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Title: Soil Respiration, Carbon and Nitrogen Leaching, and Nitrogen Availability in Response to Harvest Intensity and Competing Vegetation Control in Douglas-fir (*Pseudotsuga menziesii*) forests of the Pacific Northwest

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Stephen H. Schoenholtz

Management practices following forest harvest can affect long-term soil productivity through alteration of soil carbon (C) and nitrogen (N) pools, but processes contributing to change are poorly understood. I assessed effects of three levels of logging-debris retention in combination with initial or annual applications of competing vegetation control (CVC) following forest harvesting on soil C flux, N leaching, foliar N of planted Douglas-fir, and changes in soil N and C pools for two years at two sites with contrasting soil properties. Soil C flux was lower when heavy amounts of logging debris were retained, due largely to lower bulk soil and microbial respiration as there was no difference in dissolved organic C (DOC) flux among logging-debris treatments. Increased soil C when heavy amounts of logging debris were retained at the site with lower initial soil C reflected the lower C flux, but soil C was increased at both sites when logging debris was removed, likely due to greater decomposition of belowground organic matter (OM). There was no difference in DOC leaching or soil C between CVC

treatments at either site, despite lower OM inputs to mineral soil with annual CVC. Higher bulk soil respiration in the initial CVC treatment indicated that OM inputs from competing vegetation were rapidly consumed, and contributed little to mineral soil C. The most pronounced effects on N leaching and foliar N were associated with annual CVC, which increased Douglas-fir foliar N at both sites, and total N leaching below the rooting zone at the high-N site. However, estimated mass of leached N was small relative to the site soil N pool, and it is unlikely that the loss will negatively affect soil productivity. Logging-debris retention had little influence on Douglas-fir foliar status or N leaching, but soil N was higher at the end of the experiment when heavy amounts of logging debris were retained at the low-N site. There appears to be small potential for logging-debris removal and annual CVC to reduce soil productivity at these sites after harvesting, but logging-debris retention may improve soil productivity, particularly at sites with low initial pools of C and N.

Soil Respiration, Carbon and Nitrogen Leaching, and Nitrogen Availability in Response to Harvest Intensity and Competing Vegetation Control in Douglas-fir (*Pseudotsuga menziesii*) forests of the Pacific Northwest

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APPROVED:

Major Professor, representing Forest Engineering

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Dean of the 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.

Robert A. Slesak, Author

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Dedication

For my daughter, Maya

I hope you come to love forests as much as I do

Soil Respiration, Carbon and Nitrogen Leaching, and Nitrogen Availability in Response to Harvest Intensity and Competing Vegetation Control in Douglas-fir (*Pseudotsuga menziesii*) forests of the Pacific Northwest

Chapter 1

Introduction

Management of forests for timber production is becoming progressively more intensive due to increased demand for wood coupled with a shrinking land base available for production (FAO, 2006). Advances in technology and increasing use of biomass for energy have amplified utilization of smaller diameter trees, making shorter rotations an increasingly common practice. As the limited land available for timber production is subjected to more harvests over shorter time periods the potential for decreased site productivity over the long-term is an increasing concern. Soil is one of the key factors of site productivity, hence many have cautioned on the need to maintain soil quality if intensive forest management is to be sustainable (Nambiar, 1999; Jurgenson et al., 1997; Powers et al., 1990).

Although many soil properties susceptible to management–induced change have potential to alter soil quality (Schoenholtz et al., 2000), it is commonly believed that reduction in soil organic matter (SOM, with soil carbon (C) as a surrogate) has the greatest potential to alter soil productivity across a wide range of site conditions (e.g. regional climate, soil characteristics, relief, etc.). Soil C is a fundamental soil quality property due to its effect on nutrient cycling, biological activity, soil structure, and water holding capacity (Powers et al. 1990). Soil nitrogen (N) is also of primary interest for soil quality as N is the most common nutrient limiting to growth in terrestrial ecosystems (Johnson, 1992; Keeney, 1980). Almost all soil N is found within SOM, hence maintenance of soil C pools should result in a concurrent maintenance of soil N pools.

Forest harvest has potential to reduce soil C and N through removal of biomass and alteration of the soil environment. In intact forests, change in soil C is small due to a balance between organic matter (OM) inputs to soil and C losses arising from microbial respiration (i.e. the steady-state concept). Biomass removal at harvest can potentially reduce OM inputs to soil, which may cause a reduction in soil C if microbial respiration is maintained at the pre-harvest rate. However, microbial respiration may actually increase following harvesting (Edwards and Ross-Todd 1983), as increased soil temperatures following removal of the forest canopy are conducive to increased microbial activity (e.g. Lloyd and Taylor, 1994) as long as soil moisture is not limiting (Rey et al., 2002). The combination of reduced OM inputs and potentially increased microbial respiration could result in reduced soil C following harvest. In the case of N, greater microbial activity following harvesting can increase net N mineralization (Prescott et al. 1997, Vitousek et al., 1992), which may result in N loss via leaching if vegetative demand is lower than N supply (Vitousek and Matson, 1985).

Management practices following harvesting which promote OM inputs to soil and reduce changes in the soil environment have potential to reduce any C or N loss. Logging-debris retention has been recognized as one management practice likely to maintain or enhance SOM pools (e.g. Powers et al. 1990; Johnson et al. 2002), as it can offset reductions in litter inputs following harvest. Logging debris also shades the soil surface, mitigating changes in soil temperature and moisture (Devine and Harrington, 2007; Roberts et al., 2005), thereby potentially inhibiting any increase in microbial respiration following harvesting. Although there is clear potential for logging debris to maintain or enhance SOM following harvesting, experimental results are somewhat conflicting. The meta-analysis performed by Johnson and Curtis (2001) indicated an increase in soil C and N when logging debris was retained, with the effect largely restricted to coniferous forests. In contrast, the 10-yr summary results from 27 installations of the Long Term Soil Productivity network presented by Powers et al. (2005) suggest no effect of logging debris on soil C when the forest floor is retained. Other studies have concluded that debris retention has no effect on soil C over periods as long as 15 yr (Johnson et al. 2002, Olsson et al., 1996).

Studies have shown that at least a portion of debris C enters the soil (Mattson et al., 1987; Piirainen et al., 2002; Qualls et al., 2000; Robertson et al., 2000), which leads to the question of where does this C go if no change in soil C occurs? Obvious modes include losses due to leaching of dissolved organic C (DOC) and increased microbial respiration of soil C. Both of these C fluxes are dependent on biological and physical processes that are altered by the presence of logging debris (Devine and Harrington, 2007; Busse et al., 2006; Belleau et al., 2006; Roberts et al., 2005). Although some studies have measured DOC leaching following harvesting (Qualls et al., 2000; Piirainen et al., 2002) and attempted to assess changes in microbial respiration through measurement of bulk soil respiration (Carter et al., 2002; Toland and Zak, 1994;

Hendrickson et al., 1989), no studies have concurrently assessed DOC and microbial respiration fluxes in response to experimental manipulations of logging debris. Consequently, it is unclear if increased C flux is responsible for the paucity of detectable changes in soil C pools in response to surface placement of logging debris. An alternative possibility to increased C flux is that logging debris *does* actually increase soil C, but the change may be too small to be statistically detectable with the large background variability inherent to soil (Homann et al., 2001; Rothe et al., 2002) and additional variability associated with non-uniform application of logging-debris treatments (Meehan, 2006; Eisenbies et al., 2005).

Competing vegetation control (CVC) following harvesting also has strong potential to alter soil C and N directly, or indirectly through alterations of the effects of logging debris. Competing vegetation control is a common site preparation practice following harvesting to increase crop tree survival and growth in the initial years following planting (Harrington, 2006; Roberts et al., 2005), but CVC may also cause concurrent reductions in soil C and N pools (Shan et al., 2001). Vegetation is an important factor controlling N retention (Marks and Bormann, 1972; Vitousek et al., 1979), and several studies have documented elevated nitrate leaching following CVC (Briggs et al., 2000; Smethurst and Nambiar, 1995; Vitousek and Matson, 1985). In addition, inputs of above- and belowground OM are reduced following CVC, potentially reducing soil C if heterotrophic organisms consume more pre-existing SOM when availability of recently-fixed OM is low. A few studies from the southeastern U.S. have shown reductions in soil C following annual CVC (Echeverria et al., 2004; Miller et al., 2006), demonstrating the potential for reduced soil C when this practice is employed. However, those studies occurred at sites where initial soil C pools were low (< 20 g C kg^{-1}) and it is not clear if those results are applicable to soils with high C concentrations or in regions with different climate. More studies across a range of site conditions are needed to elucidate the main and interactive effects of CVC on soil C and N pools.

Ultimately, the goal in assessing the effect of management practices on soil C and N is to determine the potential for reductions in soil productivity over the long-term. Such assessment is inherently flawed if concurrent assessment of crop tree growth in response to the associated management practices is not evaluated. If the management practice results in increased soil quality, but decreases resource availability such that tree growth is negatively impacted, then the practice would not be suitable for sustainable forest management (i.e. non-declining yield). Alternatively, if the management practice maintains or improves soil quality, while increasing tree growth (assuming the goal of the management practice is to increase growth), then the practice may be suitable for sustainable forest management.

Resources necessary for plant growth broadly include light, water, and nutrients, but light is generally not limiting to tree growth following forest harvesting (Harrington, 2006). Of all the essential nutrients, N is commonly the most limiting to tree growth (Johnson, 1992; Keeney, 1980). Presence of competing vegetation strongly influences water and N available for crop tree uptake (Rosner and Rose, 2006; Smethurst and Nambiar, 1989), but the presence of logging debris may modify the response (Roberts et al., 2005). Experimental studies have shown the effect of logging debris on total soil N and net N mineralization to be positive (Chen and Xu, 2005; Piatek and Allen, 1999), negative (Blumfield and Xu, 2003; Smethurst and Nambiar, 1990b), or neutral (Carter et al., 2002; Li et al., 2003; Mendham et al., 2003). The effect of these changes on tree N acquisition is unclear, as total soil N and net N mineralization are at best indices of N availability (Schimel and Bennett, 2004), and provide no information regarding actual plant N acquisition. The interactive effect of logging debris and CVC on tree N acquisition is even less clear, as only a few studies have employed experimental manipulations designed to separate the influence of each practice (e.g. Roberts et al., 2005).

Study context

The Long Term Soil Productivity (LTSP) program is a national research trial under the jurisdiction of the U.S. Forest Service with the goal of determining the long-term role of soil disturbance at time of harvest on forest productivity. Started in 1989, the network of research sites includes over 100 “core” and affiliate sites that cover a wide range of site conditions and climate across North America (Powers, 2006). The central concept underlying the LTSP program is that site productivity is strongly controlled by soil processes that are readily affected by management activities (Powers, 2006). Two soil properties most likely to limit critical soil processes that may be affected by management are soil porosity and SOM (Powers et al., 1990). Consequently, the “core” LTSP sites have experimental manipulations that combine increasing OM removal with increasing soil compaction to assess how increasing management intensity influences soil porosity and SOM pools. Organic matter treatments ranged from bole-only harvests where all non-merchantable biomass material was retained on site, to extreme OM removal where all non-merchantable OM material and the forest floor were removed.

One criticism of the core LTSP design was that it only highlighted the potential for negative impacts from forest management, and did not include any experiments designed to improve site quality (Powers 2006). In particular, the compaction treatments were considered unrealistic of operational settings. Consequently, affiliate sites were invited to join the network with modified designs as long as experimental manipulation included the bole-only harvest with no compaction. Of interest to this study, was the establishment of the Fall River LTSP affiliate located in western Washington, which was designed to address the role of more contemporary forest management practices (fertilization, OM removal, competing vegetation control) on soil porosity, SOM, and tree growth in the Pacific Northwest (PNW) (Ares et al., 2007). The Douglas-fir (*Pseudotsuga menziesii*) forests of the PNW are some of the most productive in the world, and the region provides a significant portion of the wood biomass for forest products in the U.S. (Adams et al., 2006). Although the LTSP network covers most of the forested regions where active forest management is practiced, no core or affiliate sites were located in the PNW prior to establishment of the Fall River study (Ares et al., 2007).

Because the Fall River study design was only established on a single highly productive site, and there was interest in determining how these management practices influenced soil productivity across a range of conditions in the PNW, two additional affiliate LTSP sites were established near Matlock, WA on the Olympic Peninsula in western WA, and in the western Cascades near Molalla, OR. Research objectives at the Matlock and Molalla sites are narrower than that of Fall River, and are focused on the influence of OM management and CVC following harvesting on soil properties, nutrient cycling, and Douglas-fir growth. These research objectives are the foundation for research presented in this dissertation.

Objectives and Chapter Descriptions

The broad objectives of my research were to determine the effect of increasing logging-debris retention and CVC (initial or annual applications) on soil C and N pools, C and N fluxes, and the effect of these practices on Douglas-fir N acquisition at two sites that differ strongly in annual precipitation and soil characteristics (Table 1.1). Each of the main research chapters is written in manuscript style, with the intent of submission to peer-reviewed journals as presented.

Chapter 2 focuses on the effect of logging-debris retention and CVC on changes in soil C, *in situ* measures of bulk soil and microbial respiration, and the factors influencing soil respiration at each site with emphasis on the role of soil temperature. Results from controlled laboratory incubations are also presented to assess the contribution of modified substrate on microbial respiration when soil temperature and moisture are held constant. The specific objectives of the research presented in this chapter are to determine 1) if logging-debris retention or CVC treatments modify soil C, 2) if varying amounts of logging-debris retention modify microbial respiration and how the relationship varies in the presence and absence of competing vegetation, 3) the contribution of recently-fixed C to bulk soil respiration when only initial CVC is applied, 4) causal mechanisms for changes in microbial and bulk soil respiration, and 5) if changes in microbial respiration could explain any changes in soil C.

Chapter 3 focuses on the effect of logging-debris retention and CVC on the leaching of DOC and soluble N forms (nitrate, ammonium, dissolved organic N (DON)). A water balance model was used to estimate flux of these elements to provide a context for potential estimated losses below the rooting zone. The specific objectives of this chapter are to determine treatments effects on 1) DOC and N (nitrate, ammonium, DON) concentrations and flux below the rooting zone to determine the potential for leaching, and 2) production of water-leachable DOC and N under controlled laboratory incubations to infer the causal mechanisms contributing to the results observed *in situ*.

Chapter 4 focuses on the effect of logging-debris retention and CVC on N supply and acquisition by young, planted Douglas-fir crop trees. Crop tree N acquisition was assessed with the use of foliar analysis combined with vector analysis for interpretation. Treatment effects on total soil N and soil water content are also presented in this chapter. The specific objectives are to assess treatment effects on 1) foliar N status in Douglas-fir seedlings for three years following planting, 2) soil available N (KCl-extractable) and potential net N mineralization to determine if there were relationships between these variables and the foliar response, and 3) soil N to assess the potential for longer-term effects on N supply.

Chapter 5 was written as a synthesis of main research findings within Chapters 2 through 4. In particular, findings from Chapters 2 and 3 are evaluated in combination to determine the potential for C flux following application of the logging-debris and CVC treatments used in this study.

Literature Review

The following review includes literature from a review completed for my research proposal, with additional literature that was used when writing the research chapters. Much of this literature review is presented within the introduction of each chapter, but in a more narrow and concise form appropriate for each chapter's focus.

Soil Carbon

The presence of logging debris following harvesting has potential to influence soil C cycling in multiple ways. Debris can increase soil C pools through inputs of dissolved DOC or incorporation of fractured and sloughed organic material into the soil via decompositon and bioturbation. Debris can also modify the soil microsite environment, causing an indirect effect on soil C through modification of microbial activity. Inputs have potential to increase soil mineral C through sorption to mineral surfaces or transformation to stable humic forms. However, inputs also have potential to be lost from the soil profile due to leaching of DOC and /or increased rates of soil respiration (Figure 1.1).

Soil respiration

Ecosystem C flux is dominated by soil respiration (Valentini et al., 2000) which is a large C flux to the atmosphere (Reichstein et al., 2003). Bulk soil respiration (BR) arises from heterotrophic consumption of SOM (primarily microbial respiration (MR)) but also includes autotrophic sources from root respiration. The relative contribution of MR and autotrophic respiration to BR varies both temporally and spatially. In a detailed review, Hanson et al. (2000) reported a range of 10% to 90% for autotrophic respiration at forested sites with an average value of about half BR. Although BR measures are useful in estimating the contribution of soil CO_2 efflux to atmospheric C, they have limited inference for changes in soil C as root respiration arises from a C source external to the soil. Inference to soil C change requires assessment of MR in combination with estimates of OM inputs.

Methods to separate MR from BR include direct *in situ* measurement of root respiration with cuvettes, measurement of respiration from excised roots following collection, and *in situ* measurement of MR via trenching to remove root respiration (Hanson et al., 2000). Kelting et al. (1998) note the potential for artifacts associated with each method, but concluded that the excised root method was most appropriate due in part to the ease of collecting and measuring multiple samples. In studies where the primary interest is assessment of MR over time, then the trenching approach is probably more appropriate as it estimates MR under *in situ* environmental conditions with minimal soil disturbance. Trenching to estimate MR has traditionally been accomplished by trenching around an area and removing all vegetation within it (Bowden et al., 1993, Sulzman et al., 2005). Respiration measurements are then made within and outside the trenched area to determine the contribution of MR to BR (Hanson et al., 2000). Others have used PVC and steel cores (Dilustro et al., 2005; Kelting et al., 1998) as a modified trench method where the action of driving the core into the ground severs all roots and prohibits future root in-growth.

Many studies have shown a strong influence of soil temperature on MR (Reichstein et al., 2003; Kelliher et al,. 2004). The relationship has generally been found to be non-linear, where respiration increases exponentially with increasing soil temperature (e.g. Lloyd and Taylor, 1994). Moisture has also been found to have strong control on soil respiration, but the relationship is similar to Liebigs Law of the Minimum where soil moisture limits MR at high and low water content (Qi and Xu, 2001). Logging-debris retention has been shown to reduce soil temperature (Devine and Harrington, 2007; Roberts et al., 2005) and increase soil moisture (Devine and Harrington, 2007) potentially modifying MR. However the net effect of these environmental effects is unclear as reduced soil temperature would be expected to reduce MR, but increased SWC may result in greater MR during times of moisture limitation where MR would be otherwise inhibited (Rey et al., 2002). Microbial respiration may also increase due to greater substrate availability when logging debris is retained (Epron et al., 2006), possibly offsetting any effect of reduced temperature.

Although there is clear potential for logging-debris retention to modify MR, experimental studies demonstrating any effect are scant. A few studies have attempted to assess changes in MR by measuring BR, assuming that any contribution of root respiration to BR is low following harvesting. Hendrickson et al. (1989) found no difference in BR between a whole-tree and bole-only harvest of mixed stands in the Great Lakes region U.S.A, but differences in the amount of live vegetation between treatments and the presence of aspen (*Populous tremuloloides*) sprouts likely confounded the response. Similarly, Carter et al. (2002) found no difference in BR between bole-only and whole-tree-plus-forest-floor-removal harvests one year after harvest of a loblolly pine (*Pinus taeda*) stand in Louisiana when competing vegetation was controlled, but there was also no difference in soil temperature between treatments. In a controlled laboratory experiment, Edwards and Ross-Todd (1983) observed higher MR rates from a whole-tree harvest compared to a bole-only harvest, but they incubated soils at field temperatures so it is not clear if the effect was partly due to differences in substrate. In the same study, *in situ* measurements indicated greater BR from the bole-only harvest than the whole-tree harvest, which may indicate substrate-induced changes in MR, but vegetation re-growth was vigorous and the effect may have been due to differences in vegetation abundance between treatments.

Competing vegetation control has been shown to modify the soil environment (Gurlevik et al., 2004; Roberts et al., 2005), inputs of OM to both above- and belowground pools (Shan et al., 2001), and microbial biomass and population structure (Busse et al., 2001; Li et al., 2004). All of these factors influence MR, but the net effect on MR following CVC is unclear. Shan et al. (2001) observed a large decrease in BR when CVC was controlled, possibly indicating a reduction in MR. However, given the large contribution of root respiration to BR (Hanson et al., 2000), the reduction observed by Shan et al. (2001) may have been due to reduced root respiration rather than MR. There are no other studies that I am aware of that have assessed the effect of CVC on MR.

DOC leaching

Inputs of C to soil as DOC are important due to the role of DOC in soil formation, nutrient cycling, and microbial activity (McDowell, 2003). Fluxes of DOC to soil are dependent on precipitation volume, precipitation chemistry, and C source quality (i.e. amount of soluble C substrate) from throughfall, but are additionally dependent on microbial community structure and activity for forest floor inputs to mineral soil. Dissolved organic carbon is generally considered a labile C source that is either readily metabolized by heterotrophic organisms (McDowell, 2003) or is effectively retained within the soil profile by physical and chemical processes (Qualls et al., 2000). Leaching losses of DOC are generally very small compared to total soil C pools (Kalbitz et al., 2000), and likely have little influence on soil C stocks compared to the large C fluxes arising from soil respiration (Valentini et al., 2000).

A number of studies have documented increased C inputs to mineral soil as DOC following logging-debris retention (Mattson et al., 1987; Qualls et al., 2000; Robertson et al., 2000). Qualls et al. (2000) concluded that DOC inputs from logging debris following harvesting of a deciduous forest in the southeastern U.S. were effectively retained in the mineral soil, and a similar conclusion was reached by Piirainen et al. (2002) following harvest of a coniferous forest in Finland. Others have noted no effect of harvesting on DOC (McDowell and Likens, 1988), or a general decrease in DOC export (Meyer and Tate, 1983). In the Qualls et al. (2000) study, adsorption to mineral surfaces was proposed as the primary retention mechanism, but microbial consumption may have also contributed to the decrease in DOC. These studies generally indicate low potential for

DOC leaching following logging-debris retention, but it is possible that response will differ with site-specific factors. In addition, the flux must be quantified to clarify the end fate of C inputs from logging debris.

Competing vegetation control also has potential to alter fluxes of DOC due to a decrease in recently-fixed OM inputs when competing vegetation is absent. Giesler et al. (2007) measured a 40% reduction in DOC concentration following tree girdling, which they attributed to reduced transport of photosynthate to roots. Similarly, in a column study Uselman et al. (2007) demonstrated that DOC leaching at 50 cm depth was derived largely from root litter rather than aboveground inputs. Reduced root inputs when CVC is applied could cause a concurrent reduction in DOC leaching. However, CVC also modifies the soil environment (Gurlevik et al., 2004; Roberts et al., 2005) and increases soil water flux through a reduction in evapotranspiration, so the net effect of CVC on DOC flux could be to increase C loss from soil. Losses are likely to be small relative to total soil C, but determination of any change in DOC flux will help to clarify the processes contributing to soil C change (if any) following CVC.

Nitrogen leaching

Several studies have shown increased nitrate leaching following harvesting (Vitousek et al., 1997; Marks and Bormann, 1972), which can be attributed to greater N availability when vegetative N demand is low. Logging-debris retention may modify nitrate leaching, as debris can be either a source or sink for available N after harvesting. Strahm et al. (2005) observed significantly greater nitrate leaching in a bole-only harvest compared to a whole-tree harvest at a highly productive site in southwestern Washington, a response that has been observed in the Great Lakes region as well (Hendrickson et al., 1989). In contrast, both Vitousek and Matson (1985) and Carlyle et al. (1998) found decreased nitrate leaching when logging debris was retained due to an increase in microbial immobilization (Vitousek and Matson, 1985). Mann et al. (1988) found no discernable effect of varying logging-debris retention following harvesting on N leaching at l1 sites across the conterminous U.S.

Nitrogen leaching following harvesting is generally dominated by inorganic N forms (primarily nitrate), but there is also potential for increased N loss as DON (Sollins and McCorison, 1981). Carlyle et al. (1998) found a 25-30% reduction in DON leaching when logging debris was retained following harvesting in southeastern Australia, but Strahm et al. (2005) observed no effect of logging-debris retention on DON following harvest at a productive site in southwestern Washington. Dissolved organic N comprised 15-45% of total N at the Australia site, but only 2-5% of total N at the Washington site. Soil N concentration at the Australia site was low (sandy soil, 1.1 g N kg^{-1} at 0-15 cm depth), but that at the Washington site was high (silt loam soil, 4.6 g N kg^{-1} at 0-12 cm depth), possibly indicating that the response may be a function of pre-harvest soil N pools. However, the differing response could be due to any number of factors, as there is generally a poor understanding of processes that govern DON production and loss from soil (Kalbitz et al., 2000; McDowell, 2003).

Vegetation is an important factor controlling N retention following harvesting (Marks and Bormann, 1972; Vitousek et al., 1979), and several studies have documented elevated nitrate leaching following CVC (Briggs et al., 2000; Smethurst and Nambiar, 1995; Vitousek and Matson, 1985). Increase in nitrate leaching has largely been attributed to reduced vegetative uptake, but a reduction in OM inputs to soil and subsequent reduction in microbial immobilization has also been shown to contribute in some regions (e.g. Vitousek and Matson 1985). Regardless, CVC has consistently been shown to increase N leaching following harvesting, which could reduce soil productivity if losses are large, initial soil N pools are small, or inputs over the course of a rotation are insufficient to offset losses. Strahm et al. (2005) found that estimated N loss via leaching when CVC was applied was a small portion of total soil N (<1.5%), and proportional to the amount of OM retained following harvesting. It is possible that N losses are small relative to total soil N pools, but more studies are needed across a range of site conditions to determine if the response varies.

Nitrogen Availability

Logging debris, particularly foliage which has a high N concentration relative to woody debris, can potentially increase plant-available N following harvesting (Piatek and Allen, 1999). However, decomposition of logging-debris with high C:N has also been shown to immobilize N (Carlyle et al., 1998; Laiho and Prescott, 1999; Palviainen et al., 2004), possibly reducing the plant-available N pool. Logging debris also modifies the soil environment, generally reducing soil temperature and either reducing or increasing soil moisture (Devine and Harrington, 2007; McInnis and Roberts, 1995). Soil temperature and moisture are known to have large influence on N mineralization rates

(Knoepp and Swank, 2002; Zak et al., 1999), and changes in the soil environment following logging-debris retention could either be conducive to, or impede net N mineralization. Even if logging debris modifies N supply, the change may be inconsequential to early tree growth (Smethurst and Nambiar, 1990b) as N mineralization generally increases following harvesting such that plant demand is less than N supply in some situations (Prescott et al., 1997; Vitousek et al., 1992).

Presence and abundance of competing vegetation will have strong control on crop tree N acquisition following harvest (Harrington, 2006; Roberts et al., 2005) as competing vegetation is capable of utilizing substantial amounts of N (Woods et al., 1992). The effect of CVC on N supply (i.e. mineralization) is less clear, as some studies have shown either an increase in net N mineralization (Li et al., 2003; Vitousek et al., 1992) or no effect (Lister et al., 2004; Meehan, 2006) in the initial years after harvest. Increased N mineralization could result from a reduction in immobilization, which can be reduced when OM inputs are low (Vitousek et al., 1992). There is also potential for a reduction in N mineralization following CVC (Echeverria et al., 2004), and reductions in total soil N (Miller et al., 2006; Busse et al. 1996).

Estimation of available soil N for tree growth is generally done directly with soil assays or indirectly with foliar analysis. The soil assay approach includes measures of mineralizable N (usually separated into ammonification and nitrification) with aerobic or anaerobic incubations that are performed either *in situ* or in the laboratory. Potentially mineralizable N has been correlated to tree growth for both aerobic (Hart *et al.*, 1986; Maimone *et al*., 1991) and anaerobic methods (Powers, 1980). Other studies did not find such correlation (Hart *et al.*, 1986; Lea and Ballard, 1982; Timmer and Ray, 1988; Weetman *et al.*, 1992). Some investigators (Keeney, 1980; Lea and Ballard, 1982; Powers, 1984) have suggested soil nutrient spatial variability as the primary cause of relatively low correlation of soil chemical characteristics with tree growth. However, even if soil variability could be reduced, soil assays are at best indices of N availability (Schimel and Bennett, 2004). Measures of N supply may be useful to assess factors contributing to N available for tree uptake, but if actual uptake is to be assessed, a more direct measure of tree nutritional status is needed.

Foliar N analysis is a measure of tree N nutritional status that indirectly reflects N pools available for plant growth (van den Driessche, 1974). Trees act as phytometers, integrating the myriad effects of site. Foliar N concentrations reflect how successful a tree is at competing for available N under given site conditions, rather than attempting to quantify all the potential competitive factors limiting N to trees (e.g. microbial immobilization, weed competition, etc.). Foliar N concentrations by themselves may be misleading when evaluating treatment effects on available N. If factors other than N are limiting to tree growth (e.g. light, water, other nutrients) then foliar N concentration will be a function of that factor rather than available N. Roberts et al. (2005) observed no significant difference in Douglas-fir foliar N concentrations following a bole-only harvest compared with a total-tree harvest, but recognized that a compensatory effect of soil moisture may have offset changes in available N. Some of the problems with N concentration in foliage can be overcome by expressing foliar N as absolute N content to provide indication of the amount of N that a tree acquired.

Vector analysis (Timmer and Stone, 1978) is a graphical diagnostic technique that simultaneously displays response of three variables (needle mass, needle nutrient concentration, needle nutrient content) for a treatment relative to an untreated control. The directional shift of the mass-concentration-content vector between control and treated trees differentiates among alternative treatment responses (i.e. N dilution, luxury consumption, N accumulation, N concentration). Although vector analysis has been used primarily in the past to assess response to fertilization (Ngono and Fisher, 2001; Timmer and Morrow, 1984; Weetman et al., 1988), it is a viable technique to assess any management practice including vegetation competition analysis (Imo and Timmer, 1998), and logging-debris effects on tree growth (Proe et al., 1999).

Tables

Table 1.1. Site characteristics and select pre-treatment soil properties to a depth of 30 cm for study sites near Matlock, WA, and Molalla, OR.

Characteristic or property	Matlock	Molalla
Location (Latitude, Longitude)	47.206 °N, 123.442 °W	45.196 °N, 122.285 °W
Elevation (m)	118	449
Mean annual temperature (°C)	10.7	11.2
Mean annual precipitation (mm)[1]	2,400	1,600
Site index$_{50\ yr}$ (m)	34	37
Soil Texture (% sand/silt/clay)	65 / 14 / 21	37 / 34 / 29
Bulk density (Mg m^{-3})	1.45 (0.05) [2]	0.98 (0.02)
Coarse fragments by mass (%)	67.6 (1.3)	37.7 (2.2)
Total soil N (kg ha^{-1})	2,246 (88)	4,338 (173)
Total soil C (Mg ha^{-1})	66.5 (3.6)	102.2 (4.7)

[1]Precipitation was calculated from the PRISM model for period 1950-2005, (http://prism.oregonstate.edu)
[2] Standard error in parenthesis, n=8 for bulk density at Matlock, n=24 for all others).

Figures

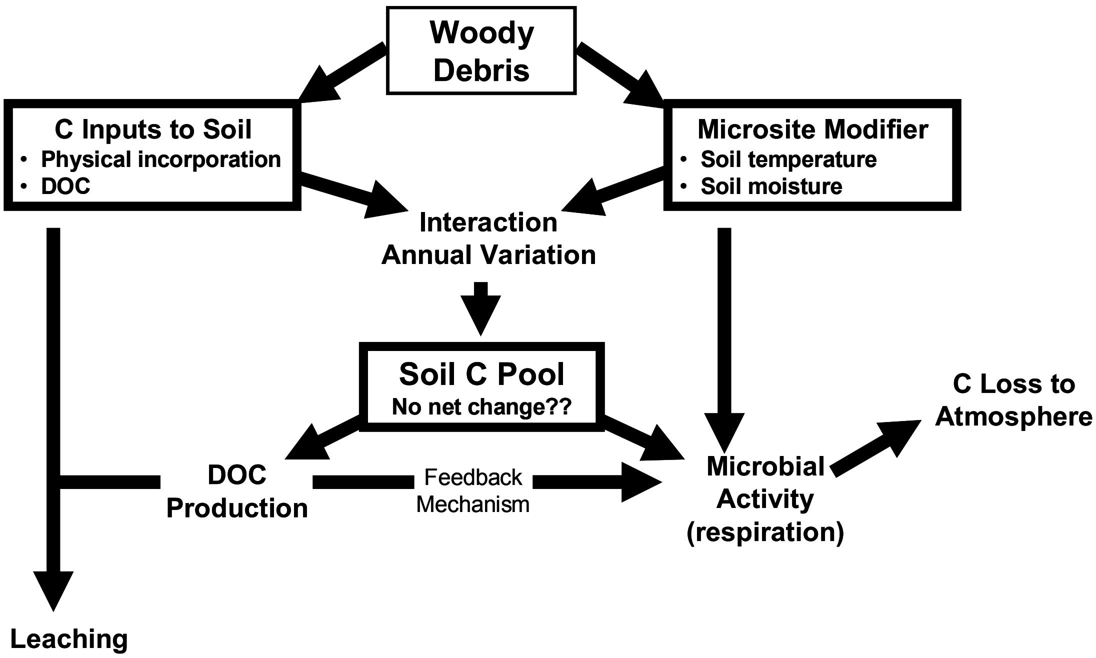


Figure 1.1. Conceptual diagram for potential effect of logging debris on soil C pools and fluxes.

Chapter 2

Response of Soil Respiration and Carbon to Logging-debris Retention and Competing Vegetation Control Following Clearcut Harvest in the Pacific Northwest.

Abstract

Management practices following forest harvest have potential to alter soil carbon (C) pools, but relatively few studies have assessed the influence of such practices on C flux arising from soil respiration. I examined the influence of varying amounts of logging-debris retention (0%, 40%, or 80% coverage) and competing vegetation control (CVC – initial or annual applications) on *in situ* bulk soil respiration (BR), microbial respiration (MR) and soil C at two Douglas-fir (*Pseudotsuga menziesi*) sites in the Pacific Northwest. Monthly measurements of BR and MR were initiated in September, 2005 (two years post-harvest) for a period of two years. Annual CVC decreased BR in most months of the growing season (April to Sept.) at both sites, which was attributed to reduced root respiration and organic matter inputs from competing vegetation. Higher soil temperature with annual CVC at one of the sites may have increased MR, but no difference in soil C between CVC treatments indicated any increase was small. A moisture limitation to BR was evident in 2006 with initial CVC, but there was little limitation in 2007 resulting in much greater BR. Logging-debris retention reduced BR, MR, and soil temperature at both sites, but the significance and magnitude of difference was variable. A soil temperature function explained between 44% and 76% of the

variation in MR and BR, but there was no effect of reduced temperature on MR in the 40% coverage, possibly due to an offset arising from favorable soil environmental conditions other than temperature that are conducive to microbial activity. A set of companion laboratory incubations indicated little difference in substrate availability among treatments, but higher MR in the 0% coverage at one incubation period suggested a transient increase in substrate availability, likely due to greater decomposition of roots from the previous stand under the 0% logging-debris coverage treatment. Soil C was significantly higher in the 80% coverage at the site which had lower initial soil C, indicating a beneficial effect of logging-debris retention under certain conditions. A significant increase in soil C at both sites when logging-debris was removed indicated belowground decomposition of OM was an important process contributing to soil C following harvesting in this region. The results suggest that root decomposition following harvesting causes an increase in soil C, which is dependent on the magnitude of logging-debris retention, its influence on the soil environment, and ultimately the microbial response. Stability of the C fraction contributing to increased soil C is unknown, and further study is warranted to determine if these increases in soil C are transient or long-term.

Introduction

Forest management is becoming increasingly intensive due to greater demand for wood products and a reduction in land available for production (FAO, 2006), and some have cautioned regarding the potential for a reduction in soil productivity and subsequent declines in forest productivity over the long-term (e.g. Fox, 2000; Nambiar, 1996). Soil organic matter (SOM) is recognized as a critical soil property controlling soil productivity because of its beneficial effects on aeration, water holding capacity, and nutrient cycling, and it has been argued that a reduction in SOM (or soil C as a surrogate) has the greatest potential to reduce soil productivity in the long-term (Jurgensen et al., 1997; Powers et al., 1990). Consequently, much effort has been spent examining the effect of forest harvest and management practices on SOM pools in intensively managed forests (e.g. Powers et al., 1990). More recently, interest in maintaining or enhancing SOM pools has been associated with the contribution of soil C efflux to atmospheric C following harvesting (Epron et al., 2006), and possible C sequestration in forest soils (Heath et al., 2003). Thus, there is great interest in maintaining or enhancing SOM pools in intensively managed forests.

Ecosystem C flux is dominated by soil respiration (hereafter referred to as bulk soil respiration, BR) (Valentini et al., 2000), which arises from respiration of living roots, microbial consumption of root-derived C compounds in the rhizosphere (e.g. turnover, exudation), and microbial respiration (MR) during consumption of SOM (Hanson et al., 2000). Changes in root and rhizosphere respiration following harvesting have low potential to alter soil C stocks as this C source is largely external to the soil. In contrast, changes in MR could cause a concurrent change in SOM and soil C. When soil C pools are in the so-called "steady-state" commonly assumed for intact forests (Bowden et al., 1993; Davidson et al., 2002b; Raich and Nadelhoffer, 1989), MR is balanced by OM inputs from above- and below-ground litter. Harvesting alters the rate and relative source contribution to total litter input, generally decreasing aboveground inputs of fresh leaf litter, but root death from the harvested stand and logging residues may offset any aboveground reduction (Epron et al., 2006). Although the net change in litter input will vary with site conditions and harvest-related management practices, increased soil temperature and modified soil moisture following harvesting are generally conducive to an increase in MR, possibly leading to an imbalance of the steady-state condition and a decline in soil C.

Management practices following harvesting which promote OM inputs to soil and reduce changes in the soil environment have potential to mitigate changes in MR. The decision to remove (or retain) logging debris at time of harvest is one likely practice to influence MR by those modes. Logging debris contains approximately 50% C by mass, and although a large portion of this C is respired to the atmosphere (Mattson et al., 1987; Palviainen et al., 2004), studies have shown significant inputs to soil as DOC when logging-debris is retained (Mattson et al., 1987; Piirainen et al., 2002; Qualls et al., 2000; Robertson et al., 2000). Logging debris also shades the soil surface, generally causing a reduction in soil temperature (Devine and Harrington, 2007; Roberts et al., 2005) which would be expected to reduce rates of MR. In a meta-analysis of 73 harvests, Johnson and Curtis (2001) found soil C in mineral soil to increase when logging debris was retained and decrease when it was removed, with the effect largely restricted to coniferous forests. The results from that analysis suggest a beneficial effect of logging-debris retention on soil C; however several long-term studies (> 10 yr) have found no lasting effect of logging-debris retention on total soil C pools following harvest (Johnson et al., 2002; Olsson et al., 1996), including summary findings from 26 installations in the Long Term

Soil Productivity (LTSP) network that covered a wide range of site and climate conditions (Powers et al., 2005).

Given that at least some C in logging debris enters the soil, and disregarding any methodological bias, there are two possibilities for the above results. First, inputs from logging debris may be offset by increased C efflux from the soil, either via increased MR or greater DOC leaching. Greater DOC loss is unlikely, as several recent studies have shown that DOC in deeper portions of the soil (>60cm) is largely derived from sources in the mineral soil rather than surface litter (Fröberg et al., 2007; Uselman et al., 2007). Greater MR could occur in response to increased substrate availability (Epron et al., 2006; Sulzman et al., 2005), or changes in the soil environment conducive to MR such as an increase in pH (Belleau et al., 2006) or greater soil moisture (e.g. mulch effect) (Devine and Harrington, 2007) under conditions when soil moisture would be expected to limit MR (Rey et al., 2002). Second, spatial variability of soil chemical properties is high, resulting in relatively large minimum detectable change for those properties following experimental manipulation (Homann et al., 2001; Rothe et al., 2002). In addition, soil C has been shown to increase following harvesting regardless of organic matter retention (Powers et al. 2005), and this increase may overwhelm any increase associated with logging debris. It is possible that logging debris does modify soil C pools, but the magnitude of the change may be too small to be statistically identifiable in the presence of background variability.

Competing vegetation re-growth (i.e. non-crop tree) may also mask or alter the effect of logging debris on MR. *In situ* separation of the MR component of BR is difficult (see Hanson et al. 2000 for a review of methods), leading some investigators to measure BR as a surrogate for MR following harvesting assuming that root contribution to BR is low (Hendrickson et al., 1989; Toland and Zak, 1994). Hendrickson et al. (1989) found no difference in BR between a whole-tree and bole-only harvest of mixed stands in the Great Lakes region U.S.A, but differences in the amount of live vegetation between treatments and the presence of aspen (*Populous tremuloloides*) sprouts likely confounded the response. Edwards and Ross-Todd (1983) found greater *in situ* BR when logging debris was retained following harvest of a deciduous forest in Tennessee U.S.A., but observed lower estimates of MR determined in lab incubations when the influence of vegetation was removed. Clearly, root respiration may mask the effect of logging debris, but competing vegetation also has potential to alter MR directly through additional modification of the soil environment (Gurlevik et al., 2004; Roberts et al., 2005), increased inputs of OM to both above- and belowground pools (Shan et al., 2001), and changes in microbial biomass and population structure (Busse et al., 2001; Li et al., 2004). Numerous studies have documented a strong control of vegetation on BR in intact forests (Campbell et al., 2004; Hogberg et al., 2001; Janssens et al., 2001), which appears to be largely driven by rates of photosynthesis (Hogberg et al., 2001; McDowell et al., 2004; Tang et al., 2005). It is possible that competing vegetation could have as great or greater influence on BR than vegetation in intact forest, possibly overriding any modification to MR associated with logging-debris retention.

I examined the effect of logging-debris retention with initial or annual applications of CVC on *in situ* estimates of BR and MR, and soil C following clearcut harvest at two sites that supported Douglas-fir (*Pseudotsuga menziesi*) forests in the Pacific Northwest to determine if the null effect of logging-debris retention on soil C observed by other studies (e.g. Powers et al. 2005) applies to this region, and whether or not changes in MR could account for any observed effect on soil C The specific objectives are to determine 1) if varying amounts of logging-debris retention modify MR and how the relationship varies in the presence and absence of competing vegetation, 2) the contribution of recently-fixed C to BR when only initial CVC is applied, 3) causal mechanisms for changes in MR and BR, and 4) if changes in MR could explain any changes in soil C. Effect of the treatments was examined at two sites that differ in soil properties and annual precipitation to see if treatment responses were altered by those site variables.

