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Julie D.H. Spears for the degree of Doctor of Philosophy in Botany and Plant Pathology presented on April 3, 2002.
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The abundance and spatial heterogeneity of coarse woody debris (CWD) on the forest floor is a prominent feature of Pacific Northwest (PNW) forest ecosystems. The effect of CWD on soil solution chemistry, nutrient cycling and availability, soil physical structure and formation of soil organic matter, however, remains unknown. Therefore, studies on the spatial and temporal imprint of CWD on forest soils are timely and can fill critical gaps in our understanding of the role of CWD in PNW forest ecosystems. I investigated the effect of CWD on soils and soil solution at the H.J. Andrews Experimental Forest in a two-part study. Mineral soils were sampled beneath CWD to a depth of 60 cm. The top 15 cm of soil was also repeatedly sampled for seasonal differences. Control leachate, CWD leachate and soil solution from control soils and from under CWD were collected from the fall of 1999 until the spring of 2001. Results indicated that CWD leachates were much more acidic than water leaching from the forest floor without CWD. Intermediate stages of CWD decomposition had

the highest concentrations of hydrophobic compounds and polyphenols of all stages of decay. Correspondingly, surface soils sampled from under well-decayed CWD were more acidic and had more exchangeable acidity and aluminum, and a lower percent base saturation than soils under the forest floor. Nutrient pools were not different under CWD, although nitrogen fluxes were slower under CWD. Although we had hypothesized that the spatial variability of CWD inputs may affect forest soils under CWD, we found that the spatial variability is much more temporal than I had hypothesized and is limited to the top five centimeters of the underlying soil.

The Imprint of Coarse Woody Debris on Soil Biological and Chemical Properties in the Western Oregon Cascades

by
Julie D.H. Spears

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Dean of Graduate School

I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

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Julie D.H. Spears, Author

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CONTRIBUTION OF AUTHORS

Two additional authors have generously contributed to the manuscripts contained in this dissertation. Mark Harmon provided the data from earlier experiments to compare the effects of different tree species chemistry on coarse woody debris leachate quality. Scott Holub is a co-author on the third chapter because he helped perform the gross nitrogen mineralization experiments and because his insights were helpful throughout the design, implementation and analysis of the experiment.

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This dissertation is dedicated to my grandmother, Mary Spears, who decided she wasn't going to wear a corset anymore and left it under the claw foot bathtub.

THE IMPRINT OF COARSE WOODY DEBRIS ON SOIL BIOLOGICAL AND CHEMICAL PROPERTIES IN THE WESTERN OREGON CASCADES

CHAPTER 1: INTRODUCTION

The spatial heterogeneity of coarse woody debris (CWD) on the forest floor is a prominent feature of Pacific Northwest (PNW) 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 nitrogen (N) fixation in CWD (Hicks 2000, 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 have been investigated (Harmon and Sexton 1995), as well as wood mineralization by invertebrates (Ausmus 1977), and the chemistry of the interstitial water in 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 PNW forest ecosystems.

Current forest management practices such as clear-cutting are dramatically decreasing the amount of CWD left on the forest floor in the Pacific Northwest (Sollins et al. 1987). A decrease in the amount of CWD may affect the amount of organic matter and nutrients stored 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, and a potential source of soil organic matter. 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, fragmented into smaller chunks, or leached as dissolved organic matter (DOM) into the soil beneath it. DOM leaching from CWD contains 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 nitrogen and phosphorus (P) in soil (Qualls and Haines 1991). Organic compounds containing N and P are mobilized through heterotrophic microbial activity as dissolved organic nitrogen and dissolved organic phosphorus 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.

The chemical nature of CWD leachate may affect its movement through the soil profile. The dissolved organic carbon (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). Yavitt and Fahey (1985) found over 80% of the DOC in CWD leachate to be acidic in nature, dominated by hydrophobic compounds, which they believe 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 a handful of studies have looked at the effect of CWD on soils and their results have been contradictory. Busse (1994) investigated the microbial community and nutrient pools in soils under CWD. He 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. However, Kayahara and others (1996) and Krzyszowska-Waitkus and others (in review) found little C input into mineral soils in a British Columbia ecosystem and lodgepole pine ecosystem in Montana, respectively.

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 were collected as often as possible from the fall of 1999 until the spring of 2001.

HYPOTHESES

Figure 1.1 represents the differences that I expect to find in soil chemistry between soils under CWD and soils under the forest floor. Chapter two explores the hypothesis that CWD alters soil chemistry differently than inputs from the forest floor through leaching inputs. Not only will the leachate chemistry differ between CWD and the forest floor, but the leachate chemistry will also differ between decay classes of CWD. CWD leachates will potentially acidify the soil beneath the CWD and leach base cations from the surface soils under well-decomposed CWD because they are more acidic and contain more total DOC, polyphenols, and hydrophobic compounds than leachates from the forest floor. Acidification will also contribute to podzolization at deeper soil depths below class 5 CWD by releasing aluminum and iron from soil matrices. I also predict

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. For comparisons as control leachates, I sampled leachates from under the forest floor with zero-tension lysimeters as well. 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) to the underlying soil via DOC production and sorption. Carbon inputs and leaching to lower horizons will increase soil C to greater depths as time of decomposition proceeds (Figure 1.1). This C influx will increase the C:N ratios of the soil which may potentially change the structure of the microbial community. Because N-fixation occurs in CWD, I also hypothesized that there would be a large amount of N leaching from the CWD into the underlying soils. This will change N pools beneath CWD. I also

hypothesized that N cycling would be slower in soils under CWD because the recalcitrant C inputs would slow microbial degradation.

To investigate these hypotheses, I used the same experimental design as in chapter 2 to collect CWD and forest floor leachates and soil solution from under CWD and from control soils without CWD. To determine the effect of CWD on soil nutrients, I analyzed leachate and soils collected from three depths under a chronosequence of CWD logs for C, N and P. In addition to N pools, I analyzed soils for nitrogen cycling rates with the isotope pool-dilution technique. I also looked at several indicators of soil microbial biomass size, activity, and community structure in surface soils over several seasons.

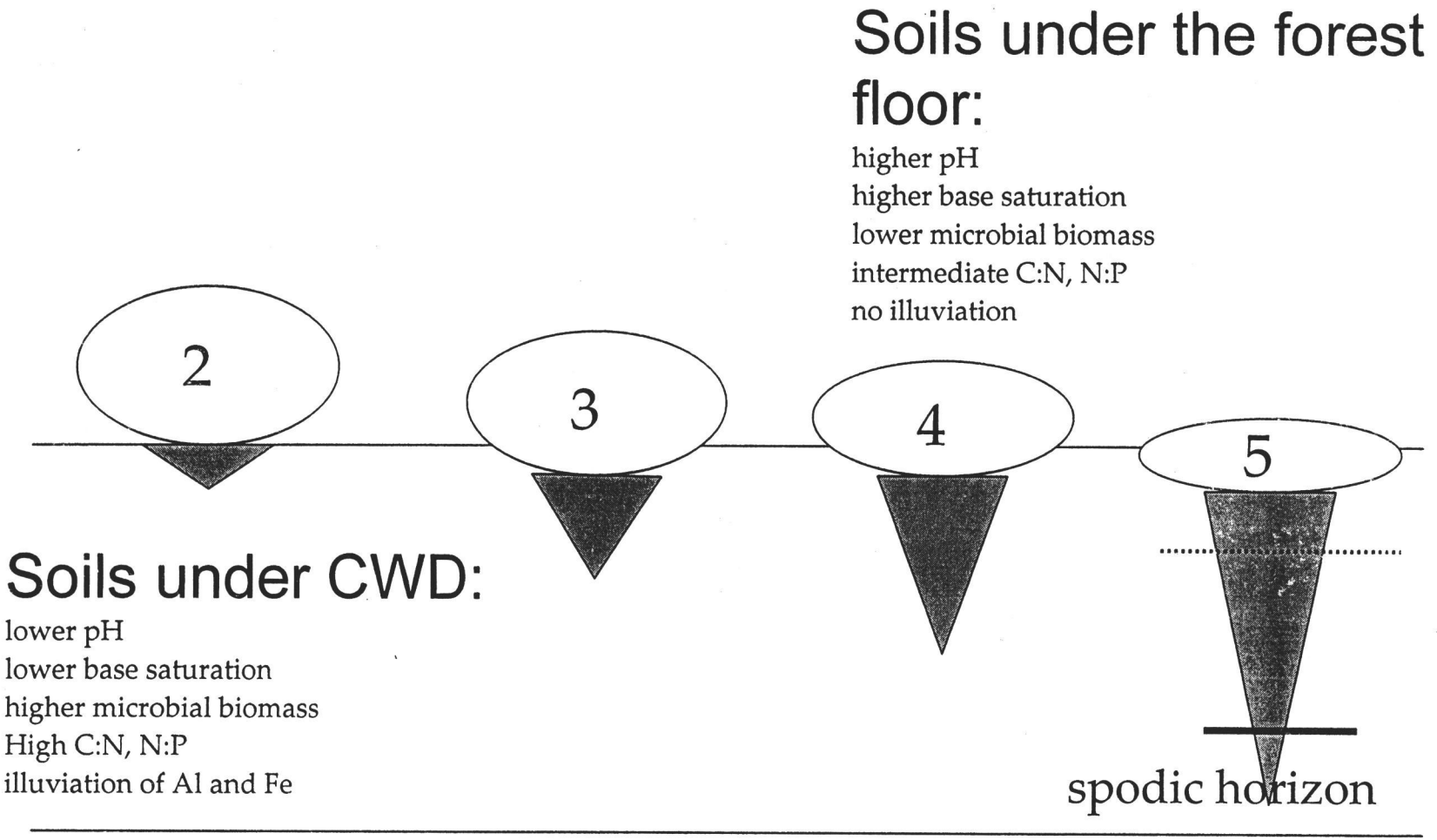

Figure 1.1. Visual representation contrasting the hypothesized chemistry of soil underlying decomposing CWD of different decay classes and soil that has not been affected by CWD.

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THE IMPRINT OF COARSE WOODY DEBRIS ON SOIL CHEMISTRY IN THE WESTERN OREGON CASCADES

CHAPTER 2

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. 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 leachates 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 other Al or Fe fractions. At depths 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 distinctive 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 (PNW) 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 alter, and oftentimes decrease, the amount of CWD contributed to the forest floor compared to historic litter inputs. This may alter the spatial mosaic of the forest floor and potentially alter PNW forest soil processes.

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). However, removing CWD may reduce soil chemical weathering in these ecosystems.

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, 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). Thus, 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.

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 others (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 and he hypothesized that it was caused by the acidic stemflow of the trees. We hypothesize that CWD leachates would have a similar effect on the underlying soil chemistry in the PNW.

