SOIL CARBON ANALYSIS

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C	carbon
CHN	carbon, hydrogen, nitrogen
DBH	diameter at breast height
DLL	dynamic link libraries
N	nitrogen
PNW	Pacific Northwest
SMC	Stand Management Cooperative
SOH	stem only harvesting
SR	stability ratio
TAB	total aboveground biomass
WTH	whole tree harvesting

EXECUTIVE SUMMARY

Our work focused on below-surface processes, which aren't often considered as they are hard to see and typically aren't the source of commodity (i.e. biofuel feedstock) in most operations for acquiring forest products. However, the soil represents the non or slowly renewable resource upon which all forests depend for many of their essential resources, including nutrients, water, support and aeration. Most of the total ecosystem carbon is located underground, and soil biodiversity greatly exceeds aboveground biodiversity, probably in all forests. The results of our stump decomposition, deep soil and nutrient sustainability studies didn't reverse what we already knew about Pacific Northwest forests. However, we did conclude that PNW forests, particularly moist coastal coniferous forests, are highly productive due partly to high belowground resource stocks and availability. We further concluded that, based on the data and studies we have, that they are likely to be resilient to additional biomass harvest removals that would provide feedstock for a biofuels and biochemical industry above current harvest levels. We do caution that this is a general conclusion, that PNW forests are highly diverse, and that resilience to additional organic matter removal is site dependent.

Tree stumps are a significant portion of the woody debris in Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) plantation forests in western Washington State. Very few studies have attempted to quantify the amount of carbon that is held in stumps or the rate at which that carbon is lost. This study assessed carbon and nitrogen concentration and stump density in a chronosequence of thinned or harvested Douglas-fir plantation forests. Using a negative log transformation of density versus age, a model for stump decay was determined. Stumps were found to have a decay rate (k) of 0.019 year^{-1}, starting at an average density of 0.35 g cm^{-3} at one year of age and declining to a density of 0.26 g cm^{-3} over 15 years. The carbon concentration of stumps decreased steadily over time, while the nitrogen content remained constant. Stumps contain up to 33% of total tree-based carbon and 22% of total ecosystem carbon in Douglas-fir plantation forests.

Soil is the primary sink for C in forest ecosystems, but is often overlooked in ecosystem C budgets. Efforts to quantify C pools often sample soils to a depth of 0.2 m despite observations that deep soil C is neither scarce nor entirely stable. This study examined the systematic sampling depth for ecosystem C analyses in the Pacific Northwest, and compared best-fit models of C in deep soil layers with laboratory measurements. Forest floor samples and mineral soil bulk density samples were collected at regular intervals from the soil surface to depths of 2.5 m from 22 sites across the coastal Pacific Northwest Douglas-fir zone. Soil samples were screened to 4.7 mm and analyzed for C content. We found that systematic soil sampling shallower than 1.5 m significantly underestimated total soil C. On average, sampling to 2.5 m compared to 0.5 m increased total C by 156% (85.3 Mg ha^{-1} to 132.7 Mg ha^{-1}), and 21% of total C within the depth range sampled was below 1.0 m. A nonlinear mixed model using an inverse polynomial curve form and predicting total C to 2.5 m given only data to 1.0 m was reliable for 20 of 22 sites; the sites that could not be accurately modeled carried the greatest C at depth and contained noncrystalline minerals. Shallow soil sampling at best provides a biased estimate and at worst leads to misleading conclusions regarding soil C. Researchers seeking to quantify soil C or measure change over time should sample deep soil to create a more complete picture of soil pools and fluxes.

Nitrogen is the primary limiting nutrient in PNW coniferous forests, and the additional removal of nitrogen through additional biomass harvest for biofuels has been cited as likely the primary limiting factor in sustainability productivity from these forests. We studied and modeled the potential for sustainable productivity in 68 intensively managed, mid-rotation, Douglas-fir stands in western Oregon, Washington, and British Columbia. The growth of the stands, all 15-25 years old, was projected to 50-55 years of age using the SMC variant of the ORGANON growth and yield simulator. From the ORGANON output, component biomass removal was estimated for stem-only harvest and a more intense whole-tree harvest. Utilizing published equations, which estimate tree component N content based on biomass and total site nitrogen from the 68 sites, nitrogen removal under the two harvest intensities is expressed as a proportion of total site nitrogen store. Based on the proportion of N removed to the total site store, the 68 sites were assigned a risk rating, and regional patterns were assessed. Half the stands appeared to have little risk based on these criteria, and half should be considered additionally. The results show that the potential for sustained productivity or need for additional nutrient management is site dependent.

INTRODUCTION

Our work focused on below-surface processes, which aren't often considered as they are hard to see and typically aren't the source of commodity (i.e. biofuel feedstock) in most operations for acquiring forest products. However, the soil represents the non or slowly renewable resource upon which all forests depend for many of their essential resources, including nutrients, water, support and aeration. Most of the total ecosystem carbon is located underground, and soil biodiversity greatly exceeds aboveground biodiversity, probably in all forests. The results of our stump decomposition, deep soil and nutrient sustainability studies didn't reverse what we already knew about Pacific Northwest forests. However, we did conclude that PNW forests, particularly moist coastal coniferous forests, are highly productive due partly to high belowground resource stocks and availability. We further concluded that, based on the data and studies we have, that they are likely to be resilient to additional biomass harvest removals that would provide feedstock for a biofuels and biochemical industry above current harvest levels. We do caution that this is a general conclusion, that PNW forests are highly diverse, and that resilience to additional organic matter removal is site dependent.

TASK 1. SOIL CARBON ANALYSIS: CARBON AND DECOMPOSITION OF DOUGLAS FIR STUMPS FOR LCA CARBON MODEL

Objective

Forests are a prominent component of the global carbon cycle, containing approximately 80% of the world's aboveground carbon stores (Sedjo, 1992). In temperate forests the potential amount of carbon sequestered is more than half a gigaton of carbon annually. In the United States, 0.15 gigatons of carbon is sequestered in the 145 million hectares of forested lands (USDA, 2001). More than 20 billion tons of this carbon can be found in the Pacific coast region of the United States, with around 200 thousand pounds acre $^{-1}$ carbon storage in Douglas-fir forests (Birdsey, 1992). According to the Washington State Department of Natural Resources (WSDNR) there are 3.8 million hectares of forested lands in western Washington. Of this, over 1.5 million hectares is controlled by private industry (WSDRN, 2007). Forested lands, including Douglas-fir plantations and other private forest, in western Washington State are an important contributor to carbon sequestration in the United States.

Forest soil carbon is essential for soil microbial activity and nutrient storage. Decomposition of woody material is one of the major routes by which organic matter enters soil (Lukac and Godbold, 2011). Decomposition rates are affected by physical, chemical and biological processes. Means et al. (1985) reported that microclimate can have an effect on decomposition. Moisture and temperature specifically affect the rates of decay but other factors such as altitude, slope, aspect and soil type may also play a role. Laiho and Prescot (2004) suggest that climate may not be as much a factor, however, since tree species prefer areas that have specific climates. Decomposing woody material partially defines a soil's structure, nutrient content, water holding capacity and supports many soil organisms including fungi, bacteria, amphibians, insects/invertebrates and, in turn, birds and mammals. Decomposition also results in some release of CO_2 from soils. The rate of decomposition of stumps, and other woody material, could be an important factor in carbon sequestration of plantation forests (Palviainen et al., 2010).

Stumps are a significant feature of a logged landscape, whether in a forest that was previously clearcut or one that has undergone thinning. Stumps can make up as much as 20% of the total carbon stores in some forests (Palviainen et al., 2010). However, very few studies have attempted to quantify the amount of carbon that is held in stumps or the rate at which the stumps decompose (Tobin, 2007; Shorohova et al., 2008).

Stumps, through decomposition, are a factor in forest soil development and productivity in terms of water storage, development of soil structure and nutrient cycling (Tobin et al., 2007). In addition, stumps have been theorized to have a higher rate of decay in comparison to other woody material due to increased moisture and greater surface area contact with the soil (Shorohova, et al., 2008). Before a tree is cut, the 'stump' portion is the connection between roots and shoots or crown. This portion of the tree serves as the main conduit for the transfer of water and nutrients to the bole and branches as well as the transfer of organic compounds to the roots. Stumps interact with soil biogeochemically and their decomposition is influenced by site environmental properties as well as soil nutrient availability and microbial abundance. Many microorganisms obtain their energy through the breakdown of cellulose and lignin thus relying on decomposing woody material in order to survive and reproduce (Harmon et al., 1986; Edmonds and Eglitis, 1989).

In western Washington, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) is the key species harvested for timber. Many studies have looked at the role of woody debris in carbon cycling of Douglas-fir forests including decomposition rates (Maser and Trappe, 1984; Edmunds and Eglitis, 1989; Stone et al., 1998; Holub et al., 2001; Weedon et al., 2009). Some studies have noted a decrease in decay rates of Douglas-fir wood due to compounds, which restrict the infestation of mycelium into the debris after death (Scheffer and Cowling, 1966). However, these studies did not examine the decomposition rates of Douglas-fir stumps, thus stump decomposition rates are largely unknown. Understanding the role of Douglas-fir stumps in carbon cycling of managed forests will fill a current gap in our knowledge.

Recent studies have attempted to quantify decay by using a resistograph and other arborist tools rather than using the traditional method of taking a cross section of the log or other material and measuring density by displacement. The resistograph measures changes in torque that correlate to changes in density, providing a relative unit of torque that could be converted to density. The resistograph has been shown to be a useful tool in determining density of coarse woody debris (Kahl et al., 2009), (Makipaa and Linkosalo, 2011). It has also been found that density determined using a resistograph has a high correlation with decay (Costello and Quarles 1999) (Kahl et al., 2009) (Rinn et al., 1996). By looking at the decrease in density using a measure of resistance of Douglas-fir woody material, decomposition rate can be calculated (Harmon et al., 2000). [This relationship can also be directly related to respiration and carbon sequestration.]

This study created a model of stump decomposition and examined changes in carbon storage over time by assessing density of stump material, stump size, volume, nitrogen content, and carbon content of Douglas-fir stumps on tree farms in a chronosequence in western Washington State. This model contributes to the understanding of changes in the ecosystem carbon pool over time and the potential of stumps to serve as a carbon sink in Douglas-fir plantation forests and fills the gap in current life cycle analysis of these forests. Various physical and environmental factors were assessed for relationships and effects on decomposition rate and C storage in stumps.

Methodology

In order to assess the decay rates and changes in carbon storage of Douglas-fir stumps over time, measurements were taken at five western Washington Douglas-fir plantation sites, which were thinned or harvested in 1992, 1999, 2006, 2010, 2012 or 2013. Sample plots were within the Stand Management Cooperative (SMC) sites, a silvicultural research coalition of state, federal, and private forest enterprises. Through the SMC, long term records of individual trees and characteristics of each site were available. All of these sites were dominated by Douglas-fir, though one site did have some western hemlock (*Tsuga heterophylla* (Raf.) Sarg.). Each site was a second or third generation plantation forest. General site characteristics such as the location, elevation, slope, average daily temperature, average temperature range and average yearly rainfall are provided in Table SC-1.1. Sites descriptions represent some of the environmental and geographical characteristics of the location. Low represents elevations below 200 m, and high represents sites at or above 300 m. Dry represents an average of less than 80 cm yr^{1} of rain, moist represents an average rainfall of 140 cm yr^{1}, wet represents a site with an average of 175 cm yr^{1} of rain and very wet represents a site with average rainfall greater than 240 cm yr^{1}. Most thinning dates corresponded to a single plot at a site. Site properties such as soil type, soil texture, established trees per hectare, thinning level, estimated trees per ha (at time of sampling), the years the site was thinned, the average stump diameter on the site and the number of samples taken per plot can be found in Table SC-1.2.

Table SC-1.1. General site characteristics. Stump sample sites ranged from high to low elevation, drier to very wet and from and flat to sloping. Average yearly temperatures among sites had a maximum difference of 3°C and an average seasonal difference of about 15°C. Site coordinates are in decimal degrees. Aspect is given with the highly-sloped site.

Site Description	N Coordinates (Decimal Degrees)	W Coordinates (Decimal Degrees)	Average Rain (cm)	Average Daily Temperature (°C)	Seasonal Temperature Range (°C)	Elevation (m)	% Slope
High, wet, level	46'27.500"	122'4.013"	190	19	15	300	5
Low, moist, level	48'19.436"	122'11.303"	130	16.5	14.5	190	0
Low, dry, level	47'53.959"	122'46.302"	80	16	12.2	180	0
Low, moist, west sloping	47'35.465"	121'43.739"	130	17.5	15	200	40, W
High, very wet, level	46'43.311"	123'25.170"	240	16.5	14	350	0

Table SC-1.2. Site properties and stump characteristics. Estimated trees ha-1 corresponds with the level of thinning and the initial number of trees ha-1 and is representative of the number of trees at the site when sampling was conducted.

Site Description	Soil Type	Soil Texture	Established Trees ha^{-1}	Thinning Level	Estimated Trees ha^{-1} (thinned)	Year(s) Thinned	Average Stump Diameter (cm)	Samples taken per plot
High, wet, flat	Cispus cindery loam	Cindery Sandy Loam/Pumice	1060	Light	250	2012	51	5
Low, moist, flat	Tokul gravelly medial loam	Gravelly Loam	1250	Light	250	2013/2010	49	20
Low, dry, flat	Sinclair gravelly sandy loam	Sandy Loam/Gravelly Sandy Loam	1110	Heavy /Moderate	200	2010/1999	30	20
Low, moist, west sloping	Blethen gravelly loam	Sandy Loam/Gravelly Loam	1110	Light to Moderate	220	2010, 2006, 1999 and 1992	34	20
High, very wet, flat	Boistfort silt loam	Silt Loam	1110	Clear Cut	0	1999	77	13

Measurements were taken on stumps ranging from 14 to 58 cm in diameter, with an average stump diameter of 39 cm. Stumps measured had been cut between 1 and 22 years ago (Figure SC-1.1). Each stump measured was tagged and given a sample number. Stumps were then measured for average height, diameter at sample height, current light exposure, bark thickness, knife penetration depth, moisture content and bark cover. Knife penetration depth is determined by thrusting a knife into the exposed stump and used to measure decay class (Tobin, et al, 2007). Some stumps were also measured for the depth to which the main cylinder of the stump extended below ground.

Figure SC-1.1. Stump appearance at ages 1, 4, 8 and 15. Note the lack of rot on the one year old stump, the white fungus on the four year old stump, the change in color on the eight year old stump and the crumbling form of the 15 year old stump.

Stumps were measured for density by proxy with an IML F300 resistograph. Stumps were first stripped of their bark at both the south and east sides. After the bark was removed, a resistograph measurement was taken on those sides at 15 cm from the top of the stump. Two resistograph measurements were taken on each stump, at the south and the east sides. The two resistograph readings per tree were then averaged to control for variability within the stump (Kahl et al. 2009). This average reading was then used to represent the stump's resistance. The readings for each stump were then converted to density. Four of the 164 resistograph readings collected were determined to be outliers using SPSS and discarded.

The conversions of resistance readings to stump density were made by taking cross sections (cookies) of stumps that were sampled and relating the displacement density of those cross sections to the readings. The density of each cross section was determined using water displacement to obtain volume and dried at 65°C until there was no change in weight (Means et al., 1985). Each cross section was taken from a stump at the same level as the resistograph reading was taken. Each cross section was taken at one site (low, moist, west sloping), with 29 total cross sections taken from stumps aged four, eight and fifteen years. The resistograph readings taken from these cross sectional portions of stump were regressed against the densities of the stump, determined using cross sections. In addition, resistograph readings from Douglas-fir wood blocks were used as an upper boundary to convert resistance to density (Figure SC-1.2). The resultant equation was cross-validated with two of the paired resistance-cross section density pairs. The difference between the calculated density and the actual density of these was only around 0.01.