Methods

Site Characteristics

This study is part of a larger investigation initiated at two sites in 2003 to assess the impact of logging-debris retention and herbicide treatments on soil properties, nutrient cycling, and Douglas-fir (*Pseudtsuga menziesii*) growth. Both study sites are affiliates of the LTSP network (Powers et al. 1990). Potential productivity (site index) is similar between sites, but large differences exist in precipitation and soil properties (Figure 2.1; Table 2.1). Site 1 (hereafter referred to as Matlock) is located on the Olympic Peninsula in WA, approximately 45 km NW of Olympia near the town of Matlock. Soil at Matlock is classified as a sandy-skeletal, mixed, mesic, Dystric

Xerorthents, formed in glacial outwash with slopes ranging from 0 to 3% (Soil Survey Staff, USDA-NRCS). Site 2 (hereafter referred to as Molalla) is located approximately 24 km NE of the town of Molalla, OR in the foothills of the western Cascades. Soil at Molalla is classified as fine-loamy, isotic, mesic Andic Dystrudepts, formed in basic agglomerate residuum with slopes ranging from 2 to 40% (Soil Survey Staff, USDA-NRCS).

The regional climate is Mediterranean, characterized by mild, wet winters and dry, warm summers with periods of drought (> 2 mo) common. Precipitation falls almost entirely as rain, but some snowfall does occur during winter months. During the study period, mean monthly air temperature at each site was similar across years, generally reaching the annual maximum in July (Figure 2.1). Maximum monthly precipitation at each site was also similar between years, occurring in January and November of 2006 (Figure 2.1). Almost no rain occurred in July and August of 2006, but rain fell in every month of the 2007 growing season (Figure 2.1).

Prior to harvest, both sites supported second-growth even-aged stands that were primarily (>95% of basal area) composed of Douglas-fir. Stand density was approximately 280 trees ha^{-1} at both sites, and basal area was 35 and 46 m^2 ha^{-1} at Matlock and Molalla, respectively. At time of harvest stand age was 45 yr at Matlock and 56 yr at Molalla. In 1994, Molalla was thinned from below, whereas in 1998 some trees were removed at Matlock following a severe ice storm.

Experimental design and treatment application

Sites were initially clear-cut harvested with chainsaws in March (Molalla) and April (Matlock) of 2003. Trees were removed with ground-based mechanized equipment along marked corridors that were evenly distributed across plots to minimize experimental error associated with soil disturbance. Following harvest, a 2x2 randomized complete block factorial design was installed at each site (Figure 2.2). The factors were harvest type (two levels - bole-only or whole-tree) and herbicide for competing vegetation control (CVC) (two levels - with initial CVC or annual CVC). The factorial combinations were replicated four times in a randomized complete block design and applied to plots 0.3 ha in size. All plots received an initial application of herbicide to reduce competing vegetation; at Molalla glyphosate was aerially applied in August, 2003 and triclopyr was applied with backpack sprayers at Matlock during September of 2003. Following this initial application, only those treatments assigned annual CVC were treated with herbicide in the spring and fall of each year as necessary to control competing vegetation. Both sites were planted with bare-root Douglas-fir seedlings in February (Molalla) and March (Matlock) of 2004 at a 3 x 3 m spacing (1,111 trees ha^{-1}).

In March of 2005, three subplots within each 0.3 ha plot were identified for application of a subplot logging-debris retention treatment (Figure 2.2). Subplots encompassed a four m^2 area centered on a single planted Douglas-fir seedling. This design modification was chosen to minimize experimental error associated with treatment application (e.g. discontinuous debris coverage at the whole-plot level) and spatial variability of soil chemical properties. Woody logging debris was randomly applied at a rate of 0%, 40%, or 80% soil surface coverage to one of the subplots in each whole-plot. For each assigned treatment application, logging debris 5.0 to 12.5 cm in diameter that was within the associated whole-plot was stacked in a systematic criss-cross fashion until the assigned coverage (+/- 10% with visual determination) was reached. In the case of the 0% treatment, all logging debris was removed from the subplot, but no attempt was made to remove legacy wood if present. Debris volume was estimated with the line-transect method (Brown, 1974) and converted to a mass estimate (assuming wood density of 0.48 Mg m^{-3}). The corresponding mass of logging debris in the 40% and 80% coverage was 13 (std. dev. = 5.2) and 30 (std. dev. = 8.5) Mg ha^{-1}, respectively, at Matlock, and 14 (std. dev. = 5.2) and 29 (std. dev. = 10.7) Mg ha^{-1}, respectively, at Molalla. The overall design is a randomized complete block split-plot with two whole-plot factors (harvest type and CVC) and one subplot factor (logging-debris coverage).

***In situ* soil respiration and associated measures**

At each subplot and at four locations within adjacent uncut forest at each site, two 15.25 cm Schedule-40 PVC respiration collars were permanently installed at random locations in June 2005. Collars were installed to a depth of 3 cm and 30 cm at each subplot. The 3 cm collar was used to estimate bulk soil respiration (BR), and the 30 cm collar was used to estimate microbial respiration (MR) with the root exclusion method (Hanson et al. 2000) where live roots are severed and future root in-growth is restricted. Several studies have determined reasonable estimates of BR and MR with the use of cores similar to those used here (Dilustro et al., 2005; Kelting et al., 1998; Vogel and Valentine, 2005). However, since it is likely that some root growth will occur below 30 cm (even in recently harvested stands), the MR estimate determined with this method is most appropriately viewed as an indication of relative differences in MR rather than absolute differences among treatments.

In addition to the above, potential error in MR estimates associated with the root exclusion method include modified soil moisture and temperature, root in-growth to cores, elevated heterotrophic decomposition of severed roots, disturbance-related artifacts at time of installation, and the buildup of a CO_2 boundary layer within the soil profile (Hanson et al. 2000). Differences in soil temperature and moisture between collars and ambient soils were assessed in the first year of measurement and found to be small (data not shown). Respiration measurements were delayed for three months following installation to reduce artifacts associated with increased heterotrophic decomposition of severed roots. Kelting et al. (1998) found that elevated heterotrophic respiration arising from severed roots in small root exclusion collars had largely dissipated after three months. Root in-growth occurred in most of the collars to some extent by the end of the study period, with in-growth being much greater in the initial CVC treatment relative to the annual CVC treatment. With the exception of the initial CVC herbicide treatment at Matlock, there was little difference among logging-debris coverages in each herbicide treatment, and there was no detectable relationship between root in-growth biomass and estimates of MR (Appendix A). Any error associated with development of a boundary layer and disturbance-related artifacts was assumed to be uniform across treatments within a site.

Beginning in September of 2005, BR and MR were measured on a monthly basis by measuring CO_2 efflux from each of the soil collars with the use of a portable infrared gas analyzer (IRGA) (LI-6250 Licor Inc., Lincoln NE) attached to a custom-built closed dynamic soil respiration chamber. The respiration chamber was fitted with a foam gasket to provide an airtight seal between chamber and collar (Hutchinson and Livingston, 2001), and an open tube was installed in the wall of the chamber to minimize pressure-related artifacts (Davidson et al., 2002a). The IRGA was calibrated to a known CO_2 standard and current barometric pressure was entered into the machine prior to a given measurement period. At the beginning of each measurement, CO_2 concentrations were scrubbed to approximately 10 ppm below ambient, and then allowed to rise during measurement. Carbon dioxide concentration was recorded over a 5 ppm change in chamber concentration at each of three consecutive measurements. Air temperature and relative humidity were measured concurrently inside the chamber with each flux measurement. Carbon dioxide flux was calculated based on ambient pressure, air temperature, relative humidity, collar surface area, and chamber volume (3,980 cm^3). The three measurements were averaged to determine a mean CO_2 flux per collar for each time period in units of $\mu mol\ m^{-2}\ sec^{-1}$.

All collars at a given site were measured over an approximate 6 hr period (1000-1600) in a single day, and the other site was generally measured in the same manner on the following day for any monthly measurement period. After each measurement, soil temperature was measured to a depth of 10 cm with a temperature probe. Volumetric soil water content (SWC) was measured at 2-hr intervals from a depth spanning 20 to 40 cm in each subplot with ECH_20 probes (Decagon Devices, Inc., Pullman, WA). Sensors were installed with a bucket auger approximately 30 cm from each tree. Calibration equations were developed in the laboratory for each of the sites by comparing known SWC with that measured by the sensor. Mean SWC on the day of measurement was used for all analyses in this study.

Laboratory incubations for microbial respiration

Soil samples were collected from half of the replications (Figure 2.2) (n=4 for each whole plot-sub plot combination at each site) in April, June, and September of 2006, and in June of 2007 to measure potential MR under controlled temperature and moisture conditions in the laboratory. Mineral soil was collected to a depth of 20 cm at three random locations in each subplot and composited. Composite samples were transported on ice to the laboratory for processing and analysis. Samples were air-dried for five days, passed through a 2mm sieve, and then stored in a cooler at 4ºC prior to analysis.

For each sample collection period, approximately 50 g of air-dried sieved soil was incubated in a microlysimeter constructed of benchtop filtration units (Falcon Filter, Becton Dickinson Labware) as described by Nadelhoffer (1990). Samples were incubated in the microlysimeters for 17 days at 25ºC and a soil water potential of -22 kPa. The target soil water potential was attained by allowing 100 ml of ultrapure water and soil to equilibrate for 30 min, and then a tension of -22 kPa was applied to each unit until water flow into the incubation collection chamber ceased. Incubation unit mass was periodically checked during the incubation period, and water was added as needed to maintain the initial soil water content following wetting. Soil subsamples were oven-dried at 105ºC for 3 days to report results on an oven-dry soil mass basis.

Microbial respiration was estimated by measuring the CO_2 evolution at days 3, 10, and 17 of the incubation period. At each measurement, CO_2-free air was allowed to flow through the incubation unit for one minute, and then the unit was capped with a rubber septa and tape was wrapped around potential leak points to ensure an airtight seal. Sealed units were allowed to incubate for approximately 1.5 hr, and then headspace gas was analyzed for CO_2 concentration on a gas chromatograph (Hewlett Packard 5700A series) with the use of standard protocols. Headspace volume, air pressure and temperature, and soil mass were used to convert CO_2 evolution to an estimate of CO_2 flux expressed on a μg-C g-soil^{-1}hr^{-1} basis. The three measures for each unit were averaged to determine a mean soil respiration rate for each replication at each incubation period.

Mineral soil carbon

Soil samples were collected from each subplot replication (Figure 2.2) in July 2005 and October 2007 to assess treatment effects on soil C during the study period. Mineral soil was collected to a depth of 20 cm at three random locations in each replication and composited in the field, taking care to remove any organic material prior to compositing. Samples were collected volumetrically in 2005 with a “push” tube sampler and in 2007 with a core sampler attached to a slide hammer. Samples were air-dried for approximately 2 wk following field collection and then sieved to pass a 2 mm mesh. Approximately 5 g of the sieved soil was separated and ground with a mortar and pestle to pass a 0.25 mm mesh. Ground samples were dried at 65 ºC for one day, and then dry combusted on a Fisons NA1500 NCS Elemental Analyzer (ThermoQuest Italia, Milan, Italy) to determine total soil C concentration.

Data analysis

A mixed model approach with repeated measures was used to assess treatment effects on the dependent variables BR, soil temperature, and soil moisture. Both block and whole-plot factors nested within block were modeled as random effects, with whole-plot factors, the subplot factor, and month modeled as fixed effects. Initial analysis showed no effect of the whole-plot factor harvest type and it was subsequently removed from the model. Effect of CVC on MR measured *in situ* was not assessed given the systematic bias associated with greater root in-growth in the initial CVC treatment (Appendix A). The relative effect of logging-debris coverage on *in situ* MR was analyzed separately for each herbicide treatment. This approach assumes that error in MR estimates associated with autotrophic inputs from roots is relatively constant across logging-debris treatments within a given herbicide treatment. Inasmuch as root contribution to MR is a function of fine root biomass, this assumption is valid at Molalla for both herbicide treatments and at Matlock in the annual CVC treatment (Appendix A). The assumption is probably not valid at Matlock site in the initial CVC treatment given the large increase in root in-growth that occurred with increasing logging-debris coverage (Appendix A).

Statistical analysis was limited to growing-season months (defined here as April-September) in each year when treatment effects were expected to be greatest. This approach was necessary to identify appropriate covariance matrices for the repeated measures analysis. The model was run with each of the candidate covariance matrices for each of the dependent variables, and the matrix that resulted in the lowest fit criteria (BIC) was then used for analysis. Examination of the residuals indicated a log transformation was necessary to meet assumptions of normality and heterogeneity for the respiration variables, but all others were deemed to have met assumptions in the untransformed state. Log transformed means were back-transformed for presentation and thus represent median values as reported. When significant interaction was observed between either the whole plot or subplot factor and time, *a priori* orthogonal contrasts were performed to test for significance of difference between 1) the absence and presence of debris (0% coverage versus both the 40% and 80% coverage treatments), and 2) the 40% and 80% logging-debris coverage treatments.

Treatment effects on MR determined in the laboratory and total soil C measurements were assessed with a mixed model approach at each sample period. Random and fixed effects were modeled as above, with the exception of the fixed time effect that did not apply. Tukey's Honestly Significant Difference test was used to determine significant differences among means for the incubation and soil C data.

The dependence of soil respiration on soil temperature was assessed with the following function as described in Lloyd and Taylor (1994):

$$CO_2 = \beta 0 * e^{\beta 1 * T} \qquad \text{(eq. 1)}$$

where CO_2 is the estimated flux of CO_2 (in μmol m^{-2} sec^{-1}), e is the natural logarithm, $\beta 0$ and $\beta 1$ are fitted parameters, and T is temperature in °C. Non-homogenous variance required log transformation of the temperature function for proper parameter estimation. Model parameters were estimated in each year for both MR and BR in the annual CVC treatment, and for BR in the initial CVC treatment. For each growing season (six-month period), comparison of regression lines was performed to determine if parameters differed by treatment. Data were fit to alternative models that tested for (1) different intercepts, (2) different slopes, (3) both different intercepts and slope, and (4) no difference among treatments. The model that resulted in the lowest fit criterion (minimum BIC) was used to determine which parameters differed by treatment. In cases where the minimum BIC resulted in a model with non-significant parameters, the next best model was chosen until a model with significant parameters was identified. Comparisons among treatments were limited to BR and MR for logging-debris treatments, and BR for herbicide treatments. Parameters were back-transformed when necessary (i.e. $\beta 0$ when regression lines were compared) for interpretation. An alpha level of 0.05 was used to assess statistical significance in all evaluations. All analyses were performed in SAS V9.1 (SAS Institute, Cary NC).

Results

Herbicide effects on BR, soil temperature, and SWC

Over the two-year study period, median BR ranged from 0.52 to 5.45 µmol m^{-2} sec^{-1} at Molalla (Figure 2.3), and from 0.58 to 4.65 µmol m^{-2} sec^{-1} at Matlock (Figure 2.4). Bulk soil respiration showed a general seasonal trend that correlated with soil temperature, being lowest during winter months when soil temperature was low and high during summer months when temperature was high. The seasonal pattern of BR from the reference stands of Douglas-fir at each site showed notable similarity over time with those from the experimental manipulations. At Matlock, BR from the reference stand was similar in magnitude to that measured in the experimental manipulations (Figure 2.4), but at Molalla BR from the reference stand tended to be higher than that from the experimental manipulations during some months of the growing season in each year (Figure 2.3).

Bulk soil respiration was significantly decreased following annual CVC at both sites in some months of the growing season during each year (Tables 2.2, 2.3; Figures 2.3, 2.4). The duration and magnitude of the effect were more pronounced at Molalla than Matlock, where median CO_2 efflux during months with significant differences was on average 1.24 (range 0.65 to 2.14) and 1.89 (range of 1.16 to 3.03) µmol m^{-2} sec^{-1} greater in the initial CVC treatment at Molalla during 2006 and 2007, respectively (Figure 2.3), compared to a greater median efflux during months with significant differences of 0.88 (range 0.65 to 1.10) and 1.27 (range 0.79 to 2.09) µmol m^{-2} sec^{-1} at Matlock for the same years (Figure 2.4). Soil temperature was significantly increased by approximately 0.8 ºC (range 0.6 to 1.1) when annual CVC was applied at Molalla during some months of the growing season (Table 2.2, Figure 2.3), but there was no discernable effect at Matlock (Table 2.3, Figure 2.4). Soil water content was significantly increased following annual CVC at both sites in 2006 and at Molalla in 2007, but the magnitude of the effect varied by month (Tables 2.2, 2.3; Figures 2.3, 2.4). Increases in SWC in the annual CVC treatment at Molalla ranged from 0.08 to 0.11 m^3 m^{-3} and 0.05 to 0.08 m^3 m^{-3} in 2006 and 2007, respectively (Figure 2.3) and increases in SWC with annual CVC at Matlock ranged from 0.05 to 0.10 m^3 m^{-3} in 2006 (Figure 2.4).

At Molalla, BR in the initial CVC treatment was greatest prior to peak soil temperature in each year, but BR in the annual CVC treatment occurred at or following peak soil temperature (Figure 2.3). The earlier reduction in BR in the initial CVC treatment occurred in both years when SWC was ≤ 0.21 m^3 m^{-3}, suggesting a water limitation to BR when SWC content is below this value. Water limitation to BR was also evident at Matlock, where minimum BR in 2006 coincided with minimum SWC in August, followed by an increase in BR in September when SWC increased and soil temperatures were still relatively warm (Figure 2.4). In the second year when rainfall extended into the growing season (Figure 2.1), greater SWC was associated with greater BR at both sites. At Molalla the effect was limited to the initial CVC treatment (Figure 2.3), but at Matlock both CVC treatments had greater median BR in 2007 than 2006 (Figure 2.4).

Logging-debris effects on BR, MR, soil temperature, and SWC

There was a significant effect of logging-debris coverage on BR at Molalla in 2006 that varied by month (Table 2.2), with a general pattern of decreasing BR with increasing logging-debris coverage (Figure 2.5). The 0% logging-debris coverage had significantly higher BR than treatments where logging-debris was retained in all months of the 2006 growing season except July (Figure 2 5). Back-transformed estimates of differences indicated relative median increases of 62% (95% CI 116%, 21%), 43% (95% CI 71%, 20%), 52% (95% CI 85%, 25%), 33% (95% CI 63%, 9%), and 23% (95% CI 56%, 3%) in BR for the months of April, May, June, August, and September, respectively at Molalla. Contrasts between the 40% and 80% coverage were not significant in any month of 2006 at Molalla. The same pattern of decreasing BR with increasing debris retention was observed in 2007 at Molalla (Figure 2.5), but differences were not significant (Table 2.2). There was no significant effect of logging debris on BR at Matlock in either year (Table 2.3), but median BR tended to be greater in the 0% coverage than either the 40% or 80% coverage in most months (Figure 2.6).

There was a significant effect of logging debris on soil temperature at both sites which varied by month (Tables 2.2, 2.3). The general pattern was similar to that observed for BR, where soil temperature decreased with increasing logging-debris retention (Figures 2.5, 2.6). The magnitude of the effect was more pronounced at Matlock then Molalla, and more pronounced in 2006 than 2007. At Matlock, differences between the absence and presence of logging debris were greatest during June in 2006 (mean difference of 3.3°C; 95% CI 2.7, 3.9) and July of 2007 (mean difference of 2.1°C;

95% CI 1.7, 2.5) (Figure 2.6). Differences between the 40% and 80% coverage also occurred during those months at Matlock, with mean increases of 1.3°C (95% CI 0.7, 1.9) and 1.2°C (95% CI 0.7, 1.7) in the 40% coverage during June 2006 and July 2007, respectively. At Molalla, differences between the absence and presence of logging-debris were greatest in July 2006 and May 2007 (mean difference of 1.4°C; 95% CI 1.0, 1.8 for both months), and differences between the 40% and 80% coverage were also greatest during those months (mean difference of 0.5°C (95% CI 0.1, 0.9) and mean difference of 0.6°C (95% CI 0.2, 1.0) for 2006 and 2007, respectively) (Figure 2.5). There was no significant effect of logging-debris coverage on SWC at Molalla (Table 2.2), but there was a significant effect at Matlock in 2007 that was independent of sample month (Table 2.3). *A priori* contrast between presence and absence of logging debris was not significant (p=0.137), but there was a significant difference between the 40% and 80% coverage (p=0.020) with the 40% coverage having 0.04 m^3 m^{-3} lower SWC across the growing season (Figure 2.6).

At Matlock, peak BR coincided with peak soil temperature in both 2006 and 2007 (Figure 2.6). As indicted with data analysis of the CVC treatments, there appeared to be a strong soil moisture limitation to BR at Matlock in July and August of 2006, but temperature may have also contributed to the response as soil temperature was decreased during those months. There was no evidence of water limitation in 2007 at Matlock as peak BR in the 40% and 80% coverage occurred during the month with the lowest SWC (Figure 2.6). At Molalla, there were several instances when BR appeared to be limited by factors other than temperature. For example, BR in the 0% coverage, 40% coverage, and reference stands decreased during the month with peak soil temperature (July) in 2006, but then increased in the 0% and 40% coverage in the following month when temperatures were lower (Figure 2.5). Water limitation to BR was unlikely during July given that BR increased in August when SWC was lower (Figure 2.5). At the same site in 2007, peak BR in all treatments occurred in July, one month prior to peak temperature. Minimum SWC coincided with peak temperature that year, possibly indicating water limitation to BR, but the SWC values were similar to those in the previous year when no water limitation to BR was evident (Figure 2.5).

Examination of the MR data by CVC treatment indicated similar effects of logging debris on MR as those observed for BR, but the magnitude and significance of the effects generally decreased due partly to a reduction in statistical power. At Molalla, there was no effect of logging-debris coverage on MR in the initial CVC treatment in either year (Table 2.4), but median MR was consistently lower in the 80% coverage (Figure 2.7). When annual CVC was applied at Molalla, the 0% coverage had significantly greater MR than both the 40% and 80% coverage in June (relative median increase of 54%; 95% CI 101%, 18%) and August (relative median increase of 29%; 95% CI 68%, 2%) of 2006 (Figure 2.7). During those months and in June of 2007, the 40% coverage had significantly greater MR than the 80% coverage (relative median increase of 46% (95% CI 98%, 7%), 39% (95% CI 89%, 2%), and 64% (95% CI 120%, 23%) for June 2006, August 2006, and June 2007, respectively)(Figure 2.7). At Matlock, there was no significant effect of the logging-debris treatment on MR when annual CVC was applied (Table 2.4), but the 0% coverage had significantly greater MR than both the 40%

and 80% coverage in the initial CVC treatment during September 2006 (relative median increase 28%; 95% CI 65%, 1%), and the 40% coverage had significantly greater MR than the 80% coverage in August 2006 (relative median increase 46%; 95% CI 110%, 2%) (Table 2.4, Figure 2.8). The general pattern of MR among logging-debris treatments was similar regardless of CVC in 2006 at Matlock, but the effect of logging debris appeared to vary with CVC in 2007 (Figure 2.8). In that year, there was little difference among logging-debris treatments for most of the growing season when annual CVC was applied (Figure 2.8B), but MR when logging debris was removed in the initial CVC treatment was greater compared to when it was retained in most of the growing season (Figure 2.8A). Deviations from these general patterns occurred in August and September of 2007 (Figure 2.8).

Temperature dependence of soil respiration

Temperature was significantly related to BR and MR at each site in both years of the study. At Matlock, the log-transformed temperature response function explained between 44 and 76% of the variation in BR across all periods within a year and CVC treatment, and 56 and 65% of the variation in MR for the annual CVC treatment in 2006 and 2007, respectively (Figure 2.9). At Molalla, between 55 and 69% of the variation in BR was explained by the log-transformed temperature function within a year, and 64 and 58% of the variation in MR when annual CVC was applied in 2006 and 2007, respectively (Figure 2.10). Clearly there is a strong temperature dependence of soil respiration, but visual examination of the data suggests high variability in the response at these sites following harvest.

Comparison of regression lines with BR data from the six-month growing season of each year indicated significant differences in estimated parameters among treatments at each site. At both sites and in both years, intercept (β0) for the log-transformed function was significantly greater in the initial CVC treatment compared to the annual CVC treatment (Tables 2.5, 2.6). Differences were greater at Molalla than Matlock. Molalla had back-transformed intercepts that were 0.49 and 1.40 μmol m^{-2} sec^{-1} higher for the initial CVC treatment than the annual CVC treatment for the 2006 and 2007 growing seasons, respectively (Table 2.5), compared to Matlock which had back-transformed intercepts that were 0.31 and 0.66 μmol m^{-2} sec^{-1} higher for the initial CVC treatment than the annual CVC treatment for the same years (Table 2.6). At Molalla, the temperature coefficient (β_1) was also significantly higher in the initial CVC treatment in 2006, but no difference existed in 2007 between CVC treatments. There were no differences in model parameters among logging-debris treatments at Matlock for either BR or MR in the growing season of either year, but there were significant differences at Molalla. During the growing seasons of both 2006 and 2007, the 0% coverage had a significantly greater estimated intercept (β0) than the 80% coverage for BR (increases between 0 and 80% coverage of 0.60 and 0.58 μmol m^{-2} sec^{-1} for 2006 and 2007, respectively), and a significantly higher intercept than the 40% coverage in 2006 (0.49 μmol m^{-2} sec^{-1} higher) (Table 2.5). Similar results were found when MR data were fit to the model in 2006 (0% coverage with intercept 0.41 μmol m^{-2} sec^{-1} higher than 80% coverage), but there were no differences among treatments in 2007 for either the intercept or temperature coefficient (Table 2.5).

Microbial respiration from incubations

Microbial respiration determined under controlled laboratory conditions showed similar patterns between sites across incubation periods. At both sites in 2006, MR was greatest in the June incubation and lowest in the September incubation (Figure 2.11). Mean MR in the June 2007 incubation (across all treatments) was approximately 43% and 30% lower than the June 2006 incubation at Matlock and Molalla, respectively (Figure 2.11). There was no significant effect of herbicide treatment on MR for any of the incubation periods at either site (Table 2.7). Mean MR tended to be greater in initial CVC treatments than annual CVC treatments, with mean (non-significant) differences greatest in April and June of 2006 at each site (Figure 2.11). There was a significant effect of logging debris on MR at Molalla in the June 2006 incubation (Table 2.7, Figure 2.12A), where 0% coverage had MR rates that were approximately 20% (0.8 μg-C g-soil^{-1} hr^{-1}) greater than either the 40% or 80% coverage. A similar pattern was observed at Matlock but differences among treatments were not significant (Table 2.7, Figure 2.12B). There was no effect of logging-debris coverage on MR at any of the remaining incubation periods (Table 2.7, Figure 2.12).

Total soil C

There was no effect of CVC treatment on soil C concentration at either site in 2005 or 2007 (Table 2.8). Differences between sample years were not significantly different from zero, but C concentrations tended to increase with time regardless of treatment. At Matlock, mean concentrations in 2005 were 66.6 (se = 4.1) and 73.3 (se =

4.0) g C kg^{-1} for the initial CVC and annual CVC treatments, respectively, and 71.9 (se = 5.0) and 80.3 (se = 4.9) g C kg^{-1}, respectively, in 2007. At Molalla, mean concentrations in 2005 were 75.9 (se = 8.4) and 83.9 (se = 8.3) g C kg^{-1} for the initial CVC and annual CVC treatments, respectively, and 84.0 (se = 7.6) and 89.8 (se = 7.5) g C kg^{-1}, respectively, in 2007.

There was no significant effect of logging-debris coverage on soil C at Molalla in either year (Table 2.8), but mean concentration in the 80% coverage tended to be higher than either the 0% or 40% coverage in 2007 (Figure 2.13A). At Matlock, there was a trend of increasing soil C concentration with increasing logging-debris coverage in 2005 (Figure 2.13B), but differences among treatments were not significant ($p=0.057$, Table 2.8). In 2007, 80% coverage had significantly greater soil C than either the 0% (14.8 g C kg^{-1} lower than 80% coverage, 95% CI: 2.2, 27.4) or 40% coverage (15.6 g C kg^{-1} lower than 80% coverage, 95% CI: 3.6, 27.6) (Table 2.8, Figure 2.13B). There was a similar pattern in soil C change during the two-year study period at both sites. Soil C was significantly increased in the 0% coverage, with a mean increase of 9.5 (95% CI: 0.6, 18.4) and 10.1 (95% CI: 0.3, 19.8) g C kg^{-1} at Molalla and Matlock, respectively. There was little change in soil C during the study period in the 40% coverage, but mean soil C tended to be higher in the 2007 period at both sites in the 80% coverage (Figure 2.13).

Discussion

Herbicide effects on BR, MR, and soil C

Greater BR in the initial CVC treatment at both sites could be due to greater root and rhizosphere respiration from competing vegetation, greater MR from increased litter inputs, or more likely a combination of both. The absence of any significant difference in potential MR (laboratory-determined) suggests no effect of initial CVC on *in situ* MR, but the lack of significance may be due to lower statistical power in the whole-plot CVC treatment (i.e. Type I error). If there was an effect of initial CVC on potential MR, it appears to have been limited to the early part of the 2006 growing season, as there was no difference in mean MR between CVC treatments during the last two incubation periods (Figure 2.10). Differences between years may be associated with the soil moisture limitation to BR that was evident in 2006, but not 2007. Substrate may accumulate during times of moisture limitation, and then be rapidly consumed when SWC increases to a point that is no longer limiting to microbial activity (Borken et al., 2006; Rey et al., 2002). Soil water content was probably high enough in 2007 to not limit MR *in situ*, resulting in greater consumption of OM inputs *in situ* such that little difference existed in potential MR when incubated under controlled laboratory conditions. Estimated intercepts of the temperature function supports this, as differences between treatments were greater in 2007, indicating greater substrate quality and availability in that year (Fierer et al., 2005). The 30% to 40% reduction in potential MR in 2007 regardless of CVC treatment also suggests greater *in situ* consumption of OM during that year. This suggests OM inputs from competing vegetation are rapidly consumed when microbial activity is not limited by soil moisture, which is supported by the lack of difference in soil C between CVC treatments at the end of the experiment. Root respiration from competing vegetation would also contribute to greater BR in the initial CVC treatment, but the relative contribution cannot be determined. Visual comparison with the reference forest BR suggests that the contribution is large, as root respiration from intact forests comprises approximately half of BR (Hanson et al. 2000).

At Molalla, higher soil temperatures in the annual CVC treatment could have led to greater rates of *in situ* MR given the strong temperature dependence of soil respiration. Using the estimated parameters in Figure 2.9 and mean temperatures from each treatment, the increased temperature observed in the annual CVC treatment accounts for approximately 10% and 5% of BR in months with significant differences in 2006 and 2007, respectively. Some of this increase would arise from root and rhizosphere sources (Boone et al. 1998), but at least a portion would constitute a response in MR. However, even if it is assumed that all of the increase in BR is due to MR, it is unlikely that greater MR in the annual CVC treatment would cause a measureable reduction in SOM given that the increase was transient and generally small. The lack of any difference in soil temperature between treatments at Matlock makes it unlikely that MR was increased following annual CVC at that site.

No difference between treatments in soil C at both sites following four years of annual CVC supports the conclusion that increased *in situ* MR in the annual CVC treatment was either small or non-existent. Soil respiration in intact forests is dominated by sources of recently-fixed C (e.g. root metabolism, leaf litter, root turnover), with only a small portion contributed from more stable, older SOM (Trumbore 2000, Giardina et al. 2004). At least at these sites, this appears to be the case following harvesting as well, implying that SOM pools are fairly resistant to change when inputs of recently-fixed C are reduced in the initial years following harvest. These results contrast with some studies from the southeastern U.S. that have shown reductions in soil C following sustained CVC (Miller et al. 2006, Echeverria et al. 2004), but those studies occurred at sites where initial soil C concentrations were low (< 20 g C kg^{-1}). The contrasting results observed here could be due to the relatively high initial soil C at these sites, or to differences in climate between the two regions. McFarlane et al. (2008) found no effect of sustained CVC on mineral soil C in Ponderosa pine (*Pinus ponderosa*) plantations across a productivity and mineral soil C gradient (15-50 g C kg^{-1}) in northern California.

Studies that have measured BR following harvesting commonly assume that the contribution of root respiration and recently-fixed litter to BR is small (e.g. Toland and Zak 1994, Hendrickson et al. 1989, Londo et al. 1999), and have concluded that increased or similar BR to uncut forest is indicative of greater SOM decomposition in harvested areas. Here, estimated BR in the initial CVC treatment and reference forest was comparable at each site, but BR in the annual CVC treatment was lower than the reference forest in most months of the growing season (Figures 2.2, 2.3). Although the relative source contribution to increased BR following initial CVC is uncertain, it is clear that the increase arose primarily from sources of recently-fixed C (i.e. fixed post-harvest). The relative increase in BR that can be attributed to recently-fixed C (calculated as the difference between treatments in BR relative to the annual CVC treatment) during the growing season was on average 49% (range 24 to 85 %) and 89% (range 29 to 133%) at Molalla in 2006 and 2007, respectively, and 39% (range 29 to 49%) and 61% (range 35 to 92%) at Matlock for the same years. These estimates are fairly conservative as they do not account for root respiration from crop trees in the annual CVC treatment. Clearly, sources of recently-fixed C from competing vegetation contribute a substantial amount to BR in the initial years following harvesting, indicating the assumption of MR-dominated BR following harvesting is not valid in all situations when annual CVC is not applied.

Logging-debris effects on BR, MR, and soil C

Although there were few significant differences in MR among logging-debris treatments, the general patterns that were observed suggest MR was modified by logging-debris. At both sites, MR in the 80% coverage was consistently lower than the 0% coverage during most months of the growing season in each year, but MR in the 40% coverage was similar to or exceeded that in the 0% coverage (Figures 2.6, 2.7). Bulk soil respiration showed a similar pattern, which likely reflects differences in MR as well. It appears that high amounts (i.e. 80% coverage) of logging-debris retention reduce MR at these sites, but moderate amounts have little effect on MR. The effect appeared to be independent of vegetation abundance at Molalla, as patterns of MR were similar between the two CVC treatments, and there was no interaction between the logging-debris and CVC treatments on BR (Table 2.2). Influence of vegetation was also limited in 2006 at Matlock, but the effect of logging debris on MR appeared dependent on CVC in 2007, probably due to the influence of greater soil moisture in that year and its effect on competing vegetation growth. The magnitude of difference among logging-debris treatments was greater at Molalla than Matlock in both years, indicating site factors (e.g. soil characteristics, annual precipitation) modified the response.

Given the strong temperature dependence of soil respiration observed here and in other studies (e.g. Lloyd and Taylor 1994), it is likely that lower MR and BR in the 80% coverage was at least partially due to the reductions in soil temperature which occurred with this treatment. No difference in potential MR under constant temperature in most incubation periods supports this. Edwards and Ross-Todd (1983) estimated greater MR following a whole-tree harvest compared to a bole-only harvest in the first year following logging that was attributed to greater surface temperatures in the whole-tree harvest. In contrast, Carter et al. (2002) found no difference in BR between bole-only and whole-tree-plus-forest-floor-removal harvests one year after logging when sustained CVC was applied, but there was also no difference in soil temperature between treatments in that study. Reduced soil temperature likely contributed to reduce MR in the 80% coverage, but evidence suggests factors in addition to temperature contributed to the difference between 0% and 80% coverage, at least at Molalla in 2006. At that site and year, potential MR determined in the July incubation and the intercept of the temperature function calculated from *in situ* data were significantly higher in the 0% coverage.

Greater microbial biomass was unlikely a factor contributing to the increase in MR in the 0% coverage, as several studies have shown either no effect or a decrease in microbial biomass following logging-debris removal (Carter et al., 2002; Chen and Xu, 2005; Li et al., 2004; Mendham et al., 2002). Nitrogen availability has been shown to influence microbial activity (Allen and Schlesinger, 2004; Gallardo and Schlesinger, 1994), but there were no differences among logging-debris treatments in either potential N mineralization or available N *in situ* (Chapter 4). Increased substrate availability from aboveground sources is unlikely when logging debris is removed, but belowground substrate availability may have increased due to an increase in temperature and subsequent greater root decomposition (Chen et al., 2000). Greater *in situ* root decomposition could result in a greater amount of partially decomposed root material during the incubation (i.e. small enough to pass a 2mm sieve), resulting in an increase in MR when incubated at constant temperature. Lack of a comparable effect in the 2007 incubation may indicate depletion of easily decomposable root C in the 0% coverage, or the smaller differences in temperature among treatments that year (Figures 2.4, 2.5) may have dampened differences in root decomposition (Chen et al. 2000).

The absence of a temperature response in MR in the 40% coverage may be associated with changes in the soil environment and its effect on microbial activity, resulting in an offset of temperature-induced reductions in MR. Phospholipid fatty acid (PLFA) analysis performed in June of 2007 (data not shown) indicated no difference in the microbial community structure among treatments, but a stress indicator (19:0cy / 18:1w7c) was significantly lower in the 40% coverage compared to the 0% and 80% coverage at both sites (Appendix B). Stress indicators have been shown to increase in response to various environmental stresses including low pH, low moisture, and reduced oxygen availability (Petersen et al., 2002). More favorable environmental conditions could be associated with higher soil pH (Appendix C), differences in SWC, which was consistently lower in the 40% coverage than either the 0% or 80% coverage at both sites (Figure 1.4, 1.5) (e.g. greater oxygen diffusion with lower SWC), or some other factor not assessed in this study. Whatever the mechanism, it is possible that reduced microbial stress resulted in greater microbial activity, causing greater MR than would be predicted from temperature alone. Other possibilities include a greater rate of CO_2 diffusion within the soil due to greater air-filled pore space in the 40% coverage (Susfalk et al., 2002), or a moisture-temperature interaction may have been more conducive to root decomposition (Chen et al, 2000).

Differences in soil C at the end of the experiment generally agree with the MR results. Soil C concentration was significantly higher in the 80% coverage at Matlock, but there was little difference between the 0% and 40% coverage which would be expected given the similar MR rates between those treatments. The similar (non-significant) pattern observed at Molalla suggests the same mechanism, but differences may not have been detectable due to the greater soil C pool at that site (Table 2.1). Nevertheless, there was no significant difference in DOC leaching at 60 cm depth from these same experimental units (Chapter 3), which in combination with reduced (or even similar) MR and increased C inputs indicates greater C accumulation in soil. Thus it appears that there is an effect of logging debris on soil C pools at these sites, but the effect is limited to situations where relatively large amounts of debris are retained and undetectable when the initial soil C pool is large. These results agree with those from a meta-analysis conducted by Johnson and Curtis (2001), who found that soil C was increased following harvesting of conifers when logging debris was retained, but decreased slightly when debris was removed.

Given the magnitude of increase in soil C in the 80% coverage at Matlock, much of the accumulation can be attributed to sources within the soil rather than inputs at the surface. Using bulk density values in Table 2.1 (corrected for coarse-fragment content), the 14.8 g C kg^{-1} increase in concentration corresponds to a C mass of 13.9 Mg C ha^{-1}, almost the same mass of C applied as logging debris assuming a C concentration of 500 g kg^{-1}. Much of the logging debris still remains at the soil surface (personal observation), and most C (>75%, Mattson et al., 1987) in logging debris is respired directly to the atmosphere, making it likely that C inputs from logging debris were a small contributor to the estimated 13.9 Mg C ha^{-1} increase. The significant increase in soil C in the 0% coverage, combined with greater MR, underscores the role of belowground decomposition to changes in soil C following harvest.

In the summary LTSP findings, Powers et al. (2005) concluded that increased soil C following harvesting was largely due to fine root decomposition, and inputs from logging debris made little, if any, contribution to the increase. My results generally support the conclusion by Powers et al. (2005), but also indicate that the relative increase is dependent on the magnitude of logging-debris retention and its influence on the soil environment. In operational settings or experimental designs where treatments are applied to large areas, the effect of logging debris would be masked as coverage is discontinuous and each of the debris treatments used in this study would be present to some extent regardless of experimental treatment or operational practice (Meehan, 2006). Mean soil C at the whole-plot level (R. Slesak, unpublished data) was also greater in the bole-only harvest compared to the whole-tree harvest at Matlock, but differences were not statistically significant, indicating that the detectable difference reported here was a result of uniform logging-debris coverage and reduced spatial variability of soil properties.

Although differences in soil C may not be apparent, a relative mass balance calculation indicates C loss from mineral soil in the 0% coverage is greater than the 80% coverage, implying greater total belowground C (mineral soil + root necromass) in the 80% coverage. Greater belowground C may have implications for soil C sequestration, but the stability of the fraction contributing to the increase (i.e. root necromass) could be low, resulting in no difference between treatments over the course of a rotation. Further work is warranted to determine the stability of increased C at these sites, and to quantify any differences in belowground OM storage following logging-debris retention or removal.

Conclusions

Presence of competing vegetation significantly increased BR in the initial years after harvesting, with the magnitude of the effect greatest when soil moisture was not limiting. The absence of any difference in potential MR determined under controlled laboratory conditions and no difference in soil C at the end of the study period suggests that recently-fixed C inputs were rapidly consumed and had little impact on soil C pools. Greater MR following annual CVC at Molalla was likely given the significant increase in soil temperature, but the estimated increase was small and had no net effect on soil C. Although annual CVC has potential to reduce soil C through a reduction in litter inputs, the results suggest a concurrent reduction in MR when litter inputs are reduced, and a net result of no change in soil C beyond that which occurs following harvest.

Logging-debris retention reduced MR under high applications, due to a decrease in soil temperature and a subsequent reduction in belowground OM decomposition. Moderate amounts of logging debris had little effect on net MR despite a reduction in soil temperature, which may have been due to greater microbial activity associated with more favorable conditions of other environmental factors. Soil temperature had strong control on BR and MR at both sites, but the above results suggest other factors may have as strong a control as soil temperature on soil respiration following harvesting. Further work is needed to clarify the influence of logging debris on the soil environment, the microbial response to such modification, and variation in response associated with site-specific factors.

Significantly greater soil C at one of the sites when high amounts of logging debris were retained indicates a beneficial effect on soil C pools, with the effect due to modification of the soil environment and its influence on belowground OM decomposition rather than an increase in C inputs to soil. Increased soil C when logging debris was removed further demonstrated the large influence of belowground OM decomposition to changes in soil C following harvesting. The results suggest that logging debris may increase soil C in some situations, but such an effect may not be apparent in the presence of large increases in soil C associated with belowground OM decomposition. The stability of the C fraction contributing to increased soil C is currently unknown, and continued monitoring will be necessary to determine the net outcome on soil C pools.

Tables

Table 2.1. Site characteristics and select pre-treatment soil properties to a depth of 30 cm for study sites near Matlock, WA, and Molalla, OR.

