) ultimately concluded that nutrient accumulation and/or release from CWD contributed little to the mineral soil and correspondingly, we found no large increase in mineral soil C beneath well-decayed logs, e.g. class 4 or 5 logs. However, class 3 CWD may have had the most profound effect on soils because it is the most active period of decomposition and had the largest concentrations of DOC in leachate. Therefore, when we focused on soil C beneath well-decayed logs, we may have missed the most active stage of C inputs to soil. Indeed, Busse (1994) found a large increase in microbial biomass C, but a large decrease in total soil C in mineral soils under Class 3 CWD. We hypothesize then, that the most active stage in C cycling in mineral soil under CWD is during the most active stage of decomposition, class 3. During this time, large amounts of labile DOM are leaching from the CWD, which are actively cycling through the microbial biomass, and perhaps even priming the degradation of the more recalcitrant C in soils. By class 4 and 5, the labile, degradable inputs from CWD have decreased and the recalcitrant DOM is replacing any previously degraded soil C. Thus we see no difference in total soil C in mineral soil under later stages of CWD decay.

We had expected that there would be more cellulose rich compounds in mineral soils under the forest floor because the forest floor is richer in more labile C substrates than CWD and that correspondingly microbial enzyme activity would be higher under the forest floor. However, there may have been more cellulose than we expected under CWD because we did see rates of beta-glucosidase activity that was comparable to the

rates under the forest floor. Throughfall inputs, or leaching from the organic layer that covers well-decayed CWD, may provide the more labile C substrates necessary to sustain the enzyme activity.

In well-decayed CWD, throughfall and leaching of the organic layer may act as an additive effect, increasing the amount of labile C to the mineral soil. WSOC was higher under CWD, which suggests that perhaps there is a persistent labile source of C under CWD that can sustain microbial activity at levels equal to the forest floor. Additionally, there were no noticeable differences in community functioning as represented by enzyme activities or the Biolog ® plates, which would suggest that CWD and forest floor DOM inputs to the mineral soil were similar in quality and quantity.

Nitrogen concentrations tend to be highest in the most decayed material while N-fixation is most active when the CWD is moderately decayed, class 2 and 3 (Hicks 2000). After bulking solution chemistry for N analysis, we found that class 2 had the highest rates of N efflux in CWD leachate as DON, while N concentrations in control leachate was consistently lower than any decay class of CWD. Nitrate and ammonium were very low in leachates and thus DON was the major form of N loss from the CWD. This agrees with Yavitt and Fahey (1985) who found DON to account for 80% of N loss from well-decayed CWD. Even at 30 and 50 cm, DIN was only a minor component of N in soil solution. We would expect then, that there would be an increased rate of N

mineralization in mineral soils beneath CWD. Contrary to this hypothesis however, there were faster rates of N cycling in mineral soils under the forest floor than under well-decayed CWD. Production of NH_4^+ was higher in control mineral soils than mineral soils under CWD. The extra-mineralized NH_4^+ was equally consumed and thus net mineralization remained equal among treatments. At the later decay stages, CWD may be contributing DON in extremely recalcitrant forms slowing the mineralization of DON and perhaps gross mineralizations would show a faster rate of cycling under CWD of earlier decay classes (class 2 or 3) when more labile C compounds are available.

As with C, we found suggestively more labile P under class 4 CWD. Soils under class 5 CWD usually contained less available P than control soils. The most plausible mechanism for this increase in earlier decay classes may be the release of organic acids during the active phase of CWD decomposition. Organic acids have been shown by many to release P from Al and Fe hydroxides by ligand exchange (Fox et al. 1990). Large differences in labile P may not be noticeable in decay class 5 because CWD DOM inputs may concurrently bind the available P as unavailable soil organic P. We did not measure organic P in this study.

Species Differences

There were few differences in leachate chemistry collected under class 1 logs from various species and this may be simply because the most active decay has yet to begin. This would explain why DOC was similar for all species in September of 1986.