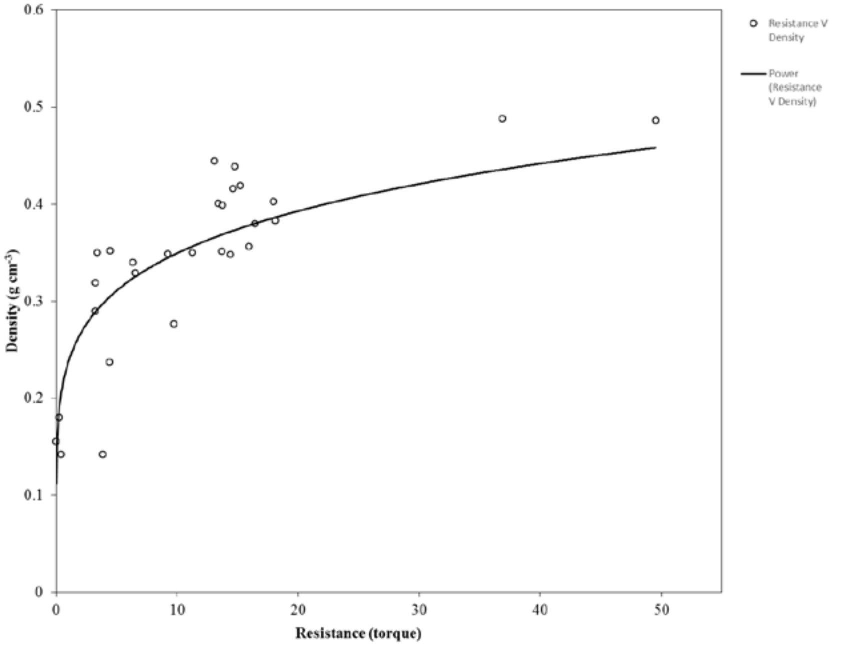

Figure SC-1.2. Resistance to displacement density. Resistance is represented in units of torque registered by the drill as the drill bit passes through the stump material. Each measurement is an average resistance for the two measurements taken per stump. The equation for the line is Y= .24X.169, where X is resistance, Y is density (g cm-3) and has an r2 of .668.

A standard 9.5 x 1 cm drill bit attached to a cordless drill was used to excavate material for CHN analysis. Excavation was done at the same location on the stump where resistograph readings were taken (the south and east sides). A maximum depth of 9.5 cm of wood was excavated. The wood samples were taken from the south side and east sides of each stump. This process of measuring stump characteristics, taking two resistograph readings and excavating material with a conventional drill, was then repeated on every fourth stump below ground. In order to obtain a below ground sample, a 30-40 cm pit was dug at the south side of the stump. However, the belowground material was only taken from the south side of each stump and the collected material was analyzed for C and N separately.

Samples were refrigerated until dried at 70°C for 48 hours weighed in the lab. The dried material was then ground to 2 mm and analyzed for CHN using a Perkin Elmer Series II CHNS/O Analyzer 2400.

The total volume of each stump was calculated by using diameter and height measurements. The below ground volume was added to this, however, the depth of each stump below ground was not measured on all stumps and thus this number was calculated by using the average depth. As a result, the final volume of these stumps was calculated by adding 30 cm to the total height of the stump.

Age of stump was calculated by subtracting the year that the tree was cut from 2014 (the year sampled). Additional site parameters were added to the data from the SMC database with information about each site and plot, including the date planted, aspect, soil type, average rainfall, average temperature, temperature range and elevation.

The final stump density data was compared to all known factors first using a Pearson correlation to establish the potential factors influencing decay. Plots of the density versus stump age were then generated to assess what type of model may be more accurate. Density was transformed using the natural log and a stepwise linear regression model of the potential significant factors against the natural log of density was performed using IBM SPSS 19. Carbon and nitrogen concentrations vs stump age and carbon and nitrogen content versus stump age were also plotted using IBM SPSS 19.

Results

Carbon concentrations in stumps remained relatively constant both by site and by age (Table SC-1.3). Concentration ranged from 0.11 to 0.68 g g^{-1} with a standard deviation of 0.05 g g^{-1} and a standard deviation of less than 0.005 g g^{-1} for stumps aged 2 years or younger. Carbon concentration was slightly higher in material from stumps age 8 to stumps aged 15 and 22 years (Figure SC-1.3).

Nitrogen concentration increased in the stump samples over time (Figure SC-1.3) with the mean concentration increasing from 0.02mg g^{-1} at 1 year of age to 0.35mg g^{-1} at 22 years since cut (Table SC-1.3). There was a gradual increase in standard deviation of nitrogen concentration with stump age, from about 0.02mg g^{-1} at 1 year since cut to 0.1 mg g^{-1} at ages 4 and 16 g g^{-1} at age 15.

Carbon to nitrogen ratios decreased over time. The average C to average N ratio for stumps aged 1 year was about 2400 whereas this ratio for stumps aged 22 years was only about 140 (Figure SC-1.3). There was a very high standard deviation of C: N ratios by age.

Table SC-1.3. Aboveground stump densities, mositure, carbon and nitrogen concentrations, and C/N ratios. Densities were calculated using the equation in Figure 2 ($Y= 0.24X^{.169}$ where X is resistance, Y is density (g cm^{-3})).

Age (Years)	Number of Samples	Average Calculated Density (g cm^{-1})	Average Moisture (%)	Average Nitrogen Concentration (mg g^{-1})	Average Carbon Concentration (g g^{-1})	Average C to Average N (g C g N^{-1})
1	21	0.35	70	0.21	0.48	2300
2	4	0.36	110	0.73	0.48	660
4	61	0.36	120	0.80	0.48	600
8	20	0.34	160	1.1	0.48	440
15	53	0.28	600	2.9	0.50	170
22	5	0.27	390	3.5	0.49	140

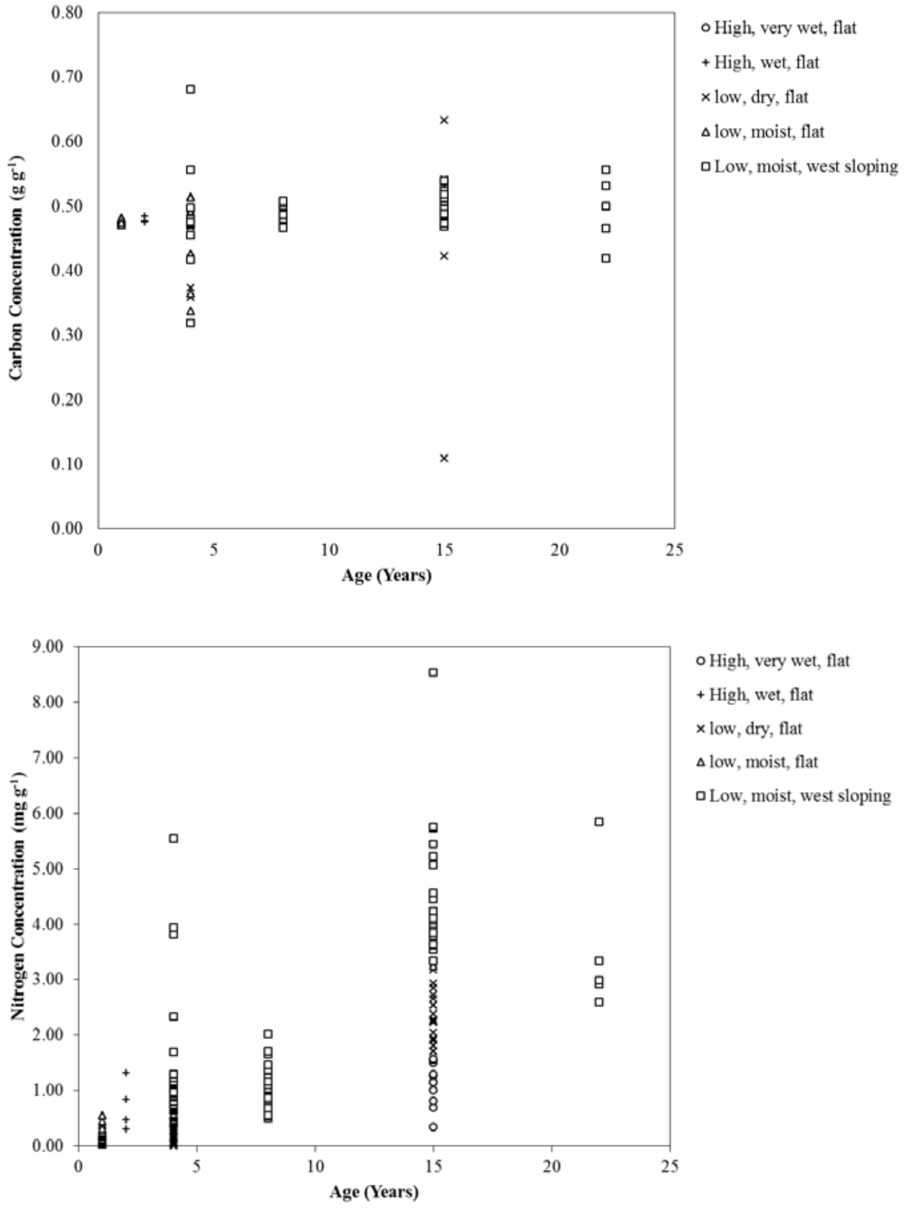

Figure SC-1.3. Stump carbon (g g-1) (upper) and nitrogen (lower) concentrations (mg g-1) by stump age (years since cut) and by site (symbol).

There was generally a lower concentration of carbon with more variation in belowground stump material in stumps aged 4 years and older. Nitrogen concentration was higher in below ground samples younger than 8 years of age. Carbon to nitrogen ratios are much lower for the belowground stump material than the above ground stump material. There was a slight difference between samples take above and samples taken belowground in carbon concentration ($p=.05$) and nitrogen concentration ($p=.04$, Table SC-1.4).

The average Douglas-fir plantation stump has about 98 kg of carbon depending on the age of the stump and the volume, with a standard deviation of 67 kg. At 1 year since cutting the average stump in this study had about 15 kg of carbon. This decreased to about 7 kg of Carbon at age 15. The average total stump volume (including belowground material) was about 625 m^3 with a standard deviation of about 430 m^3. The 15-year-old stumps were at the lower end of this size range and the 1-year-old and 22-year-old stumps at the upper end of this size range (Table SC-1.5).

There was a general decline in the measured density of stumps over time as well as an increase in variation (Figure SC-1.4). Calculated densities ranged from 0.05 to 0.40 g cm^{-3}. The average calculated densities, moisture content, carbon nitrogen concentration, carbon concentration, and carbon to nitrogen concentration are in Table SC-1.3.

The major factor affecting stump density is age. Elevation, the presence of red rot (also referred to as brown rot), and canopy cover also had high coefficients of correlation; other factors such as diameter, average precipitation, temperature fluctuation, bark cover, bark thickness, and the presence of insects did not have a high correlation (Figure SC-1.5).

Table SC-1.4. Comparison of above and belowground stump densities, carbon and nitrogen concentrations, and C/N ratios. Note that the differences in densities between above and belowground samples are minimal. N = 45

Age (Years)	Below Ground Density (g cm^{-3})	Above Ground Density (g cm^{-3})	Below Ground Nitrogen Concentration (mg g^{-1})	Above Ground Nitrogen Concentration (mg g^{-1})	Below Ground Carbon Concentration (g g^{-1})	Above Ground Carbon Concentration (g g^{-1})	Below Ground Average C to Average N (g C g N^{-1})	Above Ground Average C to Average N (g C g N^{-1})
1	0.37	0.34	1.7	0.20	0.48	0.48	280	3200
2	0.32	0.36	1.5	0.50	0.48	0.48	320	960
4	0.39	0.39	1.3	1.0	0.46	0.50	370	500
8	0.38	0.39	2.6	1.1	0.41	0.48	160	440
15	0.29	0.26	3.2	3.3	0.44	0.50	140	150
22	0.34	0.25	3.1	3.2	0.48	0.48	150	150

Table SC-1.5. Average stump volume, stumps per hectare, and kg C by age and site.

Age (years)	Site Description	Average Above Ground Stump Volume (cm^{3})	Average Below Ground Stump Volume (cm^{3})	Average Total Stump (cm^{3})	Average Stump Density (g cm^{3})	Average g C per stump (g C stump^{-1})	Stumps per Hectare	Average Total Site C (kg C ha^{-1})
1	Low, moist, level	62000	26000	88000	0.35	15000	1100	17000
2	High, wet, level	50000	52000	100000	0.36	18000	850	15000
4	Low, dry, level	30000	21000	51000	0.35	8900	900	8000
4	Low, moist, level	37000	25000	61000	0.36	10000	1100	11000
4	Low, moist, west sloping	39000	27000	66000	0.37	12000	900	11000
8	Low, moist, west sloping	20000	24000	44000	0.34	7200	900	6500
15	Low, moist, west sloping	35000	21000	57000	0.27	7800	900	7000
15	Low, dry, level	16000	14000	30000	0.29	4000	900	3600
15	High, very wet, level	80000	44000	120000	0.26	16000	850	14000
22	Low, moist, west sloping	73000	30000	100000	0.27	15000	900	14000

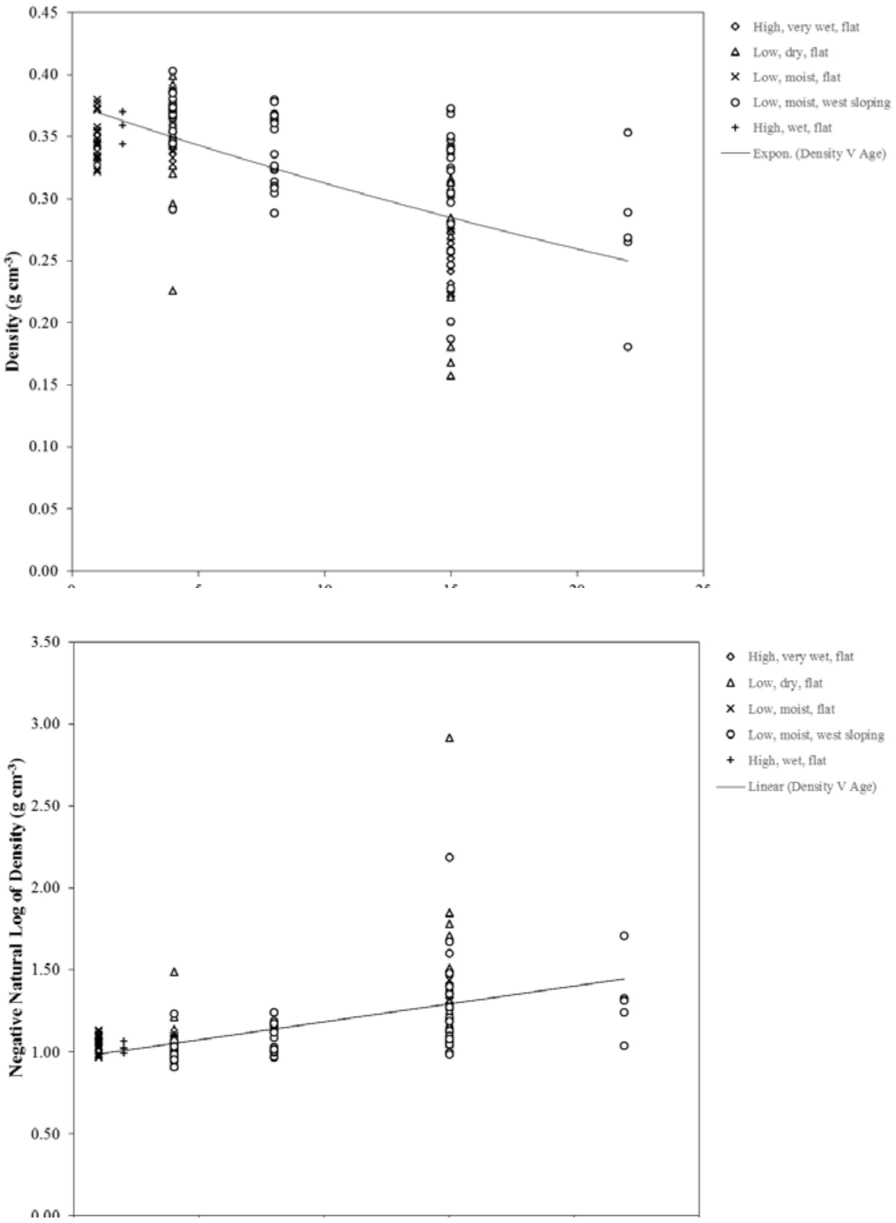

Figure SC-1.4. Stump densities (g cm-3) by stump age (years since cut), with trend line, and plot of negative natural log transformed stump densities (g cm-3) by stump age (years since cut). Densities were calculated using the equation which resulted from regressing resistograph units against the densities of cross sections measured using the resistograph (Y = .24X.169 where X is resistance, Y is density (g cm-3).