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, increasing the amount of metal-bound P in soils under CWD and limiting 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 releases 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 will be greater than concentrations of DOM leaching from the forest floor. We also hypothesize that CWD leachates should also be much more acidic and rich in hydrophobic and polyphenol compounds than forest floor leachates. These acidic inputs will bind to the soil exchange sites, complex metals, decrease soil pH and increase the exchangeable acidity in soils beneath CWD. Furthermore, base cations will be decreased as they are leached as base partners to maintain electrical neutrality with the organic ligands. Podzolization processes will control Al and Fe chemistry, binding Al and Fe to carbon compounds and eluviating these complexes to lower soil horizons. We therefore hypothesized that there will be a strong signal of podzolization as measured by the ratio of free iron compounds to free plus organically bound iron compounds.

METHODS

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 circinatum* 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 Humic Dystrodepts. 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 cmol/kg as well.

Experimental design and field methods:

Douglas-fir CWD was classified into the five decay classes of Triska and Cromack (1979) and ages were approximated using Sollins et al. (1987). Class 1 is a freshly fallen log with sound wood and intact bark. After approximately 10-11 years the bole enters class 2 (Sollins et al. 1987). A class 2 log has an initial rotting of the sapwood. After approximately 50 years, the bark begins to slough off and the sapwood decays further. The CWD is in class 3 of decay. The CWD is class 4 after lying on the ground for approximately 87 years. By this time, the sapwood has rotted, the bark has sloughed off, and the heartwood is thoroughly rotted. After about 150-200 years of decomposition and when the log has no structural integrity, has settled into the soil and is thoroughly rotted red mush, it is categorized as class 5.

We used class 2 CWD from a log decomposition experiment described more extensively in Harmon (1992). Class 3, 4, and 5 CWD were naturally occurring in the same site. Since class 2 CWD was placed at the site for a log

decomposition experiment, we chose CWD of later decay classes that were adjacent to the class 2 logs. Five sets of logs were selected, with each set containing one CWD in class 2, 3, 4, and 5 and a control with no CWD.

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 classes 2, 3, and 4. Because class 5 logs are unstructured and settled into the soil, it was not possible to sample leachates with zero-tension lysimeters. 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 that were installed at two depths (30 cm and 50 cm) beneath CWD of classes 2, 3, and 4 and in control soils without CWD (hereafter called control solutions). The first two samples of leachate and soil solution were discarded in order to allow the lysimeters time to equilibrate and to avoid artifacts from installation.

To determine if CWD decomposition affects mineral soil chemistry, bulk soils were sampled from under ten class 4 and ten class 5 logs. We used five of the class 4 logs that were used in the leachate study, although soils were not sampled near the lysimeters. An additional five class 4 logs were selected from

the site area to increase the sample size to 10. Class 5 CWD and control soils were chosen in areas adjacent to 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 samples were homogenized thoroughly within each depth.

Lab analyses

Solutions were analyzed for pH with an Orion Scientific electrode prior to filtration. Solutions were filtered using an ashed GF/F filter and stored frozen 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 soils 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.

STATISTICAL ANALYSES

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.

RESULTS

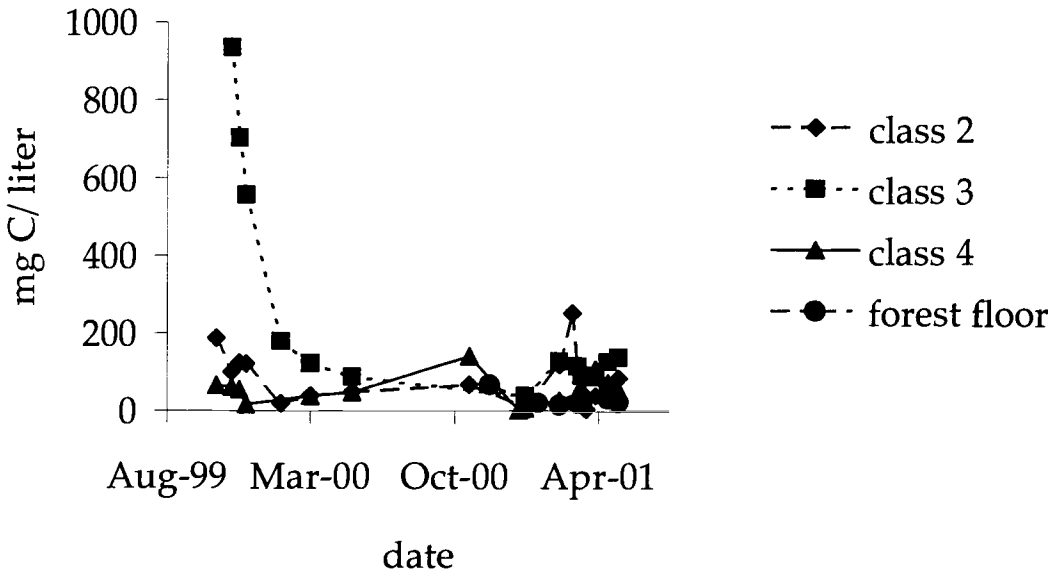
Lysimeter solutions

Dissolved organic carbon concentrations in CWD leachate from zero tension lysimeters were generally much higher than those from the forest floor (Figure 2.1a and b). DOC concentrations from zero tension lysimeters under class 3 CWD were generally highest. Class 2 CWD leachate showed considerable variation in DOC concentration with no apparent trends in relation to the other decay classes.

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.

a



b

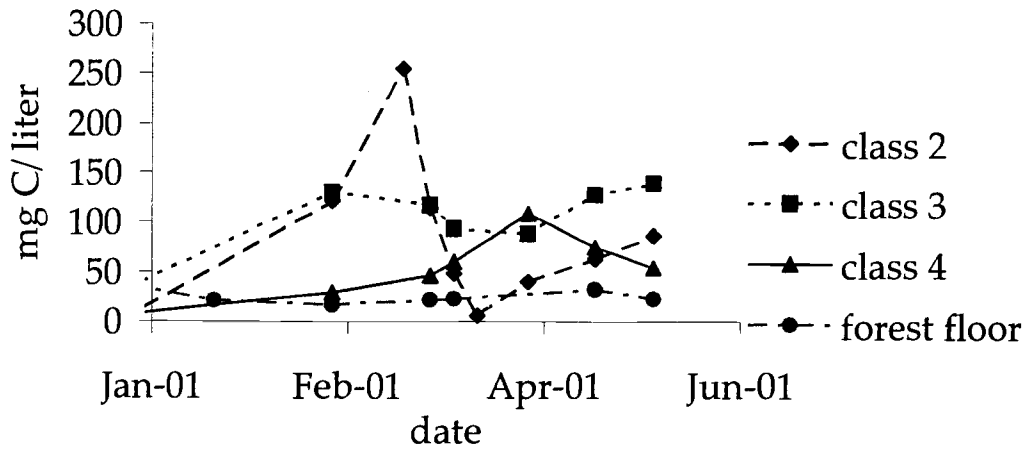


Figure 2.1a. Mean DOC concentrations collected from zero tension lysimeters under CWD of decay classes 2,3, and 4 and from under the forest floor. Forest floor samples were not collected prior to fall of 2000. Figure 2.1b shows the spring of 2001 in greater detail. Note the change in scale between Figure 2.1a and 2.1b.

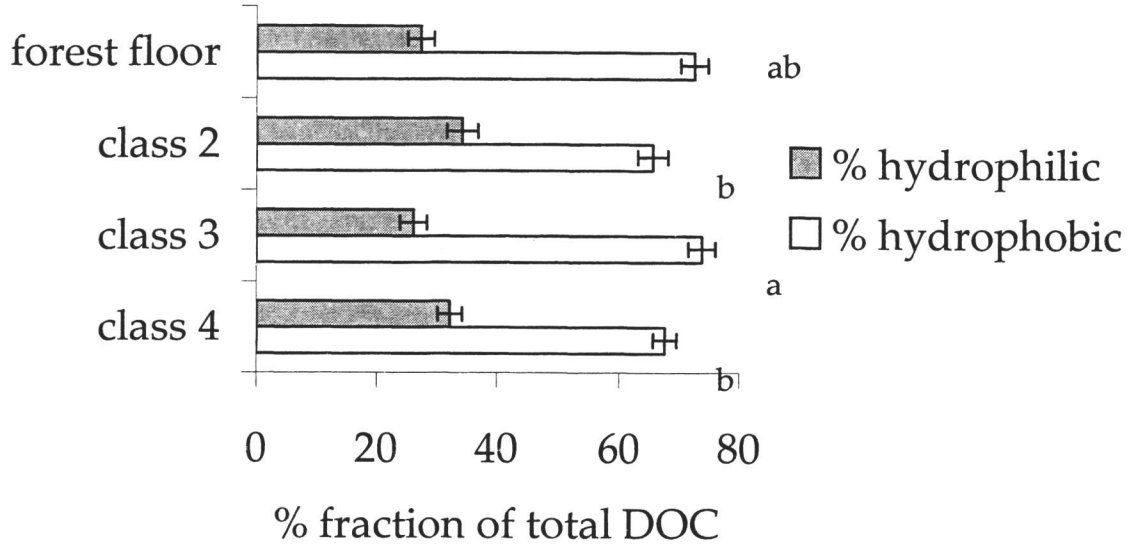


Figure 2.2. Mean percent hydrophobic and hydrophilic DOC across all collection dates in samples collected with zero-tension lysimeters from under CWD of decay class 2, 3, and 4 and from under the forest floor. Bars represent one standard error and letters represent statistical difference using Tukey's HSD multiple comparison test. Treatments with different letters are significantly different at α=0.05.

Zero-tension leachate from Class 3 CWD had higher concentrations of polyphenols than class 2 CWD, 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 although concentrations were not significantly different (Figure 2.3). 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, se=0.25) were lower than in leachates under CWD. However there was no difference among concentrations collected under various decay classes. The mean from class 2 was 5.52 (se=1.36), the mean from class 3 was 5.97 (se=1.04), and the mean from 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).