Characteristic or property	Matlock	Molalla
Location (Latitude, Longitude)	47.206 °N, 123.442 °W	45.196 °N, 122.285 °W
Elevation (m)	118	449
Mean annual temperature (°C)	10.7	11.2
Mean annual precipitation (mm)[1]	2,400	1,600
Site index$_{50\ yr}$ (m)	34	37
Soil Texture (% sand/silt/clay)	65 / 14 / 21	37 / 34 / 29
Bulk density (Mg m^{-3})	1.45 (0.05) [2]	0.98 (0.02)
Coarse fragments by mass (%)	67.6 (1.3)	37.7 (2.2)
Total soil N (kg ha^{-1})	2,246 (88)	4,338 (173)
Total soil C (Mg ha^{-1})	66.5 (3.6)	102.2 (4.7)

[1]Precipitation was calculated from the PRISM model for period 1950-2005, (http://prism.oregonstate.edu)
[2] Standard error in parenthesis, n=8 for bulk density at Matlock, n=24 for all others).

Table 2.2. Test statistics for fixed treatment effects by year on the dependent variables soil water content, soil temperature, and bulk soil respiration at Molalla.

Effect	**Soil water content**		**Soil temperature**		**Bulk soil respiration**	
	F statistic	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**
			2006			
CVC[1]	9.68	**0.017**[2]	21.32	**0.002**	16.37	**0.005**
Debris	2.11	0.141	29.74	**<0.001**	7.16	**0.003**
CVC*debris	0.58	0.568	2.53	0.099	0.20	0.818
Month	143.56	**<0.001**	1766.69	**<0.001**	20.12	**<0.001**
CVC*month	19.80	**<0.001**	8.71	**<0.001**	7.49	**<0.001**
Debris*month	0.73	0.700	8.22	**<0.001**	2.49	**0.008**
CVC*debris*month	1.29	0.238	0.74	0.690	0.76	0.666
			2007			
CVC	6.28	**0.041**	9.43	**0.018**	20.79	**0.003**
Debris	0.89	0.423	19.17	**<0.001**	2.37	0.112
CVC*debris	0.27	0.768	2.56	0.096	0.62	0.545
Month	82.98	**<0.001**	2515.52	**<0.001**	18.93	**<0.001**
CVC*month	7.90	**<0.001**	17.81	**<0.001**	8.76	**<0.001**
Debris*month	0.28	0.986	6.90	**<0.001**	0.78	0.652
CVC*debris*month	0.66	0.757	2.08	**0.028**	0.41	0.942

[1]CVC=competing vegetation control.
[2]Test statistics in bold are significant at α=0.05.

Table 2.3. Test statistics for fixed treatment effects by year on the dependent variables soil water content, soil temperature, and bulk soil respiration at Matlock.

Effect	Soil water content		Soil temperature		Bulk soil respiration	
	F statistic	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**
			2006			
CVC[1]	14.79	**0.006**[2]	0.09	0.769	2.53	0.156
Debris	1.04	0.367	66.28	**<0.001**	1.08	0.355
CVC*debris	0.78	0.468	0.79	0.462	0.71	0.501
Month	71.73	**<0.001**	416.96	**<0.001**	85.61	**<0.001**
CVC*month	4.01	**0.002**	0.93	0.463	7.83	**<0.001**
Debris*month	0.61	0.804	6.40	**<0.001**	1.38	0.194
CVC*debris*month	1.19	0.297	0.37	0.959	1.28	0.241
			2007			
CVC	0.96	0.360	0.03	0.868	13.00	**0.009**
Debris	4.35	**0.023**	39.15	**<0.001**	0.54	0.591
CVC*debris	2.50	0.101	0.20	0.817	0.46	0.634
Month	24.88	**<0.001**	689.04	**<0.001**	84.02	**<0.001**
CVC*month	2.00	0.080	2.74	**0.020**	9.42	**<0.001**
Debris*month	0.90	0.534	12.78	**<0.001**	0.85	0.579
CVC*debris*month	1.43	0.171	0.57	0.838	1.14	0.337

[1]CVC=competing vegetation control.
[2]Test statistics in bold are significant at α=0.05.

Table 2.4. Test statistics for fixed treatment effects on microbial respiration by year and competing vegetation control treatment at the Matlock and Molalla sites.

Effect	Initial competing vegetation control		Annual competing vegetation control	
	F statistic	***p* value**	**F statistic**	***p* value**
		Matlock		
2006				
Debris (df = 2, 27)[1]	1.56	0.246	0.44	0.656
Month (df=5,201)	103.86	**<0.001**[2]	45.66	**<0.001**
D * M (10, 201)	2.55	**0.009**	1.27	0.258
2007				
Debris (df = 2, 27)	0.76	0.486	0.28	0.761
Month (df=5,201)	53.30	**<0.001**	70.93	**<0.001**
D * M (10, 201)	1.33	0.223	0.54	0.858
		Molalla		
2006				
Debris (df = 2, 13)	1.78	0.207	3.17	0.073
Month (df=5, 97)	19.65	**<0.001**	31.20	**<0.001**
D * M (10, 97)	0.88	0.555	3.71	**0.001**
2007				
Debris (df = 2, 14)	0.68	0.522	0.72	0.503
Month (df=5, 105)	23.53	**<0.001**	23.52	**<0.001**
D * M (10, 105)	1.02	0.432	2.62	**0.007**

[1] Degrees of freedom for the critical F statistic in parenthesis.
[2] Test statistics in bold are significant at α=0.05**.**

Table 2.5. Model parameters for the model $CO_2 = \beta_0 e^{\beta 1*(T-10)}$ by treatment when significant differences existed among treatments at Molalla.[1]

Effect	Bulk soil respiration	Microbial respiration
	2006	
Debris		
β_1 - all	0.059 (0.042, 0.076)[2]	0.044 (0.027, 0.061)
β_0 – 0% debris	**2.585 (2.280, 2.930)**	**2.169 (1.915, 2.456)**
β_0 – 40% debris	**2.093 (1.868, 2.345)**	2.062 (1.843, 2.307)
β_0 – 80% debris	**1.984 (1.778, 2.214)**	**1.755 (1.576, 1.955)**
CVC		
β_1 – ICVC[3]	**0.093 (0.070, 0.117)**	
β_1 – ACVC[4]	**0.062 (0.040, 0.083)**	
β_0 – ICVC	**2.303 (2.052, 2.586)**	
β_0 – ACVC	**1.816 (1.602, 2.060)**	
	2007	
Debris		
β_1 – all	0.047 (0.021, 0.073)	0.055 (0.033, 0.077)
β_0 – all		1.994 (1.780, 2.235)
β_0 – 0% debris	**2.783 (2.334, 3.318)**	
β_0 – 40% debris	2.472 (2.102, 2.908)	
β_0 – 80% debris	**2.201 (1.883, 2.574)**	
CVC		
β_1 – all	0.064 (0.042, 0.086)	
β_0 – ICVC	**3.094 (2.733, 3.503)**	
β_0 – ACVC	**1.698 (1.487, 1.938)**	

[1]All coefficients are significant at α=0.05; parameters in bold within a year and treatment grouping indicate a significant treatment contrast at α=0.05;
[2]95% confidence interval is in parenthesis.
[3]ICVC=initial competing vegetation control.
[4]ACVC=annual competing vegetation control.

Table 2.6. Model parameters for the model $CO_2 = \beta_0 e^{\beta 1*(T-10)}$ by treatment when significant differences existed among treatments at Matlock.[1]

Effect	Bulk soil respiration	Microbial respiration
	2006	
Debris		
β_1 all	0.025 (0.002, 0.049)[2]	0.054 (0.034, 0.074)
β_0 all	1.917 (1.643, 2.237)	1.318 (1.164, 1.493)
CVC		
β_1 all	0.026 (0.002, 0.049)	
β_0 – ICVC[3]	**2.082 (1.762, 2.459)**	
β_0 – ACVC[4]	**1.770 (1.500, 2.089)**	
	2007	
Debris		
β_1 all	0.080 (0.058, 0.101)	0.068 (0.049, 0.086)
β_0 all	2.005 (1.776, 2.262)	1.718 (1.549, 1.906)
CVC		
β_1 – all	0.079 (0.059, 0.099)	
β_0 – ICVC	**2.375 (2.092, 2.697)**	
β_0 – ACVC	**1.714 (1.513, 1.943)**	

[1]All coefficients are significant at α=0.05; parameters in bold within a year and treatment indicates a significant treatment contrast at α=0.05
[2]95% confidence interval is in parenthesis.
[3]ICVC=initial competing vegetation control.
[4]ACVC=annual competing vegetation control.

Table 2.7. Test statistics for treatment effects on microbial respiration for each incubation period at the Matlock and Molalla sites.

Effect	**Matlock**		**Molalla**	
	F statistic	***p* value**	**F statistic**	***p* value**
		April 2006		
CVC[1] (df = 1,3)[2]	2.49	0.212	1.92	0.260
Debris (df= 2,12)	0.41	0.672	0.26	0.775
CVC*debris (df= 2, 12)	0.67	0.530	2.65	0.111
		June 2006		
CVC (df = 1,3)	0.75	0.451	5.51	0.100
Debris (df= 2,12)	1.13	0.357	4.06	**0.045**[3]
CVC*debris (df= 2, 12)	0.73	0.504	1.21	0.334
		September 2006		
CVC (df = 1,3)	0.37	0.585	1.09	0.374
Debris (df= 2,12)	0.19	0.833	0.85	0.453
CVC*debris (df= 2, 12)	1.01	0.393	1.31	0.307
		June 2007		
CVC (df = 1,3)	0.01	0.934	0.18	0.697
Debris (df= 2,12)	0.27	0.771	0.23	0.802
CVC*debris (df= 2, 12)	1.21	0.334	0.25	0.782

[1]CVC=competing vegetation control.
[2] Degrees of freedom for the critical F statistic in parenthesis.
[3] Test statistics in bold are significant at α=0.05.

Table 2.8. Test statistics for fixed treatment effects by site on soil C concentration to a depth of 20 cm in 2005, 2007, and change in soil C concentration over the two-year period for Matlock and Molalla.

Effect	**Matlock**		**Molalla**	
	F statistic	***p* value**	**F statistic**	***p* value**
2005				
CVC[1]	1.34	0.285	0.46	0.519
Debris	3.19	0.057	0.32	0.729
CVC*debris	0.53	0.594	0.40	0.677
2007				
CVC	1.45	0.268	0.30	0.601
Debris	4.21	**0.026**[2]	0.62	0.544
CVC*debris	1.42	0.260	0.66	0.527
Change in soil C				
CVC	0.06	0.807	0.77	0.409
Debris	0.95	0.399	2.00	0.156
CVC*debris	2.62	0.091	0.43	0.654

[1]CVC=competing vegetation control.
[2] Test statistics in bold are significant at α=0.05.

Figures

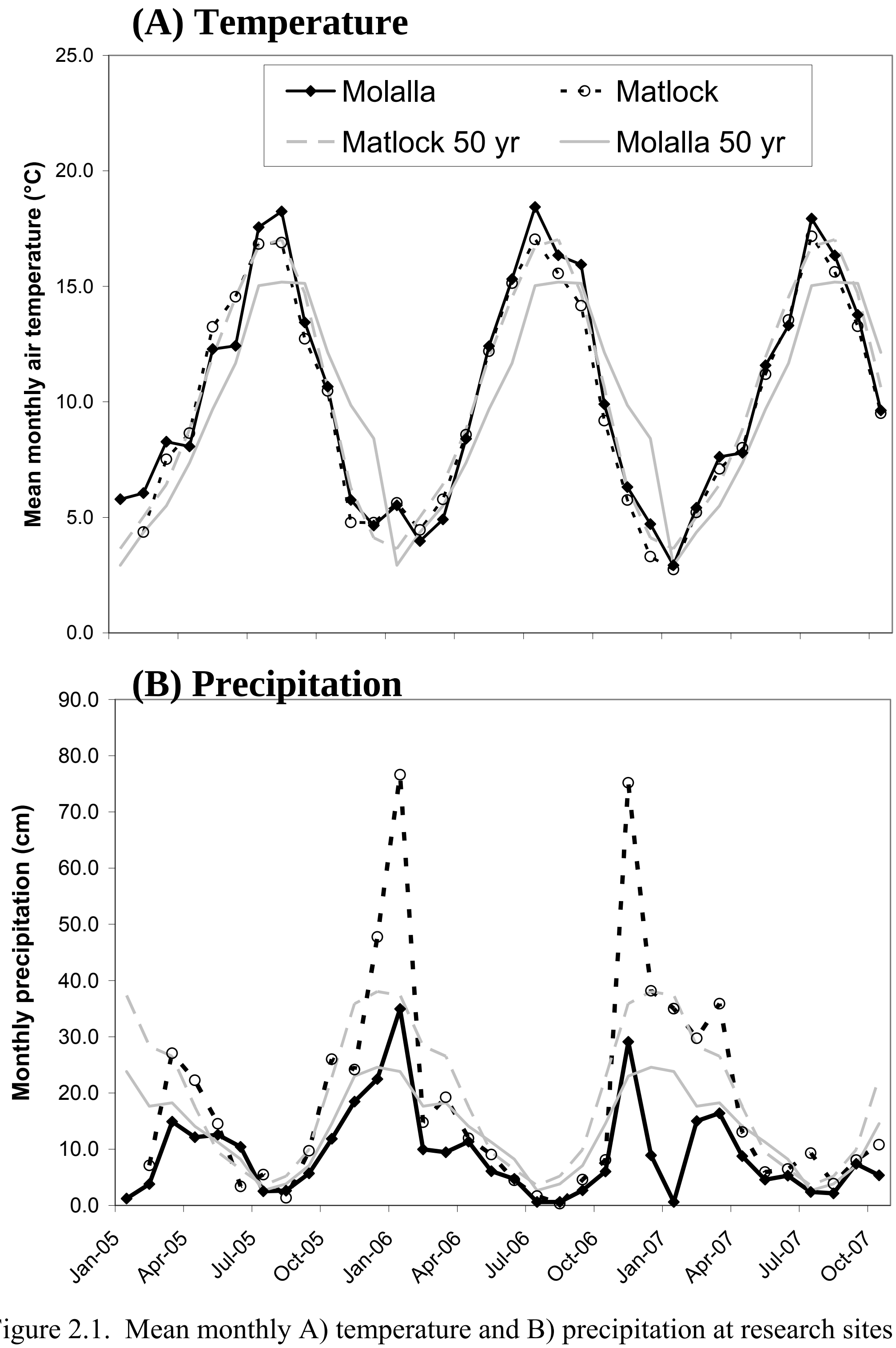


Figure 2.1. Mean monthly A) temperature and B) precipitation at research sites near Matlock, WA and Molala, OR during the period of study and the corresponding 50-yr monthly average.

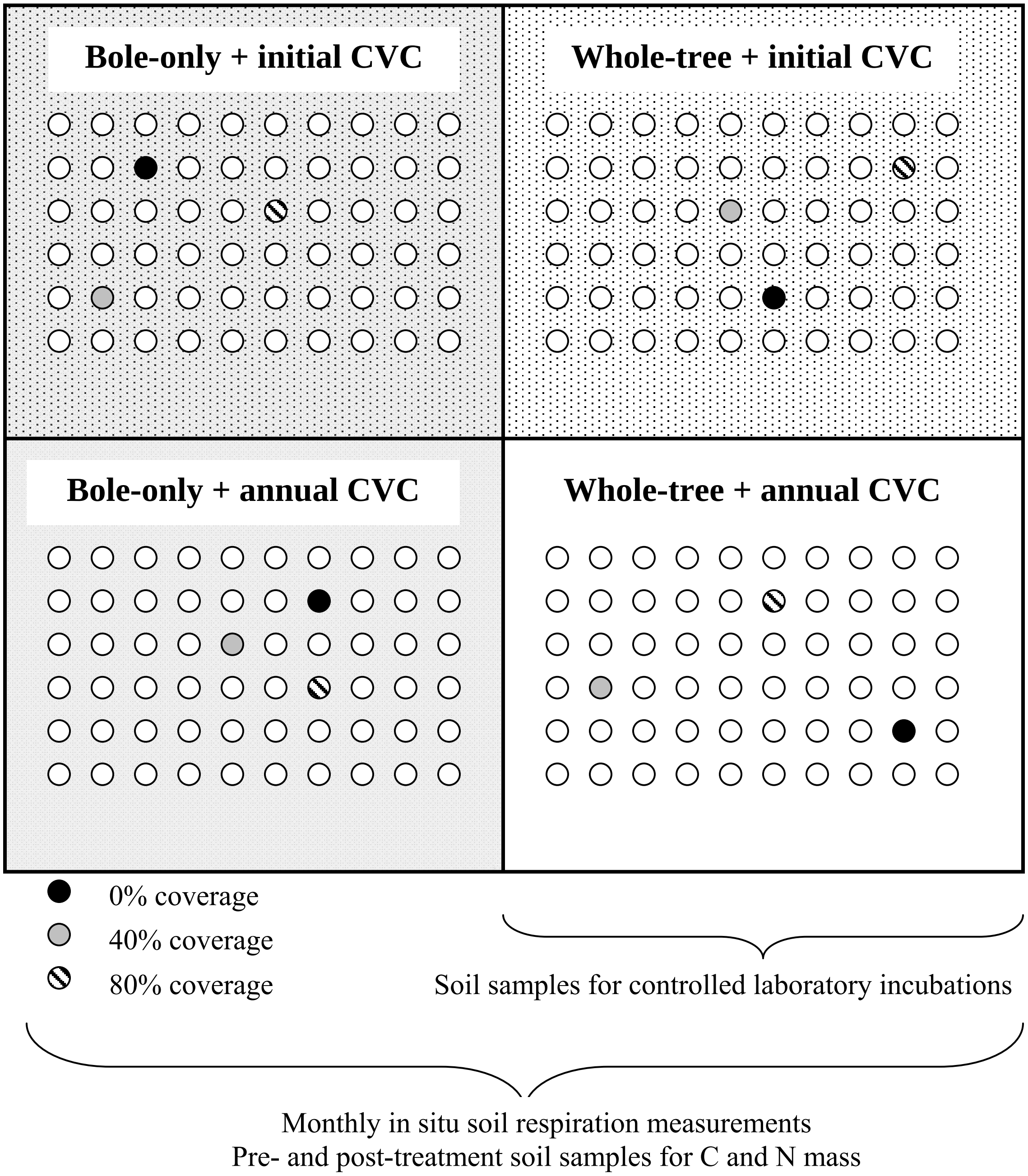


Figure 2.2. Schematic diagram of experimental design showing whole-plot factors harvest type and competing vegetation control (CVC), subplot factor logging-debris coverage, and the measures associated with each. Measures were performed in the subplots only. Schematic represents one of four blocks present at the Matlock and Molalla sites.

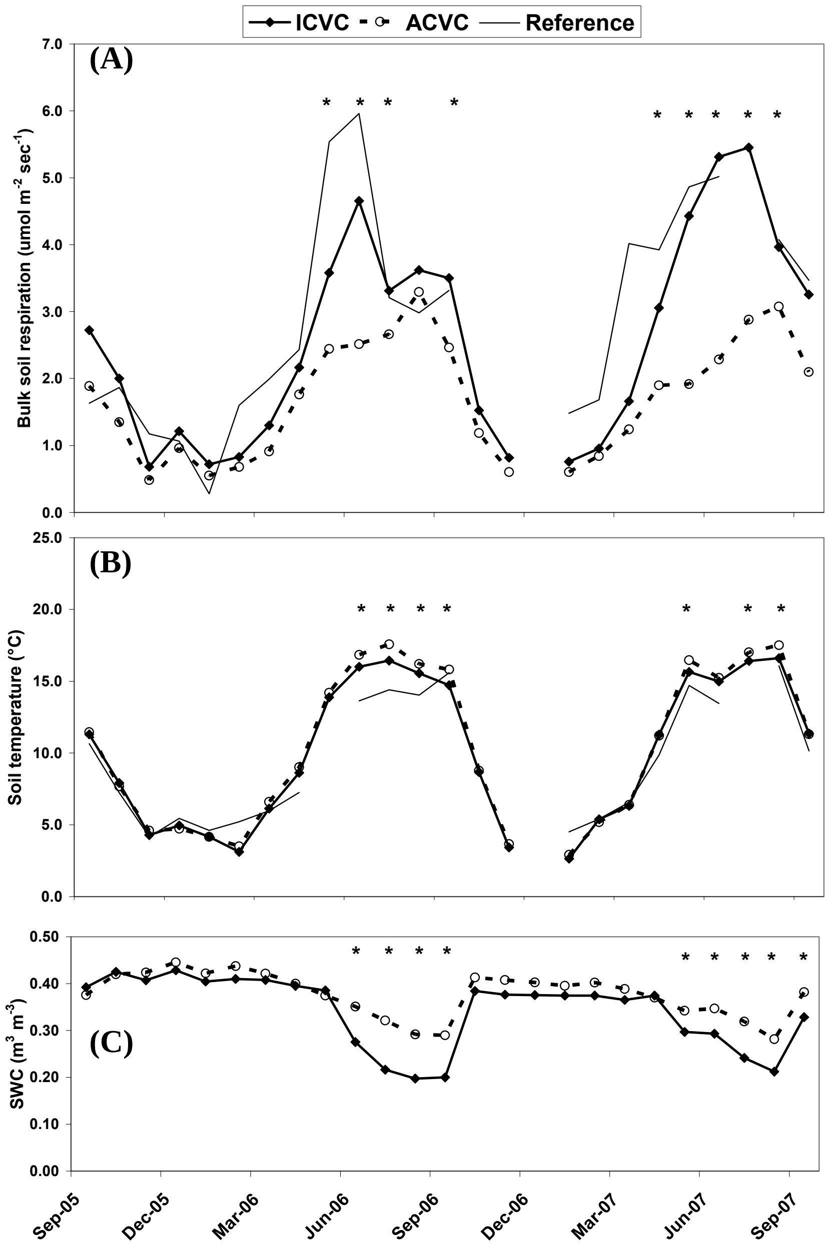


Figure 2.3. (A) Bulk soil respiration, (B) soil temperature, and (C) volumetric soil water content (SWC) by sample month and competing vegetation control treatment at **Molalla.** ICVC=initial competing vegetation control, ACVC = annual competing vegetation control. * indicates significant difference between treatments for the associated month. Only data from April to September in each year were analyzed.

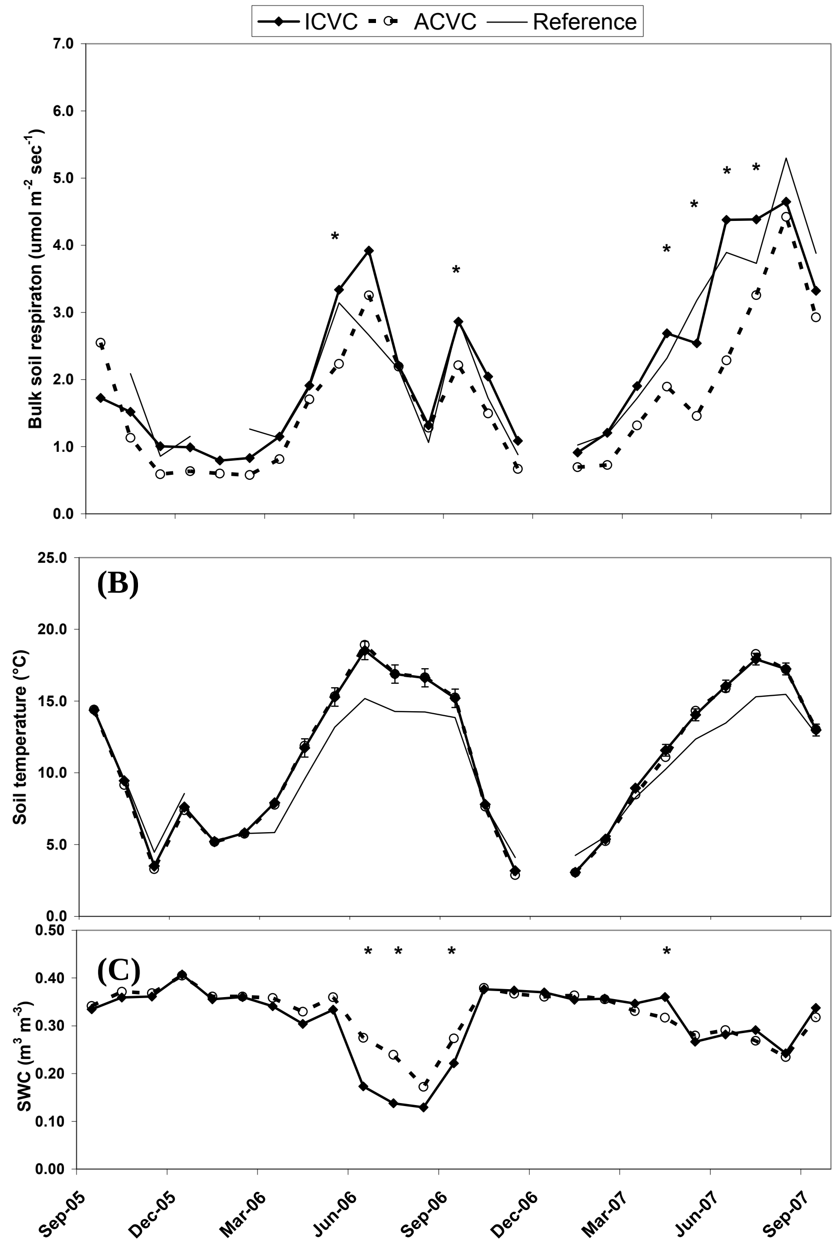


Figure 2.4. (A) Bulk soil respiration, (B) soil temperature, and (C) volumetric soil water content (SWC) by sample month and competing vegetation control treatment at **Matlock**. ICVC= initial competing vegetation control, ACVC = annual competing vegetation control. * indicates significant difference between treatments for the associated month Only data from April to September in each year were analyzed.

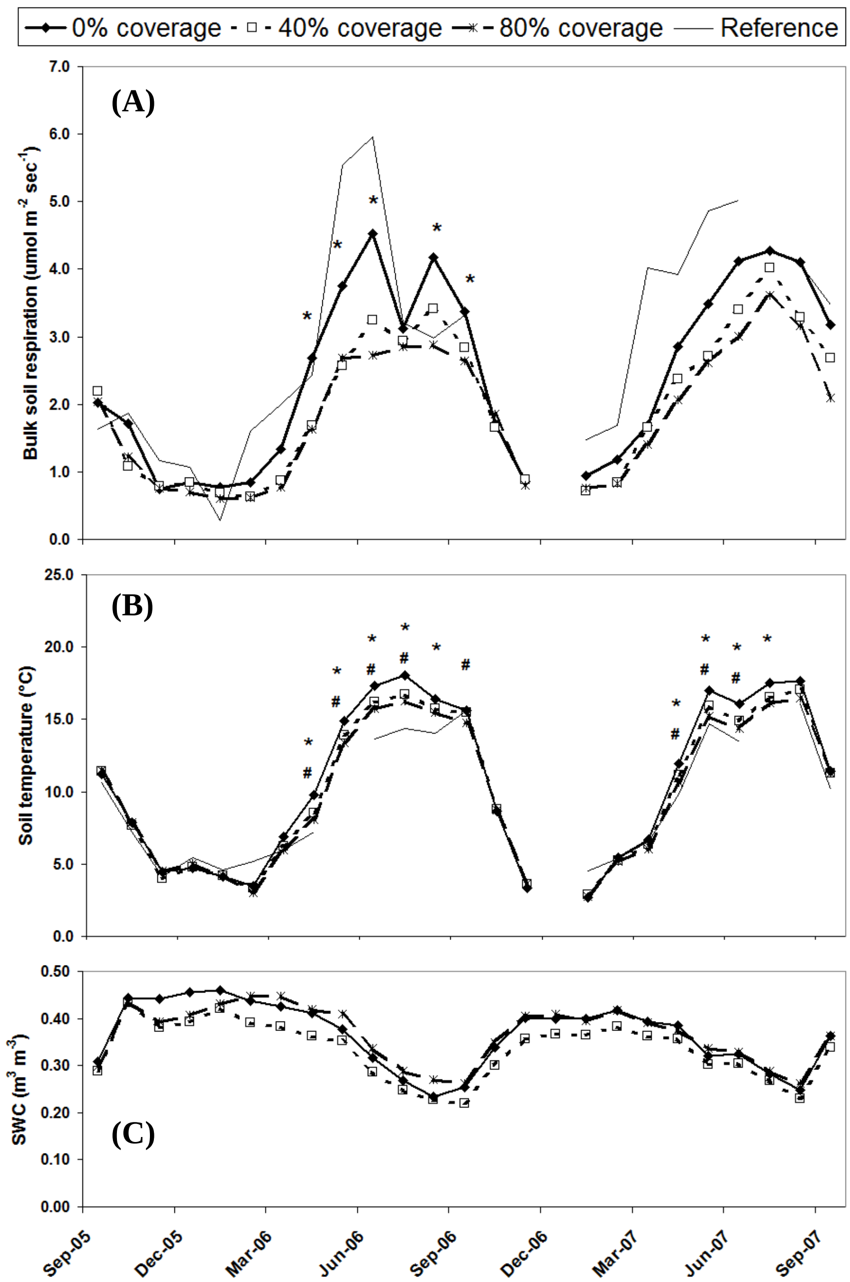


Figure 2.5. (A) Bulk soil respiration, (B) soil temperature, and (C)volumetric soil water content by sample month and logging-debris treatment at **Molalla**. * indicates significant contrast between absence and presence of logging debris, # indicates significant contrast between the 40% and 80% logging-debris coverage. Only data from April to September in each year were analyzed.

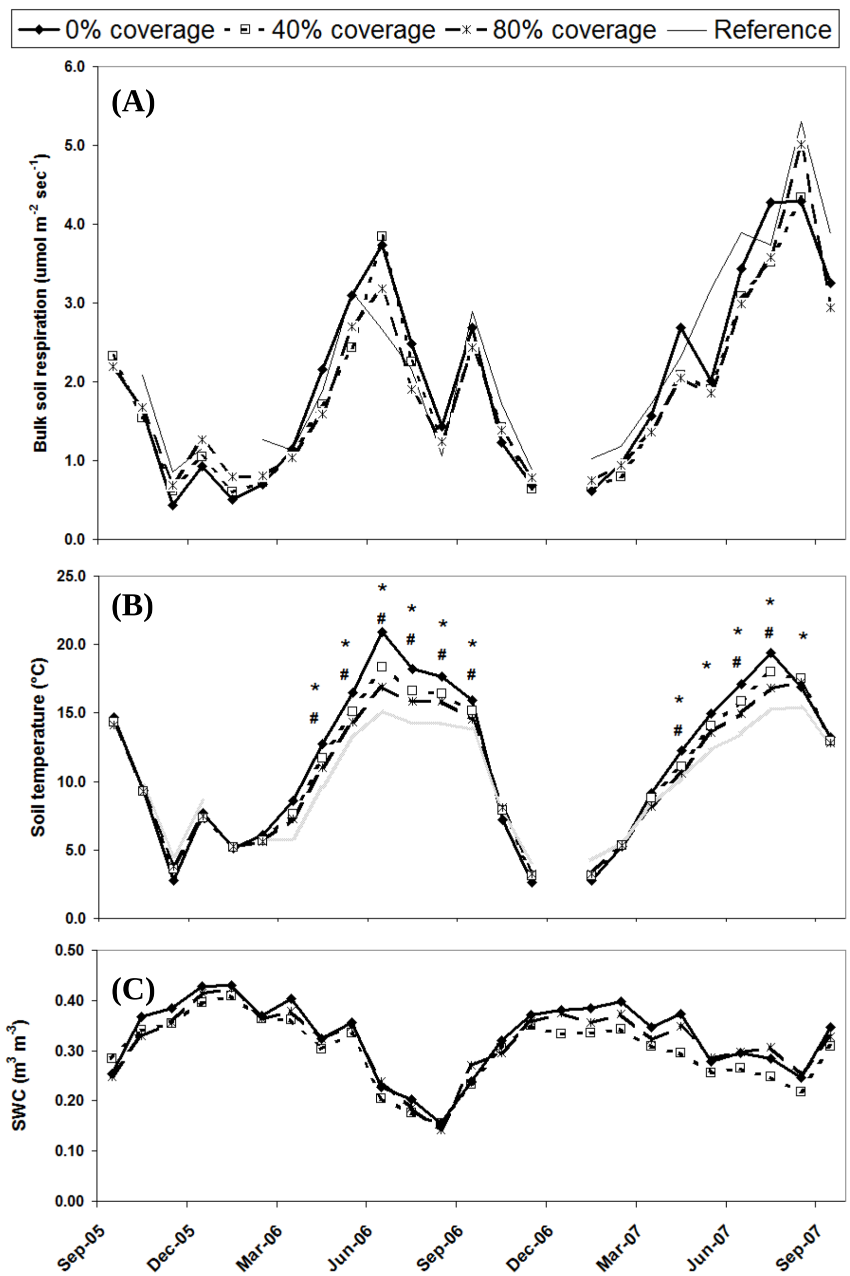


Figure 2.6. (A) Bulk soil respiration, (B) soil temperature, and (C) volumetric soil water content by sample month and logging-debris treatment at **Matlock**. * indicates significant contrast between absence and presence of logging debris, # indicates significant contrast between the 40% and 80% logging-debris coverage. Only data from April to September in each year were analyzed.

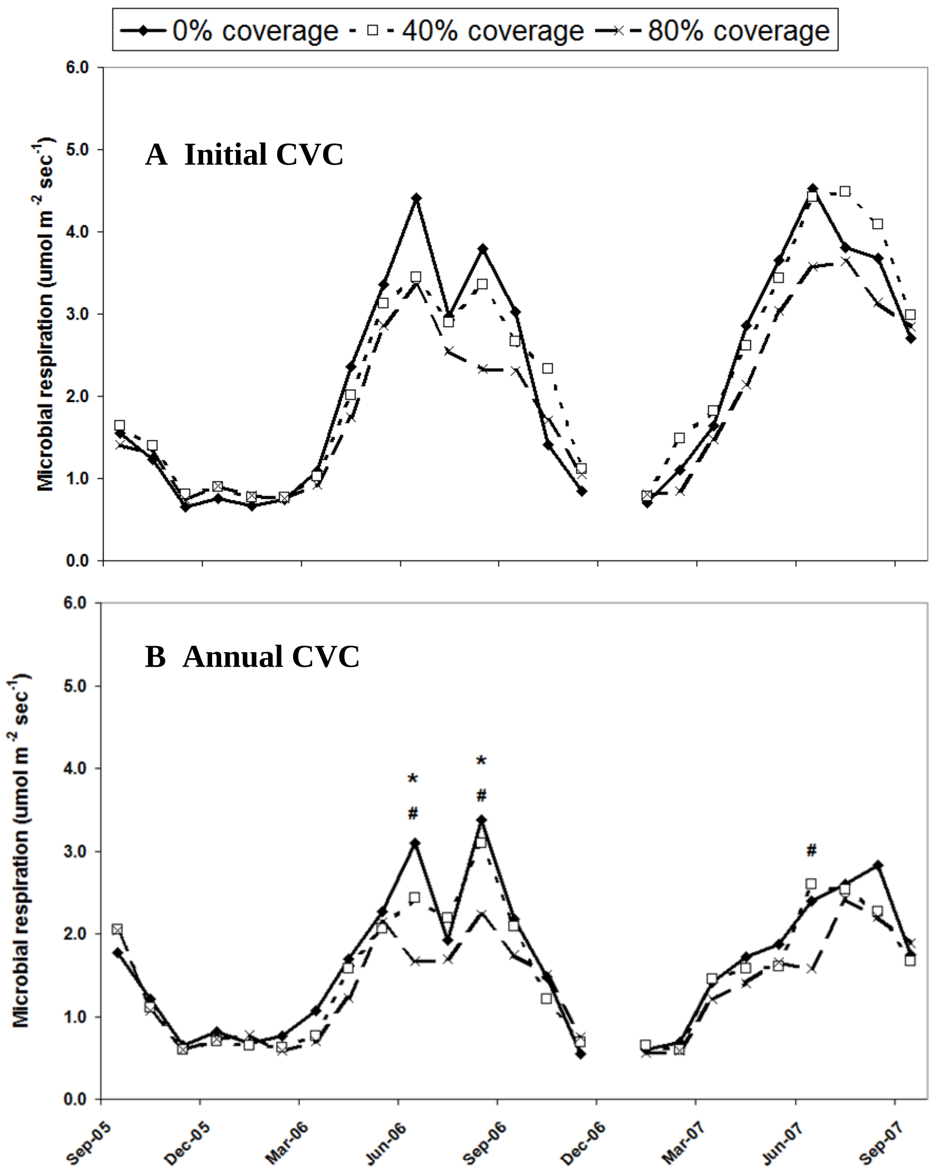


Figure 2.7. Microbial soil respiration by sample month and logging-debris treatment at **Molalla** with (A) initial competing vegetation control (CVC) , and (B) annual competing vegetation control. * indicates significant contrast between absence and presence of logging debris, # indicates significant contrast between the 40% and 80% logging-debris coverage. Only data from April to September in each year were analyzed.

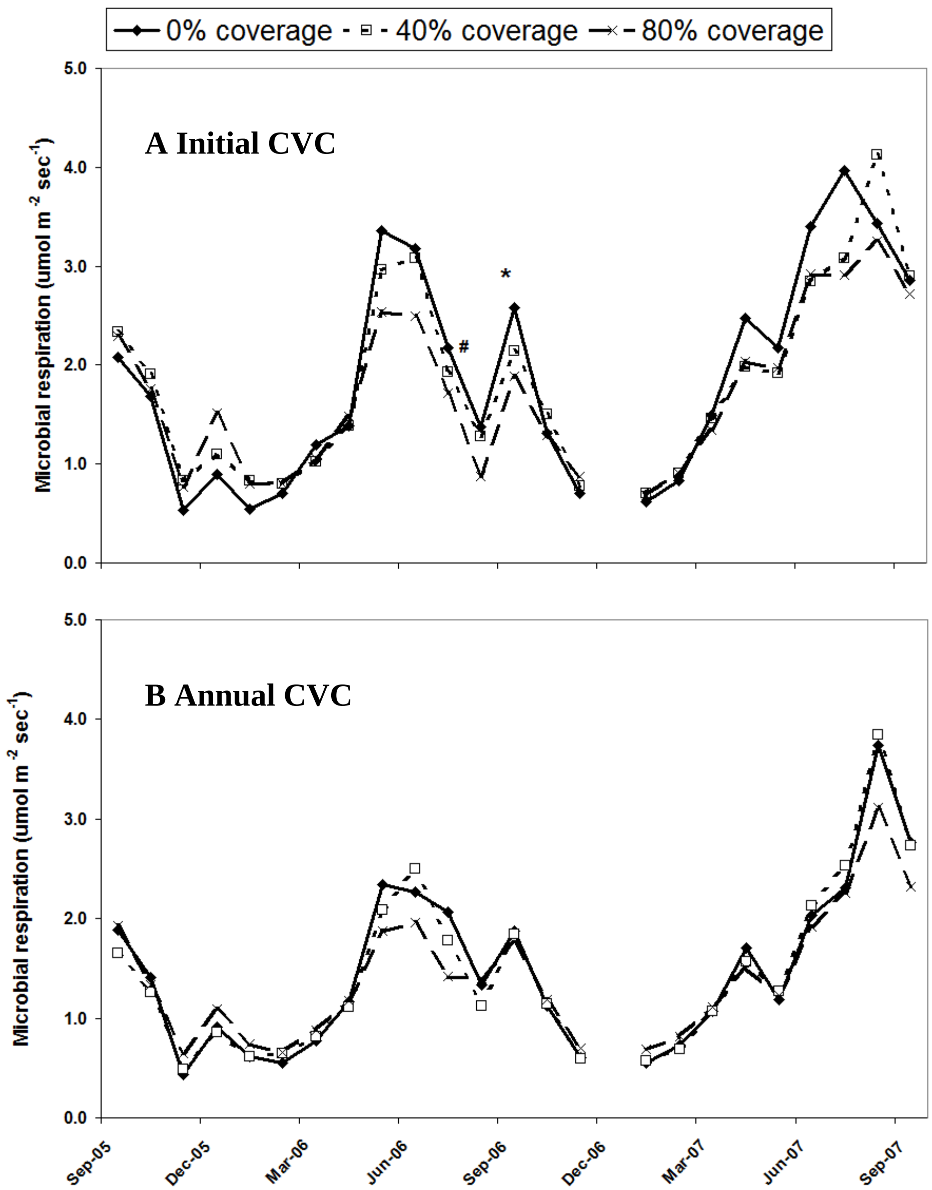


Figure 2.8. Microbial soil respiration by sample month and logging-debris treatment at **Matlock** with (A) initial competing vegetation control (CVC) and (B) annual competing vegetation control. * indicates significant contrast between absence and presence of logging debris, # indicates significant contrast between the 40% and 80% logging-debris coverage. Only data from April to September in each year were analyzed.