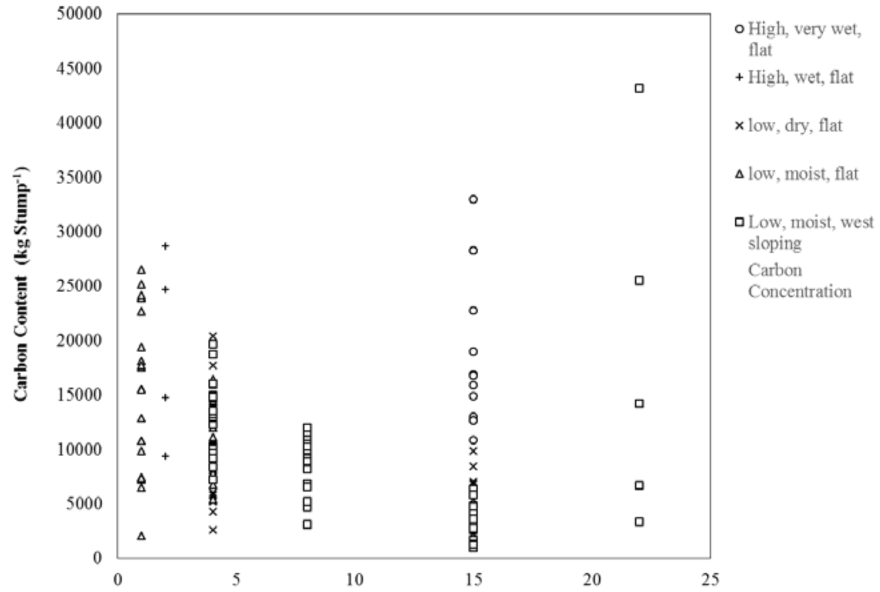

Figure SC-1.5. Plot of carbon content (kg stump-1) by age (years since cut). Concentrations are shown per stump, with each site (symbol).

Elevation, temperature, slope and the presence of red rot were also significant for determining density, however slope correlated very highly with temperature and red rot correlated with very highly age. Elevation and slope (or temperature were included in subsequent models; slope (p=0.011) explaining the greatest amount of additional variance, despite the low correlation with density. None of the models with these additional factors increased the adjusted r^2 by more than 0.04 from the model utilizing age as the only factor.

Moisture content and the presence of red rot were also significant factors in determining density, however, they all correlated highly with each other and none explained much of the remaining variance in stump density. The relationship of stump density to stump age is plotted in Figure SC-1.6.

Stumps within the first decay class were around 1 to 2 years old. Stumps within the second class were around 4 years old, stumps within the third class were around 8 years old and stumps in the fourth and fifth decay classes were 15 and 22 years old respectively, with many class 5 stumps aged 15 years.

There was no significant difference in density for stump material collected above ground vs material belowground (Table SC-1.4). Determining density by using the resistograph resulted in density data that were slightly lower than other decomposition studies of Douglas-fir boles (Harmon et al., 1986; Means et al., 1985).

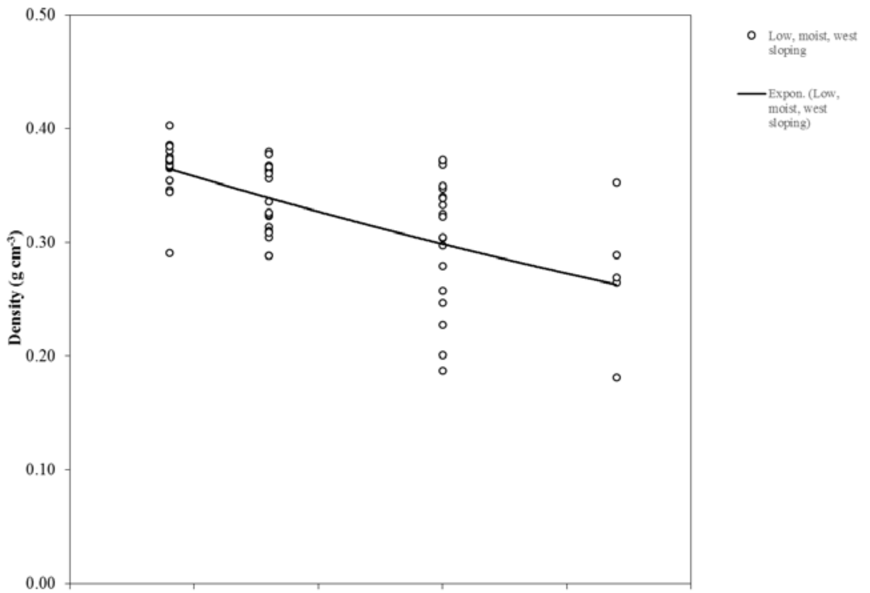

Figure SC-1.6. Stump density (g cm-3) by stump age (years since cut) of the low, moist, sloping site.

This could be due to calculating the relationship of density to resistance with fewer data points at the upper-end of the density range. It could also be because we were unable to obtain resistograph data from stumps that were freshly cut or because of the age of the trees when they were cut, as some of the Douglas-fir decay studies examined old growth logs. However, the densities ranges of this study were within the standard deviation range of densities for Douglas-fir in Harmon et al.'s study (1986) on Douglas-fir and decomposition classes.

The rate of Douglas-fir stump decay per year (k) found by modeling the natural log of density against age in this study (k=0.019) is very close to the k of Douglas-fir logs determined by Stone et al. (1998), and the fine root decomposition rate for Douglas-fir found by Chen et al. (2002). This is especially interesting as Stone's average sample diameter was the same as this study's (34cm) and with a larger standard deviation. The k for this study is in between the value for large diameter Douglas-fir logs established by other studies (Edmonds et al.,1986; Erikson et al., 1985). Yet, other studies found the decay constant to be much higher (Means et al., 1985; Edmonds et al. 1989).

The plot of density by age over all sites for this study was consistent with the findings of Edmonds et al. (1989), Erickson et al. 1985 and Harmon et al. (2000), that decay does not start for up to five years. The results from the low, moist west-sloping site show that at 8 years, however, decay has generally begun. The lag time could be explained by the fact that many stumps could potentially be alive for long periods after they have been cut; using stored nutrients or due to root grafting with live trees. Delayed death or root grafting could explain some of the higher densities on one site, where it was noted that the cambium had grown-up over a portion of many stumps after those stumps were cut.

Despite having a site with several ages and age classes with which to base a model of decay on for Douglas-fir stumps, there was still a lot of unexplained variation. However, other studies have reported high variation in wood density (Means et al. 1995). Some of the unexplained variation in the model found in this study may be due to the use of the resistograph. For example studies have found the genetics of individual trees to be a factor in the amplitude of resistograph readings and in turn could have impacted our density measurements/calculation. The trees on the sites in this study were most likely genetically different (Isik and Li, 2003; Gwaze and Stevenson, 2008). Additionally, moisture content of wood has been cited as increasing drill resistance (Kahl, 2009). Moisture content may explain why some of the trees aged 15 years still registered a very high resistance as the water content of trees of that age was, on average, 400% higher than that of the other ages. Resistograph reading changing due to wood moisture may also explain why moisture content was found to be a potentially significant factor affecting density (Shorohova et al., 2008).

The factors that determine decay are hard to predict beyond age (Barker, 2008). This study had several factors that were highly correlated with decay and are likely to still be influential in the decay process. However, due to the mild weather across the region, the minor changes in elevation across sites and other characteristic that were similar among study sites, it is hard to determine which factors in addition to age influence the rate of stump decomposition. With more variability factors such as elevation and temperature could become more significant for generating a model of decay in Douglas-fir stumps. This study found factors such as red rot to negatively correlate with density, suggesting that classification systems based on appearance could accurately determine the rate of decay. However, determining the decay constant through age classification of woody material in other studies did not yield a model that explained as much of the variation (Means et al.,1985; Edmonds et al., 1986; Stone, et al., 1998 and Harmon et al., 2000).

Carbon content decreasing over time could be due to both respiration and translocation of the carbon due to biological activities of fungi, bacterial and other flora and fauna. Carbon concentration staying more or less constant supports the idea that respiration in Douglas-fir logs does not change between decay classes (Marra and Edmonds, 1994). The loss of carbon over time, coupled with stable nitrogen content suggest that while carbon is being lost, nitrogen is neither being fixed nor lost at significant levels.

The general patterns associated with carbon and nitrogen concentrations and C:N ratios in this study were consistent with Palviainen et al. (2010) as well as Edmonds et al. (1989). However, with the high variability of C:N ratios in each age group of stumps, it is hard to determine if the average numbers are representative enough to make inferences on C:N trends.

The carbon concentration of Douglas-fir stumps was higher, on average, than that of fine roots as found by Chen et al. (2002). Whereas the nitrogen content of fine roots was far less on average than that of Douglas-fir stumps (Chen et al., 2002). This somewhat agrees with the findings that the belowground portion of the stump had a higher nitrogen concentration than the above ground portion, though the fine root concentration of nitrogen was an order of magnitude larger than this study's below ground stump material.

Using the model of decay coupled with the average volume of stumps (675m^3), and 900 stumps ha^{-1} on average, indicates that a site should have around 110,000 kg ha^{-1} of carbon when it is initially thinned, decreasing down to 83,000 kg ha^{-1} of carbon by age 15. Considering that an area that has undergone multiple thinning operations such as at the low, moist, west sloping site with trees aged 4 and 15 years the site could have about 93,000 kg ha^{-1} of carbon. Using Thies and Cunningham's (1996) equation for root biomass with the average stump diameter of 0.39 cm, we can estimate the average root biomass of our stumps to be about 83 kg stump^{-1}, resulting in an additional 37,000 kg ha^{-1} of carbon to be accounted for in stumps and roots. This puts the total amount of carbon in a freshly cut stump as high as 147,000 kg ha^{-1}with a single thinning.

The model from this study estimated that cut stumps and roots contain up to 147 Mg ha^{-1}. Total soil carbon for forest plantation sites has been estimated at 130 Mg ha^{-1} (James and Harrison, 2014). The total carbon in these Douglas-fir plantation forests can be estimated at 500 Mg ha^{-1}. Stumps may contain up to 33% of the total tree-based carbon on a Douglas-fir plantation site, 22% of the total forest carbon, when accounting for the cylinder alone and up to 29% of the total ecosystem carbon on a forest plantation sites when accounting for both the stump cylinder and the roots.

Conclusions

Douglas-fir stumps decay at a rate of 0.019 year^{-1} and are a significant carbon sink for a substantial period of time in plantation forests, with up to 33% of the total tree-based carbon and 29% of the total ecosystem carbon stored in stumps and roots. Carbon content decreases with the age of the stump leaving valuable nitrogen in forests where nitrogen is often a limited nutrient. When all other woody debris is removed, stumps help to support a recovering ecosystem while still supplying nutrients to young saplings and the trees that remain. Using a resistograph of decay classification managers can establish levels of carbon on thinned or logged sites as potential sources of carbon credits.

TASK 2. SOIL CARBON ANALYSIS: DEEP SOIL QUANTIFICATION, MODELING AND SIGNIFICANCE OF CARBON IN SUBSURFACE LAYERS

Objective

Soil represents the most important long-term carbon (C) reservoir in terrestrial ecosystems because it contains more C than plant biomass and the atmosphere combined (Jobbagy and Jackson, 2000). Despite this fact, soil remains the most under-reported element in ecosystem analyses in the primary literature (Harrison et al., 2011). In particular, detailed mechanistic explanations for soil C flux have not yet been proposed in subsurface layers, defined here as layers below 0.5 m (Fang et al., 2005; Fontaine et al., 2007).

Emissions of CO_2 and CH_4 in the United States from fossil fuel combustion and other sources totaled 5,691.8 Tg CO_2-C in 2011, and greenhouse gas emissions have increased, on average, by 0.4 percent per year since 1990 (US Environmental Protection Agency, 2013). Consequently, areas where C compounds may be removed from the climate system are particularly critical to the interests of protecting human health and preserving the ecosystem services upon which we rely (Lorenz et al., 2011). Among terrestrial ecosystems, forests have been identified as important potential sinks for C in the face of climate change (Birdsey et al., 2006). Within the contiguous United States, Pacific Northwest forests represent the single most important ecosystem for C sequestration – around 39% of total storage in all US forests, despite representing a much smaller proportion of total forested area (Birdsey, 1992). Homann (2005) found that current C storage in the Pacific Northwest is less than half of the potential total ecosystem storage, indicating that substantial C sequestration is theoretically possible in the future, depending on harvest-age trends, the occurrence of wildfires, and the rate of forest conversion.

Our ability to investigate total ecosystem C or change in C over time and across management treatments relies upon accurate measurement of the total soil C stock because it is the single largest C sink in forest ecosystems (Dixon et al., 1994; Harrison et al., 2003; Lal, 2005; Zabowski et al., 2011). A substantial portion of total soil C can be found in subsurface layers (Diochon and Kellman, 2009; Harrison et al., 2011; Kaiser et al., 2002; Zabowski et al., 2011). Jobbagy and Jackson (2000) projected that 56% of soil C globally can be found below a depth of 1m. Nevertheless, the major National Programs for forest C estimates in the US – the Forest Inventory and Analysis (FIA) and Forest Health Monitoring (FHM) programs of the United States Department of Agriculture Forest Service – sample soils only to 0.2 m (O'Neill et al., 2005). This represents a major systematic underestimation of total soil C, an error that propagates to any model or budget that relies upon this data (Hamburg, 2000).

Defense of shallow forest soil sampling has relied upon studies that identify residence times ranging from hundreds to thousands of years for C in subsurface horizons (Chabbi et al., 2009; Fang and Moncrieff, 2005; Kaiser et al., 2002). While residence times for resistant C compounds in deep soil horizons may be quite long, the assumption that this C will remain non-reactive may be less accurate than previously thought, particularly in soil with deep rooting zones. Recent research has given considerable support to the hypothesis that deep soil C may become reactive under certain conditions. Studies show that the stability of C in deep soil layers is controlled by the supply of fresh, energy rich C (Fontaine et al., 2007); that labile and resistant C pools respond similarly to changes in soil temperature (Fang et al., 2005); that nitrogen fertilization affects soil C both in surficial and subsurface horizons (Adams et al., 2005; Canary et al., 2000); and that retention of organic matter postharvest may increase the recalcitrant soil organic pool in intensively managed timber stands (Strahm et al., 2009). The resistant C in deep soil was formed under a particular set of environmental conditions that, by every account, will be altered by continuing climate change and human-influenced forcings (Fang et al., 2005; Richter, 2007; Richter and Yaalon, 2012). Natural and man-made changes in factors and processes that affect soil genesis can spur rapid movement of C into and loss of C from soil material that has remained stable for long periods of time (Fontaine et al., 2007; Harrison et al., 2011). Moreover, increases in the recalcitrant soil organic C pool are an important potential long-term sink for atmospheric C (Lorenz et al., 2007). Ignoring subsurface soil horizons effectively eliminates our ability to accurately assess C sequestration in the more recalcitrant pool.

This study, conducted in coastal Pacific Northwest Douglas-fir forests, addresses three objectives: (1) to determine the effect of systematic sampling to variable depths on estimates of forest soil C; (2) to evaluate the ability of mathematical models to accurately predict total soil C to 2.5 m based upon 1.0 m sampling (a depth usually reachable with a non-powered soil auger); and (3) to assess which soils are most important to sample more deeply.

Methodology

Twenty-two sites were selected from University of Washington Stand Management Cooperative (SMC) Type V Long-Term Site Productivity plots to cover a range of soils across the parent materials and climatic conditions of the coastal Pacific Northwest region (Table SC-2.1). All study sites were intensively managed Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) plantations within the region bounded by the Pacific Ocean and the Cascade Range, and thus under a maritime-influenced climate which is generally characterized by mild, wet winters and warm, dry summers (Figure SC-2.1). The long-term mean annual temperature across the twenty-two sites ranged from 6.1°C at the Vailton site in the Cascade foothills to 11.6°C at the Jory site in the Willamette Valley (mean annual temp across all sites was 9.0°C). The lowest mean January temperature was 0.5°C at the Tokul site in the northern end of the Puget Trough, and the highest mean July temperature was 19.8°C at the Honeygrove site in the Willamette Valley. Mean annual precipitation ranged from 813 mm at the Olete site on the east side of the Olympic Range to 3302 mm at the Hoko site on the west side of the Olympic Range (mean annual precipitation across all sites was 1780 mm).

Table SC-2.1. Soil series, subgroup, texture, range in percentage coarse fragments over all horizons, and parent material for 22 forest soils in western Washington and Oregon. Data from Soil Survey Staff (1999).