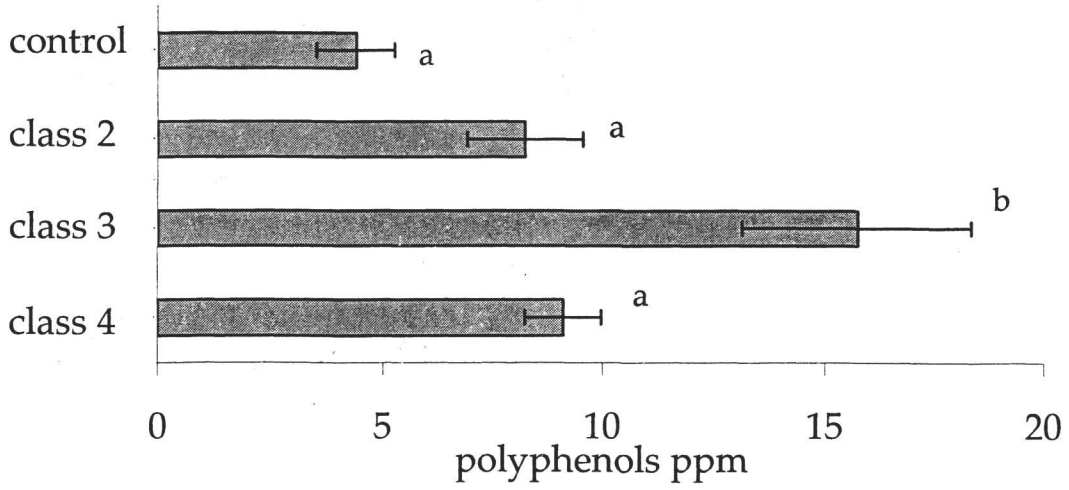


Figure 2.3. Mean polyphenol concentrations across all collection dates in samples collected with zero-tension lysimeter under CWD class 2, 3, and 4 and control leachates from under the forest floor. Bars represent one standard error and letters represent statistical difference using Tukey's HSD multiple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

CWD leachate in zero tension lysimeters was approximately thirty times 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.

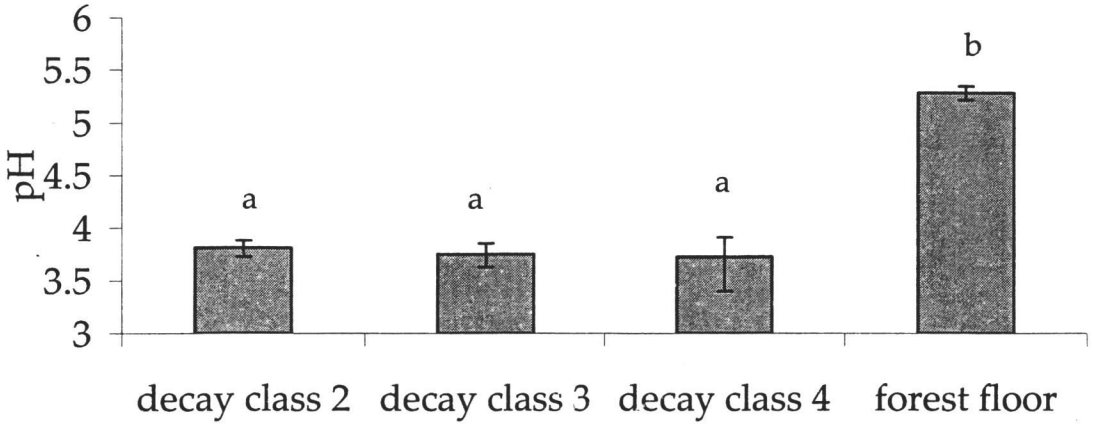


Figure 2.4. Mean pH of leachates across all dates from under CWD of decay class 2, 3, and 4, and from under the forest floor. Bars represent one standard error and letters represent statistical difference using Tukey's HSD multiple comparison test. Treatments with different letters are significantly different.

Mineral soils

Most mineral soil chemical analyses were significant only in the surface soils (Table 2.1). 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), but there was no difference between soils under class 4 and 5. Soil pH increased with depth under

all treatments and did not differ among decay classes or controls at deeper soil depths.

Chemical Analysis:	Result:	0-5 cm	5-15 cm	15-30 cm	30-60 cm
Soil pH	lower under CWD	*	NS	NS	NS
Exchangeable aluminum	higher under class 4 CWD	*	NS	NS	NS
Exchangeable acidity	higher under CWD	*	NS	NS	NS
Percent base saturation	lower under CWD	*	NS	NS	NS
Oxalate Al/ dithionite Al		NS	NS	NS	NS
Oxalate Fe/ dithionite Fe		NS	NS	NS	NS
Organic Al/ Carbon		NS	NS	NS	NS
NaOH inorganic P		NS	NS	NS	NS
NaOH organic P		NS	NS	NS	NS

Table 2.1. Summary of the soil chemical analyses from this study. Comparisons were made between soils under decay classes 4 and 5 and control soils with Tukey's HSD means comparison at each depth.
* indicates that the differences between soils under CWD and control soils were highly significant (p-value≤0.01).
NS indicates no statistical difference among soils under CWD and control soils.

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 one standard error in parentheses for soils under decay class 4 and 5 CWD and control soils at 4 depths. Letters represent statistical difference using Tukey's HSD multiple comparison test. Treatments with different letters are significantly different at α=0.05.

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 class 5 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 ($p \leq 0.002$, Table 2.3). Exchangeable acidity remained relatively high in soils under class 5 CWD to a depth of 60 cm. However, the large variability in exchangeable acidity in soils deeper than 5 cm under class 5 made it difficult to detect significant differences (see standard errors in Table 2.3).

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 one standard error in parentheses for soils under decay class 4 and 5 CWD and control soils at 4 depths. Letters represent statistical difference using Tukey's HSD multiple comparison test. Treatments with different letters are significantly different at α=0.05.

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 2.5), although differences were not statistically significant. There was a strong positive correlation between exchangeable Al and exchangeable acidity for 0-5 cm samples ($p=0.0001$, $R^2= 0.68$).

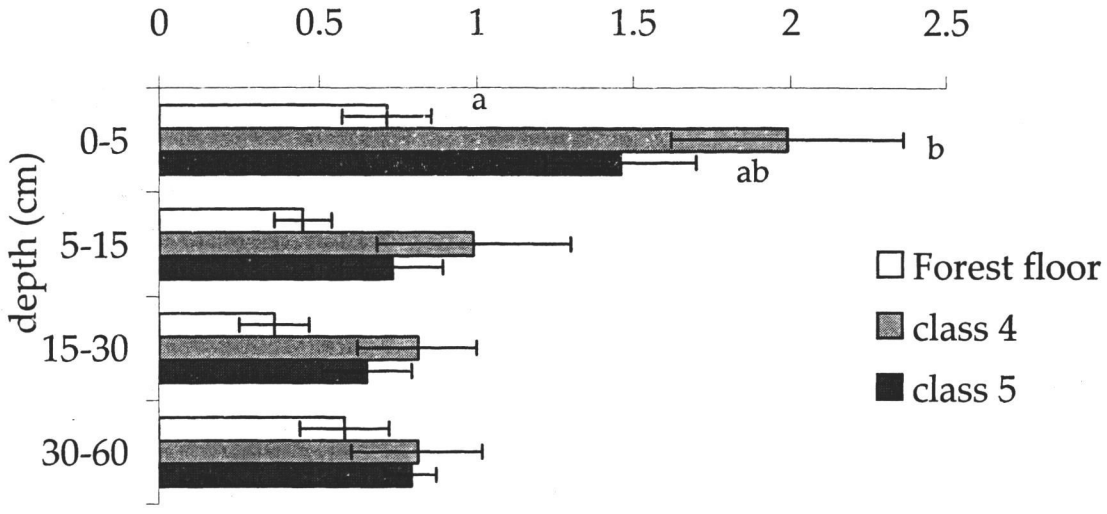


Figure 2.5. Mean exchangeable aluminum concentrations in cmol/kg from four soil depths below CWD of decay class 4 and 5 and control soils. Bars represent one standard error and letters represent statistical difference using Tukey's HSD multiple comparison test. Treatments with different letters are significantly different at α=0.05.

Free, organically bound, and amorphous Al and Fe concentrations were not significantly different between soils under CWD or soils under the forest floor (Figure 2.6 and 2.7). The ratio of organically bound iron to free iron plus organically bound iron did not indicate a spodic horizon under CWD (Figure 2.8).

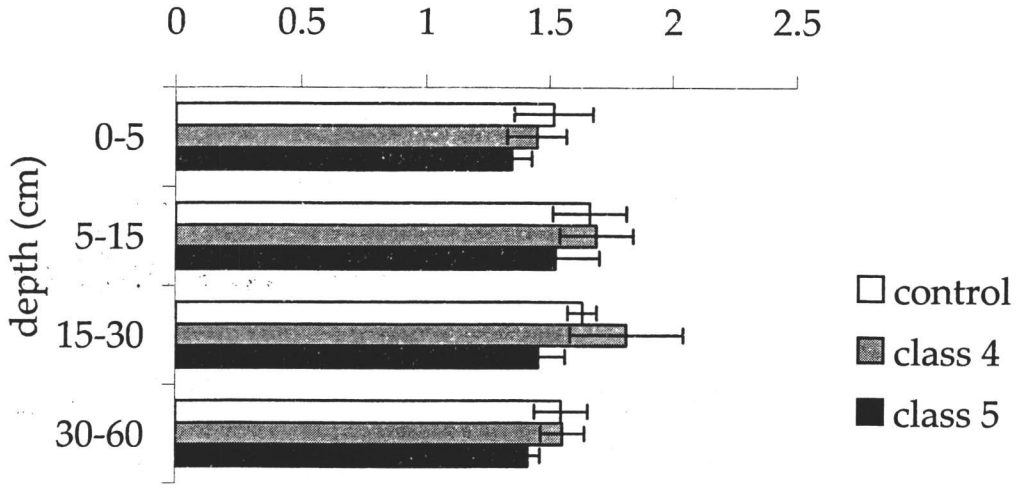


Figure 2.6. Ratio of oxalate extractable aluminum to dithionite extractable aluminum from soils under class 4 and 5 CWD and from under the forest floor. Bars represent one standard error.

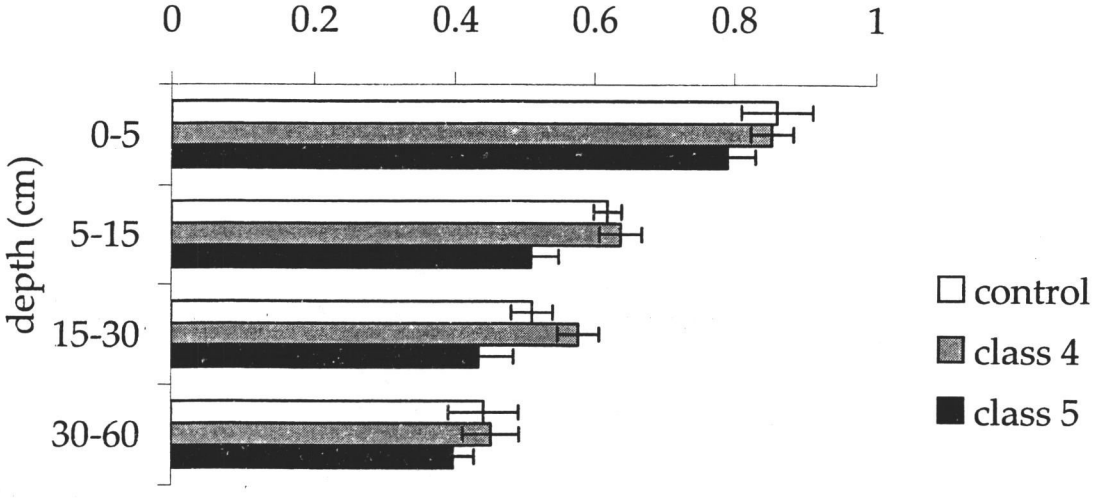


Figure 2.7. Ratio of oxalate extractable iron to dithionite extractable iron from soils under class 4 and 5 CWD and from under the forest floor by depth. Bars represent one standard error.