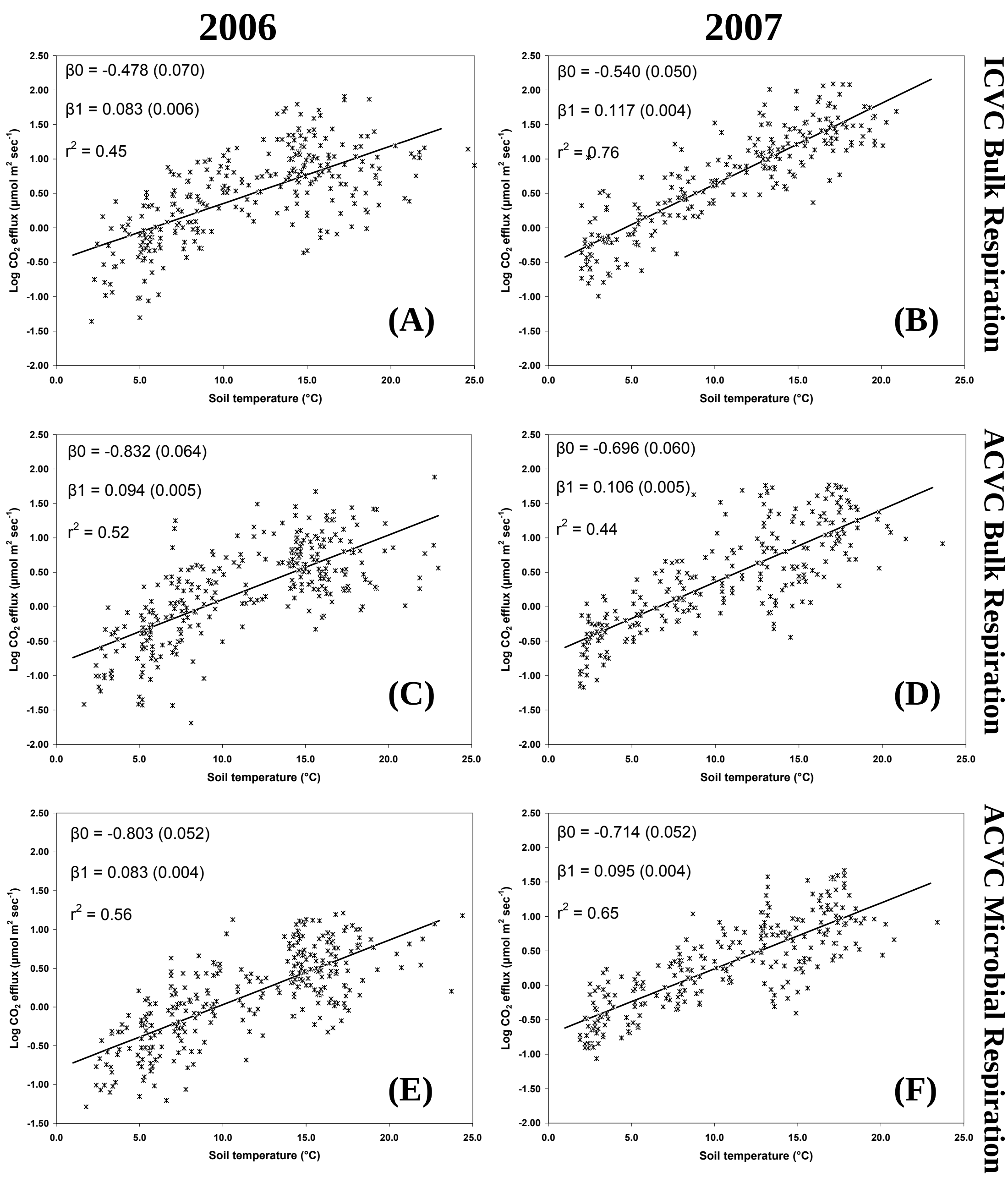


Figure 2.9. Estimated parameters for the log-transformed equation CO_2 flux = $\beta 0 e^{\beta 1 * T}$, coefficient of determination, and observed data points by competing vegetation control treatment and year at Matlock for (A and B) bulk soil respiration when initial competing vegetation (ICVC) was applied, (C and D) bulk soil respiration when annual competing vegetation (ACVC) was applied, and (E and F) microbial respiration when annual competing vegetation was applied. Approximate standard error of each parameter is in parenthesis.

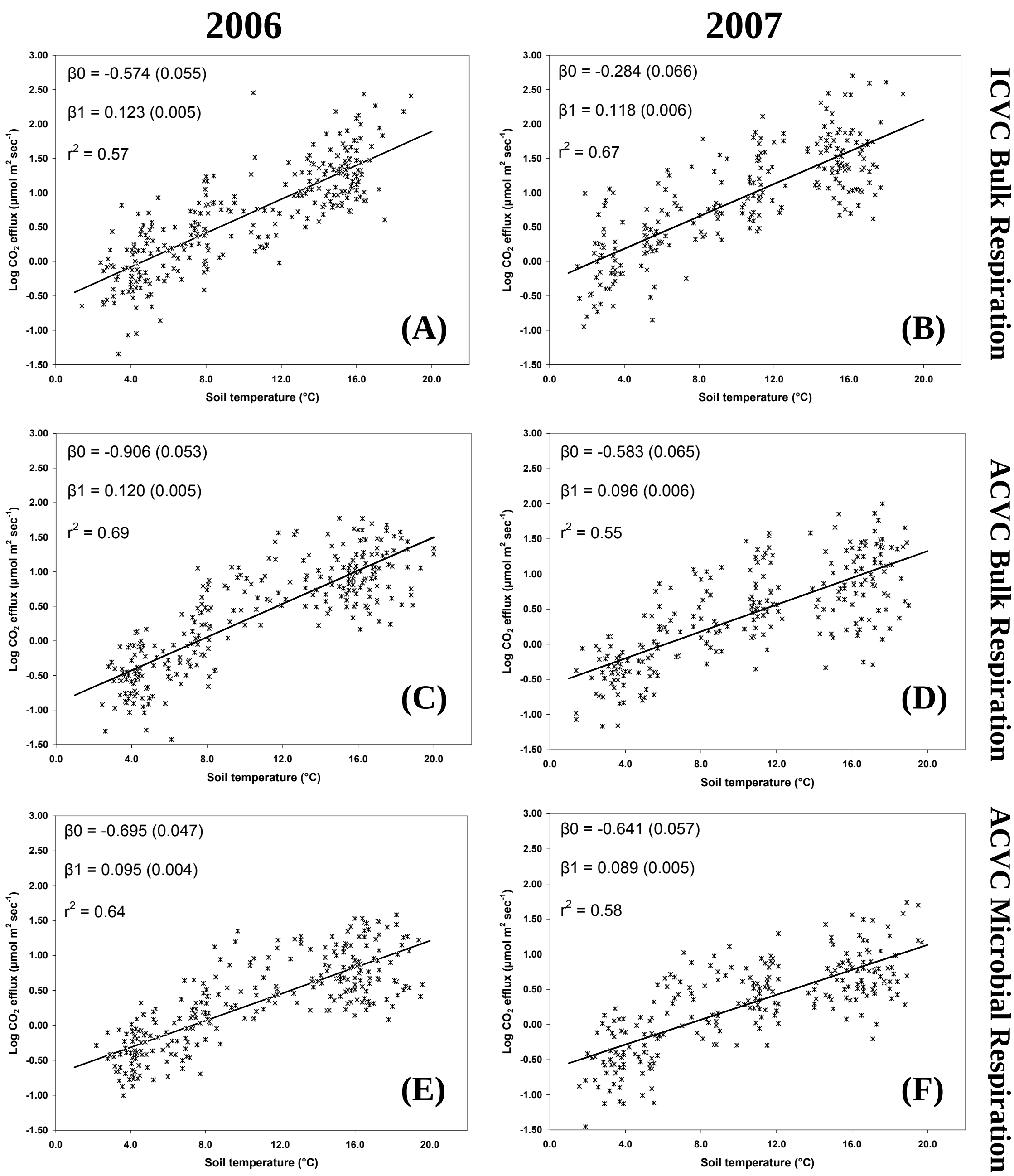


Figure 2.10. Estimated parameters for the equation CO_2 flux = $\beta 0 e^{\beta 1 * T}$, estimated coefficient of determination, and observed data points by competing vegetation control treatment and year at Molalla for (A and B) bulk soil respiration when initial competing vegetation control (ICVC) was applied, (C and D) bulk soil respiration when annual competing vegetation control (ACVC) was applied, and (E and F) microbial respiration when annual competing vegetation control was applied. Approximate standard error of each parameter is in parenthesis.

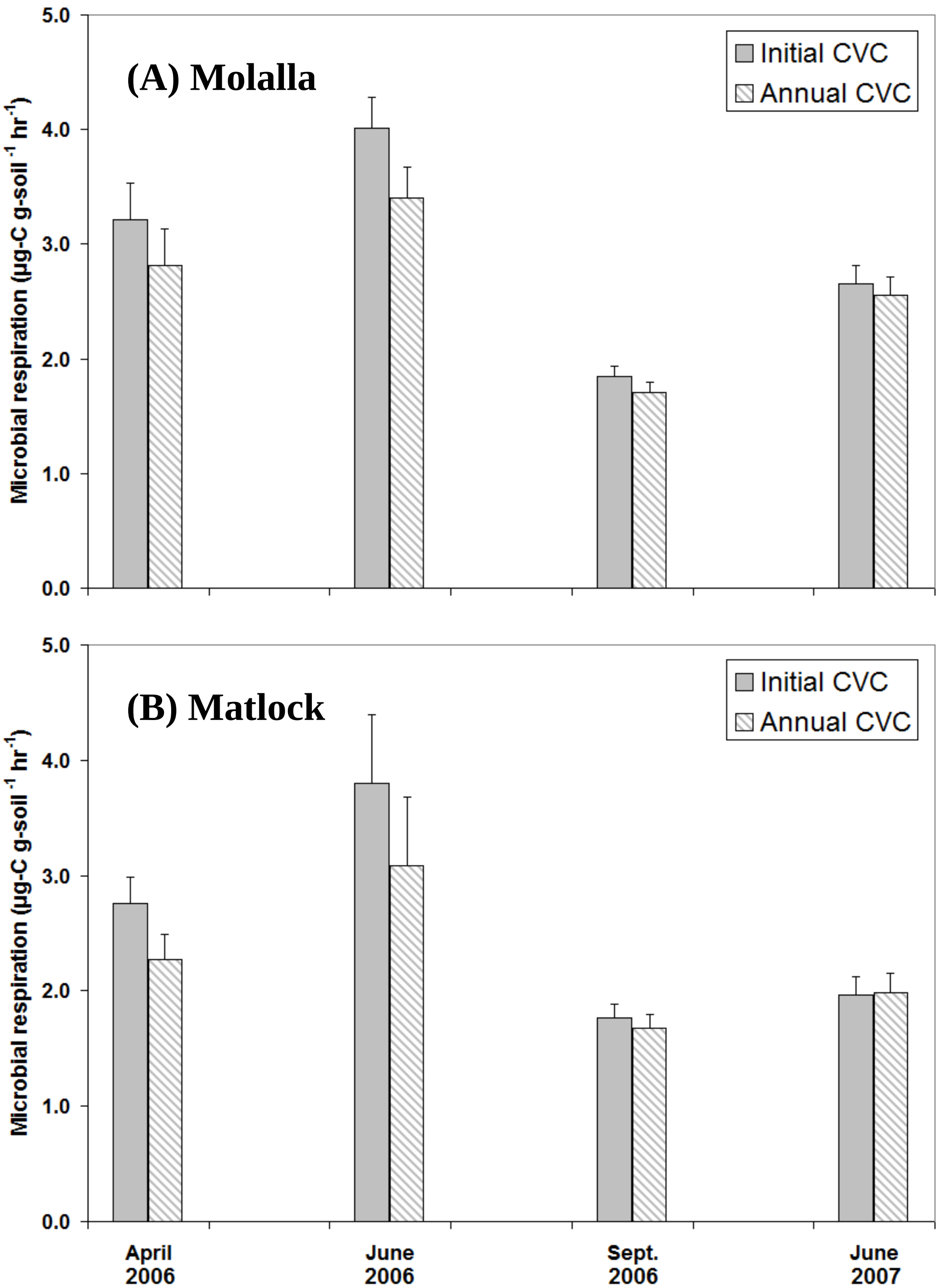


Figure 2.11. Mean microbial respiration rates determined in a 17-day incubation by competing vegetation control treatment and sample period at (A) Molalla, and (B) Matlock. CVC=competing vegetation control. Error bars are standard error of the mean. There were no significant differences between treatments for any sample period.

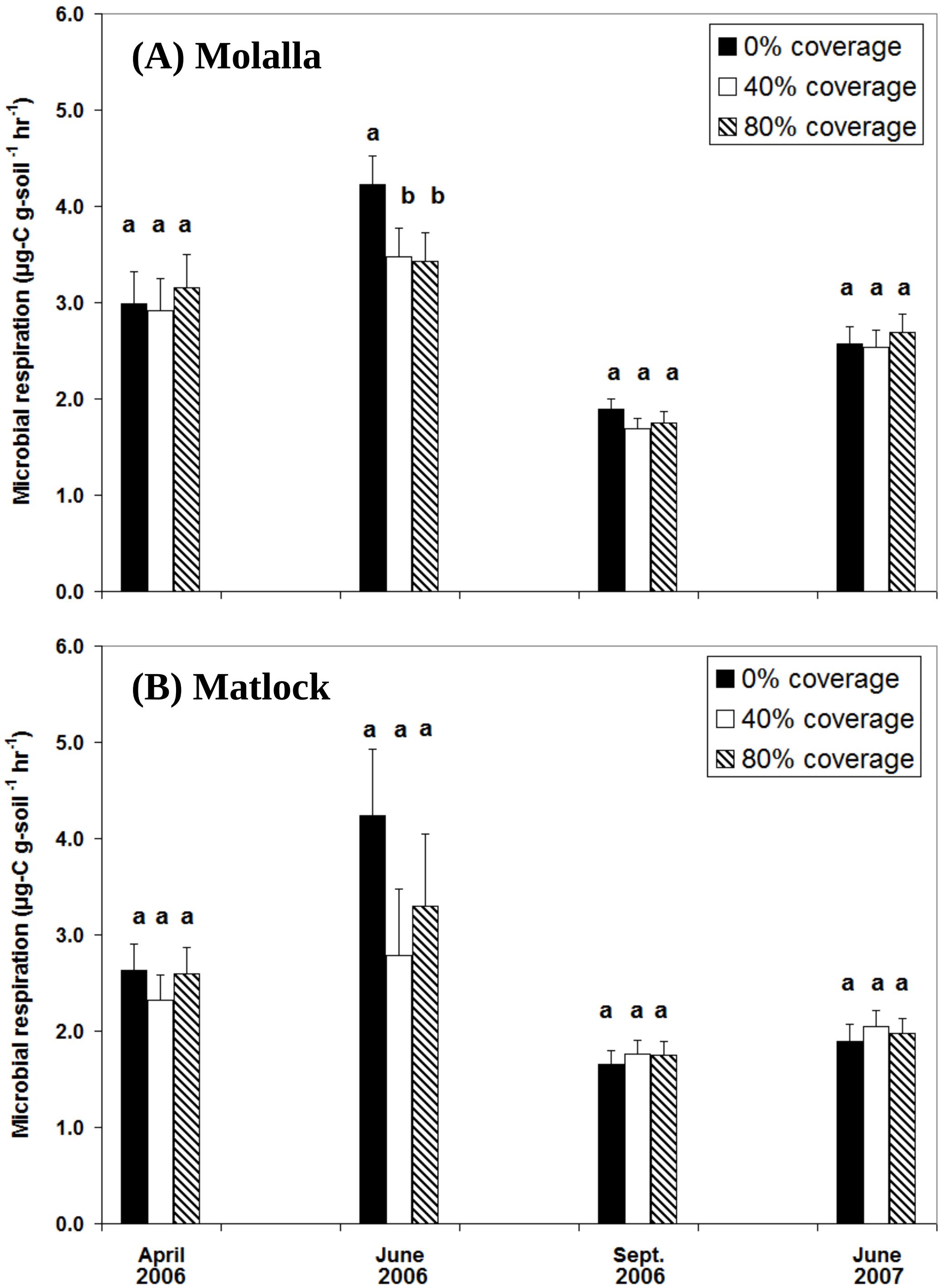


Figure 2.12. Mean microbial respiration rates determined in a 17-day incubation by logging-debris coverage treatment and sample period at (A) Molalla, and (B) Matlock. Means within a sample period followed by a different letter are significantly different at α=0.05. Error bars are standard error of the mean.

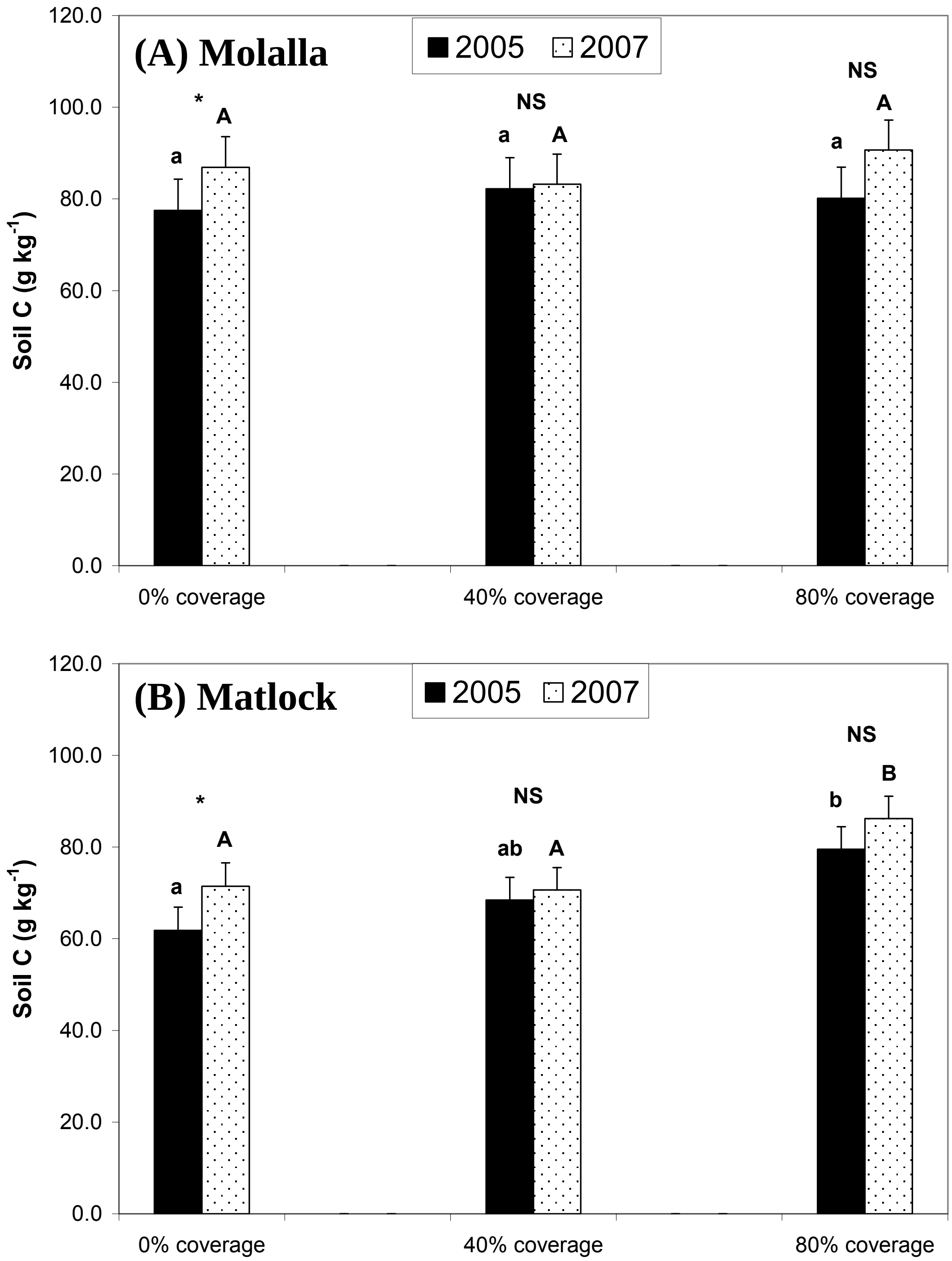


Figure 2.13. Soil C concentration to a depth of 20 cm by sample period and logging-debris treatment at (A) Molalla, and (B) Matlock. Means within a year followed by a different letter (lowercase letters for 2005 and uppercase letters for 2007) are significantly different at α=0.05. * and NS above a treatment indicate significant (α=0.05) and non-significant differences between the two measurements. Error bars are standard error of the mean.

Chapter 3

Dissolved Carbon and Nitrogen Leaching Following Logging-debris Retention and Competing Vegetation Control in Douglas-fir Stands in the Pacific Northwest

Abstract

Management practices at time of forest harvesting have potential to alter leaching of soluble carbon (C) and nitrogen (N), but the effect is likely to vary with site factors. I examined the effect of logging-debris retention and competing vegetation control (CVC, initial or annual applications) on dissolved organic C (DOC), dissolved organic N (DON), and nitrate-N concentrations in soil water at 60 cm depth to determine the relative potential of these practices to contribute to soil C and N loss. The experiment was installed in 2005 (two years post-harvest) at two sites that differ in soil characteristics and annual precipitation. Soil water was collected monthly for two years. At both sites, annual CVC resulted in significantly higher soil water nitrate-N concentration and flux, but the magnitude and duration of the effect was greater at the site with relatively high initial soil N. Differences between sites were largely due to greater water-leachable nitrate production at the high-N site, but greater N uptake by crop trees at the site with relatively low initial N also contributed. Dissolved organic N concentration was significantly increased at the high-N site in March of each year following annual CVC, but the contribution of this increase to total N concentration was small (2-4% of total N flux). Dissolved organic N comprised a large portion of total dissolved N at the low-N site such that there were no differences in total dissolved N between the two CVC

treatments despite the increases in nitrate-N when annual CVC was applied. At the low-N site, most of the increase in nitrate-N following annual CVC could be attributed to treatments where logging debris was retained, likely due to greater ammonium availability associated with higher total soil N. There was not a significant effect of logging-debris retention or CVC treatment on soil water DOC concentrations, indicating that DOC inputs from logging debris and competing vegetation were either retained or consumed in the mineral soil. Soil water DOC concentrations were consistently higher when no logging debris was retained at both sites in the first year, possibly due to greater DOC production in mineral soil associated with higher soil temperatures. The estimated increase in leaching flux of dissolved C and N associated with annual CVC or removal of logging debris was low relative to total soil pools, making it unlikely that loss of these elements at this magnitude via leaching will negatively affect future soil productivity at these sites.

Introduction

Forest management is becoming increasingly intensive due to greater demand for wood products and a reduction in land available for production (FAO, 2006). Consequently, some have cautioned on the potential for a reduction in soil productivity and subsequent declines in forest productivity over the long-term when intensive forest management is practiced (Fox, 2000; Nambiar, 1996). Past research has focused on retention of soil carbon (C) (as a surrogate for soil organic matter, SOM) and nitrogen (N) to maintain soil productivity given the critical role of SOM for nutrient supply, gas exchange, and water holding capacity (Powers et al., 1990), and the common N limitation to tree growth in many areas of North America (Johnson, 1992; Keeney, 1980). Logging debris contains significant amounts of C and N (Powers et al., 2005), and there is concern that increased biomass removal (e.g. utilization of all logging-debris residue) could result in reduced pools of soil C and N. The underlying assumption behind this concern is that changes in soil C and N following harvesting are largely dependent on aboveground inputs. However, several long-term studies (> 10 yr) have found no lasting effect of logging-debris retention on total pools of soil C and N following harvest (Johnson et al., 2002; Olsson et al., 1996), including summary findings from 36 installations in the Long Term Soil Productivity (LTSP) network that covered a wide range of site and climate conditions (Powers et al., 2005).

The above results contrast with those from several studies that measured large inputs of dissolved organic C (DOC) and dissolved organic N (DON) to mineral soil from logging debris following harvesting (Mattson et al., 1987; Qualls et al., 2000; Robertson et al., 2000), which presumably have potential to increase C and N in the mineral soil. However, there is also potential for the inputs to be balanced by increased outputs via C consumption (microbial respiration), denitrification, or leaching of DOC and dissolved forms (inorganic and organic) of N. Several studies have shown increased nitrate leaching following harvesting (e.g. Vitousek et al., 1997), but the effect of logging-debris retention on this increase is unclear. Strahm et al. (2005) observed significantly greater nitrate leaching in a bole-only harvest compared to a whole-tree harvest at a highly productive site in southwestern Washington, a response that has been observed in the Great Lakes region as well (Hendrickson et al., 1989). In contrast, both Vitousek and

Matson (1985) and Carlyle et al. (1998) found decreased nitrate leaching when logging debris was retained due to an increase in microbial immobilization (Vitousek and Matson, 1985). Mann et al. (1988) found no discernable effect of varying logging-debris retention following harvesting on N leaching at l1 sites across the conterminous U.S. The variability in leaching response is most likely a function of site-specific factors (Gundersen et al., 2006), underlying a need to evaluate the effect of logging-debris retention on N leaching under a variety of site conditions.

Nitrogen leaching following harvesting is generally dominated by inorganic N forms (primarily nitrate), but there is also potential for increased N loss as DON (Sollins and McCorison, 1981). Carlyle et al. (1998) found a 25-30% reduction in DON leaching when logging debris was retained following harvesting in southeastern Australia, but Strahm et al. (2005) observed no effect of logging-debris retention on DON following harvest at a productive site in southwestern Washington. Dissolved organic N comprised 15-45% of total N at the Australia site, but only 2-5% of total N at the Washington site. Soil N concentration at the Australia site was low (sandy soil, 1.1 g N kg^{-1} at 0-15 cm depth), but that at the Washington site was high (silt loam soil, 4.6 g N kg^{-1} at 0-12 cm depth), possibly indicating that the response may be a function of pre-harvest soil N pools. However, the differing response could be due to any number of factors, as there is generally a poor understanding of processes that govern DON production and loss from soil (Kalbitz et al., 2000; McDowell, 2003).

The potential for increased DOC loss (i.e. leaching below the rooting zone) following harvesting appears to be lower than for N. Qualls et al. (2000) concluded that

DOC inputs from logging debris following harvesting of a deciduous forest in the southeastern U.S. were effectively retained in the mineral soil, and a similar conclusion was reached by Piirainen et al. (2002) following harvest of a coniferous forest in Finland. Others have noted no general effect of harvesting on DOC (McDowell and Likens, 1988), or a general decrease in DOC export (Meyer and Tate, 1983). In the Qualls et al. (2000) study, adsorption to mineral surfaces was proposed as the primary retention mechanism, but microbial consumption may have also contributed to the decrease in DOC. These studies generally indicate low potential for DOC leaching following logging-debris retention, but the effect of logging-debris removal is less clear. Removal of logging debris causes an increase in soil temperature and a change in soil moisture (Devine and Harrington, 2007; Roberts et al., 2005), which is likely to modify rates of microbial activity and DOC production in soil. In the LTSP summary findings, Powers et al. (2005) and Sanchez et al. (2006) attributed increased soil C and N following harvesting to rapid decomposition of the root system of the previous stand, with the effect most pronounced when all surface OM was removed, presumably due to increased soil temperature. Greater decomposition when logging debris is removed could result in greater DOC leaching, but this possibility has not been evaluated.

Other management practices at time of harvest and the initial years following planting have potential to influence C and N loss via leaching. Competing vegetation control (CVC) is commonly used in the initial years after planting to increase crop tree survival and growth (Harrington et al., 1995; Newton and Preest, 1988). Vegetation is an important factor controlling N retention following harvesting (Marks and Bormann, 1972; Vitousek et al., 1979), and several studies have documented elevated nitrate leaching following CVC (Briggs et al., 2000; Smethurst and Nambiar, 1995; Vitousek and Matson, 1985). Increase in nitrate leaching has largely been attributed to reduced vegetative uptake, but a reduction in organic matter (OM) inputs to soil and subsequent reduction in microbial immobilization has also been shown to contribute in some regions (e.g. Vitousek and Matson 1985). Vitousek and Matson (1985) found that nitrate leaching in the fall after CVC treatments in the Piedmont region of North Carolina was strongly related to the nitrate pool size in late summer rather than rates of production. Their results suggest that leaching during winter months (when water flux is greatest) is largely a function of factors that control the accumulation of nitrate during the growing season when environmental conditions are conducive to nitrification. This is probably especially relevant in regions that have summer drought, as the reduction in soil water flux during times when nitrate production is high may lead to substantial rates of nitrate leaching in winter months.

Competing vegetation control also has strong potential to alter the production and transport of DOC. Both above- and below-ground litter contribute to DOC production and flux through mineral soil, but the relative contribution of each to DOC loss at deep soil horizons is poorly understood (Kalbitz et al., 2000). A recent study observed decreased DOC in mineral soil following tree girdling (Giesler et al., 2007), suggesting root inputs (e.g. exudates, turnover) are a more important source of DOC in mineral soil. Froberg et al. (2007) determined that much of the DOC collected from soil water at 70 cm was derived from sources in the mineral soil rather than aboveground inputs.

Similarly, in a column study, Uselman et al. (2007) demonstrated that DOC leaching at 50 cm depth was derived largely from root litter rather than aboveground inputs. These studies indicate practices that reduce root inputs may cause a concurrent reduction in DOC leaching. Competing vegetation control reduces both above- and below-ground C inputs to soil, which could cause a reduction in DOC leaching. However, CVC also modifies the soil environment and increases soil water flux, so the net effect of CVC on DOC flux could be to increase C loss from soil. There is potential for CVC to modify DOC leaching following harvesting, but no experimental studies have been performed to examine this possibility.

The Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests of the Pacific Northwest are some of the most productive in the world, and management intensity is expected to increase in the future, primarily in intensively managed plantations (Adams et al., 2005; Moores et al., 2007). It is likely that biomass removal will increase (magnitude and frequency) during that time, and the use of CVC will probably be more frequent and of longer duration. There is a need to understand the main and interactive effects of these practices on N and C leaching across a range of site conditions, but only a few experimental studies have been performed in the region (e.g. Strahm et al. 2005). The objectives of this research are to determine the effect of increasing logging-debris retention in combination with either initial or annual applications of CVC on 1) DOC and N (nitrate, ammonium, DON) concentrations and flux below the rooting zone to determine the potential for leaching, and 2) production of water-leachable DOC and N under controlled laboratory incubations to infer the causal mechanisms contributing to the results observed *in situ*. Two sites with contrasting soil properties and annual precipitation were used to determine if observed affects varied with those variables.

Methods

Site Characteristics

This study is part of a larger research project initiated at two sites in 2003 to assess effects of logging-debris retention and CVC treatments on soil properties, nutrient cycling, and Douglas-fir growth. Both study sites are affiliates of the LTSP network (Powers et al. 1990). Potential productivity (site index) is similar between sites, but large differences exist in precipitation and soil properties (Table 3.1). Site 1 (hereafter referred to as Matlock) is located on the Olympic Peninsula in WA, approximately 45 km NW of Olympia near the town of Matlock. Soil at Matlock is classified as a sandy-skeletal, mixed, mesic, Dystric Xerorthents, formed in glacial outwash with slopes ranging from 0 to 3% (Soil Survey Staff, USDA-NRCS). Site 2 (hereafter referred to as Molalla) is located approximately 24 km NE of the town of Molalla, OR in the foothills of the western Cascades. Soil at Molalla is classified as fine-loamy, isotic, mesic Andic Dystrudepts, formed in basic agglomerate residuum with slopes ranging from 2 to 40% (Soil Survey Staff, USDA-NRCS). The regional climate is Mediterranean, with mild, wet winters and dry, warm summers with periods of prolonged drought (> 2 mo) being common. Precipitation falls almost entirely as rain, but some snowfall does occur during winter months.

Experimental design and treatment application

Sites were initially clear-cut harvested with chainsaws in March (Molalla) and April (Matlock) of 2003. Trees were removed with ground-based mechanized equipment along marked machine trails that were evenly distributed across plots to minimize experimental error associated with soil disturbance. Following harvest, a 2 x 2 randomized complete block factorial design was installed at each site. The factors were harvest type (two levels - bole–only or whole-tree) and CVC (two levels - initial CVC or annual CVC). The factorial combinations were replicated four times in a randomized complete block design and applied to 0.3 ha plots (50 x 60 m). All plots received an initial application of herbicide to reduce competing vegetation; at Molalla glyphosate was aerially applied in August, 2003 and triclopyr was applied with backpack sprayers at Matlock during September of 2003. Following this initial application, only those treatments assigned annual CVC were treated with the same herbicides as applied initially in the spring of each year. Both sites were planted with plug+1 bare-root Douglas-fir seedlings in February (Molalla) and March (Matlock) of 2004 at a 3 x 3 m spacing (1,111 trees ha^{-1}). Each site was enclosed with a 2.5 m high fence to prevent browse damage to seedlings.

In March of 2005, three subplots within each of the whole-tree harvest treatment plots (n=8) were identified for application of a subplot logging-debris retention treatment. Subplots encompassed a 2 x 2 m area centered on a single planted Douglas-fir seedling. This design modification was chosen to reduce experimental error associated with treatment application (e.g. discontinuous logging-debris coverage at the whole-plot level) and spatial variability of soil chemical properties. Use of the whole-tree treatment plots was arbitrary (i.e. compared to use of the bole-only plots) as we only had sufficient resources for 24 subplots at each site, and could not address potential interaction between the whole-plot harvest factor and the subplot factor given the level of replication. Woody logging debris was randomly applied at a visual surface coverage of 0%, 40%, or 80% to one of the subplots in each whole-plot. For each assigned treatment application, logging debris 5.0 to 12.5 cm in diameter that was within the associated whole-plot was stacked in a systematic criss-cross fashion until the assigned coverage (+/- 10% with visual determination) was reached. In the case of the 0% treatment, all logging debris was removed from the subplot, but no attempt was made to remove legacy wood if present. The overall modified design used in this study is a randomized complete block split-plot with one whole-plot factor (CVC treatment) and one subplot factor (logging-debris coverage).

Lysimeter installation and soil water collection

Tension lysimeters were fabricated with high-flow (100 kPa) porous ceramic cups (maximum pore size of 2.5 μm)(Soil Moisture Corp., Santa Barbara, CA) and PVC tubing. Lysimeters were installed to a depth of 60 cm at each replication in early October, 2005. This depth was chosen as a liberal estimate of being below the rooting zone in the initial years after planting to ensure the absence of any root uptake at the soil water sampling depth. A steel tube with the same diameter as the lysimeter was hammered into the ground at a 45º angle to limit soil disturbance above the ceramic cup. The steel tube was removed from the hole, and a slurry was made with sieved soil from the bottom 10 cm of each hole. The slurry was guided to the bottom of each hole and lysimeters were seated firmly to create a tight seal between the ceramic cup and soil. Bentonite pellets were placed around the outer 5 cm of each lysimeter to a depth of 2.5 cm to inhibit preferential flow along the side of the lysimeter tube.

Lysimeters were primed and purged (sample discarded) at two separate months prior to initiation of sample collection. Soil water samples were collected on a monthly basis at each site beginning in December 2005 at Molalla, and in January 2006 at Matlock. A 50 kPa vacuum was applied with a hand pump approximately 2 days before collection. Lysimeters were purged of any water between collection periods. Following collection, samples were placed on ice, transported to the laboratory, and stored at 4 ºC until analysis. Samples were typically analyzed the day after collection, but some samples were stored for up to 2 days before analysis. Lysimeter data are reported as concentrations (mg L^{-1}) and mass flux (kg ha^{-1} yr^{-1}).

Soil sampling and incubations

Soil samples were collected at four periods at each site to assess treatment effects on DOC, inorganic N (nitrate-N and ammonium-N), and DON production under controlled laboratory conditions. Samples were collected in April, July, and September of 2006, and July of 2007. At each sample period, mineral soil was collected to a depth of 20 cm at three random locations in each replication and composited in the field, taking care to remove any large organic material prior to compositing. Samples were collected with a volumetric core sampler attached to a slide hammer. Composite samples were placed in plastic bags on ice, and immediately transported to the laboratory for processing. Samples were air-dried, sieved to pass a 2 mm mesh, and then stored at 4 ºC until analysis.

Approximately 50 g of air-dried sieved soil was placed in a microlysimeter constructed of benchtop filtration units (Falcon Filter, Becton Dickinson Labware) as described by Nadelhoffer (1990). Samples were incubated for 17 days at 25ºC and a soil water potential of -22 kPa. Soil in each incubation unit was initially leached with 100 ml of ultrapure water by applying a tension of -22 kPa with the use of a vacuum pump. Water was allowed to equilibrate with soil for 30 min. before the vacuum was applied. Incubation unit mass was periodically checked during the incubation period, and ultrapure water was added as needed to maintain the initial soil water content following wetting. After 17 days, soil was leached again with 100 ml of ultrapure water as described above. Leachate volume was measured with a graduated cylinder, and an aliquot was separated for chemical analyses. Incubation units without soil were processed in the same manner to serve as controls. Final incubation leachate and lysimeter samples were analyzed for inorganic N (nitrate-N and ammonium-N), total N, and DOC. Nitrate-N and ammonium-N were determined colorimetrically on a Lachat Quick Chem 4200 analyzer. Dissolved organic C and total dissolved N (TDN) were determined on a Shimadzu Total Organic Carbon Analyzer (TOC-V$_{CSH}$) attached to a total N measuring unit (TNM-1). Dissolved organic N was calculated as the difference between TDN and inorganic N (sum of nitrate-N and ammonium-N). Leachate concentration was multiplied by leachate volume to determine mass of water-leachable

DOC and N forms removed at the end of the incubation. Incubation leachate values are reported on a dry soil mass basis (mg kg-soil^{-1} 17 days^{-1}).

Soil water flux calculation

Monthly soil water budgets were calculated for estimation of C and N mass flux below the rooting zone via leaching at each site. Net soil water flux was calculated as the difference between total monthly precipitation and potential evapotranspiration (ET) estimated with the Thornthwaite method (Dunne and Leopold, 1978). Precipitation and mean monthly air temperature (measured at 0.25 m above the soil surface) were measured daily at each site with a meteorological station that was located approximately at the center of each experimental area. Daily estimates were summed (precipitation) or averaged (air temperature) to determine total monthly precipitation and mean monthly air temperature at each site, which were then used to estimate soil water flux. When precipitation exceeded potential ET (i.e. a positive soil water flux) I assumed the net difference in water moved vertically below the rooting zone. This assumption is valid if soil water storage remains at or above field capacity during the rainy season, which is likely at these sites given the frequent rainfall and low temperatures which occur during the rainy season (i.e. winter). In addition, the Thornthwaite method does not account for differences in vegetation abundance or form on potential ET, possibly leading to an underestimation of potential ET and overestimation of soil water flux (Dunne and Leopold, 1978). Given the regional climate, where rainfall is greatest when vegetation growth is low, I assumed that any error arising from unaccounted vegetation effects would be low.

Statistical analysis

A mixed model approach with repeated-measures analysis of variance was used to assess treatment effects for both *in situ* soil water and incubation leachate. Block, whole plot within block, and subplot nested within whole plot were modeled as random effects, and the whole-plot factor, subplot factor, time variable (either month or sample period), and their interactions were modeled as fixed effects. For each dependent variable at each site, the covariance matrix used for repeated measures was identified by fitting the model to all possible candidate matrices and then choosing the matrix which resulted in the lowest fit criteria statistic (BIC). Examination of the residuals after model fit indicated that assumptions of homogenous variance and normality were valid for incubation data and *in situ* nitrate-N concentrations, but the variables DOC, DON, and TDN determined *in situ* required log transformations to meet assumptions of homogenous variance. For those variables, mean values were back-transformed and thus represent median values as reported. Ammonium-N concentrations were almost always below detection limits, and were not statistically analyzed for *in situ* measurements. Flux estimates were calculated as the product of water flux and nutrient concentration. Due to missing data associated with lysimeter failure (i.e. did not hold vacuum), I was unable to sum C and N flux within each of the replications, and was therefore unable to test if total annual flux was different between treatments. Annual estimates when reported are the annual sum of the monthly product of the mean or median concentration and estimated soil water flux within a given treatment.

When significant interaction was observed between the subplot factor and time, *a priori* orthogonal contrasts were performed to test for significance of difference between 1) the absence and presence of debris (0% coverage versus both the 40% and 80% coverage treatments), and 2) the 40% and 80% logging-debris coverage. Back-transformed contrast estimates for log-transformed variables are reported as the ratio of treatment medians for the associated contrast with 95% confidence intervals. When significant interaction was observed for the time x whole plot x subplot term, treatment effects were initially assessed by slicing with time (either period or month) held constant to determine time periods with significant differences, followed by slicing with either the subplot or whole-plot factor held constant depending on the significance of lower order interactions. For some of the incubation data, Tukey's Honestly Significant Difference test was used to determine significant differences among means. An alpha level of 0.05 was used to assess statistical significance in all evaluations. All analysis was performed in SAS V9.1 (SAS Institute, Cary NC).

Results

Water budget

Net soil water flux followed a seasonal pattern, being positive and similar to monthly precipitation during winter months when precipitation was high and air temperature was low, and negative during summer months when precipitation was low or absent and air temperature was high (Figure 3.1.). Monthly precipitation and net soil water flux was much higher at Matlock than Molalla, and was greater during 2006 than 2007 at both sites. Annual soil water flux was 250 and 220 cm at Matlock in 2006 and 2007, respectively, and was 110 and 70 cm in 2006 and 2007, respectively, at Molalla. Net soil water flux was generally zero or negative from May through September of each year, with exceptions for June and May at Matlock in 2006 and 2007, respectively (Figure 3.1B).

Soil water collected *in situ*

Dissolved inorganic N, DON, and TDN

Soil water inorganic N was almost entirely composed of nitrate-N; ammonium-N was found in only a few samples at random periods and concentrations were generally at or below detection limits. There was a significant main effect of CVC treatment on soil water nitrate-N concentrations at each site in 2006, but the effect varied by month (Tables 3.2, 3.3). The magnitude of the effect was greater at Molalla where application of annual CVC resulted in nitrate-N concentrations that ranged from 1.7 to 5.4 mg N L^{-1}, whereas N concentrations in the initial CVC treatments were at or below detection limits (Figure 3.2). Corresponding total mass of nitrate-N leached below the rooting zone in the annual CVC treatment for months which were assessed in 2006 was approximately 15 kg N ha^{-1}. Similar results were observed at Matlock, but the peak nitrate-N concentration in the annual CVC treatment was only 1.5 mg N L^{-1} (Figure 3.3). However, the greater soil water flux at Matlock (Figure 3.1) resulted in a similar total nitrate-N flux of 17 kg N ha^{-1}

to that observed at Molalla. Concentrations of nitrate-N with annual CVC treatment were similar in the second year at Molalla (range from 1.7 to 4.0 mg N L^{-1}, corresponding total net flux of 14 kg N ha^{-1}), but there was an obvious decline in concentration during the second year with annual CVC treatment at Matlock. Differences in 2007 at Molalla were not statistically significant due to low statistical power for the whole-plot treatment, and greater variance in that year amongst replications. In both years at Molalla when annual CVC was applied, nitrate-N concentrations were greatest in December, decreased until March, and then were relatively constant until soil water was insufficient for sample collection (Figure 3.2). At Matlock in 2006, nitrate-N concentrations were high in January, decreased to the lowest value in February, and then consistently increased until July when lysimeters did not produce a soil water sample (Figure 3.3).