Soil Series	Subgroup	Texture	Coarse Fraction (%)	Parent material
Astoria (A)	Andic Humudept	Silt loam	6-88	Colluvium and residuum derived from sandstone and siltstone
Barneston 1 (Bn1)	Typic Vitrixerand	Gravelly coarse sandy loam	12-81	Volcanic ash and glacial outwash
Barneston 2 (Bn2)	Typic Vitrixerand	Gravelly loam	25-59	Volcanic ash and loess over glacial outwash
Blachly (Bl)	Humic Dystrudept	Silty clay loam	31-70	Residuum and colluvium derived from basic igneous and sedimentary rock
Cloquallum (C)	Aquandic Dystrudept	Silt loam	15-44	Silty lacustrine deposits with volcanic ash
Hoko (Hk)	Aquic Durudand	Gravelly silt loam	12-53	Alpine Basal till with volcanic ash
Honeygrove (Hg)	Typic Palehumult	Gravelly clay loam	0-20	Colluvium and residuum derived from sandstone, siltstone, and volcanic rock
Hoquiam (Ho)	Typic Fulvudand	Silt loam	0-50	Old alluvium over glacial drift
Jory (J)	Xeric Palehumult	Silty clay loam	0-4	Colluvium and residuum derived from basalt
Kinney (K)	Andic Humudept	Silt loam	49-80	Residuum weathered from igneous rock with a mantle of volcanic ash
Lemolo (L)	Typic Humaquept	Silt loam	7-65	Oceola mudflow deposits
Melbourne (M)	Ultic Palexeralf	Loam	0	Residuum from siltstone
O'Brien (Ob)	Typic Fulvudand	Silt loam	6-58	Glacial outwash
Olete (Ol)	Vitrandic Haploxerept	Very gravelly silt loam	16-79	Colluvium and residuum from basalt
Ovall (Ov)	Andic Humixerept	Gravelly loam	0-28	Glacial drift over residuum with volcanic ash near surface
Ragnar (R)	Vitrandic Dystroxerept	Fine sandy loam	0-16	Glacial outwash
Scamman (Sm)	Aquic Palexeralf	Silt loam	0-49	Mixed glacial and sedimentary materials
Shelton (Sh)	Typic Haploxerand	Gravelly sandy loam	36-76	Basal till with volcanic ash
Tokul (Tk)	Aquic Vitrizerand	Gravelly loam	19-77	Volcanic ash and loess over glacial drift
Tolovana (Tv)	Typic Fulvudand	Silt loam	0-74	Colluvium and residuum derived from tuffaceous sedimentary rock
Vailton (V)	Alic Hapludand	Silt loam	0-33	Residuum and colluvium from siltstone and shale mixed with volcanic ash
Windygap (W)	Xeric Haplohumult	Silt loam	0-7	Colluvium and residuum derived sedimentary rock

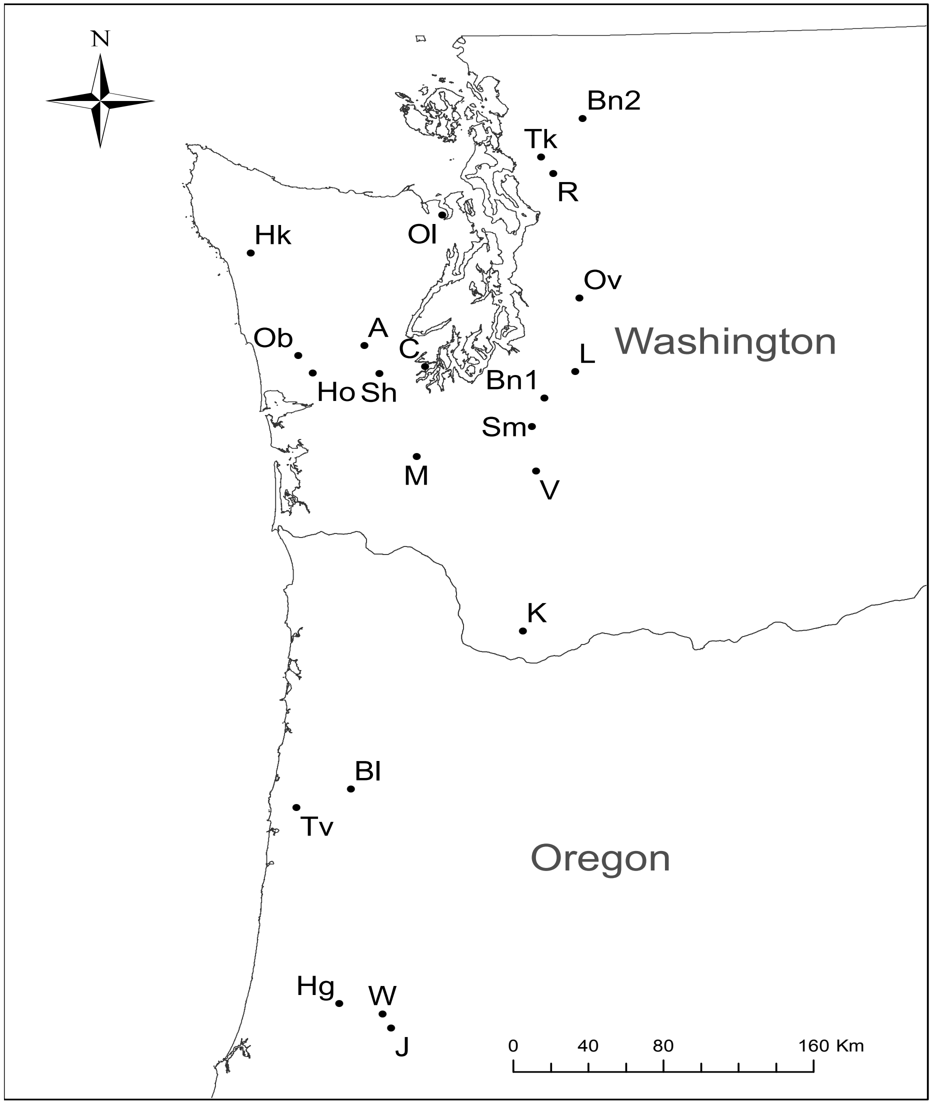

Figure SC-2.1. Locations of 22 soil series sampled in the Pacific Northwest Douglas-fir Zone of Washington and Oregon. One soil profile was excavated at each site. Map units for each series are listed in Table SC-2.1.

Soil pits were dug using an excavator to 2.5 or 3.0 m and classified using data from the US Department of Agriculture Natural Resources Conservation Service soil surveys. Bulk density samples were taken using a soil corer of a known volume in the middle of succeeding soil layers bounded by the mineral soil surface and depth intervals of 0.1, 0.5, 1.0, 1.5, 2.0, 2.5 and 3.0 m (when excavation was not impeded by bedrock or hardpan). Major horizons were identified and recorded, along with horizon thickness and profile depth (Figure SC-2.2). In all cases, soil pits extended into the C horizon, often extensively. Additionally, one forest floor sample per site was gathered from randomly placed 0.3 m x 0.3 m quadrats. Samples were sealed in plastic bags and immediately returned to the lab, where they were refrigerated at 3°C until analysis.

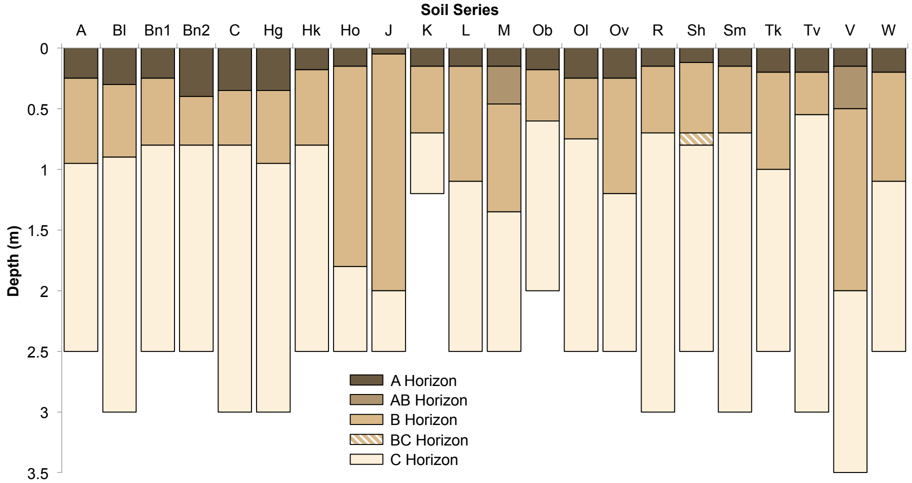

Figure SC-2.2. Major genetic horizons and profile depth in 22 soil series in Washington and Oregon. Profiles were sampled to at least 2.5 m except where bedrock or compacted material impeded further excavation.

Hamburg (2000) showed how inadequate soil sampling gives rise to inaccurate conclusions in analyses of ecosystem processes, and we followed his recommendation that measurements of soil bulk density and carbon concentration come from the same samples. The bulk density mineral soil samples were weighed, dried in an oven at 60°C to constant weight and then re-weighed. Holub (2011) demonstrated that sieving soil to <2 mm unnecessarily discards significant amounts of soil C in the 2 mm to 4.75 mm fraction. Consequently, our samples were separated into coarse and fine soil fractions using a 4.75 mm sieve. All obvious roots remaining in the sieve were discarded. The >4.75 mm fraction was weighed to determine mass and percent content of coarse and fine material at each depth interval.

Subsamples of the <4.75 mm fraction were ground with a mortar and pestle for elemental analysis. Organic horizons likewise were weighed, dried at 60°C to constant weight and reweighed. Samples were ground to <0.5 mm using a Wiley Mill. Total C analysis was performed using a PerkinElmer 2400 CHN analyzer.

The total carbon (C_t) for each sample layer was calculated from: the layer height (H) (cm) between the sample depth and the depth of the sample immediately above it or the surface; the bulk density (D_b) (g cm^{-3}) of the <4.75 mm fraction; and the C concentration (C_{con}), using the equation

$$C_t = H\, D_b\, C_{con} \qquad (1)$$

The result of this calculation for each sample was then converted from g cm^{-2} to Mg C ha^{-1} using dimensional analysis. Equation (1) assumes that C_{con} is constant throughout a layer of height H. Thus, C_{tot} calculated for any given layer will likely underestimate C from the top portion of the layer and overestimate C from the bottom portion of the layer. These differences should approximately cancel, giving a reliable estimate for C_{tot} in a given layer. However, due to the high rate of change in C_{con} near the soil surface, this method will slightly underestimate C_{tot} at the surface.

To address the question of appropriate soil sampling depth, total cumulative C was calculated for interval layers 0-0.1 m, 0-0.5 m, 0-1.0 m, 0-1.5 m, 0-2.0 m, 0-2.5 m, and 0-3.0 m. Due to the unpredictable variability of organic horizon C across sites, only mineral soil was included in these analyses. The cumulative C in each interval was calculated by addition of C from the deepest measured layer in the interval to the total of the more shallow depth intervals. For example, the 0-3.0 m interval includes all C in the measured layers from the surface to 3.0 m.

To test differences between soil C estimates for different soil sampling depths, mean total C across the cumulative depth intervals was compared using a linear mixed effects model in R (R Core Team, 2013). Using the nlme package (Pinheiro et al., 2013), depth intervals were compared as a fixed effect factor. The individual soil profiles were treated as a random block effect to account for the expected differences between different soils and to isolate the effect of sampling depth on the cumulative soil C estimates. Because samples taken from the same profile are spatially autocorrelated and thus violate the assumption of independent measures, the model incorporated a Gaussian spatial autocorrelation structure between sampling depths given the particular soil from which the samples were taken. Model-fitting with mean total C values showed strong departure from normality in the tails of residuals upon examination of the Normal Quantile-Quantile plot; a log transformation was performed on the data, and examination of the Normal Q-Q plot for the transformed data showed satisfactory agreement with the assumption of normally distributed residuals. The mean cumulative C for the 0-3.0 m layer was not included in the modeling or statistical analysis due to an inadequate number of soil profiles that could be sampled to that depth (n = 7 compared to n = 20-22 for other layers). ANOVA of the linear mixed-effects model was used to test for differences between means, and Tukey's Honestly Significant Difference (Tukey's HSD) was used to calculate multiple comparisons between the sampling depth intervals using the multcomp package in R (Hothorn, 2008). All statistical tests were performed at $\alpha = 0.01$ because the F-statistic is an approximation in ANOVA analysis of linear mixed models, and thus a greater degree of statistical confidence is desirable. Denominator degrees of freedom for the ANOVA F-test were calculated using the Satterthwaite method.

A series of functions were tested using SAS University Edition Proc NLMIXED (University Edition, SAS Institute Inc., Cary, NC, USA) to determine whether a single equation form could be fit to the soil C profiles (to a 2.5 m depth) at all 22 sites. In these functions, soil depth (i.e., the midpoint of each sampling interval) was used as the predictor of soil C content, and site was treated as a random effect. The group of functions initially tested were: logarithmic, first-degree inverse polynomial (hyperbola), type III exponential, the power function (Bennema, 1974), Schumacher's equation (Schumacher, 1939), the log-log equation of Jobbagy and Jackson (2000), the Langmuir equation, the negative exponential function (Minasny et al., 2006), the Chapman-Richards function, and the organic matter function used by Russell and Moore (1968). Where applicable, functions were fit to the C concentration profiles (concentration decreasing with depth) and to the cumulative soil C content (total content typically increasing toward an asymptote with depth). Four of the tested functions fit the data relatively well across all sites: these models always converged, were significant ($P < 0.05$), and had satisfactory distribution of residuals upon visual assessment. All four of the best-fitting functions fit cumulative soil C content rather than the C concentration profile. These functions were:

the first-degree inverse polynomial function:

$C_t = D / (a + b D)$ (Sit and Poulin-Costello, 1994)

the Langmuir equation:

$C_t = C_{max}\, a D / (1 + a D)$ (Johnson et al., 2011)

a logarithmic function:

$C_t = a + b \ln(D)$ (Johnson et al., 2011)

and a type III exponential function:

$C_t = a e^{b/D}$ (Sit and Poulin-Costello, 1994)

where C_t is total carbon content (Mg ha^{-1}) between the soil surface and depth D (cm), C_{max} is the maximum C content of the soil profile (Mg ha^{-1}), and a and b are fitted constants for the functions.

To test the capacity of these functions to predict deep soil C without sampling beyond 1.0 m, each function was then used to predict each soil's total C to a depth of 2.5 m, after fitting the function only with data from sample intervals of 0-0.1, 0.1-0.5, 0.5-1.0 m.

Results

The total C in the profile for each soil series (0-2.5 m depth) ranged from 61 to 321 Mg ha^{-1} (Figure SC-2.3). By comparison, mean organic-horizon C was 7.3 Mg ha^{-1} (with a range from 2.3 to 14.1 Mg ha^{-1}). Total C in organic horizons did not correlate with total C in the mineral soil ($R^2 = 0.04$) and thus subsequent analysis focused solely on mineral soil. ANOVA of total C at increasingly large intervals of depth showed that some mean estimates of C were significantly different than others ($F = 188.51$, $P< .001$). The F-test and p-value was calculated with 101 denominator degrees of freedom from the Satterthwaite approximation. Tukey's HSD revealed 4 groups of means that were significantly different from each other: the total soil C estimates for the 0-0.1 m and 0-0.5 m sampling depths were different from both each other and all other mean estimates; the 0-1.0 m, 0-1.5 m, and 0-2.0 m sampling depth means form a third group; and finally the 0-1.5 m, 0-2.0 m, and 0-2.5 m sampling depths form the final group of means ($P<0.005$; Figure SC-2.4). Controlling for spatial autocorrelation between samples and differences between sites, sampling to 1.0 m will give significantly less total soil C than sampling the whole profile (to a depth of 2.5 m).

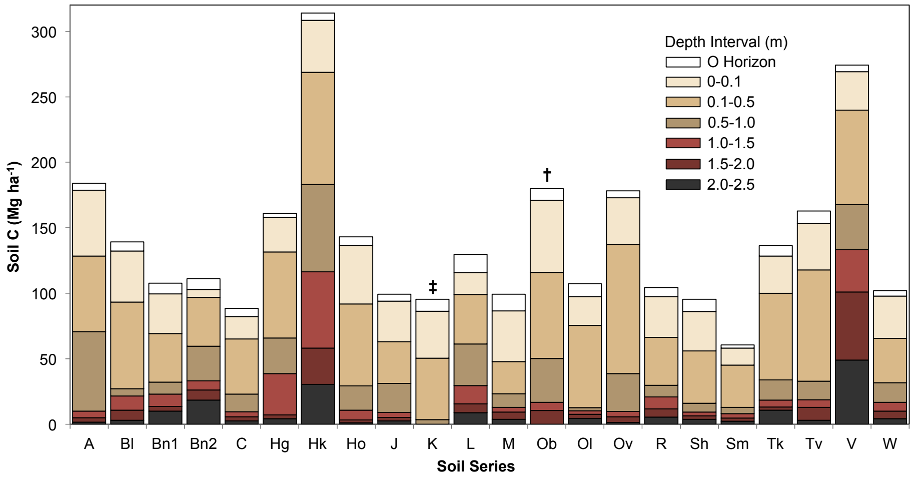

Figure SC-2.3. Soil C (Mg ha-1) within each sampled depth interval for 22 forest soils in western Washington and Oregon. † Soil pit excavation impeded by compacted glacial till at 2.0 m. ‡ Soil pit reached igneous bedrock at 1.0 m.