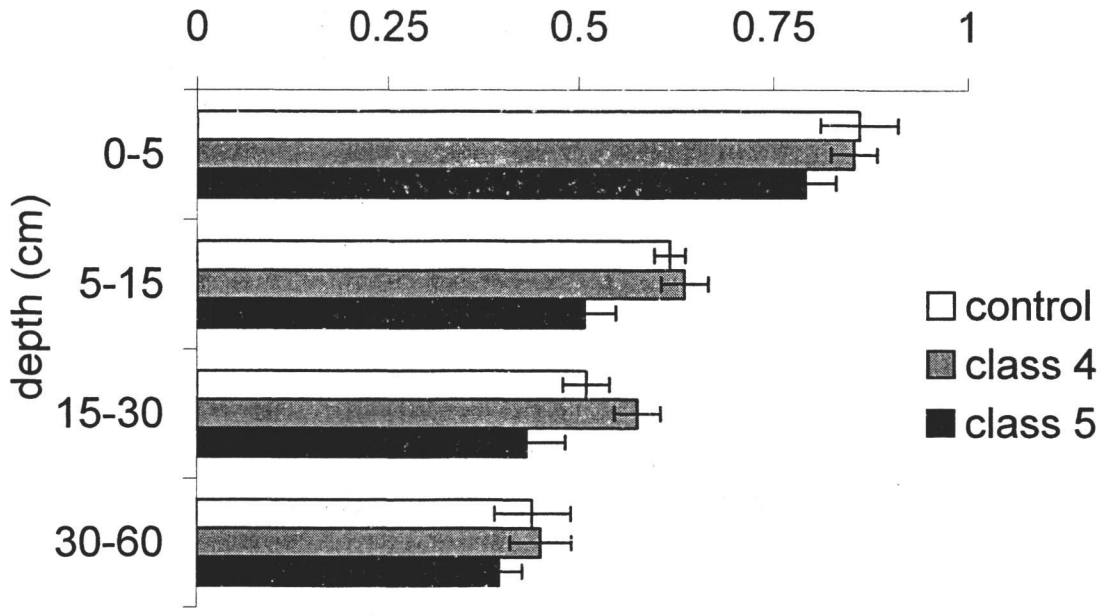


Figure 2.8. Ratio of organic iron divided by free iron plus organically bound iron from soils under class 4 and 5 CWD and from under the forest floor by depth. Bars represent one standard error.

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 soil 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 there was a large amount of variation.

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 one standard error in parenthesis for soils under decay class 4 and 5 CWD and control soils at 4 depths. Letters represent statistical difference using Tukey's HSD multiple comparison test. Treatments with different letters are significantly different at α=0.05.

The inorganic NaOH P fraction (NaOH P_i) was not significantly different among soils under class 4 and 5 CWD or control soils to a depth of 0-30 cm (Figure 2.9). At 30-60 cm, however, soils under class 4 CWD had more NaOH P_i than soils from under class 5 CWD ($p \leq 0.03$). 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 (Figure 2.10). 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.

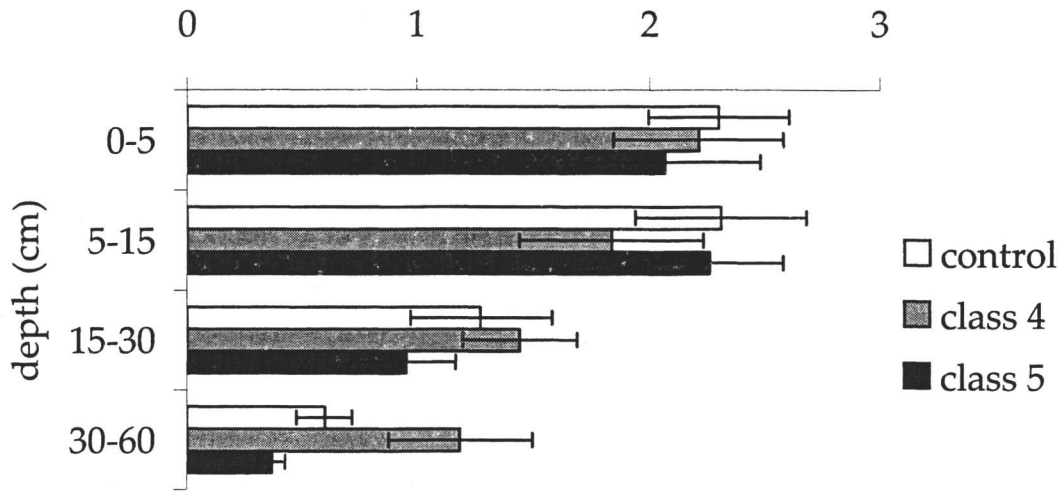


Figure 2.9. Mean inorganic phosphorus in ug P/ g extracted with NaOH for soils under decay class 4 and 5 and for soils under forest floor. Bars represent one standard error.

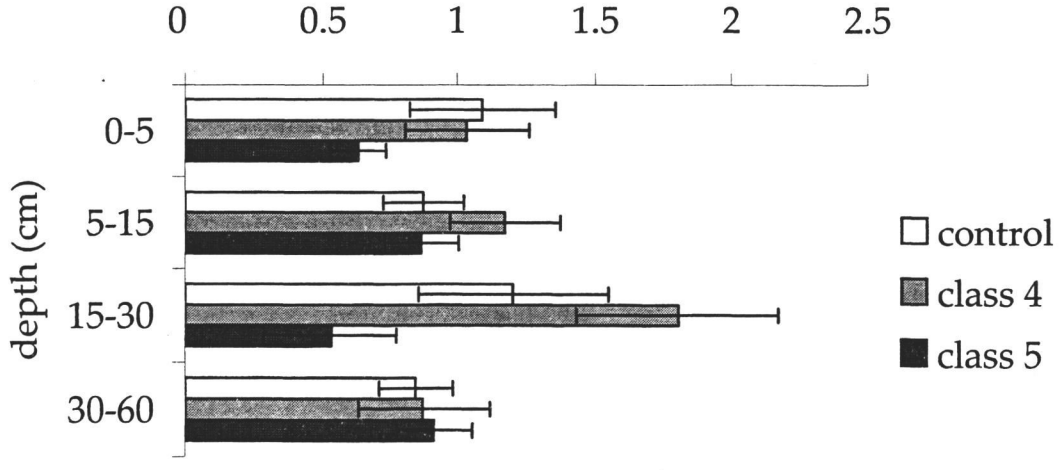


Figure 2.10. Mean organic phosphorus in ug P/ g extracted with NaOH for soils under decay class 4 and 5 and control soils. Bars represent one standard error.

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 more actively rotting (class 4) than later in decomposition (class 5). Soils under class 5 CWD were rarely different than control soils that had not had any visible evidence of CWD influence. This suggests that the earlier decay classes may have the largest effect on soil chemistry and that the affect of CWD on soil heterogeneity is not a profound, spatially discrete effect, but rather a more subtle effect that changes over time.

We hypothesized that the DOC in CWD leachates would be much more acidic, and contain more hydrophobic and phenolic 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 which was similar to the ratio of hydrophobic to hydrophilic DOC found in 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. Polyphenol concentrations were more accurate in describing the differences between CWD and forest floor leachates. 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. Although we did not measure class 5 leachates, we suspect that polyphenol concentrations in class 5 leachates would be lower than in class 4 leachates. The remaining substrate in class 5 is thoroughly degraded, lignin rich, and mostly non-soluble and therefore may not leach as readily as other CWD decay stages (Prescott et al. 1990).

Concentrations of soluble polyphenols were generally similar to those reported by Yavitt and Fahey (1985). There was a particularly large fall flush of

polyphenols in 1999 when 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 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 than in control soils. However, it was lower only in the surface soils and not in deeper soils. Acidic soils under CWD may be due to several sources, 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 the matrix is broken down during chemical weathering, creates H+ ions as the Al is rapidly hydrolyzed. Exchangeable Al was found to be much higher in soils under CWD than in soils under the forest floor, particularly in surface soils (Figure 2.5). However, exchangeable Al did not account for all of the total exchangeable acidity, and the remainder must be from the acidic functional groups on soil organic matter.

Mineral soils under CWD were high in exchangeable Al, reflecting an increased amount of Al dissolution from soil matrices, while control soils were

high in exchangeable Ca, reflecting that little leaching had occurred. We found that shallow soils under CWD consistently had a lower percent base saturation than soils under the forest floor (Table 2.4). 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 containing compounds and acidic DOC from the CWD leachates probably leach the base cations to lower soil horizons as partners to maintain electrical neutrality. Thus 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 replaces Al and Fe organic compounds on soil exchange sites, thereby causing Al and Fe to leach to the lower horizons. Thus, 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 that the concentration of hydrophobic was twice as large as the concentration of hydrophilic compounds in CWD leachates, trends indicated only a slight increase in the Fe and Al compounds in lower soil horizons.

The larger pool of exchangeable Al that we observed under CWD compared to control soils suggested that the soil P should be tied up as an unavailable form of Al hydroxide P. However, there was little difference in P chemistry in soils under CWD compared to control soils. As mentioned above, 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.

CONCLUSIONS

Soils under the most highly decayed CWD and at depth did not reflect a CWD imprint. Although we had hypothesized that CWD may have a profound spatial imprint on the underlying soils, chemical weathering of the soil beneath CWD was limited. This study suggests that the spatial effects of CWD on soil chemical 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. Leachates release Al from soil minerals, which act as a positive feedback increasing soil

acidity. Previous studies have suggested that there were more organic acids, particularly citric acid, under CWD of class 4 than in control soils (Krzyszowska-Waitkus, personal com.). However, no study has directly investigated the effect of low molecular weight organic acids on soil chemistry beneath CWD and therefore, further study is necessary to elucidate the mechanisms of soil acidification under CWD.

Although we hypothesized that well-decayed CWD would show a spatially explicit imprint on soils, the imprint was not evident under the most highly decayed CWD. Therefore, CWD may create only 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 site. More extensive research in different ecosystems is needed to determine whether the patterns that we observed are generalizable.