There was no effect of logging-debris retention on nitrate-N at Molalla, but the interaction between logging debris and month was significant at Matlock in 2006 (Table 3.3). Examination of means by CVC treatment indicated that almost all of the logging-debris effect was associated with the annual CVC treatment, where nitrate-N concentrations in the 0% coverage were significantly lower than the 40% and 80% coverage during April (by 0.4 mg N L^{-1}), May (by 0.6 mg N L^{-1}), and June (by 1.5 mg N L^{-1}) (Figure 3.4). Visual comparison of logging-debris coverage means within the annual CVC treatment (Figure 3.4) with those for annual CVC averaged across all logging-debris coverage (Figure3.3) indicated that most of the increase in nitrate following annual CVC can be attributed to the 40% and 80% coverage in that treatment

There was a significant increase in median DON concentration with annual CVC treatment in March of each year at Molalla (Table 3.2, Figure 3.2), but no differences existed in other months. Median DON increase was 0.60 mg N L-^{1} (relative increase of 300% compared to initial CVC , 95% CI 161%:551%; corresponding mass increase 0.5 kg N ha^{-1}) in 2006, and 0.27 mg N L^{-1} (relative increase of 241% compared to initial CVC, 95% CI 158%:531%; corresponding mass increase 0.4 kg N ha^{-1}) in 2007. At Matlock, there was no effect of CVC treatment or logging-debris coverage on DON concentrations in either year (Table 3.3). In 2007, there was a significant effect of logging-debris retention by month on DON at Molalla (Table 3.2) where the 40% coverage had significantly higher concentration than the 80% coverage in January (by 0.22 mg N L^{-1}), but the 80% coverage was significantly higher than the 40% coverage in June (by 0.27 mg N L^{-1}). There was no difference in DON concentration between the presence and absence of logging debris during these same months, and no consistent pattern of DON among treatments over time (data not shown). Dissolved organic N concentrations at Molalla were relatively low in both years compared to the high nitrate-N concentrations in the annual CVC treatment (Figure 3.2), but at Matlock DON concentrations were as high or higher than the nitrate-N concentrations in the annual CVC treatment in 2006, and consistently greater than nitrate-N concentrations in 2007 (Figure3.3).

There was a significant effect of CVC treatment on TDN at Molalla in both years, with the significance and magnitude varying by month, and being similar to the results reported for nitrate-N (Table 3.2). Total dissolved N concentration was higher when annual CVC was applied in all months of 2006 (median increase range of 1.1 to 4.6 mg N L^{-1}), and all months except May and June of 2007 (median increase range of 0.8 to 2.6 mg N L^{-1}) (Figure 3.5). The corresponding estimated annual increase in N mass flux below the rooting zone for months with significant differences was 22 and 11 kg N ha^{-1} in 2006 and 2007, respectively. At Matlock, there was no difference in TDN between CVC treatments in either year (Table3.3, Figure 3.6) despite the differences in nitrate-N, but there were significant differences among logging-debris treatments when annual CVC was applied in 2006 (Table 3.3). In that year, 80% coverage had significantly higher TDN than the 40% coverage in January (median increase of 3.3 mg N L^{-1}), and the 40% and 80% coverage had significantly higher TDN than the 0% coverage in July (median increase of 2.7 mg N L^{-1}) (Figure 3.7). There was no effect of logging-debris retention on TDN at Molalla in either year (Table 3.2, Figure 3.8).

DOC and DOC:TDN

There were no significant differences among logging-debris or CVC treatments on DOC concentrations at either site (Table 3.4). During the first year, concentrations ranged from 5 to 12 mg C L^{-1} at Matlock, and from 4 to 6 mg C L^{-1} at Molalla (Figures 3.5, 3.6). Concentrations of DOC tended to decrease in the second year at both sites. At Molalla, DOC concentrations were consistently higher in the 0% coverage in 2006, and a similar trend was observed at Matlock in 2006 when annual CVC was applied (Figures 3.7, 3.8). Estimated total annual DOC flux during 2006 was 53, 42, and 41 kg C ha^{-1} yr^{-1} in the 0%, 40%, and 80% coverage across both CVC treatments at Molalla, respectively, and 36, 28, and 29 kg C ha^{-1} yr^{-1} in the 0%, 40%, and 80% coverage, respectively with annual CVC at Matlock. There was little difference in DOC concentrations among treatments or between sites in 2007 (mean estimated DOC annual flux for all of 24 kg C $ha^{-1} yr^{-1}$). Concentrations of DOC were higher in the initial CVC treatment in most months of 2006 at Matlock, but the opposite trend was observed in 2007 (Figure 3.6). At Molalla, DOC concentrations were higher in the annual CVC treatment in most months of both years (Figure 3.5), but differences were small and not significant.

The ratio of DOC:TDN was significantly higher in the initial CVC treatment at Molalla in most months of both years, largely due to significantly lower TDN in that treatment (Table 3.2, Figure 3.5). Differences in DOC:TDN ranged from 10 to 20 in months with significant differences in 2006, and from 7 to 19 in 2007 (Figure 3.5). Similarly, DOC:TDN was significantly higher at Matlock in the initial CVC treatment, but the effect was only significant in the first three months of 2006 (mean difference of 11 across all three months) and not apparent in 2007 (Table 3.3, Figure 3.6). The effect at Matlock was largely due to differences in DOC concentration, rather than an increase in TDN that was observed at Molalla. There was a significant effect of logging-debris retention on DOC:TDN at Molalla in 2006 (Table 3.2), where the ratio was significantly higher in the 0% coverage than the mean of the 40% and 80% coverage by 9 and 7 during May and June, respectively (Figure 3.8). In most months of that year, there was a general pattern of decreasing DOC:TDN with increasing logging-debris retention, which appeared to be driven by differences in DOC rather than TDN. A similar pattern was observed at Matlock when annual CVC was applied in both years (Table 3.3, Figure 3.7) but differences between 0% coverage and the mean of 40% and 80% coverage were only statistically significant in February (by 6), and July (by 11) of 2006, when removal of logging debris resulted in higher DOC:TDN. As with Molalla, the difference appeared to be driven by greater DOC with 0% debris coverage rather than an increase in TDN (Figure 3.7).

Soil water in laboratory incubations

Inorganic N, DON, and TDN

There was significant interaction between CVC treatment and incubation period on nitrate-N at Molalla during soil incubations in the laboratory experiment (Table 3.5). Water-leachable nitrate-N was 7.2 and 13.4 mg N kg soil^{-1} higher in soils from the annual CVC treatment than in soils from the initial CVC treatment for the September 2006 and July 2007 incubation periods, respectively (Figure 3.9A). Mean nitrate-N was greater in the annual CVC treatment at Matlock for all incubation periods, but differences were not statistically significant (Table 3.5, Figure 3.9B). There was also significant interaction amongst logging debris, CVC, and incubation period on nitrate-N at both sites (Table 3.5). Slicing with incubation period held constant indicated the interaction was associated with the July incubation in each year at both sites. During that period in 2006 at Mollala, there was a general increase in nitrate with increasing logging-debris retention in the initial CVC treatment, but significantly higher nitrate (23 mg N kg^{-1} 17 days^{-1}) when logging debris was removed in the annual CVC treatment (Figure 3.10A). At the same site in 2007, there was no effect of logging debris in the initial CVC treatment, but nitrate increased with increasing logging-debris coverage when annual CVC was applied

(Figure 3.10A). At Matlock, there was significantly greater nitrate in the 80% coverage in the initial CVC treatment for both July 2006 and 2007 incubations (mean difference of 7.3 and 8.6 mg N kg^{-1} 17 $days^{-1}$, for 2006 and 2007, respectively) (Figure 3.10B). When annual CVC was applied, the 40% coverage produced 12.0 mg N kg^{-1} 17 $days^{-1}$ more water-leachable nitrate than the 0% coverage in the July 2006 incubation, but no differences existed amongst logging-debris coverage treatments in the 2007 incubation (Figure 3.10B).

The main effect of logging debris on ammonium-N approached statistical significance at Matlock (Table 3.5), where 2.00, 2.25, and 3.06 mg N kg-$soil^{-1}$ (SE for all=0.30) was leached on average at each incubation in the 0%, 40%, and 80% coverage, respectively. A similar pattern was observed at Molalla, where 2.06 (SE=0.37), 2.19, (SE=0.38), and 2.83 mg N kg-$soil^{-1}$ (SE=0.37) was leached on average at each incubation in the 0%, 40%, and 80% coverage, respectively. Dissolved organic N was higher in the annual CVC treatment than the initial CVC treatment in the July 2007 incubation at both sites (Figure 3.9), but the difference was only significant at Matlock (difference of 1.8 mg N kg^{-1} 17 $days^{-1}$) (Figure 3.9B). Similarly, TDN was higher in the annual CVC treatment than the initial CVC treatment in the July 2007 incubation at both sites, but the difference was only significant at Mollala (Figure 3.9A). Significant interaction amongst logging debris, CVC, and period at Matlock (Table 3.5) was associated with the July 2007 incubation, where TDN was significantly higher in the 80% (20.86 mg N kg-$soil^{-1}$, se = 4.69) logging-debris coverage relative to the 0% (8.72 mg N kg-$soil^{-1}$, se = 4.69) and 40% (6.00 mg N kg-$soil^{-1}$, se = 4.84) coverage in the initial CVC treatment.

DOC and DOC:TDN

There was significant interaction amongst logging debris, CVC, and incubation period on water-leachable DOC at both sites (Table 3.6). The effect was limited to the April 2006 incubation at Matlock, where DOC increased with increasing logging-debris retention in the initial CVC treatment. Mean DOC in the initial CVC treatment was 6.55, 8.61, and 14.05 mg C kg-soil^{-1} (SE=1.64) for the 0%, 40%, and 80% coverage, respectively. At Molalla, there was a significant effect of logging debris in the initial CVC treatment at the July 2007 incubation, where the 80% coverage (10.22 mg C kg-soil^{-1}) was significantly lower than the 0% (15.47 mg C kg-soil^{-1}) and 40% (15.23 mg C kg-soil^{-1}) coverage. With the exception of these instances, CVC and logging-debris treatments had no significant effect on DOC, and there were no apparent trends or differences between treatments at either site (data not shown).

There was a significant main effect of logging debris on DOC:TDN at Molalla (Table 3.6). The DOC:TDN ratio decreased with increasing debris, with a mean ratio across all periods of 0.60, 0.43, and 0.26 for the 0%, 40%, and 80% coverage treatments, respectively (SE for all 0.11). A similar non-significant trend was observed at Matlock with DOC:TDN ratios of 2.11, 1.31, and 0.94, for the 0%, 40%, and 80% coverage, respectively (SE=0.44). There was no effect of CVC on DOC:TDN at either site.

Discussion

Soil water N concentrations

Annual application of CVC had a pronounced effect on N leaching following harvesting. The increase in soil water nitrate-N concentration following CVC at both sites agrees with past studies (Vitousek and Matson 1985, Briggs et al. 2000, Smethhurst and Nambiar 1995), demonstrating the importance of re-vegetation following harvesting for N retention across a variety of site conditions. It is likely that the N flux observed here was similar or greater in the annual CVC treatment in the two years prior to initiation of lysimeter sampling, as net N mineralization is generally increased immediately following harvesting (Prescott et al. 1997; Vitousek et al., 1992) and tree N demand would have been low. In addition, greater TDN flux at Molalla in 2007 suggests that N availability continues to be greater than vegetation N demand at that site, which may result in continued N leaching in later years until N demand exceeds availability. However, the estimated increase in nitrate-N and TDN flux when annual CVC was applied is very small relative to total N pools present at each site (Table 3.1), and it is unlikely that the observed flux below the rooting zone would have negative impacts on future soil productivity if the observed trends continue. Strahm et al. (2005) also found that estimated N leaching fluxes following harvesting of Douglas-fir were a small portion (<1.5%) of total soil N.

The magnitude and duration of increased nitrate concentration in soil water were greater at Molalla compared to Matlock, partly due to differences in nitrate production. Water-leachable nitrate production was approximately 2-3 times greater at Molalla than

Matlock, and nitrate production was significantly increased following annual CVC at Molalla in two of the incubations. Greater nitrate production at Molalla was almost certainly associated with a larger total N pool at that site (Table 3.1), but may have also been due to differences in N immobilization between sites (Vitousek et al., 1992). The contrast between sites was probably also associated with differences in crop tree N uptake. Tree growth data (T. Harrington, unpublished data) and measures of available N during the study period indicated greater utilization of increased available N by crop trees at Matlock following annual CVC compared to Molalla (Chapter 4), which would have caused the large reduction in soil water nitrate concentrations in 2007 at Matlock.

At Molalla, the pattern of decreasing nitrate-N concentration following an early winter peak reflects transport of nitrate produced in surface soil of the previous growing season to deeper portions of the soil, and the relatively constant concentration after early spring probably reflects a balance between new nitrate production and vegetation uptake (Goodale et al., 2000; Iseman et al., 1999). At Matlock, there was also an early decrease from a peak winter concentration in 2006, but the period of decrease was only for one month, and then concentrations consistently increased until July, when the soil was too dry to produce lysimeter samples. The rapid decrease could be an artifact of lysimeter installation, as soil disturbance can result in localized soil chemistry that differs from the surrounding soil. However, the same pattern was observed at Molalla in both years, and disturbance-related artifacts would have been minimal by the second year at that site. It is more likely that a greater amount of nitrate (relative to the soil N pool size) was transported below the rooting zone at Matlock before sampling began, as soil water flux was much greater at Matlock than Molalla (Figure 3.1).

The consistent increase in nitrate concentration at Matlock following the initial reduction is more difficult to explain. Strahm et al. (2005) attributed higher nitrate concentrations during summer months in a bole-only harvest compared to a whole-tree harvest to a decrease in N dilution when soil water flux is low. Almost all of the increase in the annual CVC treatment was from replications where logging debris was retained, and it is unlikely that presence of logging debris alone would reduce soil water flux. Alternative mechanisms include increased nitrate production or decreased vegetative uptake following logging-debris retention. Tree growth at Matlock when annual CVC was applied is approximately 30% greater in the 80% coverage treatment compared to the 0% coverage (T. Harrington, unpublished data), making it unlikely that lower crop tree N uptake contributed to the increase. Incubation results from the July 2006 period showed greater mean water-leachable nitrate when logging debris was retained in the annual CVC treatment, but only the 40% coverage was significantly greater than 0% coverage. However, water-leachable ammonium increased with increasing logging-debris retention during all incubations, likely due to a significant increase in soil N at Matlock when logging debris was retained (Chapter 4). The results suggest that logging debris increased soil N, which caused a greater availability of ammonium, and subsequent higher rates of nitrate production as soil temperature increased during spring and summer months (Knoepp and Swank, 2002). The absence of any effect in the initial CVC treatment *in situ* is most likely due to greater competition for N, probably from vegetation rather than microbial immobilization. Greater water-leachable nitrate in the initial CVC treatment with 80% logging-debris coverage during the July incubations when the influence of vegetation was removed supports this conclusion (Figure 3.10).

The significant increase in DON following annual CVC at Molalla but not at Matlock indicates the potential for increased DON loss is site-specific. At Molalla, the increase was observed in March of each year, suggesting that a seasonal transitory process contributed to the response. Factors which control DON loss are poorly understood (Kalbitz et al., 2000; McDowell, 2003), but it seems likely that the effect was biologically mediated as soil temperature began to increase from low winter temperatures in March of each year (Chapter 2). The increase in soil temperature likely increased rates of microbial activity (Lloyd and Taylor, 1994), possibly increasing net DON production because of lower C availability in the annual CVC treatment (Shan et al., 2001). Whatever the mechanism, the contribution of DON to N leaching following annual CVC was relatively small at Molalla, contributing only 2% and 4% to TDN leaching estimates in 2006 and 2007, respectively. These results are similar to those for logging-debris retention, where increased concentrations of DON in the 40% and 80% coverage were small, transient, and occurred at random times. Strahm et al. (2005) also found that DON contribution to TDN loss following OM manipulation and CVC was small following harvesting of Douglas-fir in southwestern WA. It appears there is greater potential for modification of DON loss following logging-debris retention or annual CVC at Molalla than Matlock, but the contribution of this modification to TDN flux is relatively small.

Dissolved organic N contributed a larger proportion to TDN at Matlock than at Molalla, and the absolute amounts were greater as well. These differences may be associated with differences in production (e.g. substrate quality, microbial turnover, environmental effects) or retention mechanisms in soil (e.g. differences in soil mineralogy, degree of weathering). Greater retention at Molalla in deeper portions of the soil (> 20 cm depth) is likely given the generally higher water-leachable DON production from surface soils at that site under controlled laboratory conditions (Figure 3.9). Qualls et al. (2000) hypothesized that DON loss following harvesting is controlled by hydrologic and geochemical mechanisms rather than biological processes. At Matlock soil texture is coarse, making it likely that soil water contact (both actual surface area and time of contact) with mineral surfaces is much lower than at Molalla, which would decrease potential sorption of DON. Regardless, it is clear that measurement of soil water DON following harvesting is necessary at some sites if TDN flux below the rooting zone is to be accurately quantified. The significant difference between CVC treatments in nitrate-N but no difference in TDN (Table 3.3) at Matlock underscores this point.

With the exception of the initial CVC treatment at Molalla in 2006, results from the July incubations generally indicated higher mean production of water-leachable nitrate when logging debris was retained, but the significance and magnitude of the effect was variable (Figure 3.10). The variability probably reflects actual spatial and temporal variability in the field rather than experimental error. At both sites, soil temperature to a depth of 15 cm was significantly reduced when logging debris was retained (Chapter 2), which would have caused a reduction in nitrate production *in situ* (Knoepp and Swank, 2002; Zak et al., 1999). Based on the laboratory incubations, there is potential for modified nitrate production when logging debris is retained, but it is likely that treatment effects *in situ* are muted due to high soil variability and a concurrent temperature limitation to nitrification. In the case of Matlock, a significant increase in total soil N at that site when logging debris was retained and annual CVC was applied (Chapter 4) was high enough to cause an increase in soil water nitrate even though specific rates (per unit N) of nitrification were probably lower.

Clearly there were significant effects of logging-debris retention on soil water N, but these effects were small compared to that of CVC. These results generally agree with those of Mann et al. (1988) who found no significant effect of logging-debris retention on N leaching across a variety of site and climate conditions. However, other studies have found either an increase (Strahm et al. 2005, Hendrickson et al. 1989) or decrease (Vitousek and Matson 1985, Carlyle et al. 1998) in soil water nitrate following varying levels of logging-debris retention, indicating the potential for logging debris to modify N loss following harvesting. Although the results observed here agree with those from Mann et al. (1988), the lack of any pronounced effect could be due to the experimental design that was employed. Subplot logging-debris treatments were located in whole-plots that had received the whole-tree harvest treatment at time of logging, which would have had significantly less needle and small branch retention than the bole-only treatments. Since needles contain the majority of N in logging-debris (Tiarks et al. 1994), it is possible that different results would have been observed if subplots were located in the bole-only whole-plots.

Soil water DOC and DOC:TDN

Annual CVC caused a temporary reduction in photosynthesis because it eliminated shrub and herbaceous vegetation. Since photosynthesis is the ultimate source of all DOC, annual CVC could potentially cause a temporary reduction in DOC production and loss. However, the lack of any significant difference in DOC concentrations between CVC treatments at both sites indicates that DOC loss (below the rooting zone) is largely independent of organic matter inputs from recently-fixed C following harvesting at these sites. It is likely that above- and belowground DOC inputs to soil were greater in the initial CVC treatment, but these inputs were either consumed or retained in the mineral soil. Companion soil respiration measures at these same sites indicated rapid consumption of OM inputs from competing vegetation (Chapter 2), likely reducing the potential for the movement of DOC to deeper portions of the soil. However, the labile portion (i.e. readily metabolized) of DOC is generally thought to be small (Lajtha et al., 2005; Qualls and Haines, 1992), and it is likely that DOC retention via adsorption to mineral surfaces also occurred. Although the relative contribution of consumption or adsorption is uncertain, results from the laboratory incubations (i.e. no significant differences among treatments) suggest that these processes occur rapidly *in situ*.

Given the large control of C availability on net N availability (Hart et al., 1994), greater N availability following annual CVC could be an indication of microbial C limitation, which could be expected to result in a reduction of DOC production and loss. Lajtha et al. (2005) observed greater N flux, but reduced DOC flux, when root inputs were excluded in an old-growth Douglas-fir forest, which was partially attributed to greater microbial immobilization. The results observed here indicate little connection between increased N availability and soil water DOC loss at these sites. Significantly higher soil water TDN at Molalla led to significantly lower DOC:TDN in the annual CVC treatments, but there was no effect on DOC concentrations. At Matlock, the significantly higher DOC:TDN in the initial CVC treatment appeared to be a function of both lower TDN and higher DOC, but the effect was limited to the first three months of 2006. In a review, Kalbitz et al. (2000) concluded that there was no clear effect of increased N availability on DOC dynamics of intact forests. At these sites, this also appears to be the case following harvesting.

Logging debris has been shown to be a significant source of DOC to mineral soil (Qualls et al. 2000, Robertson et al. 2000, Mattson et al. 1987), and it is likely that DOC inputs to mineral soil were increased in this study following logging-debris application. The lack of any significant effect of logging-debris retention on soil water DOC concentrations below the rooting zone indicates increased inputs were retained or consumed in the mineral soil. Both Qualls et al. (2000) and Piirainen et al. (2002) concluded that DOC inputs from logging debris were effectively retained in mineral soil as well. Qualls et al. (2000) attributed the retention mechanism to adsorption to mineral surfaces, but also acknowledged microbial consumption likely contributed to the decrease. At these sites, microbial respiration was lower when logging debris was retained (Chapter 2), possibly indicating that adsorption to mineral surfaces may be a more important retention mechanism. A companion study found significantly greater soil

C content at Matlock, but not at Molalla, following logging-debris retention (Chapter 2), providing some evidence to support this possibility. The lack of any observable effect at Molalla may be associated with much larger soil C pool at that site (Table 3.1), where DOC inputs from logging debris would be small relative to the soil C pool size and likely undetectable (Homann et al., 2001).

Despite the lack of statistical significance, soil water DOC concentrations were greater in almost all months when no logging debris was retained in 2006 at both sites (Figures 3.7, 3.8), resulting in a higher estimated annual flux. In addition, DOC:TDN ratio was significantly greater in the 0% coverage at both sites (at Matlock only when annual CVC was applied) during some months of 2006, with differences being driven by greater DOC in that treatment rather than lower TDN. A few recent studies have shown that DOC in deeper soil (>50 cm) is derived from mineral soil organic matter or root litter rather than surface inputs (Froberg et al. 2007, Uselman et al. 2007), suggesting that increased production from these sources could result in increased DOC loss if not adsorbed to mineral soil. Temperature would be expected to have large control on DOC production given the central role of microbial processes on DOC formation (Kalbitz et al. 2000). Soil temperature was significantly greater in the 0% logging-debris coverage at both sites (Chapter 2), possibly causing an increase in root decomposition (Chen et al., 2000) and greater soil water DOC concentrations. Powers et al. (2005) concluded that increased soil C following harvesting was due to decomposition of roots from the previous stand, as large increases were observed in treatments where no surface litter (including the forest floor) was retained. At my sites, soil C also tended to increase during the two-year study period regardless of treatment (Chapter 2). The limited effect of logging debris on DOC production during incubations does not provide evidence to support or refute increased DOC production, given that incubations were performed at constant temperature. If DOC loss was greater in the 0% logging-debris coverage, it appears to have been a relatively short-lived response given no apparent differences between treatments in 2007.

Conclusions

Increased soil water nitrate-N following annual CVC reflects a potential for soil N loss following harvesting when this practice is employed, but the estimated N mass loss was small compared to total N pools at each site. At Molalla, which has a relatively large total soil N pool, greater production of nitrate-N led to larger soil water nitrate-N concentrations following annual CVC compared to Matlock, which has a relatively low total soil N, lower nitrate production, and greater crop tree utilization of available N (Chapter 4). Dissolved organic N was also increased following annual CVC at Molalla, but the contribution from this source was small compared to the estimated increase in annual TDN flux. Site-specific factors that control nitrate production and tree N demand appear to be strong indicators of potential N loss when annual CVC is applied.

Almost all of the increase in nitrate-N concentration at Matlock was associated with logging-debris retention when annual CVC was applied, demonstrating the potential for greater N loss at some sites when the effects of these practices interact. However, the biological significance of these losses at Matlock is unclear, as total soil N increased during the two-year study period (Chapter 4). The absence of any logging-debris effect at Molalla may be due to the greater total soil N pool at that site, as inputs from logging debris were small compared to total soil N.

The absence of any increase in DOC concentration when competing vegetation was present and logging debris was retained indicates that DOC inputs from these sources were either consumed or retained in the mineral soil. Higher DOC concentrations when no logging debris was retained may indicate greater DOC production and loss in those treatments, which would likely be due to greater belowground decomposition associated with higher soil temperature. Given that soil C content tended to increase during the two-year study regardless of treatment (Chapter 2), the significance of greater DOC loss to soil productivity is probably low.

Tables

Table 3.1. Site characteristics and select pre-treatment soil properties to a depth of 30 cm for study sites near Matlock, WA and Molalla, OR.

Characteristic or property	Matlock	Molalla
Location (Latitude, Longitude)	47.206 °N, 123.442 °W	45.196 °N, 122.285 °W
Elevation (m)	118	449
Mean annual temperature (°C)	10.7	11.2
Mean annual precipitation (cm)[1]	240	160
Site index$_{50\ yr}$ (m)	35.9	36.2
Soil texture (% sand/silt/clay)	65 / 14 / 21	37 / 34 / 29
Bulk density (Mg m^{-3})	1.45 (0.05) [2]	0.98 (0.02)
Coarse fragments by mass (%)	65.8 (1.3)	32.2 (2.2)
Total soil N (kg ha^{-1})	2,246 (88)	4,338 (173)
Total soil C (Mg ha^{-1})	66.5 (3.6)	102.2 (4.7)

[1] estimated from the period 1950-2005 with the PRISM model (http://prism.oregonstate.edu)
[2] standard error in parenthesis, n=8 for bulk density at Matlock, n=24 for all others).

Table 3.2. Test statistics for fixed treatment effects by year on the dependent variables nitrate-N, total dissolved N, dissolved organic N, and DOC:TDN at Molalla.

Effect	**Nitrate-N**		**Total dissolved N**		**Dissolved organic N**		**DOC:TDN**[1]	
	F statistic	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**
				2006				
CVC[2]	26.71	**0.014**[3]	40.47	**0.008**	4.91	0.113	17.87	**0.024**
Debris	0.76	0.489	0.63	0.551	0.04	0.959	3.56	0.064
CVC*debris	0.71	0.513	0.47	0.635	0.58	0.575	2.75	0.108
Month	5.78	**<0.001**	6.77	<0.001	5.67	**<0.001**	5.54	**<0.001**
CVC*month	4.88	**<0.001**	3.50	**0.002**	2.80	**0.010**	4.96	**<0.001**
Debris*month	1.29	0.227	1.71	0.063	0.82	0.644	1.93	**0.030**
CVC*debris*month	1.24	0.261	1.15	0.325	0.98	0.478	1.28	0.233
				2007				
CVC	8.09	0.065	16.04	**0.028**	3.31	0.167	18.69	**0.023**
Debris	0.40	0.682	1.78	0.223	0.62	0.559	1.22	0.339
CVC*debris	0.39	0.686	1.17	0.353	0.08	0.923	1.46	0.282
Month	1.39	0.233	4.08	**0.002**	5.91	**0.001**	21.05	**<0.001**
CVC*month	1.39	0.232	6.08	**<0.001**	2.30	**0.047**	9.59	**<0.001**
Debris*month	0.63	0.809	0.41	0.953	2.04	**0.037**	1.43	0.175
CVC*debris*month	0.63	0.805	0.94	0.512	0.98	0.477	1.60	0.114

[1]DOC:TDN = ratio of dissolved organic carbon to total dissolved nitrogen
[2]CVC=competing vegetation control
[3]Test statistics in bold are significant at α=0.05.

Table 3.3. Test statistics for fixed treatment effects by year on the dependent variables nitrate-N, total dissolved N, dissolved organic N, and DOC:TDN at Matlock.

Effect	**Nitrate-N**		**Total dissolved N**		**Dissolved organic N**		**DOC:TDN[1]**	
	F statistic	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**
					2006			
CVC[2]	30.43	**0.012**[3]	3.85	0.145	2.44	0.216	4.98	0.112
Debris	0.73	0.501	1.29	0.312	0.37	0.702	3.76	0.054
CVC*debris	0.54	0.595	0.54	0.594	0.59	0.572	0.60	0.565
Month	4.46	**0.002**	17.38	**<0.001**	17.53	**<0.001**	5.05	**0.001**
CVC*month	5.17	**0.001**	0.70	0.628	1.27	0.287	1.75	0.134
Debris*month	2.93	**0.005**	1.72	0.094	0.93	0.509	2.50	**0.013**
CVC*debris*month	1.76	0.088	2.25	**0.025**	0.71	0.708	2.31	**0.021**
					2007			
CVC	4.61	0.121	1.13	0.366	0.40	0.570	0.19	0.691
Debris	1.46	0.278	0.48	0.632	0.37	0.702	1.05	0.379
CVC*debris	1..50	0.270	2.58	0.117	1.24	0.331	1.38	0.289
Month	4.34	0.001	8.28	**<0.001**	6.62	**<0.001**	9.77	**<0.001**
CVC*month	2.09	0.065	2.20	0.054	0.87	0.521	1.85	0.103
Debris*month	0.71	0.734	0.82	0.625	0.37	0.970	0.80	0.646
CVC*debris*month	0.29	0.989	0.69	0.759	0.26	0.993	1.01	0.451

[1]DOC:TDN = ratio of dissolved organic carbon to total dissolved nitrogen
[2]CVC=competing vegetation control
[3]Test statistics in bold are significant at α=0.05.

Table 3.4. Test statistics for fixed treatments effects on dissolved organic carbon concentrations at 60 cm depth by year at the Matlock and Molalla sites.

Effect	Matlock		Molalla	
	F statistic	***p* value**	**F statistic**	***p* value**
		2006		
CVC[1]	1.22	0.350	0.19	0.695
Debris	0.23	0.799	0.92	0.426
CVC*debris	1.19	0.338	0.27	0.769
Month	6.63	**<0.001**[2]	5.87	**<0.001**
CVC*month	0.90	0.487	0.43	0.885
Debris*month	0.53	0.864	0.55	0.900
CVC*debris*month	1.26	0.270	0.96	0.503
		2007		
CVC	0.36	0.589	0.00	0.957
Debris	0.17	0.849	0.28	0.764
CVC*debris	0.13	0.877	0.79	0.484
Month	45.34	**<0.001**	36.80	**<0.001**
CVC*month	2.19	0.055	1.06	0.395
Debris*month	0.93	0.526	1.26	0.261
CVC*debris*month	0.66	0.782	0.92	0.532

[1]CVC=competing vegetation control.
[2]Test statistics in bold are significant at α=0.05.

Table 3.5. Test statistics for fixed treatment effects following a 2-wk incubation at four sample periods on the dependent variables ammonium-N, nitrate-N, total dissolved N, and dissolved organic N at the Matlock and Molalla sites.

Effect	Ammonium-N		Nitrate-N		Total dissolved N		Dissolved organic N	
	F statistic	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**
				Matlock				
CVC[1] (df = 1, 3)[2]	2.09	0.244	2.96	0.184	1.50	0.308	0.12	0.747
Debris (D) (df = 2, 12)	3.70	0.056	1.31	0.307	1.64	0.235	1.97	0.183
CVC * D (df = 2, 12)	1.08	0.370	0.60	0.567	0.42	0.666	0.58	0.576
Period[3] (P) (df = 3, 54)	6.04	**0.001**[4]	4.56	**0.006**	5.27	**0.003**	17.72	**<0.001**
CVC * P (df = 3, 54)	0.72	0.546	1.16	0.333	1.78	0.162	2.96	**0.040**
D * P (df = 6, 54)	1.06	0.400	0.88	0.516	1.16	0.343	2.07	0.072
CVC * D * P (df = 6, 54)	0.65	0.691	3.00	**0.013**	3.08	**0.011**	1.48	0.201
				Molalla				
CVC (df = 1, 3)	0.00	0.981	7.21	**0.075**	1.75	0.277	0.61	0.493
Debris (df = 2, 12)	1.57	0.248	1.42	0.281	0.22	0.807	0.24	0.787
CVC * D (df = 2, 12)	0.51	0.613	2.36	0.136	0.88	0.438	1.68	0.227
Period (df = 3, 54)	7.93	**0.001**	8.97	**<0.001**	2.18	0.102	12.67	**<0.001**
CVC * P (df = 3, 54)	0.81	0.495	3.51	**0.022**	2.93	**0.042**	1.33	0.275
D * P (df = 6, 54)	1.10	0.375	1.73	0.131	1.49	0.199	0.73	0.627
CVC * D * P (df = 6, 54)	1.13	0.358	4.19	**0.002**	2.18	0.059	1.11	0.367

[1]CVC=competing vegetation control treatment. [2]Degrees of freedom for the critical F statistic for each effect are in parenthesis. [3]Period=soil collection time of April 2006, July 2006, September 2006, or July 2007 for laboratory incubation. [4]Test statistics in bold are significant at α=0.05.

Table 3.6. Test statistics for fixed treatments effects by site following a 2-wk incubation at four sample periods on dissolved organic carbon (DOC) production and DOC:total dissolved nitrogen (TDN) ratio.

Effect	DOC		DOC:TDN	
	F statistic	***p* value**	**F statistic**	***p* value**
		Matlock		
CVC[1] (df = 1, 3)[2]	0.57	0.504	3.66	0.152
Debris (df = 2, 12)	1.23	0.326	1.83	0.202
CVC * D (df = 2, 12)	0.06	0.946	0.31	0.742
Period[3] (df = 3, 54)	10.56	<0.001	7.40	**0.001**[4]
CVC * P (df = 3, 54)	0.49	0.691	2.13	0.107
D * P (df = 6, 54)	1.70	0.138	1.31	0.268
C * D * P (df = 6, 54)	2.32	**0.046**	1.08	0.384
		Molalla		
CVC (df = 1, 3)	0.46	0.548	2.54	0.209
Debris (df = 2, 12)	0.82	0.465	3.82	**0.050**
CVC * D (df = 2, 12)	0.38	0.695	1.40	0.284
Period (df = 3, 54)	28.78	<0.001	6.01	**0.001**
CVC * P (df = 3, 54)	0.55	0.651	2.63	0.060
D * P (df = 6, 54)	0.42	0.861	1.90	0.098
C * D * P (df = 6, 54)	2.80	**0.019**	0.40	0.873

[1]CVC=competing vegetation control treatment.
[2]Degrees of freedom for the critical F statistic for each effect are in parenthesis.
[3]Period=soil collection time of April 2006, July 2006, September 2006, or July 2007 for laboratory incubation.
[4]Test statistics in bold are significant at α=0.05.

Figures

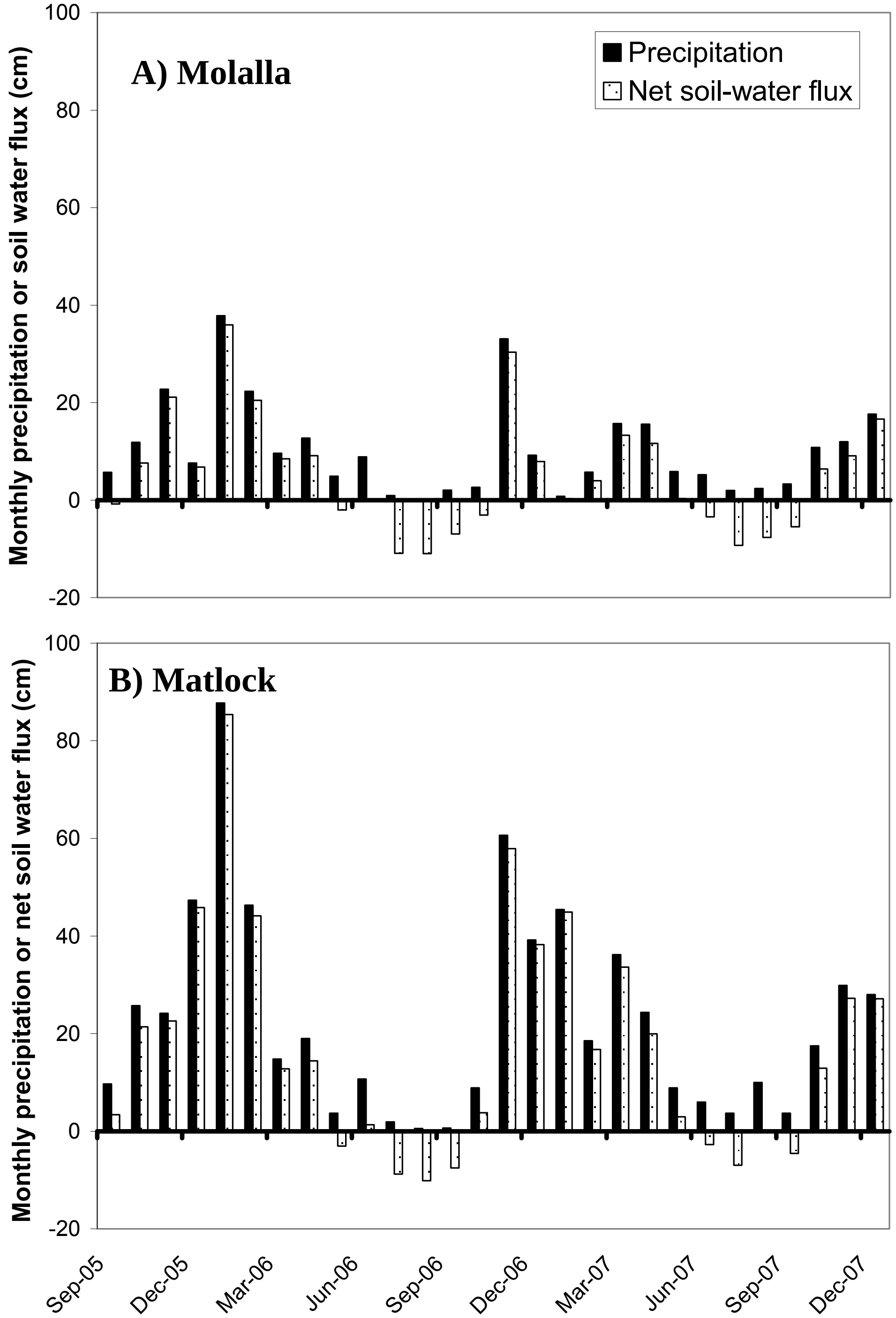


Figure 3.1. Monthly total precipitation and estimated net soil water flux at (A) Molalla and (B) Matlock sites.

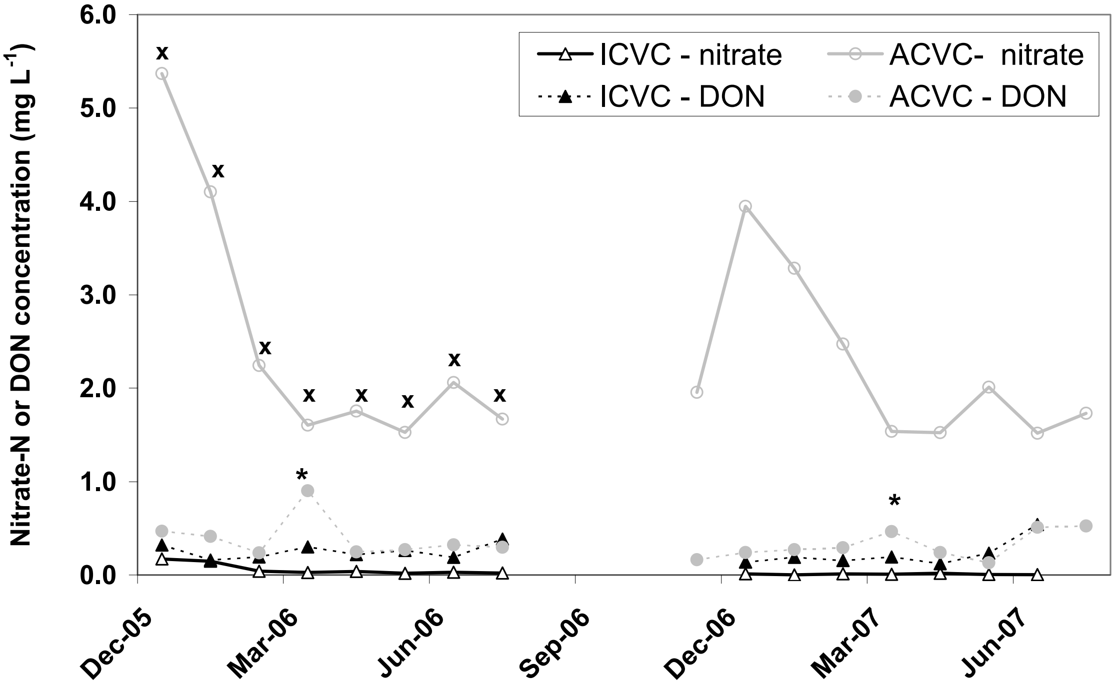


Figure 3.2. Effect of competing vegetation control treatments on soil water nitrate-N and dissolved organic N (DON) concentrations at 60 cm depth at Molalla. ICVC = initial competing vegetation control. ACVC= annual competing vegetation control. * and x indicate significant difference at α=0.05 between treatments for the associated month in DON and nitrate-N, respectively.

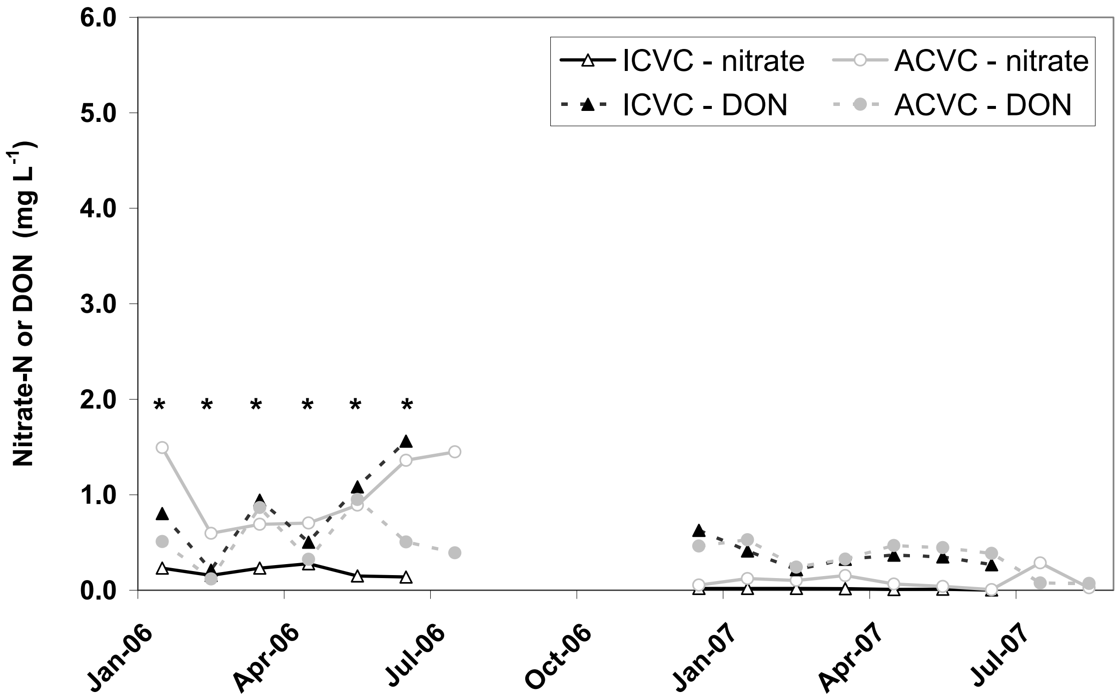


Figure 3.3. Effect of competing vegetation control treatments on soil water nitrate-N and dissolved organic N (DON) concentrations at 60 cm depth at Matlock. ICVC = initial competing vegetation control. ACVC= annual competing vegetation control. * indicates significant difference at α=0.05 between treatments in nitrate-N for the associated month.