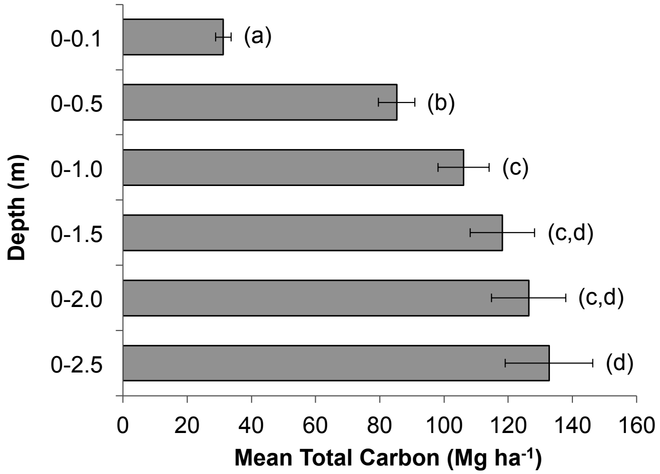

Figure SC-2.4. Mean total carbon at increasing depth intervals. Error bars represent ±1 SE. Means accompanied by the same letter are not significantly different (Tukey's HSD, P < .01).

On average, the greatest contribution to total soil C came from the 0.1-0.5 m layer (Figure SC-2.5). Though mean C concentration in this layer was less than half that of the surface to 0.1 m layer, increased bulk density and layer thickness resulted in more total C in the layer. Mineral soil C concentration decreased regularly with depth with averages between 0.32% and 0.15% in layers below 1.0 m. By comparison, average litter layer C concentration was 33.05%. Despite very low C concentrations, average soil C content in the deep layers never decreased below total C in the litter layer due to much higher bulk density. Deep soil layers contribute substantially to total soil C, even when C concentrations are low.

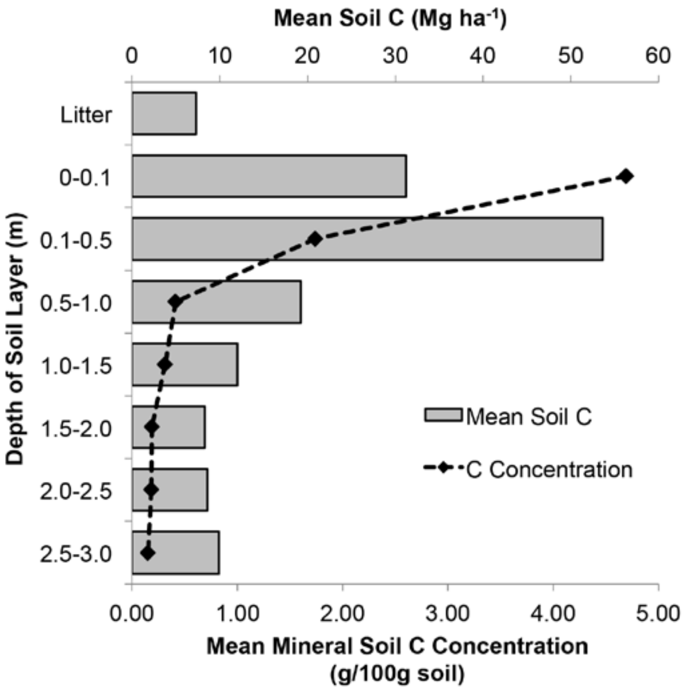

Figure SC-2.5. Mean soil C and mineral soil C concentration in succeeding soil layers from 22 soil profiles in western Washington and Oregon.

Given the need to establish a standard soil sampling depth for national C accounting, our data suggest that, for the soils in this region, sampling to at least 1.0 m would substantially increase the accuracy of total soil C estimates. However, an average of 21% of total C was located below 1.0 m, with a range from 6% in the Ovall series to 57% in the Vailton series. Jobbagy and Jackson (2000) projected that 56% of soil C globally across forest, grassland, shrubland, and other ecosystems can be found between 1.0 m and 3.0 m depth. For temperate coniferous forest soils, their estimate of soil C below 1.0 m was 41% compared to our estimate of 21% (to 2.5 m; this increases to 27% if we include the 2.5-3.0 m estimate from 7 soils). The difference between these estimates can be explained by differences in methods; Jobbagy and Jackson (2000) calculated C below 1.0 m by extrapolating from surface to 1.0 m data using a log-log equation that does not become asymptotic with depth, whereas our observed soil C profiles are best fit by functions that predict a sharper decline in soil C in deep parts of the soil profile. In general, methods that sample as deeply as possible in the soil profile will better approximate total C. Since accuracy of these estimates is critical in appropriate calculation of the total soil C reservoir (Hamburg, 2000; Whitney and Zabowski, 2004), in measuring C flux (Lorenz and Lal, 2005), and in properly modeling C dynamics over landscapes or ecosystems (Lal, 2005; Schrumpf et al., 2011), we recommend sampling soils as deeply as necessary to test defined hypotheses or meet objectives while considering budget and labor constraints.

Four soil C profiles selected to represent the range of C profiles among the 22 soils in this study are shown in Figure SC-1.6. The C profile of the Shelton soil series is typical of lowland glacial soils of the Puget Trough physiographic region. This soil is formed in young (~14,000 years) glacial materials and has a high coarse fragment content, with volcanic ash in surface horizons; cumulative C levels off significantly below the 0.1-0.5-m depth interval. Among the four mathematical functions fit to this profile, the fit of the logarithmic function was somewhat poorer than the other three functions at intermediate depths, as it did not capture the sharp decline in the rate of C accumulation below 0.5 m.

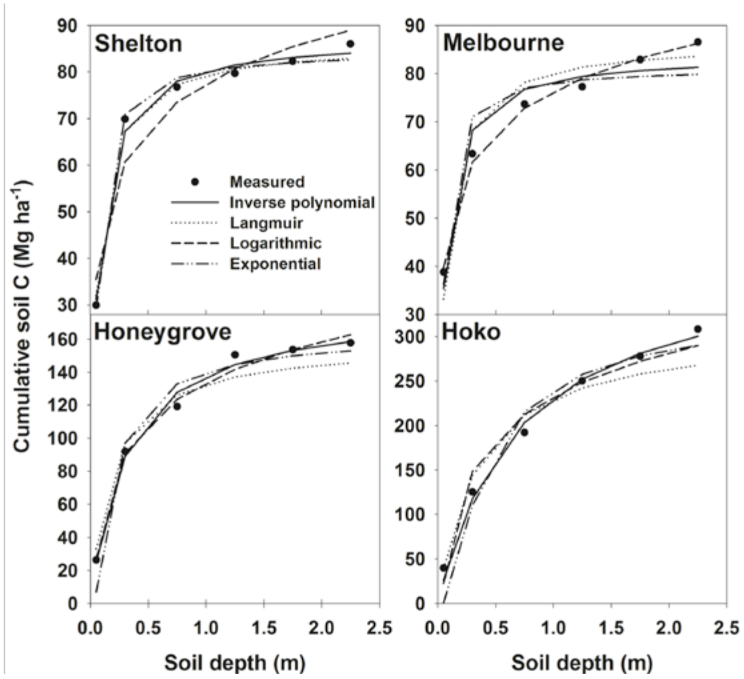

Figure SC-2.6. Cumulative soil C profiles (sample interval midpoints are plotted) for four soil series, representative of the range of profiles from 22 forest soils in western Washington and Oregon. Four functions are fit to the data for each soil series.

The Melbourne soil series is a well-drained loam formed in sedimentary residuum, found in the foothills. Although total C to 2.5 m is similar to the Shelton series, distribution of C in the profile is more even, producing a more gradual accrual of C with depth. In contrast with the Shelton series, the exponential function fit the C profile best, whereas the other three functions became asymptotic too rapidly and thus underestimated total C to 2.5 m.

The Honeygrove series is a very deep, well-drained Ultisol, with approximately twice the C content of the Shelton and Melbourne series, and an intermediate C profile shape relative to those series. The poorest fit was the Langmuir equation, which became asymptotic too early as a result of the value for the 0.5-1.0 m interval.

The Hoko series (308 Mg C ha^{-1}) is representative of the two sampled soils with very high C content; Vailton (269 Mg C ha^{-1}) is the other. Again the Langmuir equation became asymptotic too rapidly and did not account for the continuing rate of accrual lower in the soil profile.

Across all 22 soils, the inverse polynomial function had the most uniform distribution of residuals across soil depth (Figure SC-2.7; Table SC-2.2). The Langmuir equation consistently underestimated cumulative soil C at greater depths, whereas the logarithmic function tended to underestimate cumulative C near the surface and then overestimate it at a depth of 2.5 m. The exponential function overestimated cumulative C at intermediate depths but underestimated it at 2.5 m. Measured vs. predicted cumulative C to 2.5 m (i.e., curve fit at 2.5 m) appears in Figure SC-2.8. The inverse polynomial function again showed the most consistent fit across soils series. The greatest error in model fit occurred for the two soils highest in soil C. For the inverse polynomial, Langmuir, logarithmic, and type III exponential functions, the mean percentage error in predicting cumulative C at 2.5 m for the 22 soils was -2.0, -6.4, 2.7, and -5.0, respectively. Excluding the two soils highest in C, the mean percentage error for the remaining 20 soils was 1.7, 5.5, 4.0, and 4.5 for the same equations, respectively.

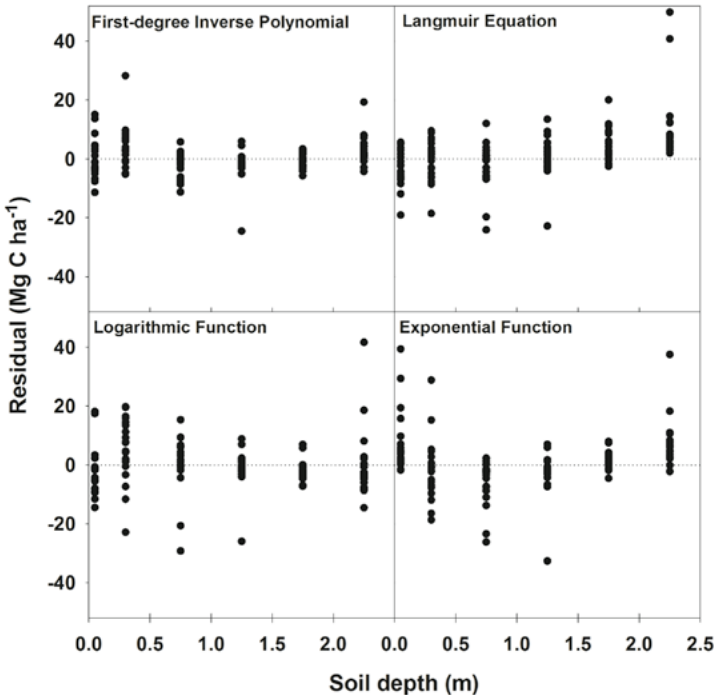

Figure SC-2.7. Residuals produced by fitting four different functions to 22 cumulative soil C profiles (0-2.5-m depth) of forest soils in western Washington and Oregon.

Table SC-2.2. Prediction error (Mg C ha-1) for four functions used to extrapolate total soil C content to 2.5 m based on samples collected between 0 to 1.0 m for 22 forest soils in western Washington and Oregon.

		Function			
Soil series	Subgroup	Inv. poly.	Lang-muir	Log.	Exp.
		Prediction error (Mg C ha^{-1})			
Astoria (A)	Andic Humudept	-18.7	-9.3	6.7	-28.4
Barneston 1	Typic Vitrixerand	-16.7	-6.1	-1.2	-20.8
Barneston 2	Typic Vitrixerand	-1.6	-15.6	-9.8	-18.8
Blachly (Bl)	Humic Dystrudept	-6.8	-5.9	17.1	-12.9
Cloquallum (C)	Aquandic Dystrudept	2.6	-4.4	15.6	-5.2
Hoko (Hk)	Aquic Durudand	-66.1	-50.1	-62.4	-115.4
Honeygrove (Hg)	Typic Palehumult	-16.0	-14.6	1.1	-33.8
Hoquiam (Ho)	Typic Fulvudand	1.2	-5.3	26.2	-7.2
Jory (J)	Xeric Palehumult	-4.0	-4.6	10.7	-11.6
Kinney (K)	Andic Humudept	9.3	-2.0	27.4	5.7
Lemolo (L)	Typic Humaquept	-2.3	-14.5	-6.5	-28.8
Melbourne (M)	Ultic Palexeralf	-11.4	-2.9	2.0	-13.4
O'Brien (Ob)	Typic Fulvudand	-4.6	-7.9	23.6	-17.4
Olete (Ol)	Vitrandic Haploxerept	3.8	-3.8	21.8	-1.7
Ovall (Ov)	Andic Humixerept	21.0	-8.5	49.3	0.6
Ragnar (R)	Vitrandic	-14.1	-5.2	0.8	-17.9
Scamman (Sm)	Aquic Palexeralf	0.3	-3.0	10.8	-3.9
Shelton (Sh)	Typic Haploxerand	-1.6	-3.1	14.3	-5.5
Tokul (Tk)	Aquic Vitrizerand	-0.4	-7.2	21.0	-11.3
Tolovana (Tv)	Typic Fulvudand	3.6	-7.5	31.4	-8.1
Vailton (V)	Alic Hapludand	-106.2	-56.8	-89.2	-128.8
Windygap (W)	Xeric Haplohumult	-11.7	-5.2	3.9	-16.7
	Mean error (Mg C ha^{-}	-10.9	-11.1	5.2	-22.8
	Mean error (Mg C ha^{-}	-3.4	-6.8	13.3	-12.8
	Mean error (%)	-5.6	-7.0	8.0	-13.9
	Mean error (%)*	-3.1	-5.8	11.5	-11.0

* Mean calculated without Hoko and Vailton series, the two series not well predicted by the models.

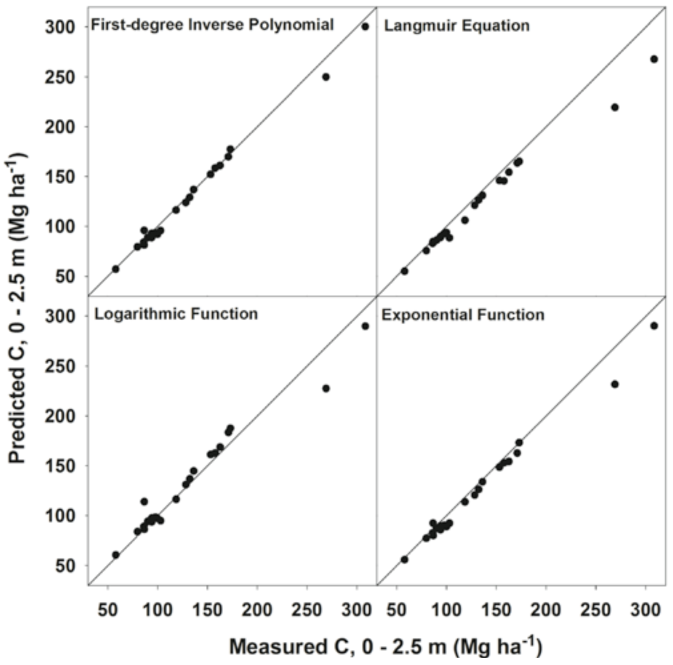

Figure SC-2.8. Predicted versus measured soil C (Mg ha-1 from 0-2.5 m) for 22 forest soils in western Washington and Oregon based on four functions fit to cumulative soil C profiles (0-2.5-m depth).

To determine the feasibility of accurately predicting cumulative soil C to a 2.5 m depth using data only to a 1.0 m depth, the four selected functions were then fit with cumulative C data from depth intervals of 0-0.1, 0.1-0.5, and 0.5-1.0 m. Using these fitted equations for each soil series, cumulative C was predicted to 2.5 m. Error increased for all four functions (Figure SC-2.9) relative to the fit with the full 0-2.5-m dataset (Figure FC-2.8). The greatest error occurred for the logarithmic and type III exponential functions (Table FC-2.2); conversely, the Langmuir equation predicted cumulative C to 2.5 m nearly as well with data from only 0-1.0 m as it did with data from 0-2.5 m. Mean percentage error in prediction across the 22 soils, and for the 20-soil sample excluding the high-C profiles, was lowest for the inverse polynomial function (Table FC-2.2).

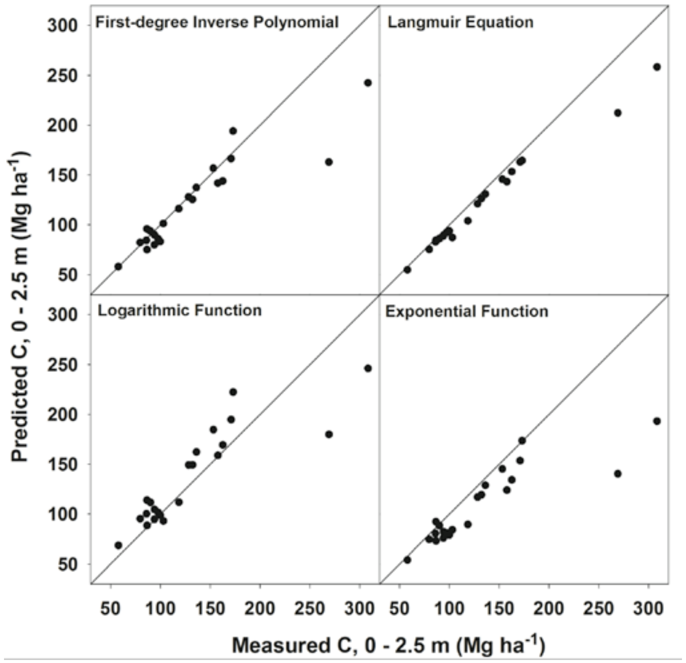

Figure SC-2.9. Predicted versus measured soil C (Mg ha-1 from 0-2.5 m) for 22 forest soils in western Washington and Oregon, based on four functions fit to cumulative soil C profiles to a 1.0-m depth that were subsequently used to predict C to a 2.5-m depth.