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LIFE UNDER LOGS; THE INFLUENCE OF COARSE WOODY DEBRIS ON SOIL BIOLOGY AND NUTRIENT CYCLING

CHAPTER 3

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

ABSTRACT

The nutrient stores and dynamics in decomposing logs have been quantified in many studies, but 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 and comparing these to forest floor leachates as controls. We measured leachates for carbon, nitrogen, and phosphorus. We measured soil from under CWD and control soils for moisture, temperature, carbon (C), nitrogen (N) , gross N mineralizations, labile phosphorus (P), and water-soluble organic carbon (WSOC). Microbial biomass and enzyme activity were measured as indices 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 leachate collected under 4 species of CWD in the early stage of decay. We found no difference in C, N, P, microbial biomass or enzyme activity between soils under CWD compared to those under the forest floor. CWD leachate and forest floor leachate were similar in dissolved inorganic and organic nitrogen, and dissolved inorganic phosphorus. Results indicate that the mineral soil under CWD was warmer, but there was no difference in soil moisture between soils under CWD and control soils. 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. Leachates differences among 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, which suggests 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

Soils are the largest pool of carbon (C) in terrestrial ecosystems (Post et al. 1990) and therefore, fluxes of C into and out of forest soils are a critical component of the global C cycle. Coarse woody debris (CWD) has been considered by some researchers as stockpiles of carbon on the forest floor that may increase soil organic carbon through leaching inputs over the long term of ecosystem development (Sollins et al. 1987, Harmon et al. 1986). If we consider that the biomass of CWD at the H.J. Andrews Experimental Forest in the Western Oregon Cascades can range from 80 to 140 Mg/ ha (Harmon et al. 1986), it is not unreasonable to assume that a large portion of this C could be contributed to the soil and therefore, that CWD may be an important source of C to forest soils in the Pacific Northwest (PNW).

CWD may also affect nutrient cycling and nitrogen (N) losses from ecosystems through microbial immobilization. Indeed, some researchers have considered that grinding up CWD and mixing the residue into the soil could abate the deleterious affects of nitrogen (N) saturation (Waring and Running 1998, Waring and Pitman 1985). 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). Pathways of soil nutrient enrichment may include: 1) nutrient losses from CWD by fungal sporocarps (Harmon et al. 1994), 2) N fixation and associated leaching of N (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).

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 by preventing freeze-thaw cycles, which disrupt biological activity and can lead to nutrient leaching losses. Thus, 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 in a lodgepole pine ecosystem. The difference between mineral soil C under CWD and control mineral soils in this study, however, was small and limited to the surface horizons. Conversely, Busse (1994) reported a decrease in total C in soils beneath CWD in an eastern Oregon lodgepole pine forest.

In addition to C, leaching of decomposing CWD may transfer nutrients 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) or approximately 1/8 of the total annual input of N to these old-growth PNW ecosystems. Nitrogen may leach from CWD as dissolved organic nitrogen (DON) to the underlying soil increasing soil N pools and influencing N cycling rates. 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). In contrast, lower amounts of mineralizable N were found in B-

horizon soils beneath well-decayed CWD than in control soils in British Columbia (Kayahara et al. 1996). In eastern Oregon, Busse (1994) also found lower concentrations of inorganic N under CWD.

In addition to N, CWD leachates may affect soil P pools. Using log chronosequences, 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 have 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.

In this paper, our objectives were to investigate the effect of CWD on soil biology and nutrient cycling by examining the nutrient inputs in CWD leachates and changes in soil nutrient pools, microbial biomass and microbial community structure using a chronosequence of decayed CWD. We hypothesized that 1) DOC concentrations of CWD leachate would be largest in the later stages of decay because there would be more C leaching as decomposition proceeds and 2)

that soils underlying well-decomposed CWD would have more C because prolonged inputs of DOC-rich leachates. We also hypothesized that there would be lower N and P than soils under the forest floor due to leaching losses of P and microbial immobilization of N in CWD. Changes in nutrient pools and N fluxes that occur as CWD ages will 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 Douglas-fir 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 the quality of CWD leachates. Species that are less resistant to decay should show a faster release of C and nutrients, whereas species that are more resistant should show a slower release.

METHODS

Site description

This study was conducted 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 type (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 circinatum* 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 annual temperature is 10° C. Soils have been classified as Humic Dystrodept. More recent chemical analyses, however, suggest that these soils may be classified as Andisols (Spears and Lajtha, in review).

Experimental design and field methods

Douglas-fir CWD was classified into the five decay classes of Triska and Cromack (1979) and ages were approximated using Sollins et al. (1987). Class 1 is a freshly fallen log with sound wood and intact bark. After approximately 10-11 years the bole enters class 2 (Sollins et al. 1987). A class 2 log has an initial rotting of the sapwood. After approximately 50 years, the bark begins to slough off and the sapwood decays further. The CWD is in class 3 of decay. The CWD is class 4 after lying on the ground for approximately 87 years. By this time, the sapwood has rotted, the bark has sloughed off, and the heartwood is thoroughly rotted. After about 150-200 years of decomposition and when the log has no structural integrity, has settled into the soil and is thoroughly rotted red mush, it is categorized as class 5.

Our experimental design included Douglas-fir CWD of four decay classes. We used class 2 CWD from a log decomposition experiment, described more extensively in Harmon (1992). We chose naturally occurring CWD of decay classes 3, 4, and 5 that were adjacent to the experimentally placed class 2 logs. Five sets of logs and associated controls were selected. Each set consisted of one log in each of class 2, 3, 4, and 5 and areas under the forest floor where there were no CWD as a control.

Leachate Sampling

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 no CWD was evident (hereafter referred to as control leachates). Prenart tension lysimeters were installed under class 2, 3, and 4 at 30 cm, 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 to avoid a disturbance signal.

Species differences

To determine if log leachate chemistry differs between tree 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, Psuedotsuga menziesii, Abies amabilis* and *Tsuga heterophilia.*

Soil sampling

In the spring of 2000, soils were sampled from under 10 Douglas-fir 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 mineralization analyses.

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 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 ascorbic acid method (Lajtha 1999). DIP in 30 cm and 50 lysimeter leachates were below detection limits of 0.001 mg l^{-1}.

Soils were sieved through a 4 mm mesh and extracted with K_2SO_4 for microbial biomass C (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 ten mmol PO_4 buffer (pH=6.8), shaken for

20 min., and allowed to settle. One hundred and fifty µl 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 Biolog ® plates do not indentify community type or functional groups, but rather indicate differences in microbial preferences and this may include only those microbes that reproduce during the incubation time.

Soils for total C and N were powdered with a spex-mixer mill and analyzed on a Carlo Erba NA1500 CHN analyzer at the Stable Isotope Biology Lab at the University of Georgia. Water-soluble organic carbon (WSOC) was extracted using a 2:1 water:soil extraction. Slurries were filtered through a GF/F filter and analyzed on a Shimadzu TOC analyzer. Labile P was extracted with resin strips 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 between CWD and control soils. 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). In order to determine if gross N transformations were also similar in soils under CWD and control soils, we used 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. Soils were incubated in the lab at 25°C after a thorough mixing of the soil. This does not represent *in situ* mineralization rates but determines potential transformation rates. Microbial biomass N and C were also analyzed on sub-samples of these samples (Voroney et al. 1999).

Lysimeter solutions from each 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 performed using SAS (SAS Institute, Cary NC, 2000). Because most lysimeters did not produce adequate samples at all dates, it was not possible to use repeated measures analysis of variance (ANOVA) to analyze all lysimeter samples for 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.

Because the lysimeter solutions for the different species of CWD were composited for two seasons, it was only valid to compare among species for differences. Thus to determine if there was a difference among species, we used ANOVA and compared the means for each season with Tukey's HSD multiple comparison test (alpha=0.05).

Results from the 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

There were few differences between soils under CWD and control soils (Table 3.1). In the spring, mean percent soil moisture in soils from 0-5 cm depths appeared 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 was not significantly different. 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). There was less diurnal variability in soil temperatures under class 3 CWD 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).

Chemical Analysis:	Result:	0-5 cm	5-15 cm	15-30 cm	30-60 cm
Soil Moisture	no difference	NS	NS	NS	NS
Water soluble organic carbon	higher under class 4 CWD	NS	*	NS	NS
Gross $^{15}NH_4$ mineralization rates	lower under CWD	*	NS	NS	NS
Percent soil moisture		NS	NS	NS	NS
Total C		NS	NS	NS	NS
Total N		NS	NS	NS	NS
C:N ratios		NS	NS	NS	*
Resin P		NS	NS	NS	NS
Sodium Bicarbonate P		NS	NS	NS	NS
Enzyme activity		NS	NS	NS	NS
Microbial biomass C and N		NS	NS	NS	NS

Table 3.1. Summary of the soil biological analyses from this study. Comparisons were made between soils under decay classes 4 and 5 and control soils with Tukey's HSD means comparison at each depth.
* indicates that the differences between soils under CWD and control soils were highly significant (p-value≤0.05).
NS indicates no statistical difference among soils under CWD and control soils.

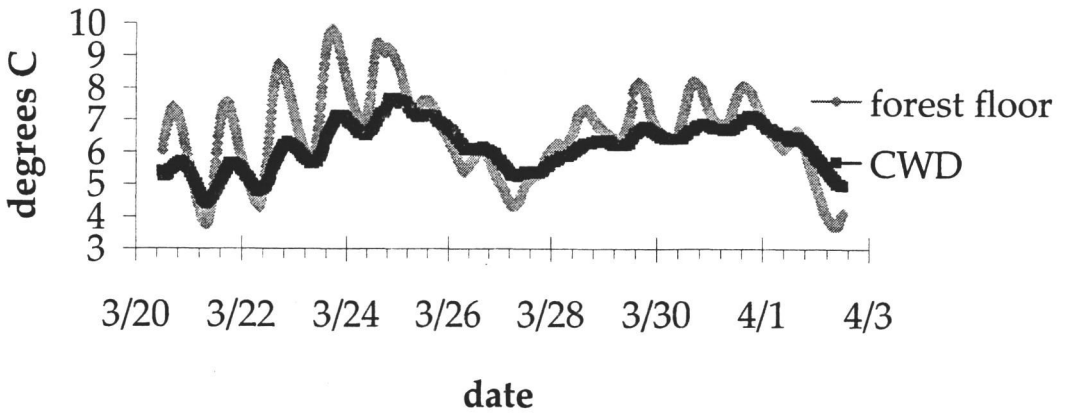
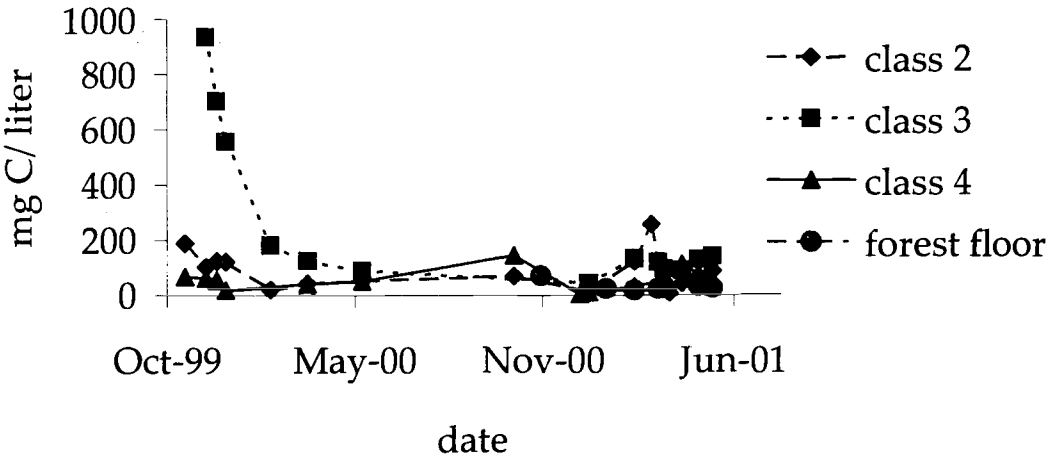


Figure 3.1. Mean mineral soil temperatures from under decay class 3 CWD and under the forest floor in the spring of 2001.