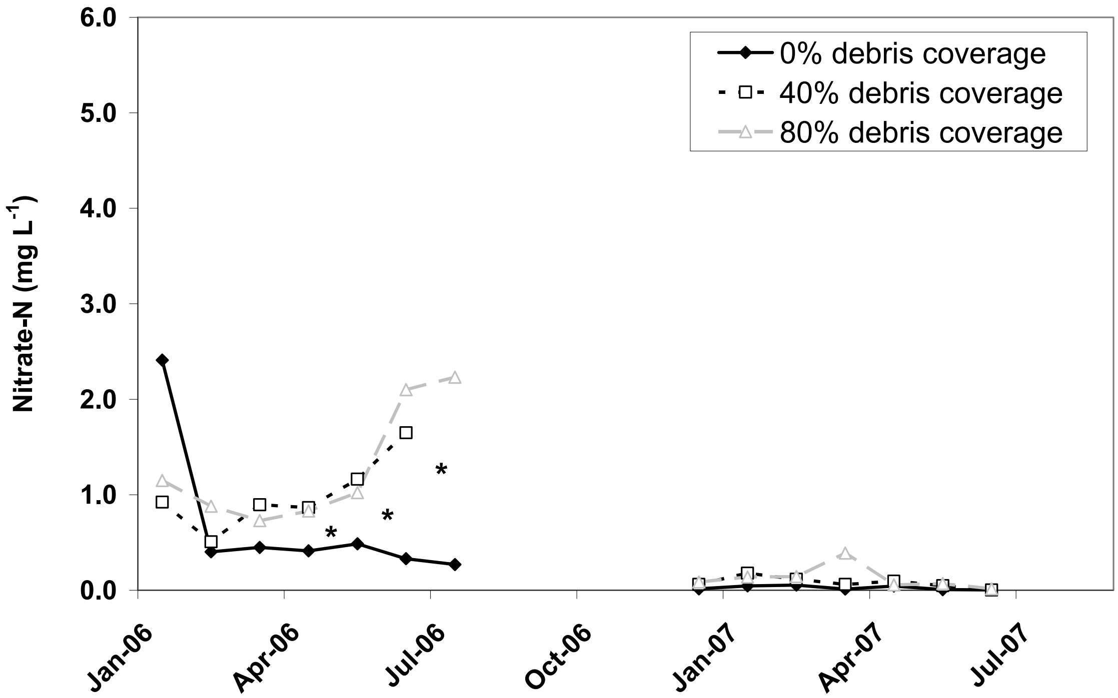


Figure 3.4. Effect of logging-debris treatments on soil water nitrate-N at 60 cm depth when annual competing vegetation control was applied at Matlock. * indicate significant difference at α=0.05 in nitrate-N between treatments for the associated month.

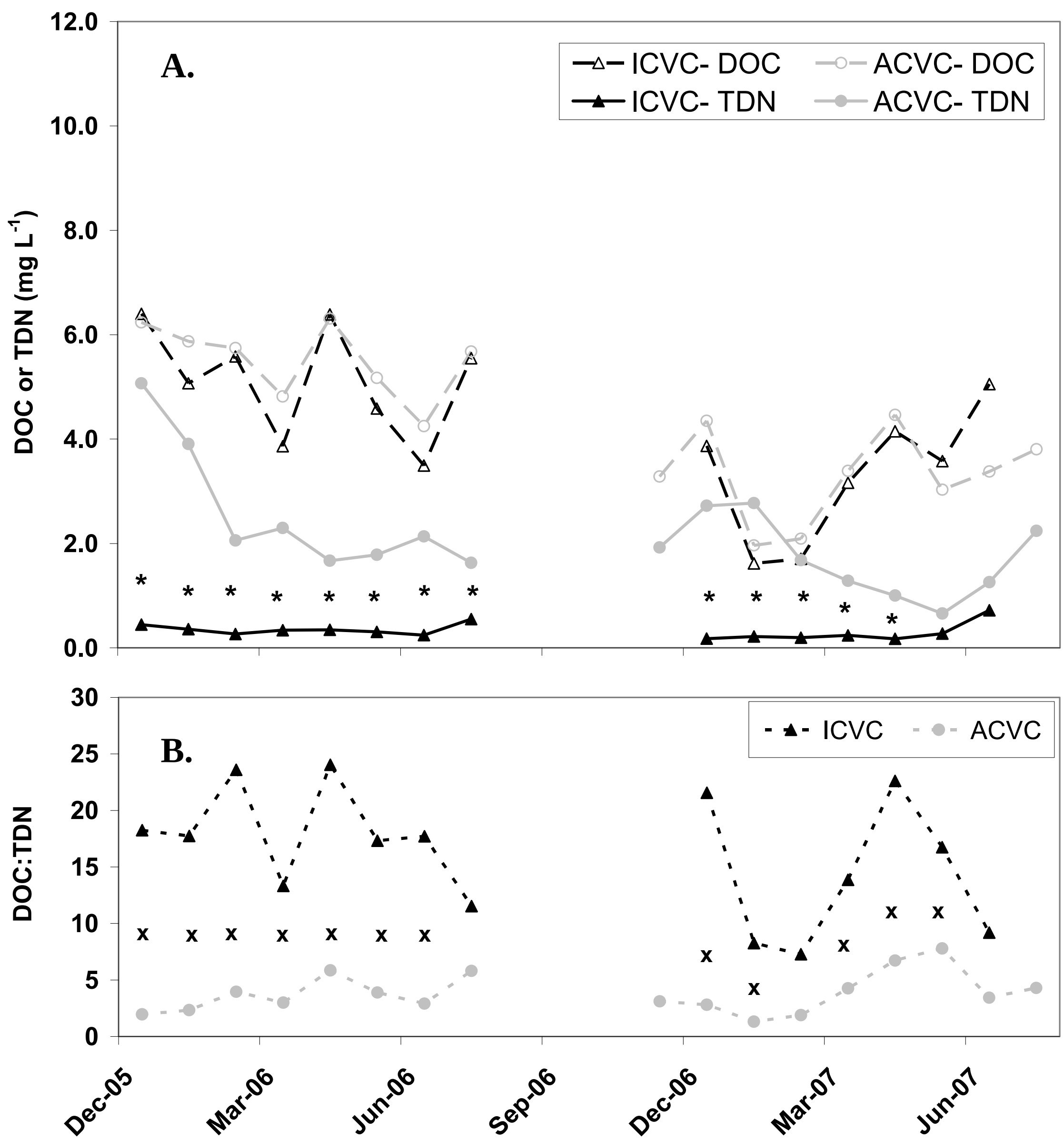


Figure 3.5. Effect of competing vegetation control treatments on soil water **(A)** total dissolved nitrogen (TDN), dissolved organic carbon (DOC), and (**B)** DOC:TDN ratio at 60 cm depth at Molalla. ICVC = initial competing vegetation control. ACVC= annual competing vegetation control. * and x indicate significant difference at α=0.05 between treatments for the associated month in TDN and DOC:TDN ratio, respectively.

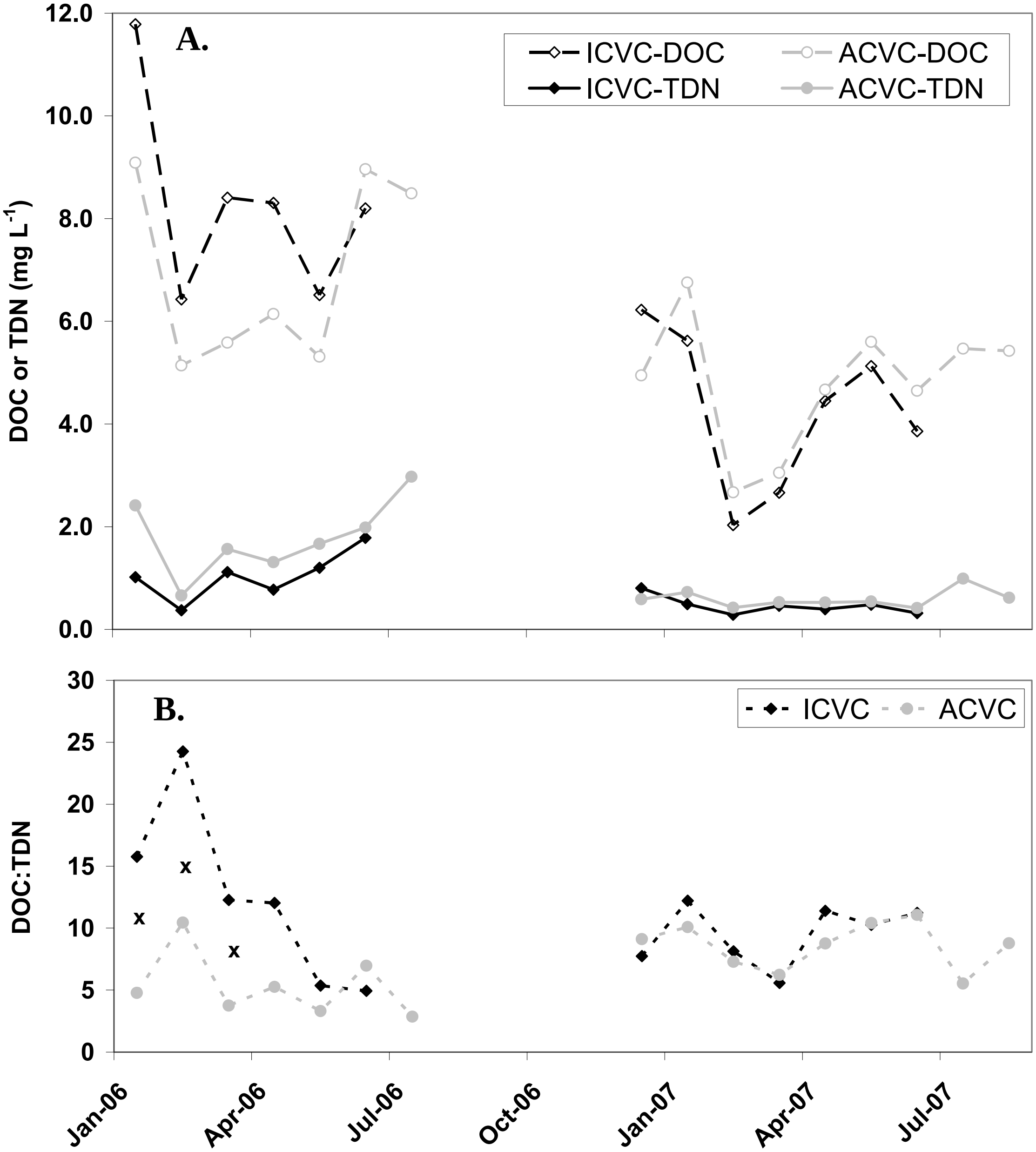


Figure 3.6. Effect of competing vegetation control treatments on soil water (A) total dissolved nitrogen (TDN), dissolved organic carbon (DOC), and (B) DOC:TDN ratio at 60 cm depth at Matlock. ICVC = initial competing vegetation control, ACVC= annual competing vegetation control. [x] indicates significant difference at α=0.05 between treatments for the associated month in DOC:TDN ratio.

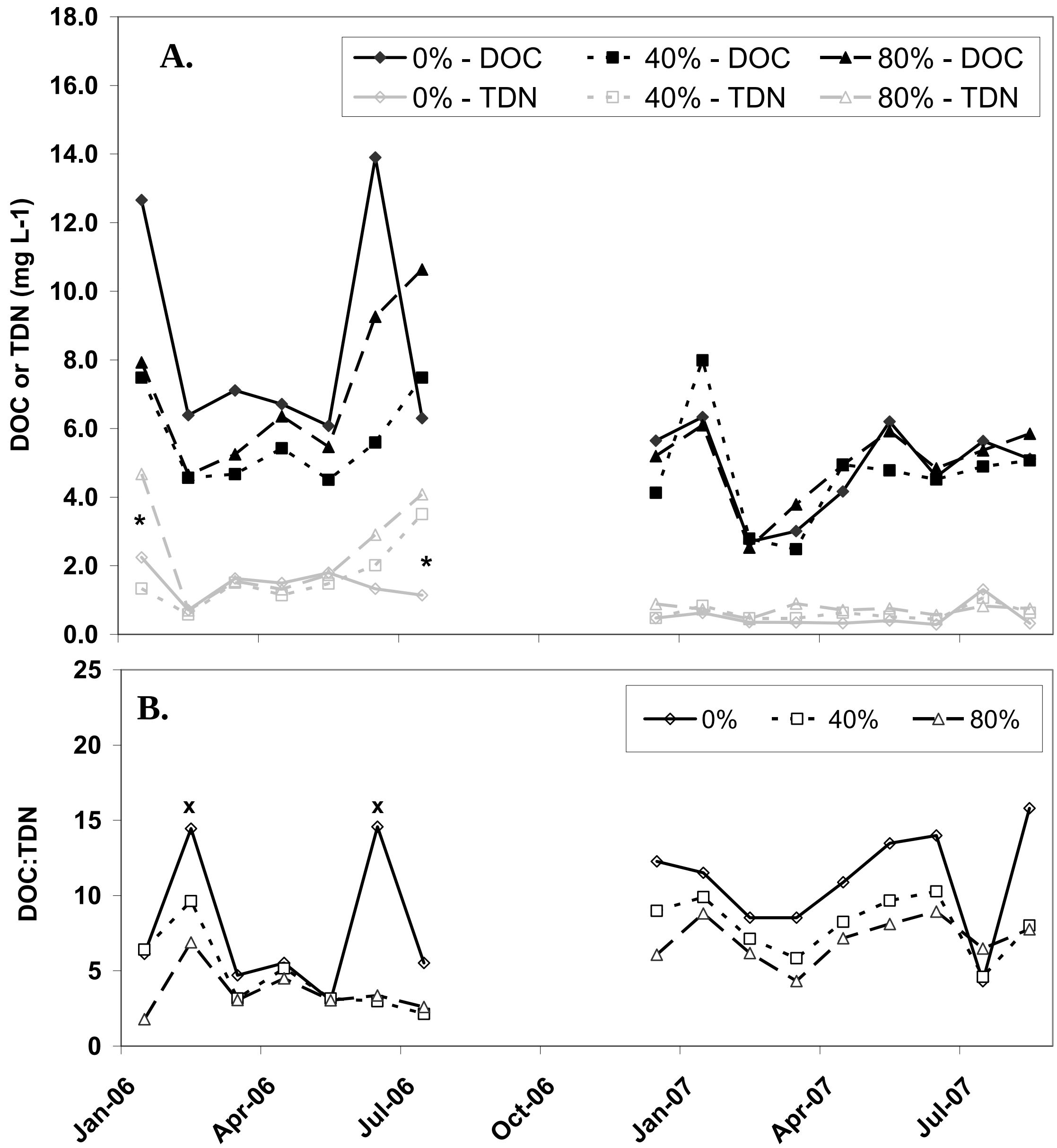


Figure 3.7. Effect of logging-debris treatments on soil water (**A)** total dissolved nitrogen (TDN), dissolved organic carbon (DOC), and (**B)** DOC:TDN ratio at 60 cm depth when annual competing vegetation control was applied at Matlock. 0%, 40%, and 80% are the respective logging-debris coverage for each treatment; * and x indicate significant difference at α=0.05 between treatments for the associated month in TDN and DOC:TDN ratio, respectively.

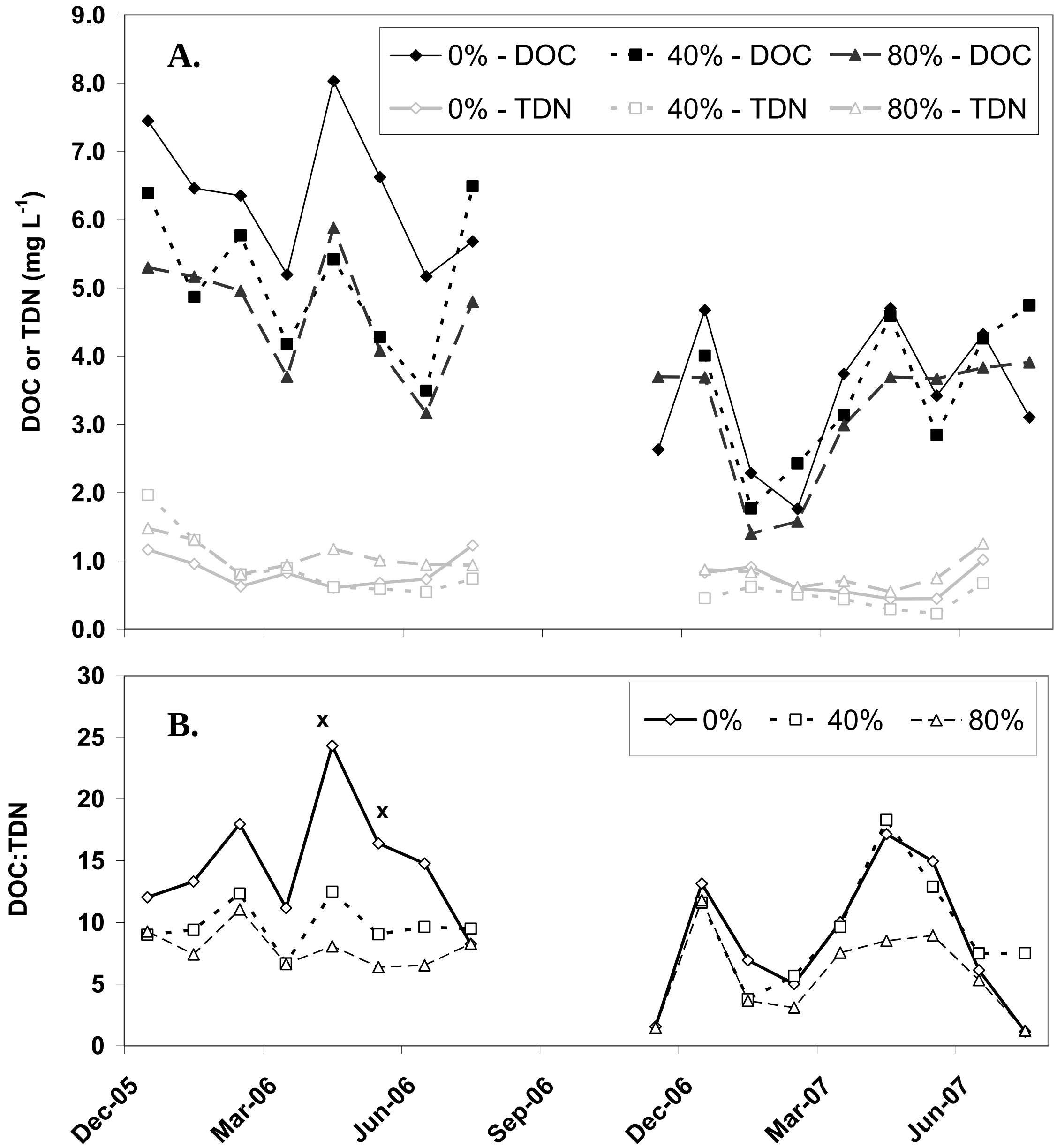


Figure 3.8. Effect of logging-debris treatments on soil water (A) total dissolved nitrogen (TDN), dissolved organic carbon (DOC) and (B) DOC:TDN from 60 cm depth at Molalla. 0%, 40%, and 80% are the respective logging-debris coverage for each treatment; ^{x} indicates significant difference at α=0.05 amongst treatments for the associated month in DOC:TDN.

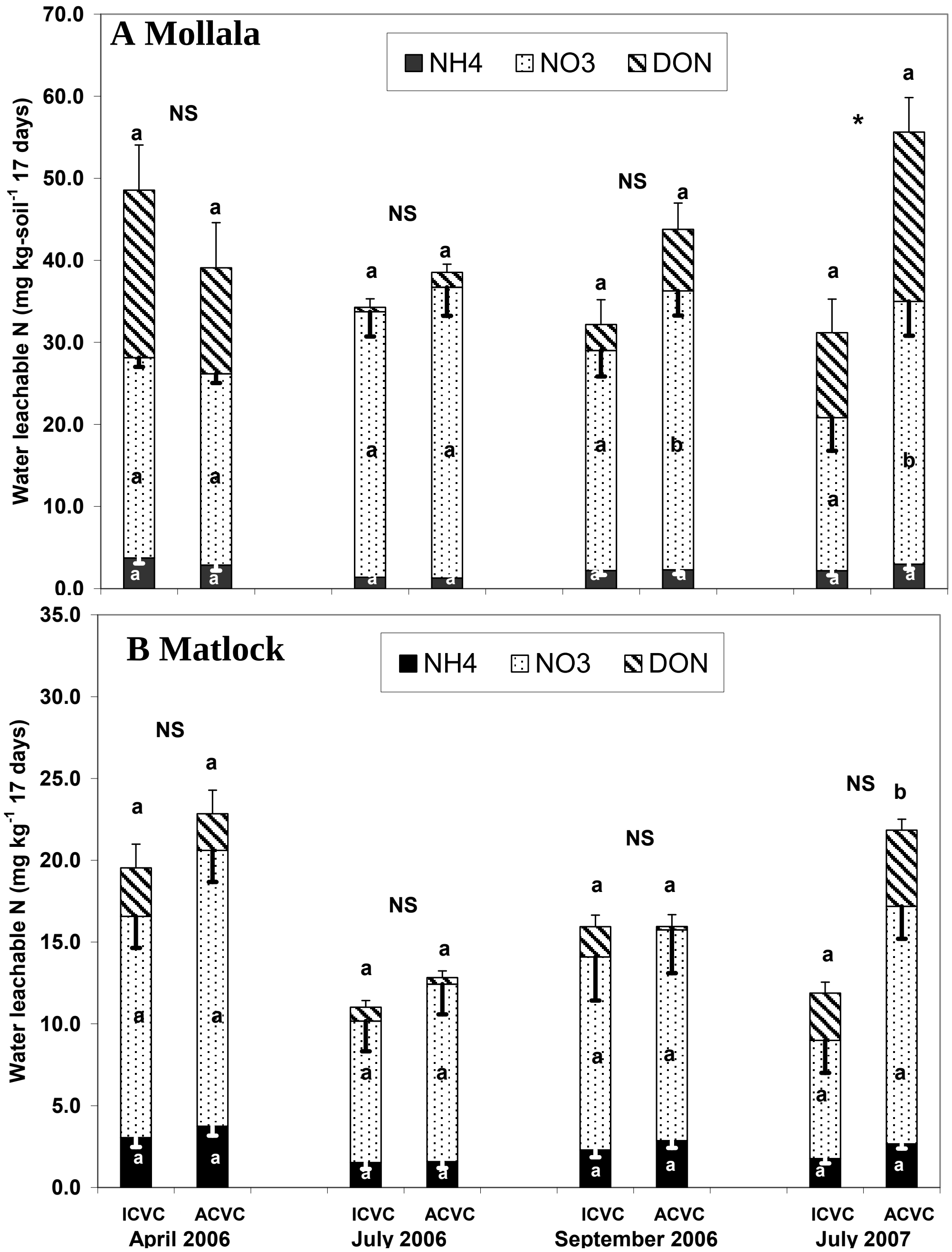


Figure 3.9. Effect of competing vegetation control and incubation period on water leachable ammonium-N, nitrate-N, dissolved organic nitrogen (DON) and total dissolved nitrogen (TDN) at (A) Molalla, and (B) Matlock. Means within an incubation period with different letters are significantly different at α=0.05. NS and * indicate non-significant and significant differences between treatments in TDN. Error bars are standard error, no error bars are shown for TDN. Note difference in scale between panels. ICVC=initial competing vegetation control and ACVC=annual competing vegetation control.

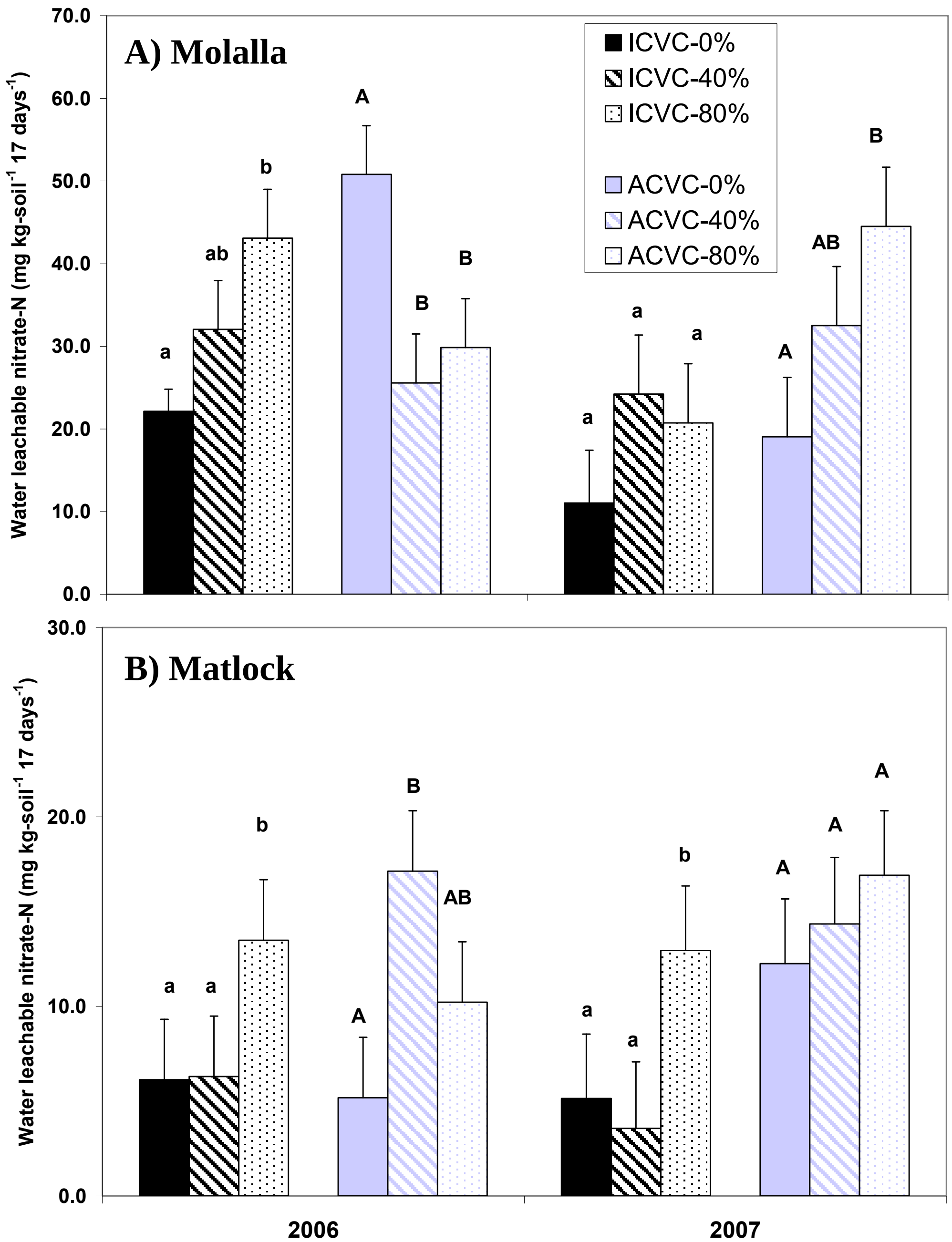


Figure 3.10. Effect of competing vegetation control and logging-debris treatments on water leachable nitrate-N following a 17-day incubation in July of each year at (A) Molalla, and (B) Matlock. ICVC= initial competing vegetation control, ACVC = annual competing vegetation control. 0%, 40%, and 80% are the respective logging-debris treatment coverages. Means with different letters within a CVC treatment and year are significantly different at α=0.05. Error bars are the standard error of the mean, n=4. Note difference in scale between panels.

Chapter 4

Soil and Foliar Nitrogen Responses to Management Practices Following Forest Harvest in the Pacific Northwest

Abstract

Logging-debris retention and competing vegetation control (CVC) can alter nitrogen (N) cycling response to forest harvest, but their short-term effect on tree N acquisition is not clear. Experimental treatments of logging-debris retention (0, 40, or 80% surface coverage) and CVC (initial or annual applications) were installed at two sites in the Pacific Northwest two years following clearcut harvest of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*) stands to assess their short-term effect on N acquisition by planted Douglas-fir and soil N supply. Foliar variables (needle N concentration, N content, and mass) and soil available N were measured for 3 years, potential N mineralization (Nmin) was measured at 4 periods in years 2 and 3, and soil total N to a depth of 20 cm was determined at the beginning and end of the experiment. Application of annual CVC significantly increased foliar N concentration and content in most years at both sites, which was associated with significantly higher available soil N and increased soil water content. Increased available N was likely due to reduced vegetative uptake, but higher nitrification may have also contributed to the increase, particularly at the site with high initial N. Potential N mineralization was significantly reduced during one sample period in 2006 at both sites when annual CVC was applied, but no effect was observed for the remaining periods. Logging-debris retention treatments had no detectable effect on any of the foliar variables or soil available N at

either site. There was no difference in soil total N between CVC treatments at either site, but there was a significant effect of logging-debris retention at the relatively low initial N site, where soil total N concentration was 0.8 g N kg^{-1} higher in the 80% coverage compared to the 0% coverage when annual CVC was applied. Competing vegetation control is an effective means to increase tree N acquisition in the initial years after planting, but the effect of logging-debris retention on tree N acquisition appears to be limited at these sites.

Introduction

Increasing demand for wood products in conjunction with a shrinking land base available for production (FAO 2000) creates expanding challenges for sustainable use of the forest resource at the global scale. Intensively managed plantations have been suggested as a potential solution to this conundrum (Fox, 2000; Nambiar, 1996), but success of such an approach is dependent on maintaining soil functions and processes critical to forest productivity and other ecosystem services. Many have voiced concern about the potential for intensive management to degrade soil (Johnson et al., 2002; Jurgensen et al., 1997; Powers et al., 1990), especially certain practices implemented at time of harvest and stand regeneration. However, results from both short- and long-term studies are equivocal, inhibiting development of policy and practice designed to maintain long-term soil and site productivity. Because of this, some have argued for a greater understanding of site-specific response to management practices, as it is likely that effects on soil functions will vary with site factors (e.g. climate, plant community and crop species, soil physical and chemical properties, etc.) (Fox, 2000; Nambiar, 1996).

Harvest-related effects (i.e. during extraction and site preparation) on total soil N and N supply have potential to alter short- and long-term site productivity given the well-documented N limitation to tree growth in many areas of North America (Johnson, 1992; Keeney, 1980). Past studies have focused on removal of logging debris at time of harvest as a likely practice that would alter the N cycle and N availability to crop trees following harvest (Powers et al., 2005). Most of these studies compared bole-only type treatments where non-merchantable material is retained on site, to whole-tree type treatments where the entire aboveground portion of the tree after felling is removed from the site. Whole-tree harvesting removes between 2 and 3 times more N than bole-only harvests (Carter et al., 2002; Powers et al., 2005), which could reduce N availability for acquisition by succeeding crop trees. However, N mineralization generally increases following harvesting (Prescott et al., 1997; Vitousek et al., 1992), and changes in N supply following logging-debris retention may be inconsequential to early tree growth (Smethurst and Nambiar, 1990b).

Logging debris can act as a source of N to soil via mobilization (e.g. simple dissolution, microbial degradation) and leaching of pre-existing N compounds within debris, or following colonization by N-fixing bacteria (Jurgensen et al., 1987; Jurgensen et al., 1984). More common is for debris to act as a N sink where N becomes immobilized during decomposition of high C:N materials (Carlyle et al., 1998; Laiho and Prescott, 1999; Palviainen et al., 2004). Logging debris also modifies the soil environment, generally reducing soil temperature and either reducing or increasing soil moisture (Devine and Harrington, 2007; McInnis and Roberts, 1995). Soil temperature and moisture are known to have large influence on N mineralization rates (Knoepp and Swank, 2002; Zak et al., 1999), and changes in the soil environment following logging-debris retention could either be conducive to, or impede net N mineralization. Considering the multiple modes and potential interactions amongst them, it is not surprising that effects on total soil N and net N mineralization following logging-debris retention have been shown to be positive (Chen and Xu, 2005; Piatek and Allen, 1999), negative (Blumfield and Xu, 2003; Smethurst and Nambiar, 1990b), or neutral (Carter et al., 2002; Li et al., 2003; Mendham et al., 2003). The effect of these changes on tree N acquisition is unclear, as total soil N and net N mineralization are at best indices of N availability (Schimel and Bennett, 2004), and provide no direct information regarding actual plant N acquisition.

Competing vegetation control (CVC) is a common site preparation practice following harvesting to increase crop tree survival and growth in the initial years following planting. Short-term effects of CVC on crop tree growth are generally positive (Harrington, 2006; Imo and Timmer, 1999; Roberts et al., 2005), and have been attributed to increased water availability (Morris et al., 1993; Roberts et al., 2005) or increased N availability (Imo and Timmer, 1999; Smethurst and Nambiar, 1989). However, effects of CVC on soil N pools, N availability, and tree acquisition of N are not as well understood. Competing vegetation control has been shown to be an important factor controlling N loss following harvesting (Marks and Bormann, 1972; Vitousek et al., 1979), which may cause a reduction in soil N if inputs (e.g. atmospheric deposition, N fixation) or initial site N are low (Miller et al., 2006). Greater N loss following CVC is mostly due to reduced vegetative competition for N (Roberts et al., 2005; Smethurst and Nambiar, 1995), but reduced N immobilization associated with reduced organic matter inputs may also contribute (Hart et al., 1994; Vitousek et al., 1992). Other factors likely to influence N supply that are modified by CVC include biomass and population structure of soil microbial communities (Busse et al., 2001; Li et al., 2004), and the soil environment (Gurlevik et al., 2004; Roberts et al., 2005). Any reduction in soil N or potential N supply following CVC could limit future productivity of the stand when tree N demand increases.

Forests in the Pacific Northwest (PNW) are considered some of the most productive in the world due largely to a favorable climate. Many of these forests respond positively to N additions as fertilizer (Hermann and Lavender, 1999), indicating limitations to growth associated with N availability. Forest management, if oriented towards maximizing crop-tree productivity, is expected to become progressively more intensive in this region (Adams et al., 2005; Moores et al., 2007) potentially increasing the magnitude of logging-debris removal and the duration of CVC following harvest. Although a number of studies have examined the influence of logging-debris retention and CVC on soil N cycling and availability (e.g. Kranabetter et al., 2006; Sanchez et al. 2005; Olsson et al., 2000; Piatek and Allen, 1999), relatively few have occurred in the PNW (e.g. Roberts et al. 2005).

The objectives of this research are to determine the effect of logging-debris retention and CVC on 1) foliar N status in Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) seedlings for three years following planting, 2) soil available N (KCl-extractable) and potential net N mineralization to determine if there were relationships between these variables and the foliar response, and 3) soil total N to assess the potential for longer-term effects on N supply. Two sites that differ in soil characteristics and annual precipitation were utilized to examine if the response to harvest practices varied with these factors.

Methods

Site Characteristics

This study is part of a larger research project initiated at two sites in 2003 to assess effects of logging-debris retention and CVC treatments on soil properties, nutrient cycling, and Douglas-fir growth. Both study sites are affiliates of the Long Term Soil Productivity (LTSP) network (Powers et al. 1990). Potential productivity (site index) is similar between sites, but large differences exist in precipitation and soil properties (Table 4.1). Site 1 (hereafter referred to as Matlock) is located on the Olympic Peninsula in WA, approximately 45 km NW of Olympia near the town of Matlock. Soil at Matlock is classified as a sandy-skeletal, mixed, mesic, Dystric Xerorthents, formed in glacial outwash with slopes ranging from 0 to 3% (Soil Survey Staff, USDA-NRCS). Site 2 (hereafter referred to as Molalla) is located approximately 24 km NE of the town of Molalla, OR in the foothills of the western Cascades. Soil at Molalla is classified as fine-loamy, isotic, mesic Andic Dystrudepts, formed in basic agglomerate residuum with slopes ranging from 2 to 40% (Soil Survey Staff, USDA-NRCS). The regional climate is

Mediterranean, with mild, wet winters and dry, warm summers with periods of prolonged drought (> 2 mo) being common. Precipitation falls almost entirely as rain, but some snowfall does occur during winter months.

Experimental design and treatment application

Sites were initially clear-cut harvested with chainsaws in March (Molalla) and April (Matlock) of 2003. Trees were removed with ground-based mechanized equipment along marked machine trails that were evenly distributed across plots to minimize experimental error associated with soil disturbance. Following harvest, a 2 x 2 randomized complete block factorial design was installed at each site. The factors were harvest type (two levels - bole-only or whole-tree) and CVC (two levels - intial CVC or annual CVC). The factorial combinations were replicated four times in a randomized complete block design and applied to 0.3-ha plots (50 m x 60 m). All plots received an initial application of herbicide to reduce competing vegetation; at Molalla glyphosate was aerially applied in August, 2003 and triclopyr was applied with backpack sprayers at Matlock during September of 2003. Following this initial application, only those treatments assigned annual CVC were treated with herbicide in the spring of each year. Both sites were planted with plug+1 bare-root Douglas-fir seedlings in February (Molalla) and March (Matlock) of 2004 at a 3 m x 3 m spacing (1,111 trees ha^{-1}). Each site was enclosed with a 2.5 m high fence to prevent browse damage to seedlings.

In March of 2005, three subplots within each of the whole-tree harvest treatment plots (n=8) were identified for application of a subplot logging-debris retention treatment.

Subplots encompassed a 2 m x 2 m area centered on a single planted Douglas-fir seedling. This design modification was chosen to reduce experimental error associated with treatment application (e.g. discontinuous logging-debris coverage at the whole-plot level) and spatial variability of soil chemical properties. Use of only whole-tree treatment plots was arbitrary (i.e. compared to use of the bole-only plots) as we only had sufficient resources for 24 subplots at each site, and could not address potential interaction between the whole-plot harvest factor and the subplot factor given the level of replication. Woody logging-debris was randomly applied at a visually estimated surface coverage of 0%, 40%, or 80% to one of the subplots in each whole-plot. For each assigned treatment application, logging debris 5.0 to 12.5 cm in diameter that was within the associated whole-plot was stacked in a systematic criss-cross fashion until the assigned coverage (+/- 10% with visual determination) was reached. In the case of the 0% treatment, all logging debris was removed from the subplot, but no attempt was made to remove legacy wood if present. The overall design is a randomized complete block split-plot with one whole-plot factor (CVC treatment) and one subplot factor (logging-debris coverage).

Soil sampling and analysis

Soil samples were collected at intervals over the 2005-07 period (Table 4.2) to assess treatment effects on available N, potential net N mineralization, and total soil N. Mineral soil was collected to a depth of 20 cm at three locations in each replication and composited in the field, taking care to remove any large organic material prior to compositing. Samples were collected volumetrically in 2005 with a “push” tube sampler and with a core sampler attached to a slide hammer in 2006 and 2007. Composite samples were collected in plastic bags, placed on ice, and immediately transported to the laboratory for processing. For periods when available N was assessed (Table 4.2), samples were stored (<1 wk) in a refrigerator at 4 ºC until processing, otherwise samples were immediately air-dried, sieved to pass a 2 mm mesh, and then stored at 4 ºC until analysis.

Available N is defined here as the amount of inorganic N (nitrate-N and ammonium-N) that can be extracted from the soil with a strong salt solution at a given sample period. For available N determination, soil at field moisture was sieved to pass a 2 mm mesh at 4 ºC. Most samples were sufficiently dry to sieve by shaking, but some with higher moisture were pushed through the mesh with a rubber stopper. Immediately following sieving, soil samples were extracted with 2M KCl at an approximate 10:1 solution to soil mass ratio. Extraction consisted of shaking a known mass of soil in the KCl solution for 30 minutes, followed by gravity filtration through Whatman #42 filter paper. Extracted salt solutions were analyzed for ammonium-N and nitrate-N concentration on a Lachat Quick Chem 4200 analyzer. Soil subsamples were dried at 105 ºC until constant mass was attained to report estimates of available N on a dry soil mass basis. The remainder of the sieved field-moist sample was air-dried and then stored at 4 ºC.

Potential N mineralization (sum of ammonification and nitrification) was assessed with an aerobic incubation approach. Initial inorganic N was extracted from air-dry soil with 2M KCl as described above. Approximately 50 g of air-dried sieved soil was placed in a microlysimeter constructed of benchtop filtration units (Falcon Filter, Becton Dickinson Labware) as described by Nadelhoffer (1990). Samples were incubated for 17 days at 25ºC and at a soil water potential of -22 kPa. Soil in each incubation unit was initially leached with 100 ml of ultrapure water by applying a tension of -22 kPa with the use of a vacuum pump. Leachate volume was measured with a graduated cylinder, and an aliquot was separated for inorganic N analyses. Incubation unit mass was periodically checked during the incubation period, and water was added as needed to maintain the initial soil water content following wetting. After 17 days, soil was again leached with 100 ml of ultrapure water, and leachate was processed in the same manner as the initial leaching.

At the end of each incubation, soil was removed from the microlysimeter, mixed thoroughly, and then inorganic N was extracted as described above for a final measurement. Soil subsamples were dried at 105 ºC until constant mass was attained for both extractions (initial and final) to report estimates on an oven-dry soil mass basis. Salt and water solutions were analyzed for ammonium-N and nitrate-N concentration on a Lachat Quick Chem 4200 analyzer. Potential net N mineralization was calculated as:

$$\text{Net Nmin} = (\text{Initial leach N} + \text{Final leach N} + \text{Final KCl N}) - \text{Initial KCl N} \qquad [1]$$

where Nmin is N mineralization, leach N is water leachable inorganic N (sum of NH_4^+-N and NO_3^--N), KCl N is KCl-extractable inorganic N (sum of NH_4^+-N and NO_3^--N), and final and initial are the time of extraction or leaching. Potential net ammonification and nitrification were determined in the same manner but with analysis of only NH_4^+-N and NO_3^--N for ammonification and nitrification, respectively.