While the predictions of the best-fit models were generally adequate for total soil C to 2.5 m, several exceptions should be noted. First, the Hoko and Vailton series (both Andisols) were not well modeled, despite the importance of these C-rich soils for C accounting. Second, considerably more soil C is located below 2.5 m in these soils, and estimates produced by these models should not be construed as the same as sampling to bedrock. Our sampling only reached bedrock at two sites, the Kinney and O'brien series soils. Finally, none of the soils modeled in this study were Spodosols; as noted by Johnson et al. (2011) Spodosols resist standard C modeling techniques due to the irregular increase of C concentration in subsurface, spodic horizons.

Several of the soils sampled in this study provide useful examples of the broad variations in factors that influence soil C profiles of the coastal Pacific Northwest region. These factors include variations in parent materials (volcanic, glacial, and sedimentary), soil age, and climatic gradients. Here we discuss typical glacial and sedimentary soils, as well as volcanic soils with the highest total soil C.

The recent glacial history of the Puget Trough was accompanied by active volcanism in the nearby Cascade Range, which influenced the formation of many soils in the region. All sites with glacial parent materials selected for this study, except the Scamman series, have been impacted by volcanic activity to some extent. The Scamman series consists of deep, poorly drained soils that formed in mixed material weathered from glacial sources and can be found on high terraces and footslopes of adjacent hills. It had the least total soil C of all sites (58 Mg ha^{-1}), of which 14% was found below 1.0 m.

Tokul series soils are formed in similar, poorly drained conditions on glacial till, but with the addition of loess and volcanic ash. These soils contain amorphic mineralogy and an Andic diagnostic layer near the surface. Both the Scamman and Tokul series occur along the western slopes of the North Cascade Range. The C profiles of the Tokul site excavated in this study contained 128 Mg C ha^{-1}, considerably more than the Scamman site despite similar age (~14,000 years), and climatic conditions (~1400 mm precipitation per year, and mean annual temperature of 10°C).

The Honeygrove and Jory series are typical of soils developed from colluvium and residuum derived from sedimentary and basic igneous rocks. Both are very deep, well drained Ultisols, with the highest mean annual temperatures of all 22 sites (11.7°C at the Jory site; 10.6°C at the Honeygrove site). Despite similar conditions and relatively close proximity (Figure SC-2.1), the Honeygrove site contained substantially more soil C than the Jory site (158 Mg ha^{-1} to 94 Mg ha^{-1}, respectively), and the C accrued more gradually down the Honeygrove profile than the Jory profile. The differences in C distribution could be accounted for by differences in topographical position between the two sites: the Honegrove site is on a toe slope in the Oregon Coast Range, whereas the Jory site is near a peak in the hills south of the Willamette Valley. Climate also differs: the Jory site receives only 1143 mm per year, while the Honeygrove site receives 1905 mm precipitation per year, which could lead to increased leaching of organic matter to deeper soil layers.

The two series with the highest total C and C below 1.0 m were the Hoko (308 Mg ha^{-1} to 2.5 m, 116 Mg ha^{-1} below 1.0 m) and Vailton series (269 Mg ha^{-1} to 2.5 m, 133 Mg ha^{-1} below 1.0 m). Despite similar soil C profiles, the Hoko and Vailton series soils differ in many key environmental factors. The Hoko series is a young (~14,000 years), moderately deep, moderately well drained soil formed in glacial till mixed with volcanic ash. The excavated site is located on the west side of the Olympic peninsula at low elevation (61 m) and has the highest annual precipitation of all 22 sites (3302 mm). The Vailton series is considerably older (~40,000,000 years) and consists of deep, well-drained soils formed in volcanic ash and colluvium and residuum from siltstone and shale. These soils are found exclusively at somewhat higher elevations; the excavated site is 634 m above sea level in the Cascade foothills. Annual precipitation for Vailton series soils is slightly above average (2032 mm compared to mean 1780 mm across 22 sites), and the annual average temperature is the lowest from all 22 sites (6.1°C). The primary similarity between these soils is the presence of large quantities of noncrystalline minerals, including allophane, imogolite, and ferrihydrite (Soil Survey Staff, 1999). These amorphous or nanocrystalline minerals exhibit high adsorption of humic material, particularly humic and fulvic acids (Kennedy et al., 2002; Oades, 1988; Tisdall and Oades, 1982), and can preferentially preserve certain organic compounds from biological degradation for long periods of time (Baldock and Skjemstad, 2000; Calabi-Floody et al., 2011; Marschner et al., 2008).

The models tested in this study lacked the ability to adequately account for continuing, non-asymptotic accrual of C in deep layers, and thus poorly predicted total soil C for the Hoko and Vailton series (Table SC-2.2). In such cases, fully sampling the deep soil (> 1.0 m) remains the only satisfactory means for estimating total soil C. We suggest a simple diagnostic to evaluate where such deep (and expensive) sampling should take place: in soils of the Andisol order, particularly those with a histic epipedon (such as the Aquand Great Group), or those with mineralogy classes that are characterized by amorphous minerals or Al and Fe oxides. Examples include the halloysitic, amorphic, and ferrihydritic mineralogy classes (Soil Survey Staff, 1999). This diagnostic is by no means flawless; while it would successfully identify the Hoko and Vailton series (ferrihydritic and amorphic classes, respectively), it would also identify the Tokul (amorphic, Aquic Vitrixerand) and Hoquiam series (ferrihydritic, Typic Fulvudand) as important to sample more deeply. Both the Tokul and Hoquiam sites carried 8-14% of total soil C below 1.0 m, and were well modeled by the inverse polynomial equation. However, for the objective of strategically sampling deep soil C at a landscape scale, some false positives (i.e., sites wrongly suspected of high deep-soil C content) are preferable to randomly located deep sampling and overlooking soils with large quantities of deep soil C.

Sampling of these volcanically influenced soils for C should endeavor to be as deep as possible, or until a restrictive layer is reached. Though the analyses here are limited to the top 2.5 m of the soil profile, our soil pit at the Vailton site reached 3.5 m. Total soil C for this site was 317 Mg ha^{-1} to 3.5 m, and 15% of this total was found below 2.5 m. If substantial adsorption of organic matter to amorphous or clay minerals had occurred in a soil, sampling to bedrock will provide the most accurate assessment of total soil C.

Deep soil sampling is both expensive and labor intensive (Harrison et al., 2003). However, the difficulty or expense of a method is no justification for inadequate alternatives (Harrison et al., 2011). Sampling protocols that call for many shallow soil pits could be improved by, at minimum, sampling more deeply in a targeted subset of soils that are more likely to have substantial C in subsurface (>0.5 m) layers. While we believe sampling to 1.0 m – and preferably to 2.5 m – provides a superior alternative to conventionally shallow soil sampling, we also recognize that it fails to capture a truly complete picture of total soil C. The development of methods to sample soil accurately down to bedrock would be a boon to biogeochemistry and soil science generally.

Conclusions

1. Rocky forest soils of the coastal Pacific Northwest have substantial total soil C below 0.5 m. When sampling depths were increased from 0.5 m to 1 m and 2.5 m, average total soil C increased by 20.9 and 47.5 Mg ha-1, respectively.

2. A nonlinear mixed effect model using an inverse first-degree polynomial was developed that adequately predicted total soil C to 2.5 m on a soil-specific basis given sampling to 1.0 m. Reasonable predictions were made for soils ranging from Entisols to Ultisols and across glacial, sedimentary, and volcanic parent materials; the exceptions were deep volcanic soils with noncrystalline minerals throughout the profile.

3. Andisols that feature noncrystalline mineralogy contain more soil C than other soil types particularly at depths below 1 m.

Soil sampling protocols in the coastal Pacific Northwest need to take these factors into account. How deep to sample soil depends on the objectives of the investigation and the soils in question. Shallow soil sampling (< 1m depth) will underestimate soil C stocks and will provide incomplete information on the magnitude and direction of soil C flux. Sampling protocols that call for a large number of shallow samples can be improved by adding a sub-set of deep soil samples targeted to soils where substantial C is likely to be stored in subsurface horizons. Sampling deeper in the soil profile is more expensive and time consuming but it will provide better estimates of ecosystem C budgets and fluxes, particularly for soils with andic properties.

TASK 3. SOIL NITROGEN ANALYSIS: POTENTIAL FOR SUSTAINED PRODUCTIVITY WITH ADDITIONAL BIOMASS REMOVAL FOR FEEDSTOCK

Objective

In the 1970's, it was broadly recognized that the biomass of traditional logging residue was going to be an important resource. Demand for energy and wood products was increasing due to the rise in the global population and the development of the non-industrialized world, while at the same time supply dwindled as the land base for production forestry shrank and fossil fuel energy reserves were being depleted (Kimmins, 1977; Sabourin et al., 1992; Wells and Jorgensen, 1979; White and Harvey, 1979). In a keynote paper delivered at a 1979 symposium on the impact of intensive harvesting on forest nutrient cycling, George Staebler predicted, "When [oil] gets to some ridiculous value, say $100 a barrel (and maybe I'm myopic to say $100 is ridiculous), I guarantee the last shreds of biomass will come out of the woods." Staebler pleaded with researches to investigate the value of retaining biomass on site so that it could be compared with the economic value of using biomass for fiber and energy (1979).

In recent years the increase in oil prices has reminded some of the 1970's energy crisis, but we haven't yet depleted the forests of biomass. However, harvesting forest residuals for energy production has become an increasingly regular practice, particularly in European countries attempting to meet Kyoto protocol requirements (Jacobson et al., 2000; Saarsalmi, 2010; Stupak et al., 2007). Biomass from forests is generally considered carbon neutral when used for energy because it is renewable and replanted trees sequester the CO_2 released by the utilization of the previous rotation. Furthermore, when biomass is used for energy it offsets the use of fossil fuels that are a net source for atmospheric CO_2 (Lattmore et al., 2009; Thiffault et al., 2011). New technologies for converting woody biomass from forests into liquid fuels also encourages accelerated biomass utilization. Liquid fuels from woody biomass have an advantage over corn-derived ethanol because they don't displace a potential food source (White, 2010). These recent regulatory and technological demands for forest biomass increase pressure on production forests. This rising demand may increase stress on soils and heighten concerns about sustainable productivity of plantation forestry (Evens, 1999; Fox, 2000). Conversely, actual production from intensively-managed forests has risen over time, as has the efficiency of forest product utilization (Howard, 2007).

The use of wood and other biomass for energy is projected to grow more than any other renewable energy source in the next two decades (U.S. Department of Energy, 2009). The largest source of available biomass for energy not currently being utilized in the US comes from forest product residues (White, 2010). With logging residue removal for bioenergy projected to increase we still have a very limited quantifiable understanding of the value of that material if it is left on site.

Managers considering biomass harvest will have to weigh the costs and the benefits of removing logging residues. The potential economic benefits of exporting logging residues will depend on developing bioenergy markets, government subsidies, and the cost of traditional energy sources, which are constantly fluctuating. The benefits in terms of reducing CO_2 emissions are being investigated by other researchers through forest product life-cycle assessments and carbon budgets (Puettmann et al., 2010). Therefore, this thesis will focus on a regional evaluation of the long-term productivity costs of intensifying forest harvests for biomass utilization.

With the development and increased use of mechanized whole-tree harvesting (WTH), a number of researchers immediately expressed concerns about the increased export of nutrients due not only to the higher total export of biomass from the site but also due to the higher nutrient concentration of branches and foliage that were traditionally left on site compared to stem wood (Benjamin, 2010; Kimmens, 1977; Malkonen, 1976; Marion, 1979; Sabourin et al., 1992, Wells and Jorgensen, 1979; White, 1974). In a review of European studies of different tree species Malkonen (1976) found that WTH removed 2-4 times as much N, 2-5 times as much P, 1.5-3.5 times as much K, and 1.5-2.5 times as much Ca as stem-only harvesting (SOH). Despite the increase in losses, a number of researchers using nutrient budget evaluations found that soil reserves, weathering, and atmospheric inputs were enough to replace most nutrients lost during WTHs over a standard rotation length, with the common exception being Ca (Boyle et al., 1973; Johnson, 1983; Turner, 1981; Weetman and Webber, 1972). Nitrogen is typically the most limiting nutrient in many forest systems (Fisher and Binkley, 2000), but it is not as frequently identified as a nutrient that is at risk of being depleted as a result of higher-intensity harvesting (Johnson 1983; Evans 1999). However, for a Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) stand in western Washington, Cole (1978) predicted N depletion following WTH would result in adverse growth impacts on an inceptisol soil.

In studies that looked at the growth of successive rotations following varying harvest intensities Bigger and Cole (1983) found increased harvest intensity only negatively impacted growth of two-year-old Douglas-fir seedlings on very low productivity sites, while other sites suffered no loss of production. Similarly, Jacobson et al. (2000) found a 5% and 6% reduction of growth in *Pinus sylvestris* and *Picea abies* respectively following WTH compared to SOH, which they attributed to reduced N supply. In contrast Saarsalmi et al. (2010) found no growth reduction in *Pinus sylvestris* 22 years following WTH compared to SOH. Re-growth one year following WTH and SOH was greater after WTH on two out of three sites reviewed by Mann et al. (1988). In a summary of 26 sites in North America, Powers et al. (2005)

found no significant difference in biomass accumulation as a result of different removal intensities after 10 years. However, in a related study in North Carolina, *Pinus taeda* stands on the lowest quality sites had an 18% volume reduction following WTH after only five years (Scott et al., 2004).

The high variability in observed response of forest growth to elevated levels of biomass removal has many potential causes, including 1) type of tree species, 2) total soil nutrient stores, 3) soil nutrient availability, 4) soil physical characteristics, 5) climate, 6) stand stocking and development, 7) method of site preparation, 8) rotation length and 9) harvesting practices, all of which can affect subsequent tree growth (Benjamin, 2010; Grigal, 2000; Johnson, 1983). While not all sites are likely susceptible to productivity loss following WTH, there is reason for concern in low-productivity stands, particularly those with existing nutrient deficiencies (Lattimore, 2009).

In addition to WTH, biomass demands may push managers toward shorter rotations. Shorter rotations put additional stress on a site because younger trees have a larger ratio of nutrient rich foliage, branches, and bark to low nutrient wood. The result is that WTH of a young stand exports more nutrients per unit of biomass than it would if the stand were harvested when it was more mature (Ranger et al., 1995; Wells and Jorgensen, 1979). In addition, while younger stands have lower total nutrient demand, they primarily utilize site reserves while older stands often depend more on internal cycling as a source of nutrients for growth. Researchers have suggested that the internal nutrient cycling of mature stands can actually lead to a net accumulation and higher reserves of site nutrients (Ranger et al., 1995; Turner, 1981; White and Harvey, 1979).

Due to high productivity and standing reserves of forests, the coastal Pacific Northwest (PNW) is one of the key areas in the US for potential energy production from biomass (White, 2010). In 2010, approximately 6 billion board feet of timber was harvested in Oregon and Washington (Warren, 2011). Associated with the region's timber harvests is approximately 6 million tons of utilizable biomass in the form of logging residue normally left on site during harvest (White, 2010). The main production species, Douglas-fir, is a good candidate for WTH because it has a relatively-high nutrient use efficiency compared to other forest plantation species in other regions (Marion, 1979). However, in much of the PNW tree growth is limited by N, which potentially increases the risk for N depletion and loss of productivity following WTH (Chapell et al., 1991; Miller et al., 1986; Turner, 1977). Stem-only harvests in the PNW should be sustainable (Fox, 2000; Talbert and Marshall, 2005) due to low nutrient removals, but there is evidence that WTH of Douglas-fir on poorer quality sites may lead to a loss in productivity in some cases (Bigger and Cole, 1983; Compton and Cole, 1991). Compared to the tropics, much of Europe, or the Eastern U.S., N inputs through atmospheric deposition and N-fixation are low in the PNW (Bormann et al., 1989). Furthermore, the ability of Douglas-fir to grow under poor nutrient conditions coupled with continued enhancements in tree stock and silviculture may mask the decline of site N stores until after significant depletion has occurred (Lattimore et al., 2009).