Dissolved organic carbon concentrations in zero-tension lysimeters were often highest under class 3 CWD, although concentrations varied over time (Figure 3.2 a and b). Peak concentrations for lysimeters under class 2 CWD were 250 mg C l^{-1}, while class 3 CWD had a one-time peak of 935 mg C l^{-1} that decreased with time. Control lysimeter DOC concentrations were lowest of all treatments throughout the year, averaging 20 mg C l^{-1}.

a



b

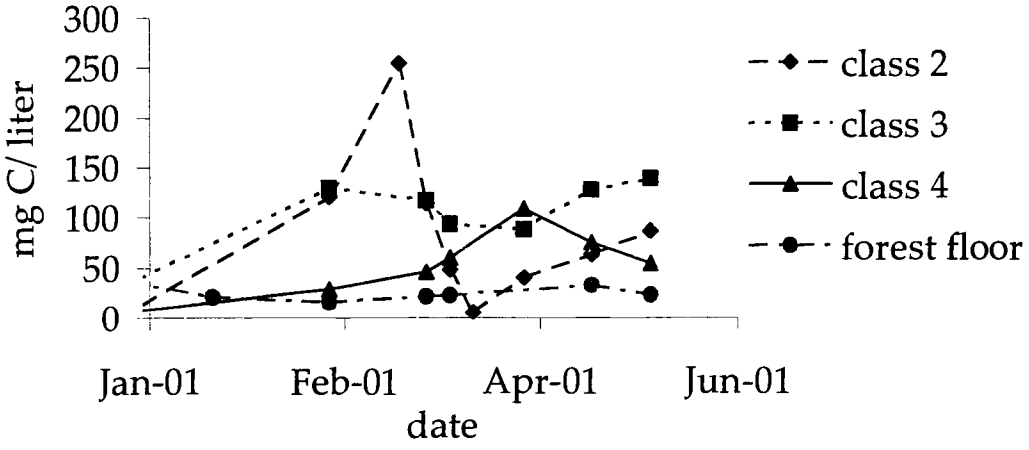


Figure 3.2. (a) Mean dissolved organic carbon concentrations from zero-tension lysimeters under CWD of decay classes 2, 3, 4 and controls under the forest floor. Forest floor lysimeters were not sampled until November of 2000. (b) A close-up view of dissolved organic carbon concentrations from zero-tension lysimeters under CWD of decay classes 2, 3, 4 and from controls under the forest floor.

Lysimeter DOC concentrations at 30 cm were often highest under class 3 CWD and lowest in control solutions (Figure 3.3a and b). Concentrations of DOC at 30 cm under 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 CWD 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.

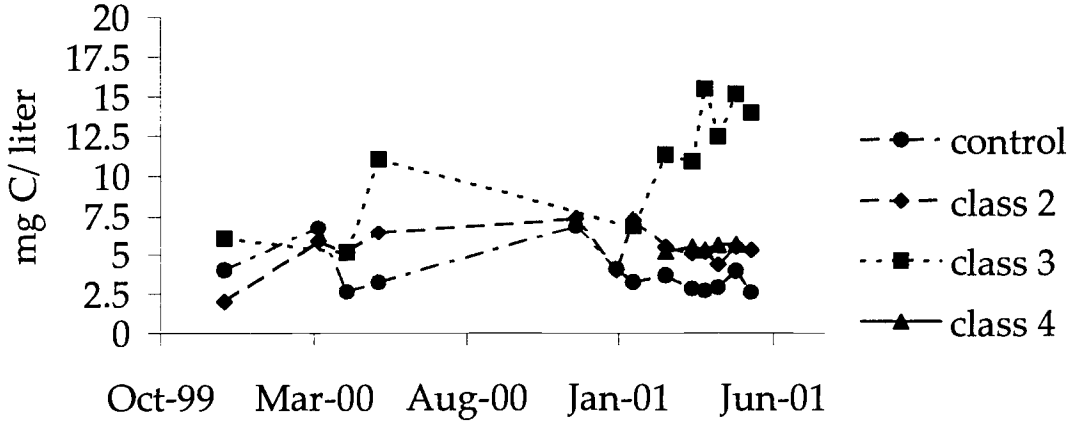


Figure 3.3. Dissolved organic carbon concentrations at 30 cm under decay classes 2, 3, and 4 CWD and control soils. Class 4 lysimeters did not provide sample until November of 2000.

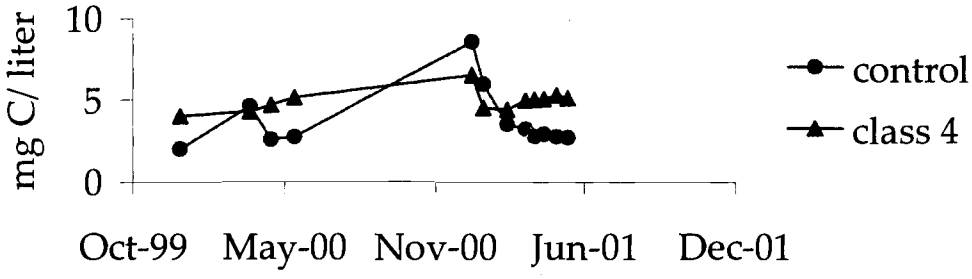


Figure 3.4. Dissolved organic carbon concentrations from soils 50 cm under CWD of decay class 4 and in control soils.

Dissolved organic nitrogen concentrations from zero-tension lysimeters and in tension lysimeters at 30 and 50 cm showed no clear pattern and variation between treatments was large (Table 3.2). Ammonium and nitrate in zero-tension leachates were near detection limits at all depths. Ammonium and nitrate concentrations at 30 cm and 50 cm showed no consistent differences with regard to CWD decay classes or controls. Zero-tension lysimeter DIP was similar among treatments and was extremely low for solutions at depths.

Depth	Decay	DON	NH4	NO3	DIP
0 cm	**control**	0.54 (0.07)	0.02 (0.01)	0.012 (0.002)	0.52 (0.07)
(leachate)	**class 2**	1.27 (0.29)	0.17 (0.04)	0.22 (0.13)	0.58 (0.09)
	class 3	0.81 (0.12)	0.04 (0.01)	0.04 (0.01)	0.68 (0.18)
	class 4	0.60 (0.06)	0.08 (0.02)	0.17 (0.06)	0.38 (0.08)
30 cm	**control**	0.04 (0.02)	0.005 (0.003)	0.009 (0.006)	0.08 (0.06)
(soil					
solution)	**class 2**	0.14 (0.09)	0.007 (0.006)	0.03 (0.02)	0.002 *
	class 3	0.21 *	0.009 *	0*	NA
	class 4	0.17 (0.08)	0.02 (0.01)	0.016 (0.007)	0.19 (0.18)
50 cm	**control**	0.07 (0.03)	0.03 (0.03)	0.63 (0.42)	NA
(soil					0.002
solution)	**class 4**	0.12 (0.03)	0.001 (0.001)	0 (0)	(0.000)

Table 3.2. Mean lysimeter concentrations (mg/L) and standard errors for dissolved organic nitrogen (DON), ammonium (NH4), nitrate (NO3) and dissolved inorganic phosphorus (DIP). Samples were averaged over two water years after visual inspection suggested no effect of date on concentrations.
NA= no sample available
*= only one sample analyzed; no standard error

Total soil C and N did not differ among soils under decay classes 4 and 5, and control soils at any depth (Table 3.3). Soil C:N ratios under class 4 were significantly higher than in control soils only at 30-60 cm ($p<0.05$).

depth	Percent total C			Percent total N			C:N ratio		
	Control	class 4	class 5	Control	class 4	class 5	Control	class 4	class 5
0-5	4.67	5.80	4.55	0.19	0.18	0.17	25	31	27
cm	(0.37)	(1.07)	(0.59)	(0.01)	(0.02)	(0.01)			
5-15	2.88	3.11	2.85	0.14	0.13	0.13	21	24	22
cm	(0.15)	(0.31)	(0.35)	(0.01)	(0.006)	(0.01)			
15-30	1.57	2.03	1.58	0.096	0.10	0.09	17	20	18
cm	(0.19)	(0.28)	(0.12)	(0.01)	(0.008)	(0.003)			
30-60	1.03	1.31	0.81	0.07	0.07	0.06	14*	17*	14
cm	(0.16)	(0.17)	(0.06)	(0.01)	(0.006)	(0.004)			

Table 3.3. Mean percent total C and N and C:N ratios from soils at four different depths under decay class 4 and class 5 CWD and from control soils. Standard errors are in parentheses.

There were no statistically significant differences in WSOC among soils at 0-5 cm, but at 5-15 cm soils under class 4 CWD had higher WSOC than soils under class 5 CWD ($p<0.02$, Figure 3.5).

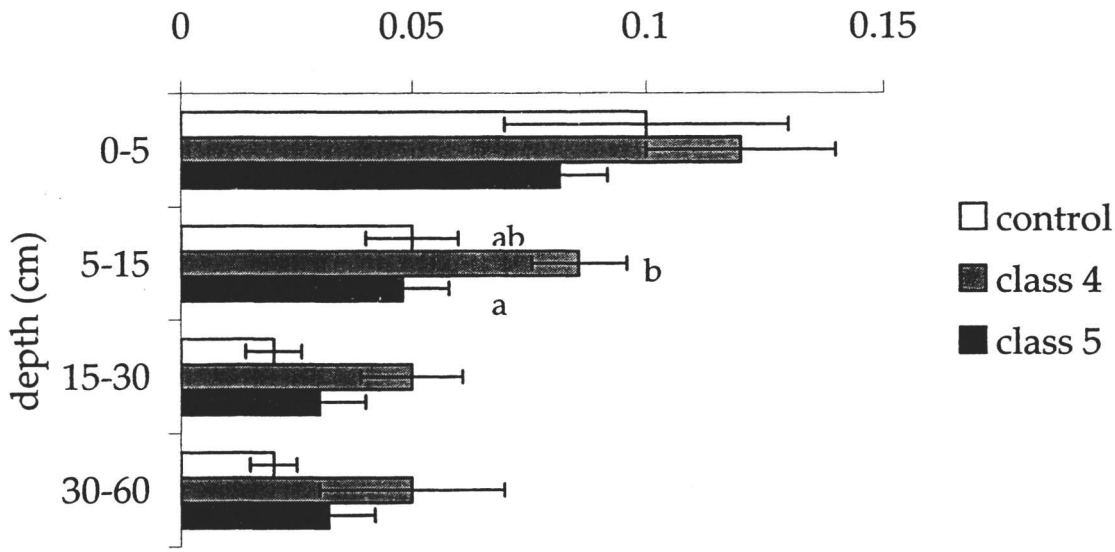


Figure 3.5. Mean water-soluble organic carbon in mg/ g soil from under CWD of decay class 4 and 5 and control soils to a depth of 60 cm. Bars represent one standard error.