Total soil N was determined for samples collected in the first and last sampling period in 2005 and 2007, respectively (Table 4.2). Approximately 5 g of air-dry soil was separated from the bulk sample and ground with a mortar and pestle to pass a 60 mesh sieve (0.25 mm). Samples were dried at 65 ºC for 24 hrs, and then dry combusted on a CNS analyzer (Leco CNS-2000 Macro Analyzer, St. Joseph MI) to determine total soil N concentration.

Volumetric soil water content (SWC) was measured at 4-hr intervals from a depth spanning 20 to 40 cm below the soil surface at each replication with ECH_20 probes (Decagon Devices, Inc., Pullman, WA). Sensors were installed with a bucket auger approximately 45 cm from each tree. Calibration equations were developed in the laboratory for each of the sites by comparing gravimetrically measured SWC with that measured by the sensor. For interpretation of treatment effects, SWC was averaged first by day and then by month to determine a mean SWC for each replication and treatment.

Foliar sampling and analysis

In the fall of each year (Table 4.2), foliage samples were collected from each tree to determine treatment effects on foliar N concentration, N content, and needle mass. At each collection period, three current-year shoots were randomly collected from the second whorl of each tree and placed in a paper bag. Samples were transported to the laboratory, buds were removed, and the remaining portion of each sample was dried in paper bags at 65 ºC to a constant weight. After the initial drying, needles were stripped from each shoot, and thoroughly mixed. A subset of 100 intact needles was separated from the bulk sample, returned to the oven at 65 ºC for several hours, and then weighed to determine oven-dry mass of 100 needles. The remaining sample was placed in a plastic bag and analyzed for foliar N. Samples were ground in a Wiley mill to pass a 1 mm mesh, dried at 65 ºC until constant mass was attained, and then analyzed on a Leco CNS-2000 Macro Analyzer for N concentrations Nitrogen content was calculated as the product of needle mass and N concentration.

Vector analysis (Haase and Rose, 1995; Timmer and Stone, 1978) was used to assess foliar response patterns amongst treatments. Vector analysis is a graphical diagnostic technique that simultaneously displays response of three variables (foliar mass, foliar nutrient concentration, foliar nutrient content) for a treatment relative to a control. The directional shift of the foliar mass-concentration-content vector between control and treated trees differentiates among alternative treatment responses (i.e. N dilution, luxury N consumption, N accumulation, N concentration, see Figure 4.1). For this analysis, means were plotted relative to the 0% logging-debris treatment for each CVC treatment to assess any effect of logging debris on foliar response.

Statistical analysis

A mixed model approach with repeated measures was used to assess treatment effects on foliar variables (needle N concentration, N content, and needle mass), net ammonification, net nitrification, net N mineralization, available N (nitrate-N, ammonium-N, total inorganic N), and SWC. Block, whole plot within block, and subplot nested within whole plot were modeled as random effects, and the whole-plot factor, subplot factor, time variable (either year or sample period), and their interactions were modeled as fixed effects. For each dependent variable at each site, the covariance matrix used for repeated measures was identified by fitting the model to all possible candidate matrices and then choosing the matrix which resulted in the lowest fit criteria statistic (BIC). Soil N concentrations in 2005 and 2007 were analyzed separately with a mixed model approach, where block, whole plot within block, and subplot nested within whole plot were modeled as random effects, and both the whole-plot and subplot factors were modeled as fixed effects. Examination of the residuals for each variable indicated that assumptions of homogenous variance and normality were valid.

When significant interaction was observed between the subplot factor and time, *a priori* orthogonal contrasts were performed to test for significant differences between 1) the absence and presence of debris (0% coverage versus both the 40% and 80% coverage), and 2) 40% and 80% logging-debris coverage. When significant interaction was observed for the time x whole-plot x subplot term, treatment effects were initially assessed by slicing with time and the whole-plot factor held constant, followed by slicing with time and the subplot factor held constant. Tukey's Honestly Significant Difference test was used to determine significant differences among means for the soil N data. Pearson correlations were calculated between foliar N content and available soil N forms in each year, and between foliar N content and N supply (nitrification, ammonification, mineralization) measures estimated from the July incubations in 2006 and 2007 to determine if any relationship existed between these variables. An alpha level of 0.05 was used to assess statistical significance in all evaluations. All statistical analyses were performed with SAS V 9.1 (SAS Institute, Cary NC).

Results

Volumetric soil water content

At both sites and in each year, there was significant interaction between CVC treatment and month on SWC (Table 4.3), where plots with initial CVC had significantly lower SWC then those with annual CVC during some summer months (from June to September) (Figure 4.2). At Matlock during months where significant differences existed, mean SWC in the initial CVC treatment was approximately 0.08, 0.05, and 0.02 $m^3\ m^{-3}$ lower than the annual CVC treatment for 2005, 2006, and 2007, respectively. At Molalla, mean SWC in the initial CVC treatment was approximately 0.08, 0.09, and 0.06 $m^3\ m^{-3}$ lower than the annual CVC treatment for 2005, 2006, and 2007, respectively.

Effects of logging debris were generally less pronounced than those for CVC treatments at both sites. There was significant three-way interaction amongst CVC, logging debris, and month at Matlock in 2005 (Table 4.3). For most months during June through September of that year, there was a general trend of decreasing SWC content with increasing logging-debris coverage when annual CVC was applied, but neither of the *a priori* contrasts were significant for any month (Figure 4.2). The opposite pattern was observed in the initial CVC treatment, where 0% logging-debris coverage was approximately 0.05 $m^3 m^{-3}$ lower than both 40 and 80% coverage in June and July of 2005 (Figure 4.2).

At Molalla, there was a significant main effect of logging-debris coverage on SWC in 2006, where annual mean SWC content decreased in the order 0% (mean 0.40 $m^3 m^{-3}$, SE=0.02), 80% (mean=0.36 $m^3 m^{-3}$, SE=0.02), and 40% coverage (mean=0.33 $m^3 m^{-3}$, SE=0.02). The same pattern was observed in 2005 and 2007, but differences were not significant (Table 4.3). In 2007, 0% coverage was significantly greater than either 40% or 80% coverage during most months when annual CVC was applied (Table 4.3), with differences during the growing season ranging from a high of 0.09 $m^3 m^{-3}$ in April, to a low of 0.03 $m^3 m^{-3}$ in September (Figure 4.2). In contrast, in the initial CVC treatment in 2007, 80% coverage had significantly higher SWC than the 40% coverage in most months, with the difference ranging from 0.09 $m^3 m^{-3}$ in April to 0.05 $m^3 m^{-3}$ in September. Visual examination of data indicates differences between the 0% and 80% coverage were similar to those between the 40% and 80% coverage.

Total soil N

There was a significant main effect of logging debris on soil N at Matlock for both sample periods ($F_{2,12}$ = 4.66, p=0.032 in 2005, $F_{2,12}$ = 6.36, p=0.013 in 2007). In 2005, mean soil N increased with increasing logging-debris retention, with N concentration in the 80% coverage being 0.6 g N kg^{-1} (se = 0.2) greater than the 0% coverage (Table 4.3). Interaction between logging debris and CVC was significant in 2007 ($F_{2,12}$ = 4.30, p = 0.039) mostly due to the 40% cover being lower by 0.9 g N kg^{-1}

(se = 0.3) than 0% and lower by 1.2 g N kg^{-1} (se = 0.3) than 80% when initial CVC was applied. When annual CVC was applied, the pattern was the same as that observed in 2005, with a difference of 0.8 g N kg^{-1} (se = 0.3) higher in the 0% coverage than the 80% coverage. In contrast to Matlock, there was no significant effect of logging debris on soil N for either sample period at Molalla ($p > 0.40$), and no apparent trend amongst means (Table 4.4). There was no effect of CVC treatment on total soil N at either site in 2005 or 2007 . Regardless of treatment, mean soil N tended to increase at both sites over the two-year period (Table 4.4).

Potential N supply and available N

There was significant interaction between incubation period and each of the treatments on net ammonification, nitrification, and total N mineralization at both sites (Table 4.5), but most of the effects were associated with CVC treatment. Total net N mineralization (sum of net ammonification and nitrification) was significantly lower following annual CVC at both sites during the July 2006 incubation, with estimated reduction of 0.91 mg N kg-$soil^{-1}$ day^{-1} (se = 0.42) at Matlock (Figure 4.3A) and 1.60 mg N kg-$soil^{-1}$ day^{-1} (se = 0.48) at Molalla (Figure 4.3B). No differences were found between CVC treatments for any other incubation period (Figure 4.3). The effect of annual CVC in July 2006 was largely driven by a reduction in ammonification at both sites (Figure 4.3). Nitrification was significantly increased by 0.91 mg N kg-$soil^{-1}$ day^{-1} (se=0.42) following annual CVC in the September 2006 incubation at Molalla (Figure 4.3B), and by 0.44 mg N kg-$soil^{-1}$ day^{-1} (se=0.21) in the July 2007 incubation at Matlock

(Figure 4.3A). With the above exceptions, no general treatment or temporal patterns in ammonification and nitrification were observed at Matlock (Figure 4.3A), but mean ammonification tended to be lower, and nitrification higher, following annual CVC at Molalla for all incubation periods relative to initial CVC (Figure 4.3B).

Significant interaction between incubation period and logging-debris treatment was observed for nitrification at Molalla, and three-way interaction amongst period and both treatment factors was observed at Matlock for nitrification and ammonification (Table 4.5). Most of the interactions were associated with lower N supply in the 40% coverage. At Molalla, nitrification was significantly lower in the 40% logging-debris coverage relative to the other coverage treatments in the April 2006 incubation (mean of 3.11, 2.13, and 3.49 mg N kg-soil^{-1} day^{-1} for the 0%, 40%, and 80% coverage, respectively), with a similar pattern observed in the July 2006 incubation (mean of 2.94, 2.12, and 2.75 mg N kg-soil^{-1} day^{-1}for the 0%, 40%, and 80% coverage, respectively). At Matlock in the initial CVC treatment, ammonification was significantly lower in the 40% logging-debris coverage compared to the other two coverage treatments in the April 2006 incubation (lower by 1.62 and 2.31 mg N kg-soil^{-1} day^{-1} compared with the 0% and 80% coverage, respectively), and significantly lower (by 1.19 mg N kg-soil^{-1} day^{-1}) than the 80% coverage in the September 2006 incubation. Nitrification was approximately 0.43 mg N kg-soil^{-1} day^{-1} higher in the 80% coverage than either of the other levels when initial CVC was applied at Matlock for the July 2007 incubation.

Main effect of CVC treatment on available N in July of each year was significant at both sites, but the magnitude varied by year for most of the N-availability variables

(Table 4.6). In each year and at both sites, available nitrate-N and total available N were significantly higher when annual CVC was applied compared to initial CVC (Figure 4.4). The increase in total available N was estimated as 16.6, 11.5, and 5.9 mg N $kg\text{-}soil^{-1}$ (se all 2.2 mg N $kg\text{-}soil^{-1}$) in 2005, 2006, and 2007, respectively, at Matlock, and as 18.0, 23.1, and 13.1 mg N $kg\text{-}soil^{-1}$ (se all 2.3 mg N $kg\text{-}soil^{-1}$) for 2005, 2006, and 2007, respectively, at Molalla There was also a significant increase in ammonium-N with annual CVC at both sites, but differences were only significant in 2005 and 2006 (Figure 4.4). For all sample periods at Molalla, nitrate-N contributed more to total available inorganic N than ammonium-N when annual CVC was applied (56%, 56%, and 93% of total available inorganic N in 2005, 2006, and 2007, respectively) (Figure 3.4B), but the same effect was only observed in 2007 at Matlock, where nitrate-N contributed 81% of the total available inorganic N (Figure 3.4A). In contrast, ammonium-N contributed more to total available inorganic N in the initial CVC treatment at both sites in 2005 (84% at Molalla; 90% at Matlock) and 2006 (93% at both sites) (Figure 4.4). There was no effect of logging-debris retention on available N at either site or any sample year (Table 4.6).

Foliar analysis

Significant increases in needle N concentration and N content were observed at both sites following annual CVC, but increases in needle mass were only observed at Molalla (Table 4.7). Interaction between CVC treatment and year was significant for most of the foliar properties at each site (Table 4.7). At Matlock, annual CVC increased needle N concentration by 5.5 and 3.4 mg N g^{-1} in 2005 and 2007, respectively, in comparison with initial CVC (Table 4.8). At Molalla, needle N concentration was significantly higher following annual CVC in all years with mean increases of 2.9, 5.0, and 3.5 mg N g^{-1} for the 2005, 2006, and 2007 sample years, respectively, when compared with initial CVC (Table 4.9). For these same periods, total needle N content at Molalla was significantly increased by 2.5, 1.0, and 1.7 mg N in 2005, 2006, and 2007, respectively, with annual CVC compared with initial CVC, largely due to the increase in N concentration (Table 4.9). Significant effects on needle mass at Molalla were only observed in 2006 when needle mass was significantly lower in the annual CVC treatment relative to the initial CVC treatment (Table 4.9).

There was no significant effect of logging-debris retention on any of the foliar variables at either site (Table 4.7). Needle N concentration and content tended to increase with increasing logging debris at both sites in 2005 and 2006, but such a trend was not observed in 2007 (Tables 4.8, 4.9).

Vector analysis indicated variable effects of logging-debris retention on needle N by site, year, and CVC treatment (Figures 4.5, 4.6). Retention of logging debris at Molalla caused relatively higher needle N concentration, N content, and mass in the annual CVC treatment in 2005 (Figure 4.5A) and for the initial CVC treatment in 2006 (Figure 4.5B), which may indicate increased N supply relative to the 0% coverage. Patterns in other years were more subtle or inconclusive. For example, in the annual CVC treatment at Molalla in 2006 and 2007, 80% coverage had little effect on needle N concentration and mass, but 40% coverage had relatively lower N concentration and content (Figure 4.5A) possibly indicating a reduction in N supply relative to 0% coverage.

At Matlock, vector analysis indicated a general increase in needle mass and N content when logging debris was present for most years in the annual CVC treatment (Figure 4.6A; arrow indicates the general trend). In contrast, when logging debris was present in the initial CVC treatment, needle mass tended to decrease and N concentration tended to increase relative to no logging debris (Figure 4.6B). Two exceptions to this general pattern with initial CVC were the 40% coverage in 2006 which exhibited signs of luxury consumption (increased N uptake with no change in needle mass), and the 80% coverage in 2007 which had lower needle mass, N concentration, and N content relative to the 0% coverage (Figure 4.6B).

Relationships between foliar N and soil N measures

Across all years and treatments, needle N content was positively correlated with both available N forms at Matlock (ammonium $r = 0.33$, $p=0.006$; nitrate $r = 0.43$, $p<0.001$). When calculated for each year, these same variables were strongly correlated in 2005 (ammonium $r = 0.74$, $p<0.001$; nitrate $r = 0.85$, $p<0.001$), not correlated in 2006 ($p > 0.5$), and only significantly correlated between foliar N content and available ammonium in 2007 ($r = 0.43$, $p=0.048$). At Molalla, only available nitrate was positively correlated across all years to foliar N content ($r = 0.34$, $p=0.003$), with similar correlation observed in 2005 ($r = 0.42$, $p=0.039$) and 2007 ($r = 0.53$, $p=0.007$) when calculated for each year.

There was no significant correlation between any N supply measure and needle N content across both years at Molalla, but ammonification was negatively correlated with N content in 2006 ($r = -0.60$, $p=0.002$), and nitrification was positively correlated with needle N content in 2007 ($r = 0.47$, $p=0.021$). Ammonification was positively correlated to needle N content at Matlock when calculated across both years ($r = 0.32$, $p=0.030$), but there was no correlation between any N supply measure and needle N content within a given year.

Discussion

Responses of foliar N, available N, and N supply

Competing vegetation control

The benefits of CVC to crop tree growth can be generally attributed to decreased competition for available resources required for growth including light, water, and nutrients, but light is generally not limiting to growth immediately following harvesting (Harrington, 2006). Application of annual CVC significantly increased needle N status (concentration and content) at both sites in most years after planting. Such an effect has been noted in previous studies (Harrington, 2006; Perie and Munson, 2000; Roberts et al., 2005; Woods et al., 1992), demonstrating the broad effectiveness of annual CVC to increase N acquisition in crop trees. Volume growth of trees receiving annual CVC at these sites is 2 to 4 times greater than that in initial CVC treatments (T. Harrington, unpublished data), indicating that the increase in N status was biologically significant.

Increased foliar N status could be a result of increased N available for uptake, increased N assimilation associated with increased available water, or a combination of both as they generally occur in concert following CVC (Ludovici and Morris, 1997; Powers and Reynolds, 1999; Roberts et al., 2005). Both SWC and available N were significantly affected by CVC at each site and in all years, thereby limiting evaluation of the causal factor contributing to the increase in foliar N status. For example, at both sites the greatest difference in foliar N concentration (Tables 4.10, 4.11) between the two treatments occurred in years with the greatest difference in SWC and available N (Figures 4.2, 4.4). The relative importance of each factor probably varies temporally, where available N is more important when soil water availability is sufficient, and available soil water is more important during severe drought (Roberts et al., 2005). Nevertheless, the variation in foliar N response amongst years at both sites indicates a complex interaction between soil water and available N on needle N status.

Reduced vegetative uptake was likely a major source of increased available N at both sites, as studies have shown that herbaceous and woody vegetation are capable of competing for substantial amounts of N (Roberts et al., 2005; Woods et al., 1992). The contribution of increased net N supply to the increase in available N is more difficult to assess. Studies conducted *in situ* have observed an increase (Li et al., 2003; Vitousek et al., 1992), or no effect (Lister et al., 2004; Meehan, 2006) on net N mineralization following CVC. Here, there was no significant increase in potential net N mineralization following annual CVC, but potential nitrification was significantly increased at both sites in one of the incubation periods, possibly due to a reduction in microbial immobilization (Vitousek et al., 1992). Mean nitrification was also (non-significantly) higher in plots treated with annual CVC relative to initial CVC plots for all incubations at Molalla

(Figure 4.3), and available nitrate *in situ* dominated the total available pool in the annual CVC treatment at that site (Figure 4.4). Increased soil moisture and temperature in the annual CVC at Molalla (Chapter 2) treatment would have most likely resulted in greater mineralization rates *in situ* as well (Knoepp and Swank, 2002; Zak et al., 1999). When increased available N has been observed following CVC it is usually because of increased nitrification (Li et al., 2003; Vitousek et al., 1992). It seems plausible that nitrification *in situ* was increased following annual CVC during the study period, with the effect most pronounced at Molalla.

Increased nitrification may have resulted in greater N acquisition by crop trees given its greater mobility in soil than ammonium. At both sites maximum foliar N concentration in the CVC treatment occurred in the year with the greatest measure of available nitrate (Tables 4.8, 4.9; Figure 4.4), providing some evidence that greater nitrate availability improves N acquisition. However, correlation analysis between needle N content and both available N forms and potential N supply revealed no consistent pattern. Needle N content was positively correlated to both available ammonium and nitrate in at least one year at each of the sites, with similar results observed for N supply measures. The sampling approach used here probably contributed to the mixed results because available N varies considerably over time (Li et al., 2003; Vitousek et al., 1992), and mineralizable N is a poor measure of plant-available N (Schimel and Bennett, 2004). If nitrate was a more important N pool for tree uptake than ammonium, the effect was probably more pronounced at Molalla given the greater effect of annual CVC on nitrification and available nitrate at that site, and the tendency for nitrate and nitrification to be more frequently correlated with needle N content at Molalla compared to Matlock.

Relative differences between CVC treatments for SWC and N availability were similar between sites, but the absolute changes were greater at Molalla than Matlock. These differences are not surprising given the much higher total soil N and total pore space present at Molalla compared to Matlock (Table 4.1). At Molalla, increased SWC and available N was observed in all years following annual CVC, but at Matlock the magnitude of the effect decreased with each successive year (Table 4.3, Figure 4.2). These results could indicate either degradation of soil properties that control SWC and available N over time (e.g. change in bulk density (Lister et al., 2004), reduction in total N (Miller et al., 2006)), or increased resource acquisition by crop trees in response to increased growth. Volume growth data (T. Harrington, unpublished data) indicate that tree growth is approximately 30% higher at Matlock than Molalla, suggesting that the apparent decrease in resource availability at Matlock with time is actually accounted for by increased acquisition by crop trees. The strong positive correlation between available N and needle N content at Matlock in 2005, but not during the remaining years, supports this possibility. If true, this implies greater efficiency of annual CVC at Matlock than Molalla, with increased resource availability being more fully utilized to maximize growth at Matlock. Such an effect may be due to higher soil temperatures at Matlock which would stimulate root uptake earlier in the growing season (Pregitzer et al., 2000), or differences in ectomycorrhizae abundance and diversity (Outerbridge and Trofymow, 2004), but is also likely associated with generally lower resource availability inherent to soil at Matlock compared to Molalla.

Potential N mineralization has been proposed as an important chemical indicator of soil quality (Burger and Kelting, 1999; Schoenholtz et al., 2000), and at least one study has equated a reduction in potential N mineralization following CVC to a reduction in soil quality (Echeverria et al., 2004). Here, significantly lower net N mineralization was observed following annual CVC at both sites in the July 2006 incubation, but no significant difference between CVC treatments was observed for the remaining periods. The variation in response could be associated with a number of factors including temporal variation in the microbial community structure (Rogers and Tate, 2001) or variation in organic matter quality (ease of mineralization) and quantity (Whalen et al., 2000). The causal factors cannot be evaluated with the data available, but the variation in response within and between years casts doubt on the usefulness of potential net N mineralization to assess reductions in soil quality at the site level. Clearly there are relationships between N mineralization and productivity at large spatial scales (Pastor et al., 1984; Reich et al., 1997), and conceptually there should be a relationship between N mineralization and soil quality. However, the utility of potential net N mineralization to assess changes in soil quality appears to be limited at the site level due to high spatial and temporal variability found at such scales (Laverman et al., 2000).

Logging-debris retention

A number of studies have documented increased N mineralization following forest harvesting (Smethurst and Nambiar, 1990a; Vitousek et al., 1992; Prescott et al., 1997), which has been attributed to a modified soil environment, surface soil mixing, and decreased vegetative uptake. This increase in available N supply could potentially be affected by logging debris either negatively (reduced) via a reduction in soil temperature (Devine and Harrington, 2007; McInnis and Roberts, 1995) and/or increased microbial immobilization (Hart et al., 1994; Vitousek et al., 1992), or positively (increased) via mobilization of N within debris and subsequent transport to soil. With few exceptions, the lack of any significant effects of logging-debris treatments on foliar N characteristics, available N, and potential net N mineralization indicates that logging-debris retention has little immediate influence on tree N nutrition and N supply at these sites. These results generally agree with past studies that have found no significant effect of logging-debris retention on either foliar N status (Kranabetter et al., 2006; Roberts et al., 2005; Thiffault et al., 2006) or net N mineralization (Carter et al., 2002; Fox et al., 1986; Kranabetter et al., 2006; Li et al., 2003; Roberts et al., 2005; Thiffault et al., 2006; Vitousek and Matson, 1985).

At both sites, tree volume growth has been significantly higher when 80% logging-debris coverage was present and annual CVC was applied (T. Harrington, unpublished data), indicating logging debris provides some resource which is limiting to growth that can be utilized when competing vegetation is not present. Vector analysis revealed several instances where logging-debris retention appeared to have a positive effect on foliar N status, which may have contributed to the increase in growth. At Molalla, when annual CVC was applied, there was greater relative foliar N concentration and content when logging debris was retained in 2005 (Figure 4.5). At Matlock, there was a general increase in foliage mass and N content following logging-debris retention when annual CVC was applied in 2006 and 2007 (Figure 4.6). It appears that trees are acquiring and allocating more N to foliage when logging debris is retained, but only in certain years and in small quantities. Given this, it seems unlikely that increased growth in the 80% coverage when annual CVC was applied is primarily due to increased N availability and greater crop tree uptake. It is more likely that the response is associated with increased availability of some other growth-limiting nutrient, or some factor not assessed in this study.

Although the effect of logging debris on needle N status and available N at these sites was weak or non-existent, there were several instances when logging-debris retention significantly modified potential N supply. At Matlock in 2006, ammonification was reduced with 40% coverage and initial CVC treatments, and nitrification was increased in 2007 with 80% coverage and initial CVC treatments. Examination of the vector graphs for these same treatments and years shows that the relative foliar response during these times was opposite what would be expected given an increase or decrease in N supply (Figures 4.5, 4.6). A similar result was observed at Molalla in 2006, where lower nitrification in the 40% coverage and initial CVC treatment resulted in a vector indicative of luxury consumption and increased N supply. It appears there is potential for logging debris to modify N supply, but the presence of competing vegetation alters the foliar response of crop trees. In situations were N supply is low, the available N pool may be insufficient to support competing vegetation, allowing crop trees to more actively compete for N. When N supply is high, a greater abundance of competing vegetation may establish near crop trees, reducing the amount of N available for tree uptake. Whatever the process, it is evident that competing vegetation has strong control on N uptake by crop trees.

A significant amount of logging debris is still present at the sites, making it possible that available N or foliar N status could be modified at longer time periods than assessed here. Olsson et al. (2000) found a significant reduction in foliar N concentration following whole-tree harvesting compared to stem-only harvesting at sites in Sweden 8-10 yr after harvest, but not at later sample periods (up to 24 yr). Similarly, Piatek and Allen (1999) found significantly lower N mineralization in whole-tree harvests compared to bole-only harvests 15 yr post-harvest, even though no clear difference was observed two yr post-harvest at the same site (Vitousek and Matson, 1985). Logging-debris retention could have a significant effect on N availability in the future, but the likelihood of such a possibility is probably low given the large amount of N found in the mineral soil at these sites (Table 4.1).

Total soil N response

Increased N mineralization coupled with reduced vegetative uptake following harvesting can result in increased N loss from the soil (Bormann et al., 1974; Vitousek et al., 1979; Strahm et al. 2005), potentially causing a reduction in total soil N if losses are large, inputs are small, or initial site N is relatively low. Several studies have documented a reduction in surface soil N following CVC (Miller et al., 2006; Busse et al. 1996). The potential for reduced soil N would be expected to increase with increasing duration of CVC application, given the longer period of reduced vegetative uptake and reduction in organic matter inputs. The lack of any significant difference between CVC treatments after four years of treatment application indicates the potential for reduced soil N following annual CVC is low at these sites. To the contrary, mean soil total N actually tended to be greater at each site when annual CVC was applied.

Logging-debris retention has been shown to reduce (Carlyle et al., 1998) or increase (Strahm et al., 2005) N loss via leaching, but the effect on total soil N has generally been found to be negligible or non-significant. In a summary of 5-yr findings from 19 LTSP sites across 5 regions in the U.S., Sanchez et al. (2006) found no significant effect of logging-debris removal on total soil N at any of the sites, but there were significant increases in total soil N at most sites following harvesting which were attributed to decomposition of the root system of the previous stand and/or adsorption of dissolved organic matter (DOM) to mineral surfaces. In my study there was a general increase in soil N concentration during the two-yr study period, and the initial measures in 2005 were also higher than pre-harvest levels in 2003 (increases of 1.9 and 2.4 g N kg^{-1} for Matlock and Molalla respectively; Meehan, 2006), which is in accordance with the results from Sanchez et al. (2006). However, the significant positive effect of logging-debris retention on total soil N at Matlock has only been previously observed at sites with very low initial soil N concentration (0.4 g kg^{-1}) (Chen and Xu, 2005).

Spatial variability of soil chemical properties in surface soils is generally high, and minimum detectable change is often larger than observed differences in mean values following experimental manipulations (Homann et al., 2001). Here small experimental units were used with the intent to reduce spatial variability in soil properties, which may have contributed to detection of a statistically significant effect of logging-debris retention on soil N. However, the fact that such an effect was observed at Matlock but not Molalla indicates influence of a site-specific factor. Differences in soil mineralogy and initial site N may explain the difference. Strahm and Harrison (2007) found significant quantities of allophone and imogolite in soil from the Matlock site, which are able to adsorb greater amounts of DOM due to their high surface area (Harsh et al., 2002) and variable charge (Parfitt, 1980). The relatively lower soil C and N at Matlock could also result in more free absorption sites at mineral surfaces than at Molalla. Adsorption of DOM to mineral surfaces may be greater following logging-debris retention, either due to greater affinity of DOM produced from logging debris than that of DOM produced in soil or greater equilibrium exchange due to an increase in DOM quantity when logging debris is retained. Other possible site-specific factors which could have contributed to the increase in soil N in response to increased logging debris at Matlock are the greater precipitation at Matlock which may increase dissolved N inputs from logging debris to the mineral soil, or the more frequent moisture limitation to microbial activity observed at Matlock (Chapter 2) which would cause a general reduction in N mineralization.

The increase in soil N at each site following harvesting and additional increases at Matlock due to logging-debris retention could result in greater N acquisition to crop trees if maintained over the course of the rotation. However, the stability and availability of increased soil N is unclear given the short time of this study. Johnson et al. (2007) documented large changes in soil N over a 30-yr period following harvesting which they were unable to attribute to losses, change in biomass, or sampling error, suggesting that soil N pools may be subject to high temporal variability. The extensive LTSP summary by Sanchez et al. (2006) found that increased soil N following harvesting was maintained for at least 5 yr, and increased with time at some of the sites. The increases observed here may follow a similar pattern, or be inconsequential to tree acquisition later in the rotation when C inputs increase, causing an increase in N immobilization (Hart et al. 1994). Further sampling at these sites will clarify the outcome.

Conclusions

Annual applications of CVC had a strong effect on Douglas-fir foliage N status (concentration and content), soil available N, and SWC at both of the sites in the initial years after harvesting and planting. It is likely that the increase in foliage N status following annual CVC is due to both increased soil water and N availability, but the contribution of each probably varied by year, depending on inter-annual variability in climate. The increase in soil available N was mostly due to decreased vegetative uptake, but a significant increase in net nitrification at one sample period indicated reduced microbial immobilization also contributed to the increase. Net N mineralization was significantly reduced in the annual CVC treatment during one sample period but not during other periods, indicating that a possible reduction in this component of soil quality was ephemeral or non-existent. Increased foliage N status was associated with a large increase in tree volume growth, demonstrating the effectiveness of annual CVC to increase crop tree growth.

The lack of any significant effect of logging-debris retention on foliage N status and available soil N indicates that logging debris has a relatively small influence on crop tree N acquisition in the initial years after planting. Vector analysis indicated improved foliage N status following logging-debris retention in some instances, but the effect was limited and generally small. Increased growth when 80% logging-debris was retained and annual CVC was applied indicated greater N uptake, but this increase appears to be driven by a growth-limiting variable other than N.

The increase in soil N at both sites following harvesting, regardless of treatment, could be due to decomposition of the root system of the previous stand or incorporation of the forest floor into the mineral soil. Significant increases in soil N with increasing logging-debris retention at Matlock but not at Molalla indicated a site-specific response which may be due to differences in soil mineralogy, but could also be due to a number of other variables that differ between the two sites. The importance of these increases to crop tree N acquisition in the future is uncertain, and effort should be made to determine the long-term biological significance of the short-term changes observed here.

Tables

Table 4.1. Site characteristics and select pre-treatment soil properties to a depth of 30 cm for study sites near Matlock, WA and Molalla, OR.

Characteristic or property	Matlock	Molalla
Location (Latitude, Longitude)	47.206 °N, 123.442 °W	45.196 °N, 122.285 °W
Elevation (m)	118	449
Mean annual temperature (°C)	10.7	11.2
Mean annual precipitation (cm)[1]	240	160
Site index$_{50\ yr}$ (m)	35.9	36.2
Soil texture (% sand/silt/clay)	65 / 14 / 21	37 / 34 / 29
Bulk density (Mg m^{-3})[2]	1.45 (0.05)	0.98 (0.02)
Coarse fragments by mass (%)	65.8 (1.3)	32.2 (2.2)
Total soil N (kg N ha^{-1})	2,246 (88)	4,338 (173)
Total soil C (Mg C ha^{-1})	66.5 (3.6)	102.2 (4.7)

[1] Precipitation was calculated from the PRISM model for period 1950-2005 (http://prism.oregonstate.edu).
[2] Standard error in parenthesis, n=8 for bulk density at Matlock, n=24 for all others).

Table 4.2. Collection dates for soil and foliar samples at the Matlock and Molalla study sites.

Sample type	**Collection date**	**Measurements**
Soil	July 29-30, 2005	Available N Total soil N
Soil	Apr. 20-21, 2006	N mineralization
Soil	July 22-23, 2006	Available N N mineralization
Soil	Sept. 28-29, 2006	N mineralization
Soil	July 3-4, 2007	Available N N mineralization
Soil	Oct. 4-6, 2007	Total soil N
Foliage	Oct. 4-5, 2005	N concentration Needle mass
Foliage	Sept. 11-12, 2006	N concentration Needle mass
Foliage	Oct. 29-30, 2007	N concentration Needle mass

Table 4.3. Test statistics for fixed treatment effects on volumetric soil water content by year at the Matlock and Molalla sites.

Effect	2005		2006		2007	
	F stat.	***p* value**	**F stat.**	***p* value**	**F stat.**	***p* value**
			Matlock			
CVC (df = 1,3)[1]	4.23	0.132	2.09	0.244	2.41	0.218
Debris (df= 2,12)	1.10	0.368	0.52	0.608	1.48	0.270
C * D[2] (df= 2, 12)	4.02	**0.049**[3]	3.49	0.064	3.22	0.079
Month (df = 9,141)	253.13	**<0.001**	176.12	**<0.001**	89.22	**<0.001**
C * M (df= 9,141)	38.71	**<0.001**	15.13	**<0.001**	7.02	**<0.001**
D * M (df= 18, 141)	1.19	0.280	0.92	0.575	1.07	0.389
C * D*M (df= 18, 141)	2.24	**0.007**	0.91	0.578	1.49	0.104
			Molalla			
CVC (df = 1,3)	0.58	0.501	2.63	0.204	1.27	0.342
Debris (df= 2,12)	3.14	0.083	5.14	**0.027**	1.32	0.303
C * D (df= 2, 12)	1.30	0.312	0.19	0.828	2.20	0.154
Month (df = 9,141)	51.59	**<0.001**	164.97	**<0.001**	158.48	**<0.001**
C * M (df= 9,141)	7.88	**<0.001**	12.98	**<0.001**	7.74	**<0.001**
D * M (df= 18, 141)	1.17	0.300	0.68	0.859	1.03	0.429
C * D*M (df= 18, 141)	1.16	0.311	0.32	0.998	1.71	**0.044**

[1] degrees of freedom for the critical F statistic in parenthesis.
[2]C=competing vegetation control (CVC) treatment, D=logging-debris treatment, M=month.
[3] Significant treatment effects at α=0.05 are shown in bold

Table 4.4. Mean soil N concentration to a depth of 20 cm by treatment and sample year at the Matlock and Molalla sites.

Sample year		Logging-debris coverage			Competing vegetation control[1]	
		0%	40%	80%	ICVC[1]	ACVC
		g kg^{-1}				
		Matlock				
2005		2.05 a[2]	2.44 ab	2.63 b	2.25	2.50
		(0.14)[3]	(0.14)	(0.14)	(0.11)	(0.11)
2007	ICVC	3.02 a	2.13 b	3.34 a	2.83	2.91
		(0.03)	(0.03)	(0.03)	(0.27)	(0.27)
2007	ACVC	2.48 a	2.93 ab	3.32 b		
		(0.03)	(0.03)	(0.03)		
		Molalla				
2005		3.14	2.98	3.12	2.77	3.41
		(0.37)	(0.37)	(0.37)	(0.46)	(0.46)
		4.02	3.53	4.01	3.54	4.17
2007		(0.42)	(0.42)	(0.42)	(0.52)	(0.52)

[1] ICVC = initial competing vegetation control, ACVC = annual competing vegetation control.
[2] Means within a row followed by different letters are significantly different at α=0.05.
[3]Standard error in parenthesis, n=8 for logging-debris treatments, n=12 for competing vegetation control treatments.

Table 4.5. Test statistics for fixed treatment effects on ammonification, nitrification, and potential net N mineralization at the Matlock and Molalla sites.

Effect	Ammonification		Nitrification		N mineralization	
	F stat.	*p* value	F stat.	*p* value	F stat.	*p* value
			Matlock			
CVC (df = 1,3)[1]	1.75	0.278	0.58	0.503	0.07	0.809
Debris (df= 2,12)	1.02	0.394	1.78	0.219	1.28	0.320
C * D[2] (df= 2, 12)	2.05	0.179	0.72	0.512	1.24	0.330
Period (df = 3,53)	13.38	**<0.001**[3]	15.87	**<0.001**	18.06	**<0.001**
C* P (df= 3,53)	3.28	**0.027**	1.03	0.388	3.44	**0.023**
D * P (df= 6, 53)	1.30	0.273	1.49	0.200	0.48	0.820
C * D *P (df= 6, 53)	3.12	**0.010**	2.51	**0.032**	0.50	0.803
			Molalla			
CVC (df = 1,3)	3.80	0.147	1.41	0.320	0.83	0.430
Debris (df= 2,12)	1.63	0.237	1.97	0.182	3.06	0.085
C * D (df= 2, 12)	0.54	0.595	3.37	0.067	0.65	0.540
Period (df = 3,53)	18.80	**<0.001**	9.55	**<0.001**	3.79	**0.016**
C * P (df= 3,53)	1.85	0.150	2.91	**0.043**	4.90	**0.004**
D * P (df= 6, 53)	0.73	0.628	2.33	**0.045**	0.86	0.531
C * D *P (df= 6, 53)	1.25	0.298	2.09	0.069	1.56	0.178

[1] degrees of freedom for the critical F statistic in parenthesis.
[2]C=competing vegetation control (CVC), D=debris, P=period.
[3] Significant treatment effects at α=0.05 are shown in bold.

Table 4.6. Test statistics for fixed treatment effects on available ammonium, available nitrate, and total available N at the Matlock and Molalla sites.

Effect	**Available ammonium**		**Available nitrate**		**Total available N**	
	F stat.	***p* value**	**F stat.**	***p* value**	**F stat.**	***p* value**
			Matlock			
CVC (df = 1,3)[1]	3.36	**0.012**	49.55	**0.006**	38.28	**0.008**[3]
Debris (df= 2,12)	1.81	0.205	3.33	0.071	1.50	0.261
C * D[2] (df= 2, 12)	0.14	0.872	2.18	0.156	1.41	0.283
Year (df = 2, 36)	69.52	**<0.001**	9.56	**0.001**	38.91	**<0.001**
C * Y (df= 2, 36)	12.60	**<0.001**	9.06	**0.001**	19.12	**<0.001**
D * Y (df= 4, 36)	1.86	0.139	0.51	0.729	0.99	0.425
C * D *Y (df= 4, 36)	0.75	0.563	0.65	0.628	0.26	0.904
			Molalla			
CVC (df = 1,3)	34.77	**0.010**	129.54	**0.002**	129.39	**0.001**
Debris (df= 2,12)	1.51	0.260	0.28	0.758	0.97	0.408
C * D (df= 2, 12)	0.95	0.413	1.18	0.342	1.66	0.230
Year (df = 2, 36)	160.19	**<0.001**	9.39	**0.001**	10.77	**<0.001**
C * Y (df= 2, 36)	17.23	**<0.001**	1.88	0.168	5.87	**0.006**
D * Y (df= 4, 36)	1.66	0.182	1.25	0.308	2.13	0.098
C * D *Y (df= 4, 36)	1.94	0.126	0.79	0.542	1.22	0.321

[1] degrees of freedom for the critical F statistic in parenthesis
[2]C=competing vegetation control (CVC), D=debris, P=period.
[3] Significant treatment effects at α=0.05 are shown in bold.

Table 4.7. Test statistics for fixed treatment effects on the dependent variables needle N concentration, needle N content, and mass of 100 needles at the Matlock and Molalla sites.

Effect	Needle N concentration		Needle N content		Needle mass	
	F statistic	***p* value**	**F statistic**	***p* value**	**F statistic**	***p* value**
			Matlock			
Year (df = 2, 34)[1]	12.66	**<0.001**[3]	3.00	0.063	1.59	0.218
CVC (df = 1, 3)	48.66	**0.006**	20.21	**0.021**	1.48	0.311
Y*C[2] (df = 2, 34)	16.33	**<0.001**	13.88	**<0.001**	3.26	0.051
Debris (df = 2, 12)	1.16	0.346	0.54	0.595	0.03	0.967
Y * D (df = 4, 34)	0.48	0.747	0.75	0.568	0.64	0.634
C * D (df = 2, 12)	1.94	0.186	1.32	0.304	2.41	0.132
Y * C * D (df = 4, 34)	1.12	0.363	0.51	0.725	0.25	0.906
			Molalla			
Year (df = 2, 36)	54.56	**<0.001**	27.27	**<0.001**	10.88	**0.001**
CVC (df = 1, 3)	39.82	0.008	20.20	**0.021**	0.00	0.975
Y*C (df = 2, 36)	3.65	**0.036**	1.76	0.186	6.27	**0.005**
Debris (df = 2, 12)	0.48	0.629	0.64	0.546	0.38	0.692
Y * D (df = 4, 36)	0.73	0.578	1.78	0.154	2.32	0.076
C * D (df = 2, 12)	1.44	0.275	1.09	0.367	0.72	0.507
Y * C * D (df = 4, 36)	1.76	0.158	2.13	0.097	0.81	0.527

[1] degrees of freedom for the critical F statistic in parenthesis
[2] Y=year, C=competing vegetation control(CVC), D=debris.
[3] Significant treatment effects at α=0.05 are shown in bold.

Table. 4.8. Effect of competing vegetation control on mean needle N concentration, needle N content, and needle mass by year at **Matlock**.