Regional risk assessment models of nutrient depletion following whole-tree and conventional harvesting are necessary for land managers to make informed decisions about resource utilization (Sollins et al., 1983). Methods that have been utilized to assess the risk of WTH on productivity are costly, time consuming, and typically site-specific. These methods generally fall into two categories: 1) nutrient balance assessments, or 2) assessment of soil nutrient stores (Hansen et al., 2007). Nutrient balance assessment methods involve inventorying inputs and outputs of a particular nutrient to a system including harvesting export under different measured or simulated intensities. If inputs are equal to or exceed outputs then nutrient depletion is not a concern, and if outputs exceed inputs you can divide the total accessible soil pool of the nutrient in question by the difference to estimate the number of rotations that can be supported by the site. The limitations of this method are that it is site specific, inputs and outputs are difficult to measure and may fluctuate with stand development, and it is very difficult to assess all input and output mechanisms as shown by so called "occult inputs" of N (Binkley et al., 2000). The problem of stand development has been addressed using chronosequence studies, but these studies also come with inherent problems because of site variation (Johnson and Curtis, 2001).

Soil nutrient store assessment methods involve sampling nutrient pools at fixed points in time. For example, soil nutrient pools could be sampled before and after harvesting of different intensities to determine if there are reductions and whether or not any reductions are different by treatment. To account for differences in stand development this method should be employed over a full rotation or longer (Hansen et al., 2007). Soil nutrient store studies are also site specific, and they only show problems after nutrient depletion has already occurred.

Recently attempts have been made to produce region-wide predictive models using geospatial analysis (Akselsson et al., 2007; Kimsey et al., 2011). These models utilize regional soil surveys, geological surveys, climate data, and topography to identify areas with high risk of nutrient depletion following intensive harvesting. These models can be very powerful tools for land managers because they are easy to use and identify areas at high risk for nutrient depletion without any field sampling. The shortcomings of these models are that they lack precision; they are limited to data that are included in survey maps and the substantial inaccuracy inherent in such maps.

Evans (1999) proposed an alternative risk assessment metric developed to evaluate "narrow-sense" sustainability, which has potential to be applied on a regional scale. Evans (1999) described sustainability in the "narrow-sense" as the ability to grow trees on the same site, rotation after rotation, indefinitely without losing productivity. Evans (1999) promoted a "stability ratio" as one way for evaluating forest plantation sustainability. The stability ratio (SR) is the amount of a given nutrient removed by harvesting relative to the total site store of that nutrient. A stability ratio of 0.1 for N would mean removing the equivalent of 10% of the site store of N at harvest. Evans (1999) further suggests that an SR less than 0.1 posed little or no risk to long-term productivity. An SR greater than 0.3 potentially represents a significant risk to productivity but the realization of nutrient depletion or productivity decline may not become evident immediately, and an SR greater that 0.5 will likely result in significant and immediate site productivity decline (Evans 1999). This simple risk assessment was applied to simulated harvests of different intensities of 68 intensively managed Douglas-fir stands across the coastal PNW region. The objectives of this study are to present an easily replicated site-specific method of assessing the sustainability of SOH and WTH using the SR and apply that method to 68 representative sites within the Douglas-fir regional of the coastal PNW to identify region patterns in sustainable harvesting.

Methodology

Beginning in 2008 the Stand Management Cooperative (SMC), a research cooperative of universities, government agencies, and private timber companies, began installing paired-tree fertilizer trials in 15-30 year old Douglas-fir plantations in western Oregon, Washington, and British Columbia (Littke et al. 2011). The objectives of that study were primarily to understand relationships between site factors and productivity, and to potentially predict sites with high likelihood of response to fertilization. The 68 sites cover a range of latitudes and site conditions representative of Douglas-fir plantations in the region (Littke et al., 2011). At each plot, 19-40 Douglas-fir trees from the dominant or co-dominant canopy class were selected. For all study trees the diameter at breast height (DBH), average breast height age, height, and height to live crown were measured. A 0.0078 ha circular plot was established centered on each of the study trees and the diameter of all trees within that plot were also measured (K. Littke personal communication). Soil characteristics including total site N were measured and calculated as described by Littke et al. (2011).

ORGANON-SMC, an individual tree computer growth model specifically developed from stands within the same region as the study sites, was used to simulate the future growth of the stand at each plot (Hann, 2011). Given basic tree measurement data and site information ORGANON simulates the growth of a stand in five year intervals using multiple species specific growth models developed from tree age, height, and DBH measurements collected throughout the coastal Pacific Northwest. Some tree parameters such as height and live crown ratio can be estimated by ORGANON from DBH using regional and species-specific allometric relationships. Hann (2011) provides more details on inputs required by ORGANON and how missing values are estimated. The stand level inputs for ORGANON used were Bruce (1981) site index for Douglas-fir, stand age, and breast height age.

Site index values were calculated for each of the 68 plots using King (1966) methods from the heights of the 10 largest DBH trees. King (1966) site index values were converted to Bruce (1981) site index values using an iterative method. Bruce (1981) site index was used because the SMC version of ORGANON is based on Bruce (1981) values.

Plots with 38-40 trees were separated into two equal and adjacent groups, all other plots had all trees in a single group resulting in the total area represented by each group ranging from 0.15-0.29 ha which provided a similar scale to the 0.20 ha plots used for developing the SMC ORGANON variant. After the growth simulation of individual trees the data from plots that had been split were merged.

The heights of individual Douglas-fir trees that were not measured were estimated using the allometric equation:

$$\ln(ht-1.5)=a+b(1/DBH)$$

Where ln=the natural logarithm, ht=height (m), and DBH=stem diameter (cm) at 1.5 m. The a and b terms were derived for each stand using all measured tree heights from that stand. For some plots, the slope of the function was very small owing to low variability of trees with measured heights. The small slopes resulted in unrealistic projections of height for very small diameter trees, so in these instances unmeasured heights were left blank for ORGANON to estimate using regionally and species specific equations built into the software. The heights of all species other than Douglas-fir were also estimated by ORGANON.

The ORGANON simulation was carried out in R statistical software using dynamic link libraries (dll) as described by Gould and Marshall (2011). The ORGANON dlls were edited and compiled using free FORTRAN software “gfortran” to be compatible with R statistical software. Running the simulation in R allowed for the customization of output data and allowed us to automate ORGANON input procedures so that the files for the 68 plots could be run without having to manually interface with ORGANON between runs. The R statistical software interface also allowed us to alter the ORGANON volume calculations so Bruce and DeMars (1974) total stem volume estimates for Douglas-fir were used rather than the default ORGANON stem volume estimates. In previous work throughout the same land base, Bruce and DeMars (1974) volume was found to be the best estimator of actual volume (E. Turnblom, personal communication). For all species other than Douglas-fir, the default ORGANON stem volume estimates were used with the log top diameter set to zero.

The biomass of tree components was estimates by first determining the biomass of the stemwood:

$$B_{stem}=1000*SG*SV$$

Where B_{stem} is the biomass of the stemwood (kg), 1000 is the mass in kg of one m^3 of water, SG is the specific gravity of Douglas-fir in the specific region containing the plot estimated with the regional map of Douglas-fir density in the western wood density survey (USDA, 1965), and SV is the Bruce and Demars (1974) stemwood volume (m^3). The estimated stemwood biomass was then divided by the stemwood ratio derived from Jenkens et al. (2003) biomass equations for softwood species to estimate the total aboveground biomass (TAB) for all trees. Finally, the TAB was used to estimate the biomass of all other tree components using Jenkins et al. (2003) equations for softwood species component biomass.

For species other than Douglas-fir, appropriate component biomass was estimated from Jenkins et al. (2003) equations adjusted for volume as described above, only different specific gravity values were used and the volume was estimated with the default volume model in ORGANON. The specific gravity used for hardwood species was an average of specific gravities of all hardwood species in the study with published average specific gravity values (Miles and Smith, 2009). The same method was applied to all softwoods not coded as Douglas-fir some species within the plots; furthermore, many species were already grouped in order to match the limited species ORGANON recognizes. Species other than Douglas-fir were a small component of most plots, making up on average less than 4% of the stems on a plot with a range from 0-18%, with only three plots having more than 15% of species other than Douglas-fir.

ORGANON simulated tree growth on a five-year interval so each plot was simulated to a stand age of 50-55 years. Estimates of total above-ground biomass and stemwood+stembark biomass of each tree were summed for each plot. The N content of Douglas-fir TAB and stemwood+stembark was estimated using equations from Augusto et al. (2000) at each plot. N export from the site was estimated assuming the removal of TAB of all trees to simulate WTH and total stemwood+stembark of Douglas-fir to simulate SOH. The estimated N exported for both simulated harvest intensities was expressed as a percent of total site N store (soil N to hardpan or 1m+forest floor N).

The N content of all other species were estimated using the same equations, which are specified for Douglas-fir. The lower precision involved in estimating N content of species other than Douglas-fir was deemed acceptable because they were only a small portion of the study trees. Furthermore, using Douglas-fir specific equations for all species should result in overall conservative estimates of the sum of N accumulated in the above ground part of trees because Douglas-fir has relatively-high N use efficiency (Augusto et al., 2000; Marion, 1979; Palviainen and Finer, 2011). Also, appropriate biomass and nutrient content equations do not exist for species in the region other than Douglas-fir.

Results

The stand stemwood volume estimates ranged from about 300 to 1130 m^3/ha with a mean of 830 m^3/ha for the 68 plots. Yield tables for managed Douglas-fir in the region predict a total stand volume of 297-1246 m^3/ha for 50-55 year-old unthinned stands planted at 300-400 trees/ha for the range of site index values in this study (Curtis et al., 1982). Normal yield tables from McArdle et al. (1949) suggest a lower range, but they were developed from unmanaged naturally-regenerating stands that had lower levels of productivity than modern intensively-managed Douglas-fir plantations.

The TAB estimated from stemwood volumes and the Jenkins et al. (2003) biomass equations ranged from 220 to 810 Mg/ha with a mean of 580 Mg/ha, and stemwood+stembark biomass estimates ranged from 160 to 620 Mg/ha with a mean of 430 Mg/ha. Figure SC-3.1 shows that the large range of TAB closely follows site index. Table SC-3.1 shows all biomass values from the literature of Douglas-fir stands between the ages of 45 and 60.

Estimated harvest exports of N from WTH ranged from 366 to 1218 kg/ha of N with a mean export of 886 kg/ha of N. Stem-only harvest N export estimates ranged from 165 to 737 kg/ha of N with a mean of 495 kg/ha of N. Table SC-3.1 compares the N export estimates of this study with those of similar aged stands in the literature.

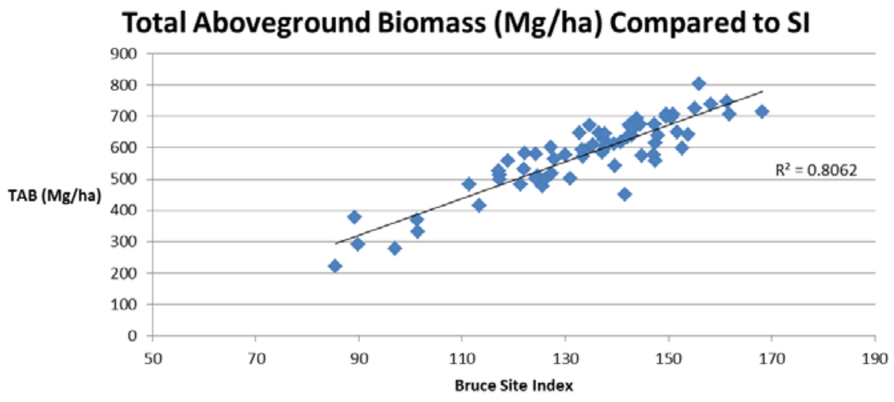

Figure SC-3.1. Total aboveground biomass increases with site index. Total aboveground biomass (TAB) is equal to the sum of all above ground parts of all trees in a plot. The relationship between TAB and Bruce (1981) 50-year site index for Douglas-fir is significant ($p<0.01$).

Table SC-3.1. Comparison of biomass and N content reported in the literature with that found in this study

Source	Stand Age	Trees/ha	Stand Volume (m^3/ha)	Stemwood Biomass (Mg/ha)	Total Aboveground Biomass (Mg/ha)	Total N content of trees (kg/ha)	50 yr Site Class	Specific Gravity
Ares et al., 2007	47	DF=303 WH=324 Total=627	DF=500 WH=414 Total=914	DF=182.2 WH=126.2 Total=308.4	DF=234.2 WH=158.6 Total=392.8	DF=375.9 WH=228.6 Total=604.5	High 2	DF=0.36 WH=0.30
Bigger and Cole, 1983	55			281* 134*	318 165	728 325		
Ponette et al., 2001	54	243	747	293	363	440		0.39
Homann et al., 1992	50	1100		275	216			
Ranger et al., 1995	60	312		307	418	694		
Heilman, 1961**	52	1000	339	148	216	361	4	0.43
Turner, 1980	50	1110		319	404	737		
Turner and Long, 1975	49	1070		178	234		High 4	
Range for this study (mean)	50-55	353-1280 (698)	302-1120 (829)	162-624 (429)	220-804 (577)	366-1220 (886)	Low 4-High 1 (High 2)	0.44-0.47 (0.45)***

Comparison of all studies found with biomass calculations of Douglas-fir stands between the ages of 45-60. When stand volume was included in the publication it was used to calculate the specific gravity by the equation described in Methods. Fifty-year site class is King (1966). DF is Douglas-fir, WH is western hemlock, and values in parenthesis are the mean values from this study. The two numbers from Bigger and Cole are from two different stands of Douglas-fir.

*includes bark

**natural stand

***based on USFS 2965

Figure FC-3.2 shows harvest export estimates relative to site nutrient stores and lines representing the critical SR values of 0.1 and 0.3. Eighteen of the 68 sites would exceed an SR of 0.1 with SOH by removing more than the equivalent of 10% of site N stores, whereas WTH is estimated to exceed an SR of 0.1 at 33 of 68 sites. A SR of 0.3 was exceeded at only one site under SOH (0.47), and at six sites under WTH (0.37, 0.40, 0.45, 0.45, 0.49, and 1.02) by our estimates. Table SC-3.2 summarizes SR estimates.

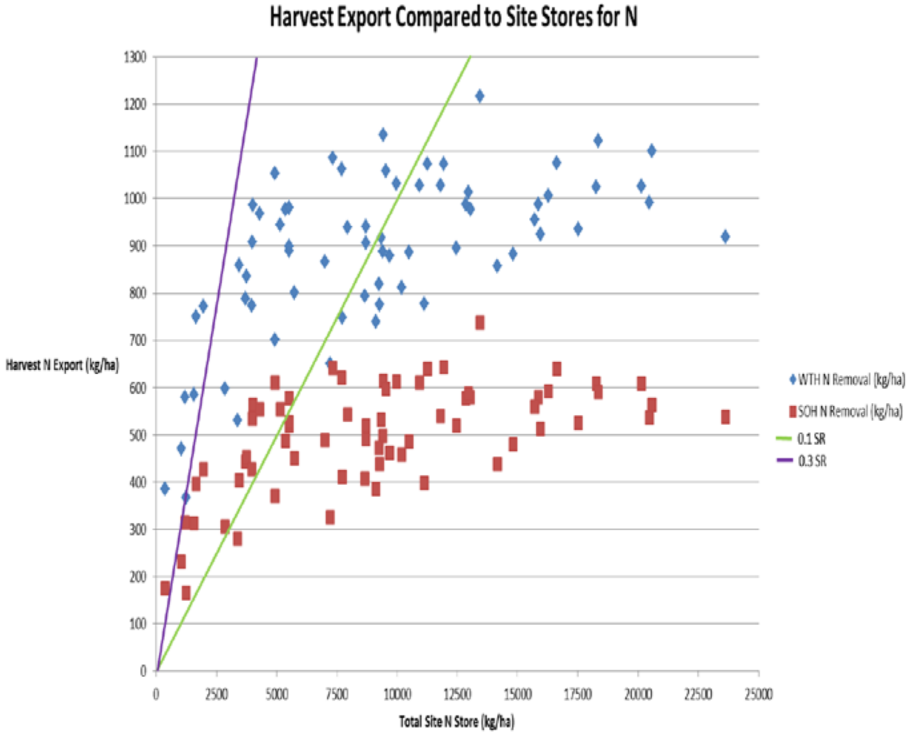

Figure SC-3.2. Harvest export of N under whole-tree harvesting (WTH) is assumed to be equivalent to total aboveground N content of all trees in a plot. The total site N store is the total N in the mineral soil to one meter depth (or impenetrable layer) plus the forest floor. If a marker is above or to the left of the 0.1 or 0.3 SR line it represents a site that exceeds that stability ratio.