There was no difference in labile phosphorus concentrations among decay classes and control soils (Figure 3.6). Resin P concentrations increased in 5-15 cm control soils but decreased with depth thereafter. Labile phosphorus extracted with sodium bicarbonate was not different among treatments and decreased with depth for all treatments as well (Figure 3.7).

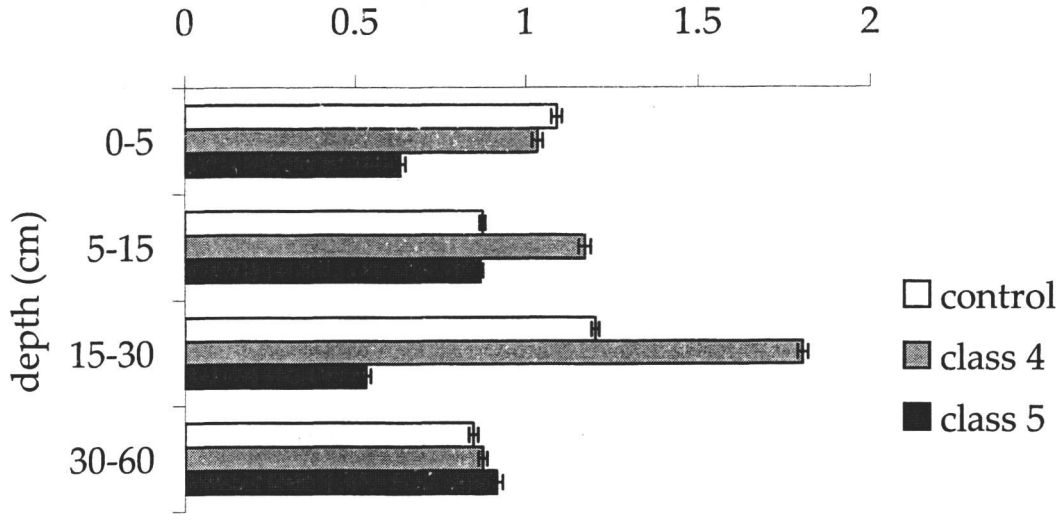


Figure 3.6. Mean resin phosphorus concentrations in ug P/ g soil from under class 4 and 5 CWD and from control soils. Bars represent one standard error.

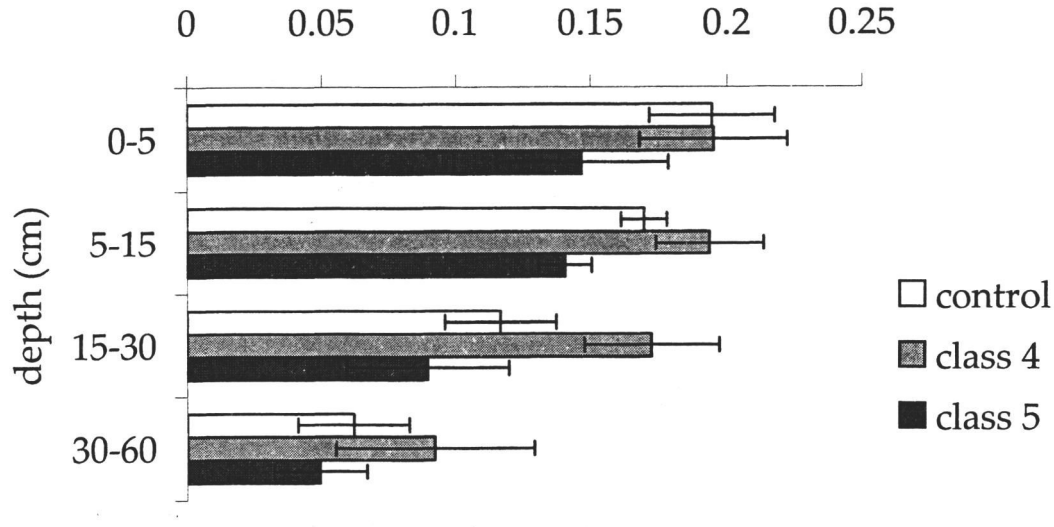


Figure 3.7. Mean sodium bicarbonate phosphorus concentrations in ug P/ g soil from under class 4 and 5 CWD and from control soils. Bars represent one standard error.

Gross $^{15}NH_4^+$ mineralization rates 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} lower 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). There was no difference in Beta-glucosidase, phosphatase activity and microbial biomass carbon among soils under CWD or control soils at any depth(Table 3.5).

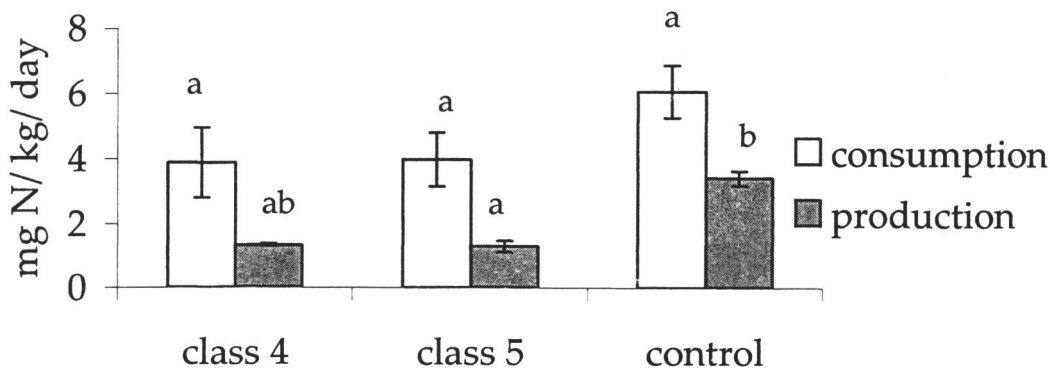


Figure 3.8. Mean ^{15}N-NH_4 gross production and consumption rates for soil under decay class 4 and 5 CWD and control soils. Bars represent one standard error.

	mg Microbial Biomass C/g	mg Microbial Biomass N/ g	Microbial biomass C:N ratio
control	0.671246 (0.061)	0.192226 (0.88)	6.7
class 4	0.610842 (0.104)	0.347861 (0.123)	4.7
class 5	0.552728 (0.112)	0.088072 (0.036)	9.3

Table 3.4. Mean microbial biomass carbon and nitrogen values for soils used for ^{15}N gross mineralizations. Data are means of five replicates in the fall of 2001. Standard errors are in parentheses.

season	depth and treatment	Microbial Biomass C	Beta-glucosidase activity	Phosphatase activity
Spring	0-5 cm Forest floor	0.51 (0.05)	11.91 (2.11)	42.98 (5.46)
	0-5 cm Class 4	0.41 (0.06)	7.34 (2.56)	36.85 (5.42)
	0-5 cm Class 5	0.38 (0.05)	6.57 (1.93)	31.22 (4.60)
	5-15 cm Forest floor	0.36 (0.06)	17.11 (2.83)	53.96 (6.94)
	5-15 cm class 4	0.29 (0.05)	15.75 (5.51)	36.21 (4.42)
	5-15 cm Class 5	0.25 (0.07)	19.05 (6.45)	52.53 (11.53)
Fall	0-5 cm Forest floor	0.24 (0.02)	10.75 (6.97)	36.82 (15.13)
	0-5 cm Class 4	0.16 (0.03)	9.27 (6.47)	34.68 (28.33)
	0-5 cm Class 5	0.17 (0.02)	5.15 (4.08)	17.52 (8.45)
	5-15 cm Forest floor	0.26 (0.12)	4.38 (2.05)	18.21 (5.44)
	5-15 cm Class 4	0.14 (0.03)	3.29 (2.12)	14.9 (11.42)
	5-15 cm Class 5	0.13 (0.03)	1.9 (0.93)	9.61 (3.15)

Table 3.5. Mean percent microbial biomass C and enzyme activity from 0-5 and 5-15 cm soils under class 4 and 5 coarse woody debris and from control soils. Standard errors are in parentheses.

Results from the Biolog plates indicate no clear differences in substrate specificity between soils under class 5 CWD and control soils from under the forest floor. 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 was attributable to seasonal rather than treatment differences.

Solution concentrations for different tree species

There were no significant differences among species for either TKN or DOC (Table 3.6). Calcium concentrations in *Thuja plicata* leachates were higher than other species in September of 1986 ($p<0.005$). *Thuja plicata* had higher concentrations of Mg in leachate than *Psuedotsuga menziesii* ($p<0.02$), but it was not statistically different than *Abies amabalis* or *Tsuga heterophilia* 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 Thuja plicata leachate were significantly lower than *Abies amabalis* and *Tsuga heterophilia,* but not *Psuedotsuga menziesii* ($p<0.03$). In February of 1987, concentrations of Mn in

Thuja plicata leachate were lower than *Tsuga heterophilia* ($p<0.05$) but were similar to concentrations of MN in *Psuedotsuga menziesii* and *Abies amabalis* leachates.