Foliar property	**Logging-debris coverage**			**Competing vegetation control**[1]	
	0%	**40%**	**80%**	**ICVC**	**ACVC**
2005					
N concentration (mg g^{-1})	13.5 (0.4)[3]	14.2 (0.4)	14.1 (0.4)	**11.2**[2] **(0.3)**	**16.7 (0.3)**
N content (mg)	6.9 (0.5)	7.0 (0.5)	7.2 (0.5)	**4.9 (0.5)**	**9.2 (0.5)**
Needle mass[4] (g)	0.50 (0.03)	0.48 (0.03)	0.50 (0.03)	**0.44 (0.03)**	**0.55 (0.03)**
2006					
N concentration (mg g^{-1})	13.3 (0.8)	14.2 (0.8)	13.7 (0.7)	13.5 (0.7)	13.9 (0.6)
N content (mg)	5.5 (0.9)	6.4 (0.9)	6.6 (0.8)	6.4 (0.8)	5.9 (0.7)
Needle mass (g)	0.40 (0.06)	0.44 (0.06)	0.48 (0.06)	0.45 (0.05)	0.43 (0.05)
2007					
N concentration (mg g^{-1})	12.0 (0.60)	12.8 (0.6)	11.5 (0.6)	**10.4 (0.5)**	**13.8 (0.5)**
N content (mg)	6.2 (0.8)	6.8 (0.8)	5.5 (0.8)	**5.3 (0.7)**	**7.1 (0.7)**
Needle mass (g)	0.51 (0.05)	0.52 (0.05)	0.47 (0.04)	0.49 (0.04)	0.51 (0.04)

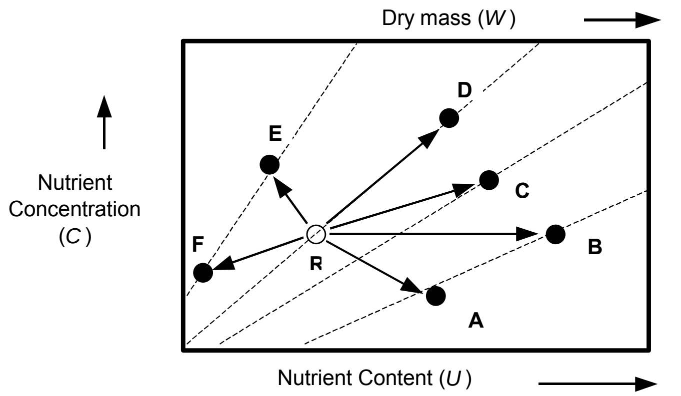
[1] ICVC = initial competing vegetation control, ACVC = annual competing vegetation control.
[2] Means within a year, row, and treatment category in bold are significantly different at $\alpha = 0.05$.
[3] SE in parenthesis, n=12 for CVC, n=8 for logging debris.
[4] Needle mass based on 100 needles.

Table. 4.9. Effect of competing vegetation control on mean needle N concentration, needle N content, and needle mass by year at **Molalla**.

Foliar property	**Logging-debris coverage**			**Competing vegetation control**[1]	
	0%	**40%**	**80%**	**ICVC**	**ACVC**
2005					
N concentration	13.2	13.5	13.7	**12.0**[2]	**14.9**
(mg g^{-1})	(0.8)[3]	(0.8)	(0.8)	**(0.6)**	**(0.6)**
N content	6.4	7.8	7.5	**6.0**	**8.5**
(mg)	(0.7)	(0.7)	(0.7)	**(0.6)**	**(0.6)**
Needle mass[4]	0.49	0.58	0.53	0.50	0.57
(g)	(0.05)	(0.05)	(0.05)	(0.04)	(0.04)
2006					
N concentration	11.4	12.5	13.0	**9.8**	**14.8**
(mg g^{-1})	(0.6)	(0.6)	(0.6)	**(0.5)**	**(0.5)**
N content	4.7	5.5	5.4	**4.7**	**5.7**
(mg)	(0.3)	(0.3)	(0.3)	**(0.2)**	**(0.2)**
Needle mass	0.43	0.46	0.42	**0.48**	**0.39**
(g)	(0.04)	(0.04)	(0.04)	**(0.04)**	**(0.04)**
2007					
N concentration	16.8	16.4	16.8	**14.9**	**18.4**
(mg g^{-1})	(0.7)	(0.7)	(0.7)	**(0.6)**	**(0.6)**
N content	7.5	6.6	7.1	**6.2**	**7.9**
(mg)	(0.5)	(0.5)	(0.5)	**(0.4)**	**(0.4)**
Needle mass	0.45	0.40	0.42	0.42	0.43
(g)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)

[1] ICVC = initial competing vegetation control, ACVC = annual competing vegetation control.
[2] Means within a year, row, and treatment category in bold are significantly different at $\alpha = 0.05$.
[3] SE in parenthesis, n=12 for CVC, n=8 for logging debris.
[4] Needle mass based on 100 needles.

Figures



Vector shift	Change in relative *W*	*U*	*C*	Nutritional effect	Nutrient status	Possible diagnosis
A	+	+	-	Dilution	Non-limiting	Growth dilution
B	+	+	o	Accumulation	Non-limiting	Sufficiency, steady-state
C	+	+	+	Accumulation	Limiting	Deficiency response
D	o	+	+	Accumulation	Non-limiting	Luxury consumption
E	-	-, +	+	Concentration	Excess	Toxic accumulation
F	-	-	-	Antagonism	Limiting	Induced deficiency by E

Figure 4.1. Example vector diagram for foliar analysis with potential vectors and possible diagnosis associated with each (adapted from Timmer and Morrow, 1984).

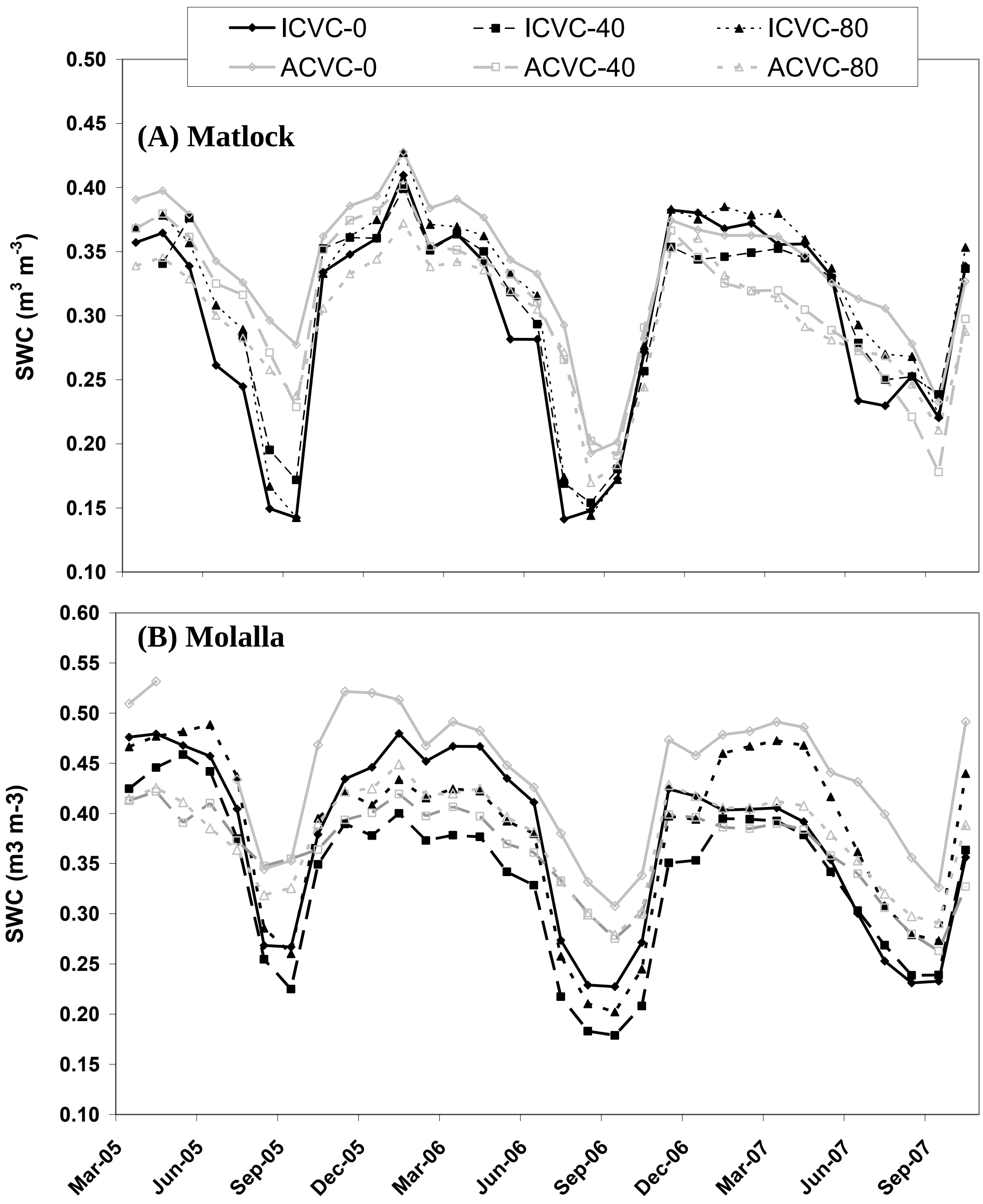


Figure 4.2 Volumetric soil water content (SWC) as influenced by logging-debris and competing vegetation control treatments at (A) Matlock and (B) Molalla. ACVC = annual competing vegetation control, ICVC= initial competing vegetation control; -0,-40,-80 = 0, 40, and 80% logging-debris coverage, respectively.

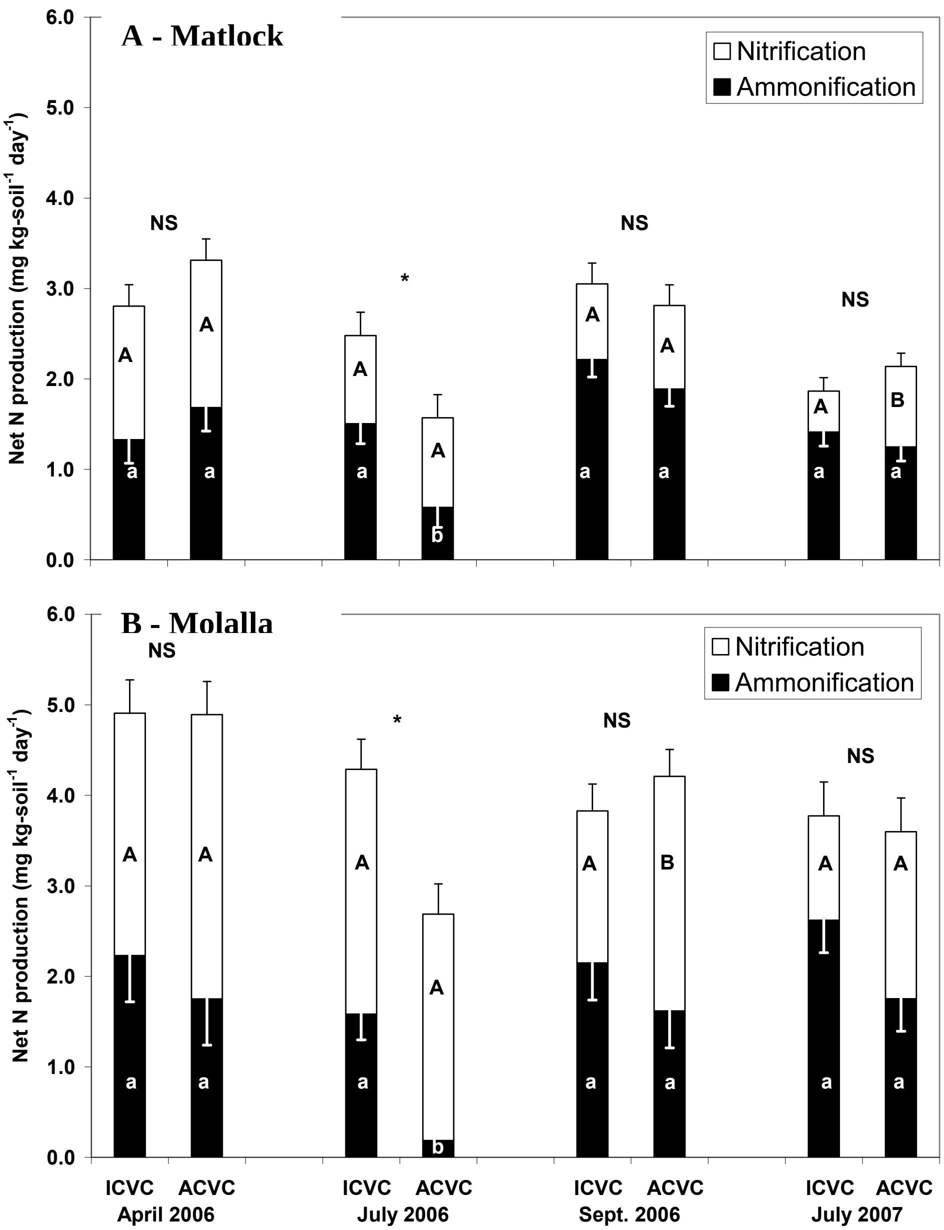


Figure 4.3. Effect of competing vegetation control treatments on net ammonification, nitrification, and total N mineralization by incubation period at (A) Matlock and (B) Molalla. Means within a date with different letters are significantly different at α=0.05. Lowercase letters are for ammonification, uppercase letters are for nitrification. * and NS indicate a significant or non-significant difference, respectively, between total N mineralization for each date. ACVC = annual competing vegetation control, ICVC = initial competing vegetation control. Error bars are one SE of the mean.

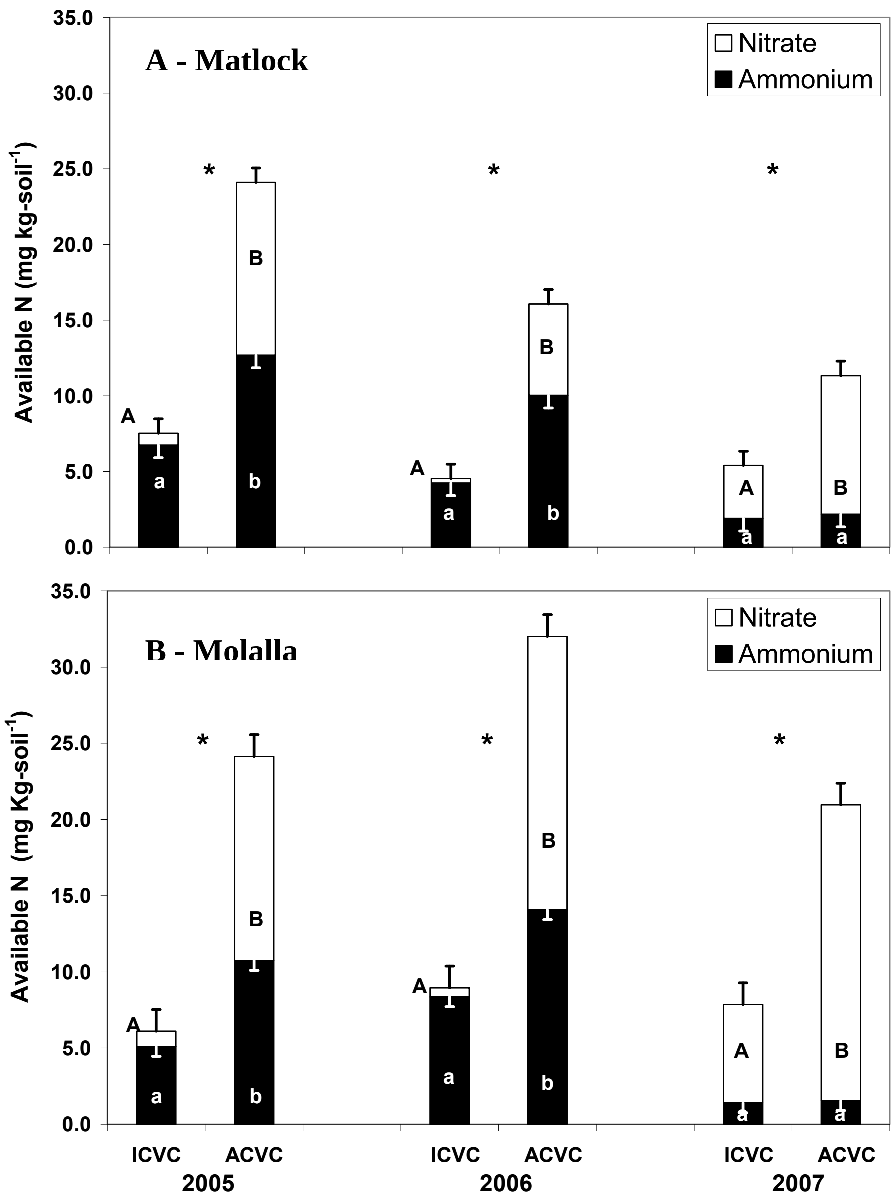
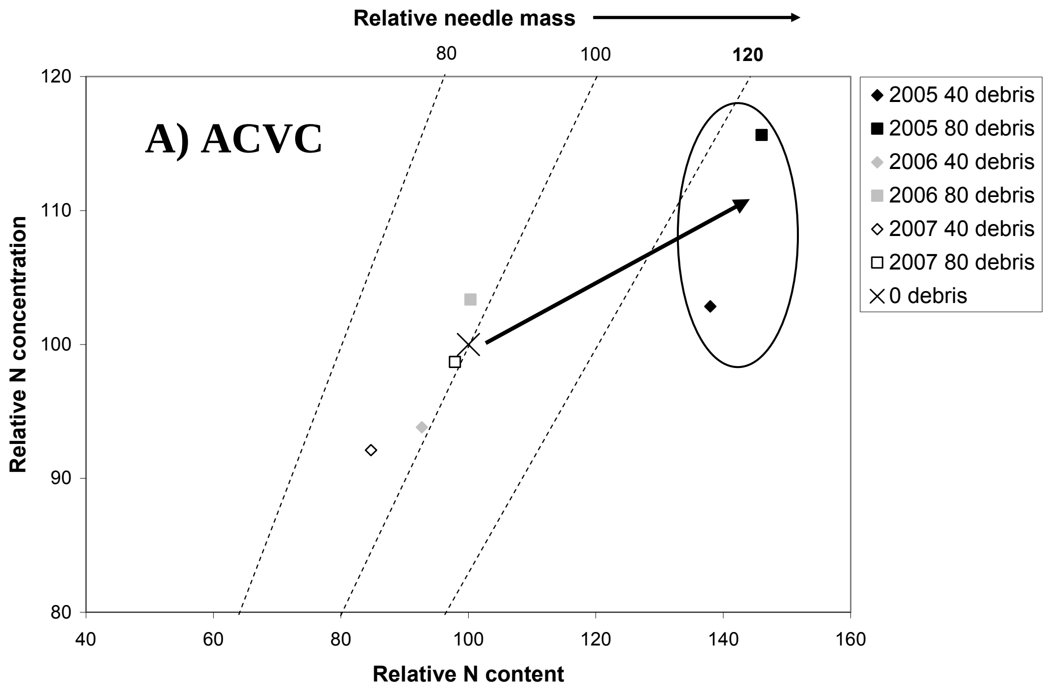


Figure 4.4. Effects of competing vegetation control treatments on soil available N concentrations at time of sample collection in July of each year at (A) Matlock and (B) Molalla. Means within a year with different letters are significantly different at α=0.05. Lowercase letters are for ammonium, uppercase letters are for nitrate. * and NS indicate a significant or non-significant difference, respectively, in total inorganic N for each year. ACVC = annual competing vegetation control, ICVC = initial competing vegetation control. Error bars are one SE of the mean. Note difference in scale between panels.



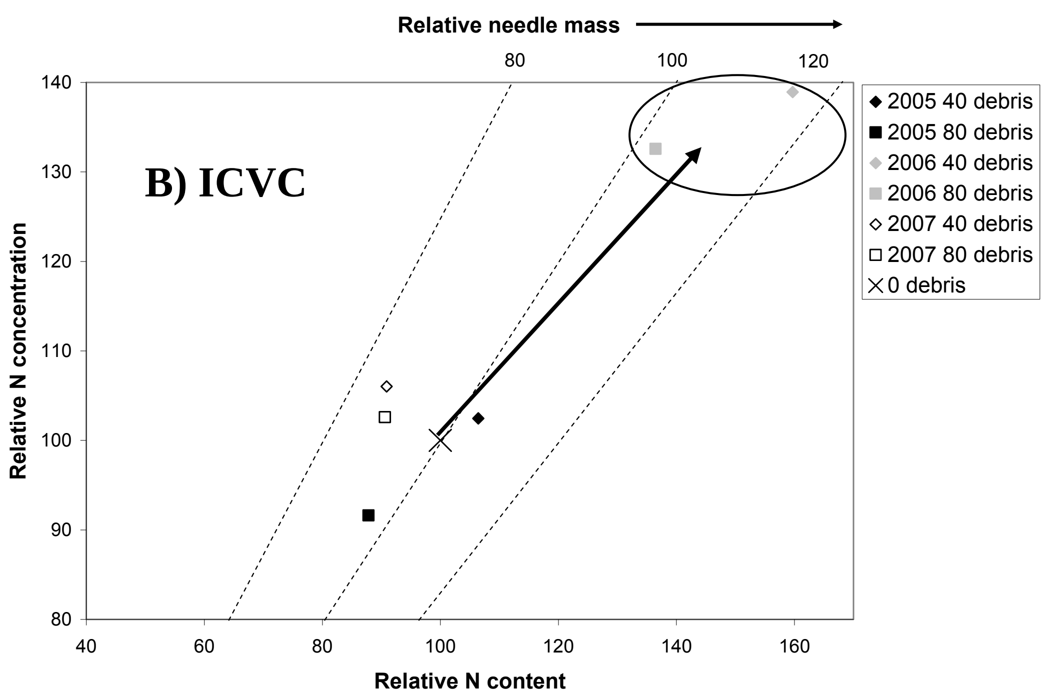


Figure 4.5. Vector graphs by logging-debris treatment and year at **Molalla** with (A) annual competing vegetation control (ACVC), and (B) initial competing vegetation control (ICVC). All values are relative to corresponding mean values of the 0% coverage treatment for that year and normalized to a value of 100.

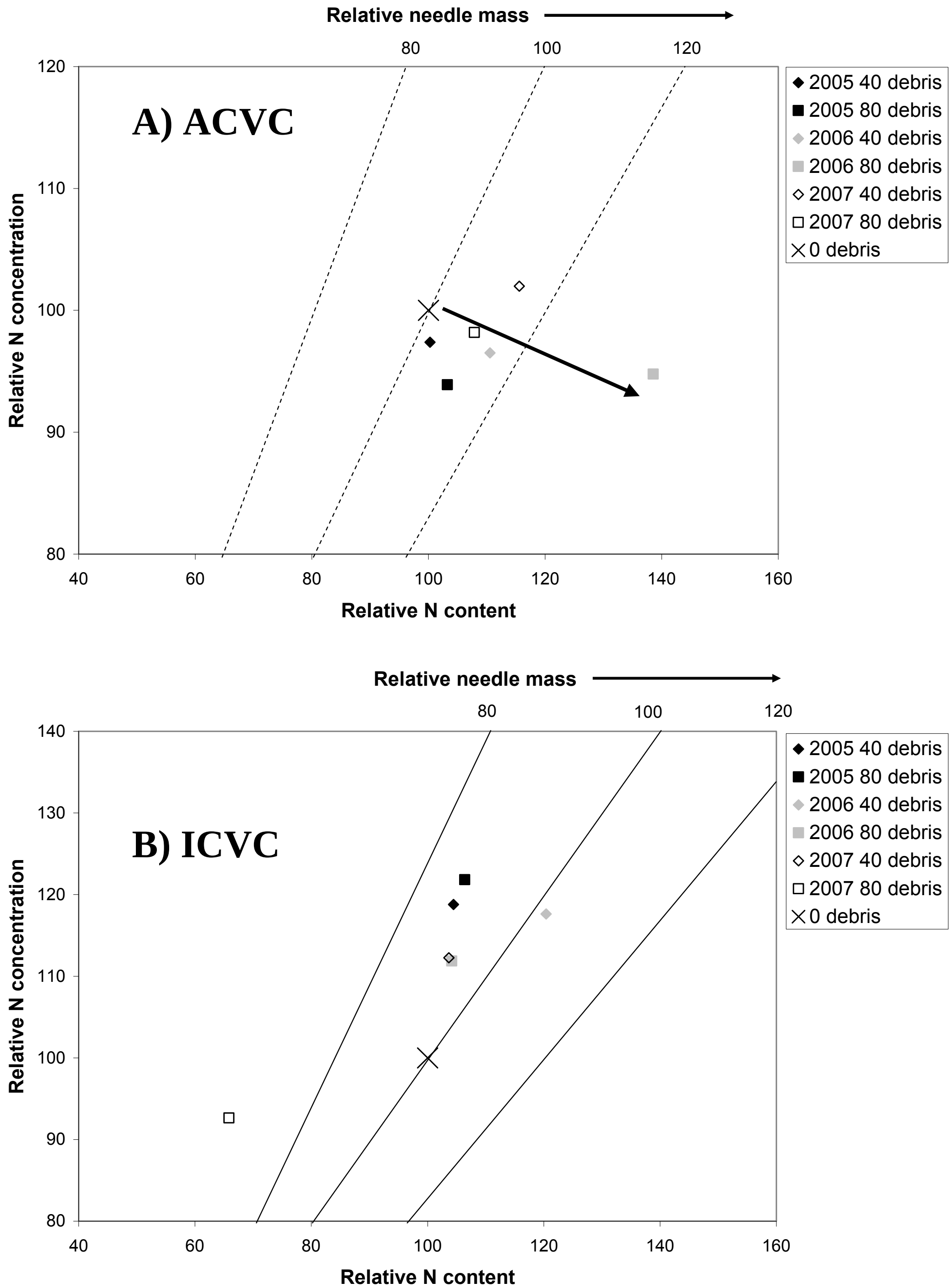


Figure 4.6. Vector graphs by logging-debris treatment and year at **Matlock** with (A) annual competing vegetation control (ACVC) and (B) initial competing vegetation (ICVC). All values are relative to corresponding mean values of the 0% coverage treatment for that year and normalized to a value of 100.

Chapter 5

Conclusions

Effects of Competing Vegetation Control and Logging-Debris Treatments on Carbon

Logging-debris retention reduced MR but had no effect on DOC leaching in the initial years after harvesting at these sites. Much of the effect on MR was due to reductions in soil temperature when logging debris was retained, but several lines of evidence suggested the response was more complicated than being controlled by temperature alone, likely involving modification of available substrate and other factors in the soil environment. In the case of moderate logging-debris retention (i.e. 40% coverage), these factors interacted such that no reduction in MR occurred despite lower soil temperatures. The lower total C flux (MR and DOC) when heavy amounts of logging debris (i.e. 80% coverage) were retained indicates that soil C was increased, which was observed at Matlock, but not Molalla. The absence of any effect at Molalla is likely due to the greater C pool at that site, where any C inputs would have been small relative to the pool size and probably undetectable. Belowground decomposition was a large contributor to increased C regardless of logging-debris retention, and differences between sites in the factors that control belowground decomposition may have also contributed to no observable effect at Molalla when logging debris was retained. Logging debris appears to have a beneficial effect on soil C in the initial years following harvesting, but only when heavy amounts of logging debris are retained, and the increase is only detectable under certain site conditions. The effect may not be apparent in operational settings, as spatial variability of logging-debris coverage is large following harvest. Taken together, these results explain why no observable effect on soil C is commonly found after experimental manipulation of logging-debris retention.

Given that soil C tended to increase following harvest regardless of logging-debris treatment, it is reasonable to conclude that logging-debris removal will have no detrimental effect on soil C pools and long-term productivity. However, greater MR, higher annual DOC flux, and reduced OM inputs when logging debris was removed indicated that the potential for net soil C loss is higher when logging debris is removed compared to logging-debris retention. Increased belowground decomposition of the root system of the previous stand almost certainly contributed to the increase in C loss when logging debris is absent, and it is unclear what such an effect will have on soil C over the long-term. It may be that greater initial belowground decomposition results in a reduced total belowground C pool, or belowground decomposition may be reduced relative to when logging debris is retained in later years, resulting in no difference between treatments over the course of a rotation. Further study is warranted to determine the long-term effect of logging-debris manipulation on mineral soil C and belowground OM pools.

No difference in soil C between CVC treatments indicated little potential for a reduction in soil C when annual CVC is applied, even though OM inputs were likely reduced. Absence of any difference between CVC treatments in potential MR determined under controlled laboratory conditions and presence of much greater BR in the initial CVC treatment indicated that OM inputs from competing vegetation were rapidly consumed *in situ*. The magnitude and duration of greater BR in the initial CVC treatment was dependent on inter-annual variability in climate and its effect on soil water content, with much greater BR when soil water was not limiting to microbial and root respiration. Given the influence of soil moisture at these sites on MR and BR and the greater SWC following annual CVC, it is possible that belowground decomposition of roots from the previous stand was higher in the annual CVC treatment, potentially offsetting any reductions in soil C associated with reduced OM inputs. However, differences in soil temperature between CVC treatments were low or absent, and there was no indication for greater DOC loss in the annual CVC treatment as was observed when logging debris was removed, suggesting that belowground decomposition was not different between treatments. Although annual CVC has potential to reduce soil C through a reduction in litter inputs, the results suggest a concurrent reduction in MR when litter inputs are reduced, and a net result of no change in soil C beyond that which occurs following harvest. Competing vegetation control appears to have little potential for C-related reductions in soil productivity at these sites.

Effects of Competing Vegetation Control and Logging-Debris Treatments on Nitrogen

Logging-debris retention had a limited influence on soil N availability and supply, foliar N nutrition of Douglas-fir, and N leaching below the rooting zone at both sites in the initial years after harvest. Soil N was higher following logging-debris retention in combination with annual CVC at Matlock, and this higher soil N was likely responsible for increased soil water nitrate-N concentrations in the annual CVC treatment in 2006. Increased soil water nitrate-N indicates that tree N demand is lower than N supply, possibly indicating improved N availability to crop trees when logging debris was retained and annual CVC was applied at Matlock. However, no effect on needle N content or concentration at Matlock indicated any improvement in N availability was of little consequence to tree nutrition.

Differing effect of logging-debris retention on soil N between sites may be due to differences in soil mineralogy, or more favorable environmental conditions for belowground decomposition at Matlock when logging debris was retained. If the greater soil N observed at Matlock following logging-debris retention and annual CVC is maintained into the future, then the increase may have an important positive influence on stand productivity given the relatively low soil N present at this site.

In contrast, annual applications of CVC had a marked influence on available N, Douglas-fir foliar N nutrition, and N leaching below the rooting zone at both sites. Much of the effect was due to decreased vegetative competition for N, but increased N supply associated with reduced microbial immobilization likely also contributed, particularly at Molalla where the soil N pool is much greater relative to Matlock. Effect of annual CVC was beneficial for crop tree N acquisition and growth at both sites, but detrimental to site N pools at Molalla as N loss below the rooting zone was increased when annual CVC was implemented. However, the estimated N flux was small relative to the total soil N pool, and there was no difference between CVC treatments in total soil N at the end of the experiment at either site. Annual CVC is an effective means to increase crop tree N nutrition and growth, with low potential for N-related reductions in soil productivity at these sites.

Implications for forest management

Logging-debris retention reduced net soil C flux, indicating a beneficial effect on soil C pools, but no detrimental effect of logging-debris removal on soil C and N pools was found. Retention of logging debris may improve soil quality and long-term soil productivity, particularly at sites with relatively low initial pools of C and N, but there appears to be little potential for a reduction in soil quality associated with reduced C and N if logging debris is removed. If improvement of soil quality is a management goal, results of my research suggest that logging debris be retained following harvesting. In terms of soil C storage (e.g. C sequestration), reduced soil C flux when logging debris is retained may result in greater total belowground soil C, but the net effect on atmospheric C flux is unclear as any increase in the soil will likely be accompanied by additional C flux from logging debris to the atmosphere.

Annual and initial applications of CVC appear to have little potential to negatively affect soil quality through reduction in soil C and N at these sites, even with increased N loss via leaching when annual CVC was applied. Although annual CVC has low potential to reduce soil quality in the initial years after planting, N loss from the soil may negatively affect nearby aquatic systems if this N is transported to streams and lakes. The potential for N transport to aquatic systems will be dependent on site-specific factors (e.g. proximity to streams, soil characteristics, annual precipitation), and mangers should decide on a site-specific basis if annual CVC is appropriate for a given set of site characteristics.

Annual CVC was an effective means to increase foliar N status and growth of young Douglas-fir, with the effect most pronounced at the Matlock site which had relatively low resource availability (i.e. available water, nitrogen). Given the low potential for reductions in soil quality when this practice was employed, it appears that annual CVC is an effective means to increase crop tree growth and maintain soil quality at these sites.

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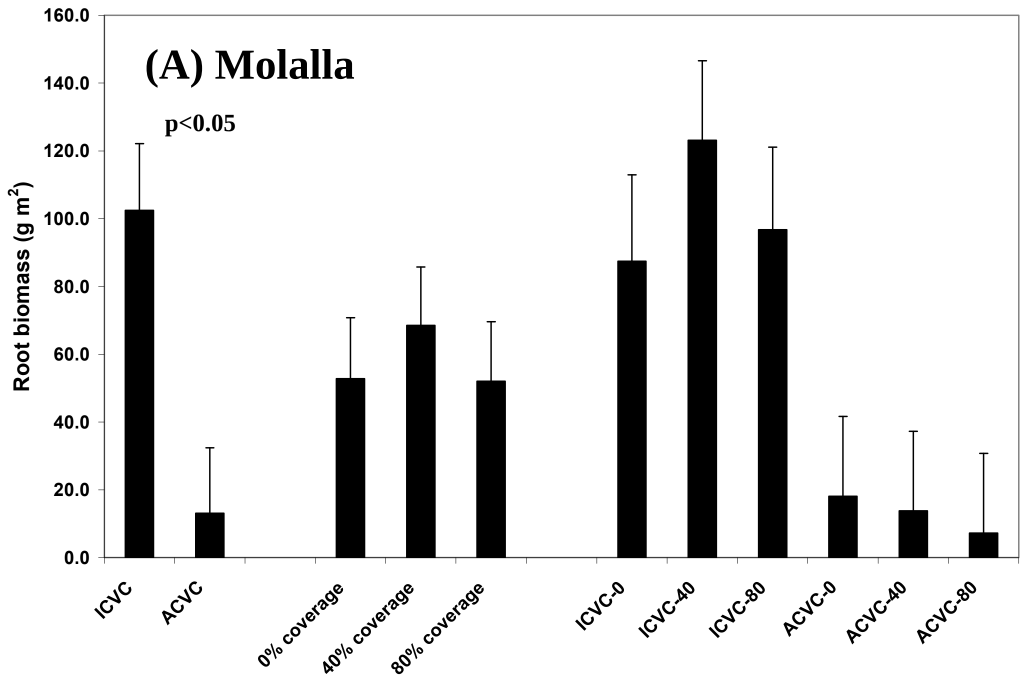
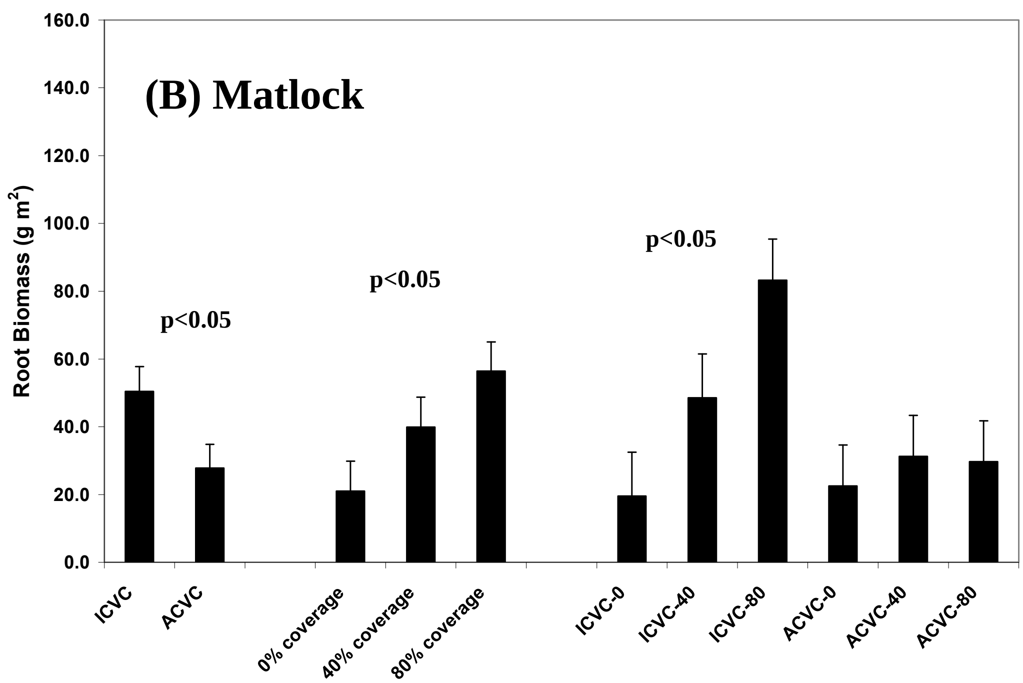
Appendices

Appendix A. Assessment of root in-growth to respiration collars

Root in-growth to collars used for MR estimation was assessed at the end of the study period. At each replication, the 30 cm deep core was removed from the soil in October 2007 and placed intact into a plastic bag for transport. Cores were transported to Oregon State University and stored at 4 °C until analysis (approximately 2 months). For analysis, soil was gently pushed from each core and live roots were separated from soil by hand if present. All roots were less than 2 mm in diameter (i.e. fine roots). Roots were washed in water to remove any attached soil, dried at 105 °C until constant mass was attained, and then weighed to determine oven-dry mass.

At both sites, most cores had roots present at the end of the two-year study (42 of 47 at MA, 39 of 47 at MO). Root in-growth was significantly greater in the initial CVC treatment at both sites (Matlock $F_{1,9} = 5.21$, p=0.048; Molalla $F_{1,9} = 18.36$, p=0.002)(see figure below). At Matlock, there was a significant main effect of logging-debris retention on root in-growth ($F_{2,22} = 4.26$, p=0.027), but the effect was largely limited to the initial CVC treatment (see figure below). There was no significant effect of logging-debris on root in-growth at Molalla ($F_{2,22} = 0.53$, p=0.598). Root in-growth estimates for the annual CVC treated plots were generally low compared to estimates of fine root biomass in Douglas-fir forests of the PNW (generally <10%) (see table below). There was no correlation between fine root in-growth and mean in situ MR by herbicide treatment during the summer in 2007 at either Matlock (ICVC r =-0.09, p=0.70; ACVC r = 0.33, p=0.12) or Molalla (ICVC r = 0.23, p=0.31; ACVC r = -0.04 p=0.86).





Appendix A Figure A1. Root biomass in 30 cm deep root exclusion collars at the end of the two-yr study by treatment at (A) Molalla, and (B) Matlock. ICVC – initial competing vegetation control, ACVC = annual competing vegetation control. 0%, 40%, and 80%

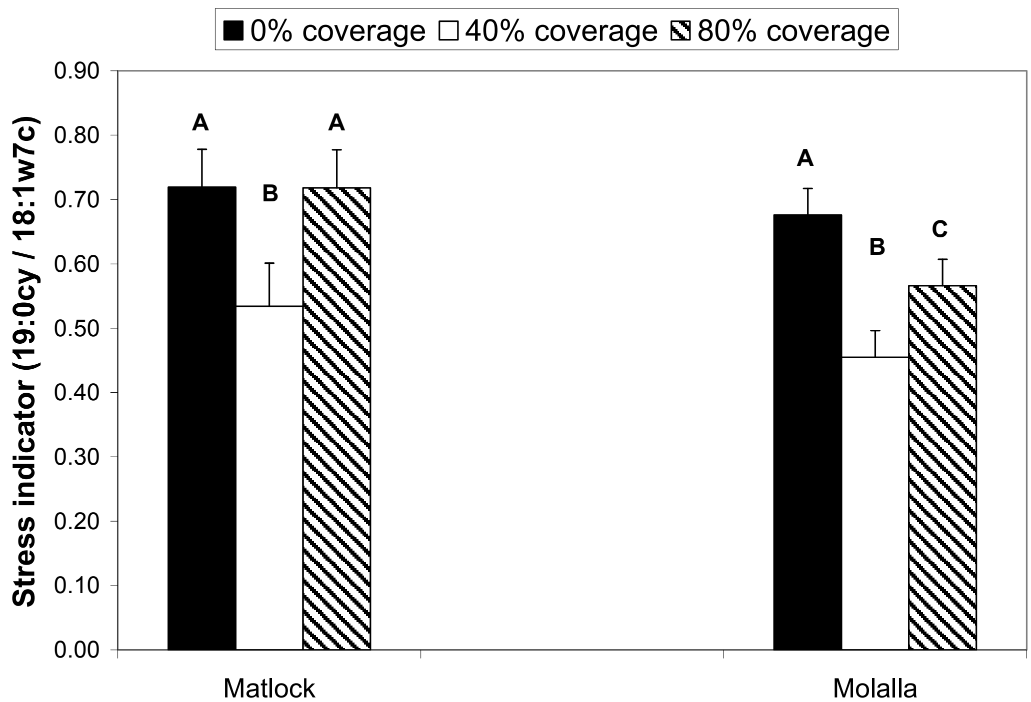
coverage is the respective logging-debris coverage treatment. Error bars are standard error of the mean).

Appendix Table A1. Estimates of fine root biomass in Douglas-fir stands from several studies in the Pacific Northwest.

Location	**Stand age (yr)**	**Depth of sampling (cm)**	**Fine root biomass (g m^{-2})**	**Citation**
Olympic Mountains, WA	> 290	20	600	Lee et al. 2004
West Cascades, OR	13-20	100	600[1]	Campbell et al. 2004
Coast Range, OR	12-14	100	400[1]	Campbell et al. 2004
West Cascades,WA	22	20	700[2]	Klopatek 2007
High productivity stands, West Cascades, WA	13-14	15	100-280	Vogt et al. 1987
Low productivity stands, West Cascades, WA	11-12	15	580-730	Vogt et al. 1987

[1] Assuming an average root C concentration of 500 g kg^{-1}.
[2] Root in-growth bags after three years.

Appendix B. Microbial stress indicator



Appendix B. Microbial stress indicator measured in the July 2007 incubation by logging-debris treatment and site. Means within a site with different letters are significantly different at α=0.05. Error bars are standard error of the mean, n=8 per treatment.

Appendix C. Mean soil pH by treatment

Appendix C. Mean soil pH by treatment and site measured in the second year of the study.[1]

Site	Logging-debris coverage			Competing vegetation control[1]	
	0% debris	40% debris	80% debris	ICVC[2]	ACVC[3]
Matlock	5.36a (0.09)	5.57b (0.10)	5.45ab (0.09)	5.60a (0.08)	5.32a (0.08)
Molalla	5.75a (0.06)	5.83a (0.06)	5.69a (0.06)	5.83a (0.08)	5.68a (0.08)

[1]Means within a treatment grouping and site with different letters are significantly different at $\alpha=0.05$; standard error in parenthesis; n=8 for logging-debris treatments, n=12 for herbicide treatments.
[2]ICVC=initial competing vegetation control.
[3]ACVC=annual competing vegetation control.