However as decay proceeds, the trees that are less resistant to decay, such as ABAM, begin to leak larger concentrations of DOC as seen in February of 1987. We had hypothesized that species that are less resistant to decay will show a faster release of C and nutrients, whereas species that are more resistant will show a slower release. Major constituents of wood chemistry are similar among coniferous species, but secondary compounds can differentially influence the decomposition of CWD and thus influence the quality of CWD leachates. However, our results did not support this hypothesis. The most resistant tree species to decay was THPL and indeed, THPL lost less Mn in leachates that other species. However in contrast to our hypothesis, THPL lost more Mg and Ca through leaching than other species. The least resistant to decay tree species was ABAM and so we expected ABAM to shown the largest nutrient and DOC losses. For example, K is a very mobile cation and leaches quite readily from CWD during decomposition (Holub et al. 2001) and therefore, K should have been most prevalent in leachate from ABAM. In September of 1986 K was higher in concentration in ABAM leachate than other species in accordance with our hypothesis, but in February of 1987, concentrations of K were highest in TSHE leachate which is contradictory to our hypothesis. These contradicting results may be due to the short time span in which lysimeters were sampled. Two sampling seasons may not adequately represent the differences in nutrient dynamics between species.

Conclusion

Studies of CWD decomposition, leaching and its effects on soils have suggested that CWD does affect soils moderately, but that this effect is differential depending on the species of CWD, the time at which the CWD is investigated and the ecosystem in which the CWD is located. We used a chronosequence of Douglas-fir logs to substitute for decay time and it was evident from this work that the largest effect of decomposing wood on soils may be during the middle decay classes when the Douglas-fir CWD is most actively decomposing.

Large inputs of DOC via leaching were slightly retained by the mineral soil, but overall CWD contributed little to mineral soil C when compared to control soils. It may be however, that it is difficult to find control soils that have never been affected by CWD or fine woody debris over long-term pedogenesis. We were confident that our control soils had no evidence of CWD to a depth of 60 cm, but this does not discount the possibility that CWD leachate may flow laterally to areas adjacent to the CWD, or that fungi may translocate C and nutrients to adjacent control areas. Additionally, we have no idea what the ultimate fate of CWD is and thus control areas may not really be controls after all.

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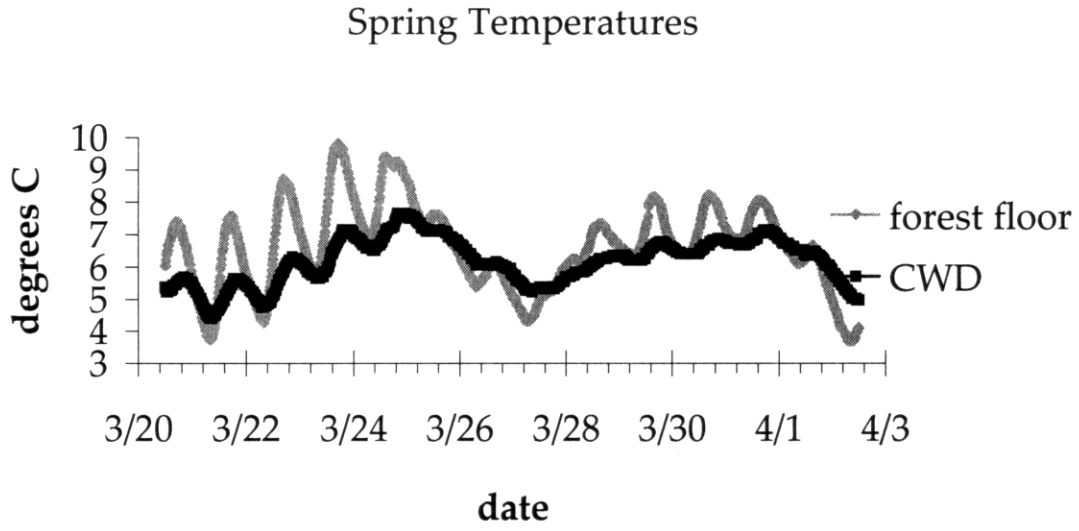

Figure 3.1. Mean spring mineral soil temperatures from under decay class 3 CWD and under the forest floor in the Spring of 2001.