		Abies amabalis	*Psuedotsuga menziesii*	*Thuja plicata*	*Tsuga heterophilia*
Sept. 1986	Al	0.27 (0.17)	0.10 (0.00)	0.10 (0.00)	0.10 (0.03)
	Fe	0.05 (0.05)	0	0	0
	Ca	0.63 (0.08) a	0.55 (0.04) a	1.30 (0.29) b	0.43 (0.09) a
	K	6.17 (2.09)	2.67 (0.49)	3.00 (0.52)	3.83 (0.75)
	Mg	0.18 (0.03) ab	0.10 (0.00) b	0.30 (0.07) a	0.16 (0.03) ab
	Mn	0.08 (0.02) a	0.03 (0.02) ab	0.0 (0.00) b	0.08 (0.03) a
	P	1.48 (0.67)	0.4 (0.12)	0.38 (0.10)	0.87 (0.23)
	S	1.60 (0.26)	1.53 (0.30)	1.77 (047)	1.45 (0.43)
	DOC	18.04 (2.91)	18.81 (1.64)	19.74 (1.25)	19.22 (4.36)
	TKN	0.30 (0.03)	0.20 (0.02)	0.26 (0.02)	0.24 (0.03)
Feb. 1987	Al	0.09 (0.03)	0.25 (0.03)	0.15 (0.03)	0.25 (0.06)
	Fe	0.02 (0.01)	0.05 (0.01)	0.050 (0.01)	0.04 (0.01)
	Ca	0.73 (0.26)	1.15 (0.23)	2.24 (0.93)	0.95 (0.25)
	K	0.55 (0.27)	0.38 (0.17)	0.34 (0.18)	1.02 (0.49)
	Mg	0.14 (0.03)	0.19 (0.04)	0.36 (0.14)	0.26 (0.09)
	Mn	0.04 (0.01) ab	0.08 (0.02) ab	0.03 (0.01) b	0.16 (0.05) a
	P	1.46 (0.57)	0.73 (0.16)	0.73 (0.35)	1.47 (0.37)
	S	1.27 (0.43)	0.71 (0.23)	0.90 (0.25)	0.58 (0.12)
	DOC	40.72 (15.24)	32.42 (5.97)	26.08 (3.53)	33.17 (4.76)
	TKN	0.53 (0.21)	0.31 (0.03)	0.42 (0.10)	0.31 (0.04)

Table 3.6. Mean percent dissolved organic carbon, total Kjeldahl nitrogen, and cations from zero-tension lysimeters under class 1 species from four tree species. Standard errors are in parentheses. Letters following the mean indicate that the solutions are significantly different for that element at alpha=0.05.

There was no difference in ammonium or nitrate concentrations among species (Table 3.7).

		Abies amabalis	*Psuedotsuga menziesii*	*Thuja plicata*	*Tsuga heterophilia*
NH4	**Dec-90**	0.26 (0.05)	0.17 (0.02)	0.46 (0.29)	0.43 (0.11)
	Apr-91	0.97 (0.35)	1.24 (0.48)	0.70 (0.14)	1.19 (0.28)
	Oct-91	4.94 (1.40)	2.96 (1.16)	1.79 (0.75)	4.26 (2.40)
	Dec-91	0.62 (0.26)	0.64 (0.23)	0.99 (0.40)	0.80 (0.29)
	Apr-92	2.01 (0.49)	3.63 (1.17)	2.80 (0.89)	2.05 (0.85)
NO3	**Dec-90**	0.22 (0.04)	0.21 (0.04)	0.30 (0.08)	0.19 (0.04)
	Apr-91	0.07 (0.01)	0.10 (0.01)	0.14 (0.04)	0.14 (0.07)
	Oct-91	0.09 (0.01)	0.08 (0.01)	0.09 (0.02)	0.11 (0.02)
	Dec-91	0.16 (0.06)	0.10 (0.01)	0.09 (0.01)	0.17 (0.04)
	Apr-92	0.16 (0.06)	0.15 (0.04)	0.19 (0.05)	0.10 (0.02)

Table 3.7. Mean NH_4-N and NO_3^-N concentrations from zero-tension lysimeter solutions collected under four class 1 CWD species. Standard errors are in parentheses.

DISCUSSION

Dissolved organic carbon in CWD leachates was approximately three times higher in concentration than leachates from the forest floor. This indicates that unless there is a large amount of microbial degradation of DOC in mineral soils under CWD, there should be a very large input of C via sorption to the underlying soils. Even at 30 cm, solution from class 3 CWD had three times higher DOC concentrations than control soil solutions.

From our class 2 through class 4 lysimeter concentrations, we calculated the volume-weighted amount of C that could be lost from CWD via DOC leaching. For each age class, we calculated the leaching flux of DOC per year and multiplied that by the number of years that the log remains in the respective age class. We used the average residence time of a log in an age class as published by Sollins et al. (1987). The total amount of C lost from CWD as DOC in leachate over 87 years was 0.0045 Mg C m^{-2}.

In order to calculate the total C store of a log to compare with the amount of C lost via leaching, we used a class 1 log with an average diameter of 1 m (Sollins et al. 1987). If we assume a density of 0.45 g cm^{-3}, the total C stores of the log are 0.18 Mg c m^{-2}. If we further assume that 50% of the log C is lost by the time is reaches class 4, the total C lost via respiration or DOC leaching over 87

years is 0.09 Mg C m^{-2}. Comparing this number to the amount of C that we calculated was lost via DOC leaching, we find that the amount of C lost via DOC leaching only accounts for 5% of the total C lost from the bole. CWD leachate DOC concentrations are large when compared to forest floor leachates, but are only a minor proportion of the total amount of C lost from CWD. This suggests that most of the C leaves CWD as CO_2 via respiration, and potentially explains why we found no increase in total soil C under CWD compared to soil under the forest floor. Other researchers have found small, if any, increases in total soil C under CWD as well (Krzyszowska-Waitkus et al., in review, Kayahara et al. 1996, Busse (1994).

Class 3 CWD may have the most profound effect on soils because it had the largest concentrations of DOC in leachate and it has been found by other researchers to be the most active stage of decomposition (Krankina et al. 1999). 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. Perhaps 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 has 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 labile C compounds such as water soluble organic matter 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 labile C than we expected under CWD because we did see rates of beta-glucosidase activity that were comparable to the rates under the forest floor. Water-soluble organic carbon was higher under well-decayed 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. 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. Additionally, there were no noticeable differences in microbial 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, e.g., class 2 and 3 (Hicks 2000). After averaging the soil solutions over time 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.

Because of the high organic N inputs, we would expect 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 mineralized NH_4^+ was equally consumed and thus net mineralization rates remained similar among treatments.

There were few differences in leachate chemistry collected under class 1 logs from various species. This may be simply because the time period of most active leaching comes in later stages of decay. 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 less resistant to decay would release of C and nutrients more rapidly. 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 species least resistant to decay 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.

CONCLUSIONS

Studies of CWD decomposition, leaching and its effects on soils have suggested that CWD does affect soils moderately, but that the effect of CWD on soil depends on the species of CWD, the time at which the CWD is investigated and the ecosystem in which the CWD is located. In a chronosequence of Douglas-fir logs, 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 under CWD, but overall CWD did not increase total soil mineral C when compared to control soils. It may be however, that it is difficult to find control soils that have never been affected by coarse or fine woody debris over long-term pedogenesis. We were confident that our control soils showed no visual evidence of CWD to a depth of 60 cm and thus, that no CWD had influenced the control soils over the timescale of log decomposition. However, we have no idea what the ultimate fate of CWD is and thus control areas may not really be

controls after all. The possibility exists that CWD may have been present at one time or another during pedogenesis, but that the CWD has decomposed completely leaving only a signature that would therefore be indistinguishable from soils currently underlying CWD.

Although we had hypothesized that the wood imprint on soils would be spatially discrete and visible under the most highly decayed CWD, this study suggests that CWD may affect the underlying soil in two possible ways: 1) CWD has no long-term effect or 2) the effect of CWD is so long-term that no spatial affect is noticeable because all soils have been affected by CWD at one time or another. Therefore, no implications for management can be made from this study, i.e., it is not possible to directly infer that the results of this study indicate a cause and effect relationship between CWD and soil processes. Results from this study can only be used to discuss the spatial heterogeneity.

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CHAPTER 4: CONCLUSIONS

The results of this study, summarized in Figure 4.1, suggest that CWD: 1) does affect soil chemistry in the surface soils, but not in deeper soils or in the last stages of decay and 2) does not affect the aspects of soil biology that were examined beneath CWD compared to soils beneath the forest floor. 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 concentrations of DOC leaching from the Douglas-fir CWD are the highest.

Large inputs of DOC via leaching were slightly retained by the mineral soil under CWD, but overall CWD did not increase total soil mineral C when compared to control soils. It may be however, that it is difficult to find control soils that have never been affected by coarse or fine woody debris over long-term pedogenesis. We were confident that our control soils showed no visual evidence of CWD to a depth of 60 cm and thus, that no CWD had influenced the control soils over the timescale of log decomposition. However, we have no idea what the ultimate fate of CWD is and thus control areas may not really be

controls after all. The possibility exists that CWD may have been present at one time or another during pedogenesis, but that the CWD has decomposed completely leaving only a signature that would therefore be indistinguishable from soils currently underlying CWD.

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 the wood imprint on soils would be spatially discrete and visible under the most highly decayed CWD, this study suggests that CWD may affect the underlying soil in two possible ways: 1) CWD has no long-term effect or 2) the effect of CWD is so long-term that no spatial affect is noticeable because all soils have been affected by CWD at one time or another. 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.

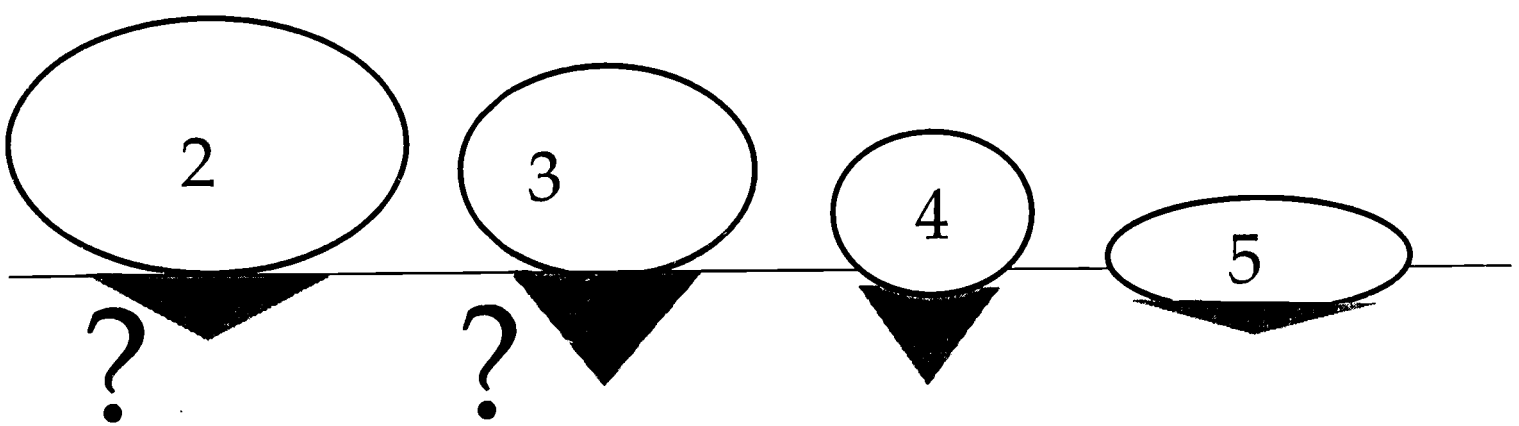

Figure 4.1. Visual representation of the results of this study contrasting soil underlying decomposing CWD of different decay classes and soil that has not been affected by CWD.

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