Table SC-3.2. Summary of Stability Ratio findings for the region. A SR of 0.1 means the quantity of N removed at harvest is equivalent to 10% of the site N store (forest floor N+mineral soil N to a depth of 1m or impenetrable layer)

	Whole-tree Havesting	Stem-Only Havesting
# of sites w/SR ≥ 0.1	33	18
# of sites w/SR ≥0.3	6	1
% of sites w/SR ≥0.1	49	26
% of sites w/SR ≥0.3	9	1
Bellow Ground N (kg/ha) above which SR is predicted to be <0.1	9113	4127

Spatial patterns are apparent in the distribution of SRs throughout the coastal PNW based on these 68 sites. The only large area that shows a potential concern for nutrient depletion or productivity loss using SOH based on SR is in northern British Columbia (BC)(see Figure SC-3.3). With WTH, coastal BC, large parts of the Puget Sound and Olympic Peninsula, and a southern section of Oregon are at a potential elevated risk for N depletion based on our evaluation of SR. These regions correspond to younger glacial soils in areas covered by the Vashon Ice-Sheet during the last ice age, except for the area in southern Oregon.

A strong relationship between site N stores (soil+forest floor N) and estimated SR was also found (see Figure SC-3.4). This relationship permits prediction of SR for stands 50-55 years of age. Using the regressions in Figure SC-3.4 sites are predicted to have an SR less than 0.1 and be at low risk of N depletion or productivity loss if there is more than 9113 kg/ha of total belowground N for WTH and more than 4127 kg/ha total belowground N for SOH.

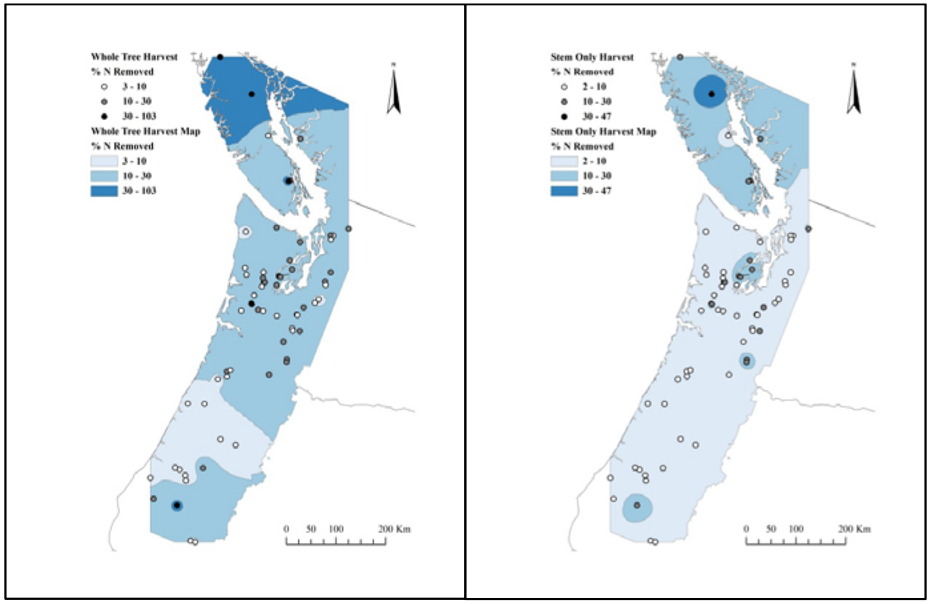

Figure SC-3.3. Spatial distribution of N removal expressed as a percentage of total site N store for whole-tree harvesting (WTH), left and stem-only harvesting (SOH), right. Percent N removed is the amount of nitrogen removed at harvest expressed as a percentage of the total site store of N and it is equivalent to the stability ratio (SR)*100. Dots indicate the location of sites. The maps do not take into account any variables other than the percent of N removed. Site N store is defined as the sum of soil N to a depth of 1 m (or impenetrable layer) and forest floor N.

The average and maximum resulting biomass for stemwood and TAB exceed all values found in other studies that evaluated stand level biomass and nutrient pools for similar aged trees. The minimum values estimated in this study, however, compare well with the lowest values found in other studies of Douglas-fir. Part of this discrepancy can be explained by the low site quality associated with most of

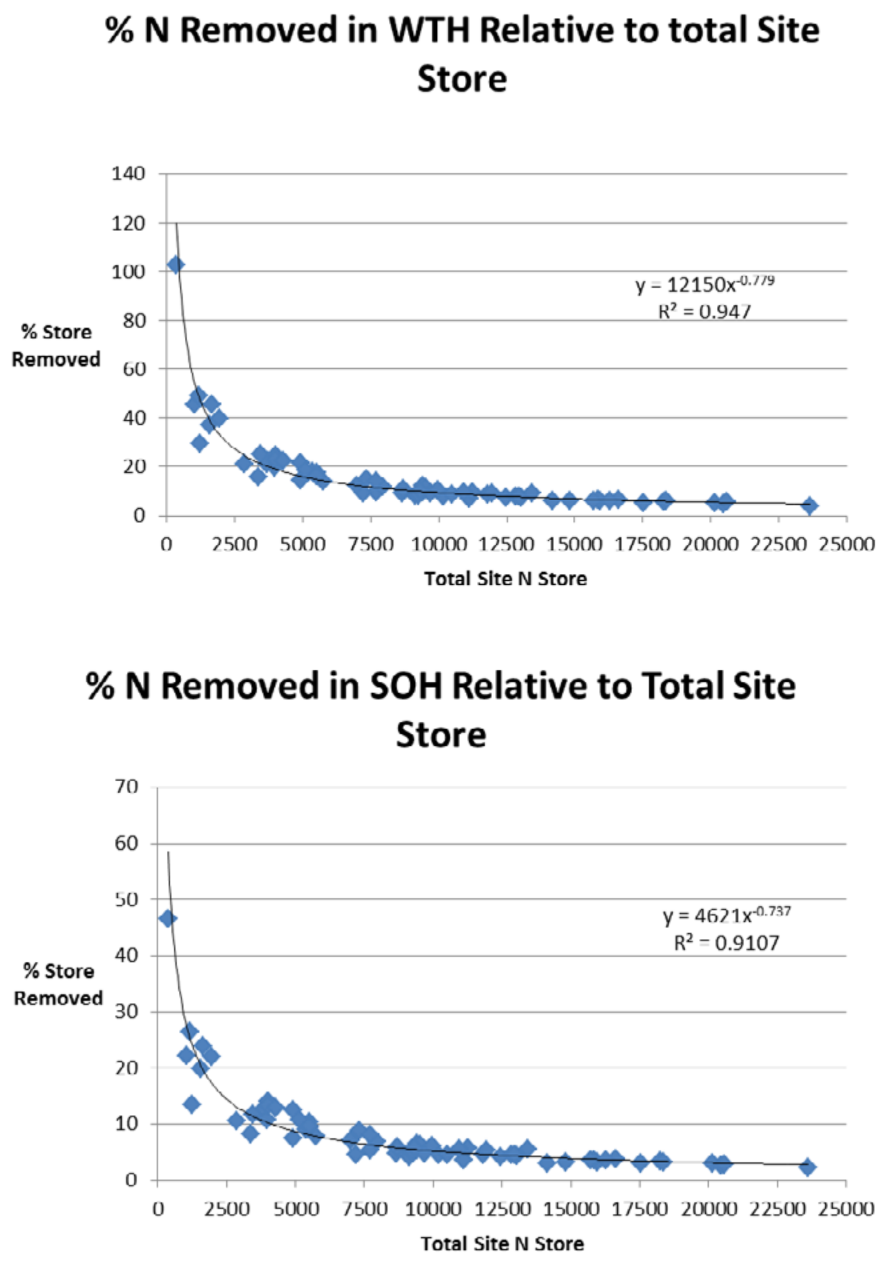

Figure SC-3.4. Percent of nitrogen removed vs. total site store for different harvest intensities. Percent Store Removed is equal to the stability ratio (SR)*100. Total site N store equals the total N in the mineral soil to one meter or impermiable layer plus the forest floor and is shown in units of kg/ha of N. The relationship of percent store removed to total site N store is significant for both stem-only harvesting (SOH) and whole-tree harvesting (WTH) with $p<0.01$ for both.

the published numbers. Although Homann et al. (1992), and Bigger and Cole (1983) do not report the site class of the stands in their studies, they are likely very low quality sites based on the soil type (inceptisol) and nutrient content reported (less than 3000 kg/ha of N). Turner (1980), is also likely on a low site class based on the soil N content reported, and the stand was also in Australia, outside the native Douglas-fir range which makes it difficult to compare to sites in the coastal PNW USA.

Both the Ranger et al. (1995) and Ponette (2001) studies were on stands growing in fertile soils, but there were no site indexes reported and the stands had very few trees/ha as a result of thinning, which would contribute to low total stand biomass at a specific age. The Ranger et al. (1995) and Ponette et al. (2001) studies were also conducted in France, outside of the natural range of Douglas-fir. The only in-region study on a higher productivity site was by Ares et al. (2007), and the stand has similar trees/ha, volume, and site index to the plot averages found in our simulations, yet the biomass estimates are much lower.

The discrepancy between our findings and Ares et al. (2007) can be explained by wood density. The specific gravity of stemwood in the Ares et al. (2007) stand (calculated by dividing the stemwood biomass by the volume of the stand and mass of water per unit of volume) is 0.36 for Douglas-fir and 0.30 for western hemlock. The western wood density survey (USDA, 1965) suggests the specific gravity of Douglas-fir for that area should be 0.43-0.45 and Miles and Smith (2009) report that the average wood density for Douglas-fir and western hemlock are 0.45 and 0.42 respectively. The smallest specific gravity found in the western wood density survey (USDA, 1965) for Douglas-fir was 0.33. Faster growing genetic stock and high occurrences of precipitation and fertilizer additions may account for the low specific gravity of the Ares. et al. (2007) site. However, Yoursy et al. (2011) measured the wood density of 32 year-old Douglas-fir with diverse genetics and a range of silvicultural treatments on site class I (King, 1966) stands in British Columbia and found specific gravity averaged 0.44 with a range of 0.34-0.54 which suggests that the range of values and mean specific gravity of the western wood density survey (1965) is reasonable for high quality sites, modern Douglas-fir strains, and intensive silvicultural practices. Gartner et al. (2002) also found specific gravity values close to 0.45 in 35 year old Douglas-fir from stands that had been thinned and fertilized. Kantavichai et al. (2010) found the mean specific gravity of 55 year-old Douglas-fir in western Washington treated with biosolids was 0.47 eleven years after application, while trees in the control group had a mean specific gravity of 0.51 for the same growth period. These other studies suggest the specific gravity calculated for the Ares et al. (2007) study is very low for the region. If the Douglas-fir in the Ares et al. (2007) study had the average specific gravity for the region (0.45), the stemwood biomass would be 411 kg/ha which is similar to our average estimated stemwood biomass of 429 kg/ha.

Another contributing factor to the lower biomass estimates in the literature may be the methodology employed by other studies to calculate biomass. All the studies listed in Table SC-3.1 used a regression model with DBH as the only input to estimate the biomass of the stand. Homann et al. (1992) used Gholtz et al. (1979) for regional equations, which Kantaviachai et al. (2010) found underestimated biomass by 45%. All the other studies except for Ares et al. (2007), developed site-specific regression models to estimate biomass based on a subset of destructively sampled trees that were selected by DBH or basal area. Sampling by DBH or basal area and developing regression equations with DBH as the only input for estimating biomass assumes all trees with the same DBH will have the same biomass and ignores height variability (Kantaviachai et al., 2010), as well as variability in specific gravity resulting from silvicultural treatments.

There is also the potential that utilizing specific gravity values from the western wood density survey (1965) may be erroneous because of changes to Douglas-fir production. Improved genetic stock, silviculture, and fertilizer additions all result in faster growing trees (Talbert and Marshall, 2005). Climate change has altered temperature and likely precipitation patterns over the same time (IPCC, 2007). Age, growth rate, temperature, precipitation, and water balance changes over time all affect specific gravity (Kantavichai et al., 2010). However, exactly how these variables interact and the potential net effect on specific gravity is unclear, particularly on a regional scale. As stated above, contemporary studies on the specific gravity of Douglas-fir throughout the coastal PNW, over a range of site classes, genetics, and silvicultural regimes support the range of values found in the western wood density survey (1965). Since there is no other regional assessment of Douglas-fir wood density the western wood density survey (1965) is the best available source for estimates of specific gravity on this scale.

In most studies, nitrogen content is estimated by multiplying the concentration of N in different sampled tree components by the biomass of each component. Similarly, our method of estimating kg/ha of N in TAB and stemwood was determined by a regression model that uses component biomass as an input (Augusto et al., 2000). Because our maximum biomass estimates are higher than those reported in the literature for reasons discussed above, our average and maximum N content was also higher than previously-reported estimates.

If we assume that a site with an SR less than 0.1 is at low risk for N depletion or loss of productivity, the majority of the coastal PNW is at a low risk of damaging site productivity with SOH. However, under WTH nearly half the plots in the study had SRs greater than 0.1 and 9% had SRs greater than 0.3, suggesting a possible decline in site productivity over time. It is important to note that SR's are a simplified method of assessing risk of productivity loss and have not been widely validated. A useful and conservative approach to SR is to consider further nutrient inventory on sites with SR greater than 0.1 before planning WTH for multiple rotations.

The spatial relationship of sites with higher SR seems to be related to soil age. Sites on less developed glacial soils are at a greater risk of losing productivity following intensive harvesting than sites on older soils in the region. The exception is in southern Oregon where there is a higher concentration of sites with elevated SR estimates that were not covered by glaciers during the last ice age. This area in southern Oregon does have a geological history distinct from the rest of the region with metamorphic and sedimentary parent material which may account for the anomalous concentration of higher SR estimates (spatial distribution patterns of SR estimates correspond to the strong relationship of SR with site stores since younger glacial soils have lower site N stores than older soils derived from sedimentary or igneous parent material). The correlation of SR with site stores of N also allows reasonable estimates of SR based on soil N content alone is significant because it gives land managers the power to predict the SR of a given stand planted at approximately 700 trees per hectare at harvest age 50-55 without having to run a growth and yield simulation.

When promoting SRs as a method to assess the risk of harvesting regimes, Evans (1999) emphasizes the difficulty of defining a “nutrient store.” In this study the nutrient selected was N because it is the most commonly limiting nutrient for forests in the region. We defined the store as the total N in the mineral soil to a depth of 1m or an impenetrable layer plus the forest floor. The nutrient store could have been made larger if we included the N in the standing trees, understory, and roots. Likewise the total site store could have been made smaller by estimating the amount of N mineralized during a rotation rather than total N. A larger store would have resulted in smaller SR estimates for harvest removal and a smaller store would have resulted in larger estimates.

Because the goal of the study was to determine nutrient depletion risk under a plantation system with continuous rotations it did not seem appropriate to include standing tree N as part of the site store. Rather we considered it a temporary pool that is always going to be removed. For the SOH treatment the N content of limbs and foliage may have been added to the store, but for almost all the sites it was a very small amount compared to the soil pool and some of it is likely to be rapidly mineralized and leached following harvest (Ares et al., 2007). Roots were not included as part of the site store because there are no equations in the literature for estimating root N content that are not stand specific. A rough calculation using Jenkins et al. (2003) biomass equations for coarse root biomass and the conservative assumption that roots have an N content proportional to their biomass compared to the whole tree results in no plots shifting below an SR of 0.1 for SOH or WTH. If roots and branches and foliage are added to the total site store for SOH, 5 plots that previously had SRs greater than 0.1 and the one SOH site that exceeded an SR of 0.3 fall below those thresholds.

For this study a rotation length of 50-55 years was used. Many land managers in the region currently utilize shorter rotations, and the SR for a shorter rotation will be smaller for the same site. However, as previously discussed the risk of site degradation may increase with repeated shorter rotation periods because young trees are disproportionately rich in nutrients. This being the case it is difficult to compare different rotation lengths. One possibility is to divide the harvest export of N by the age of the stand at harvest to get annual N removal. This method is useful for comparing absolute N export, but not for assessing SRs. An alternative would be to divide N export at harvest by the ratio of the stand age to 50 years and then dividing by the site store to get an SR at 50 year harvest estimate. This method is easy to use and allows for a direct comparison of SR at different harvest ages, but should be used cautiously because trees at different stages of development affect N cycling differently. For example, young stands are thought to consume soil stores of nutrients while mature stands may depend more on internal cycling to meet their nutrient demands and may actually contribute nutrients to a site over time (Ranger et al., 1995; Turner, 1981; White and Harvey, 1979).

Nutrient balance methods have been employed in the past in this region to estimate N accumulation over time and could prove useful when considering rotation length. If the accumulation of N over a 50-55 year rotation is less than the quantity of N removed by harvesting at the end of a rotation the harvest method may not be sustainable indefinitely on the site. Annual inputs from atmospheric deposition and precipitation in the region are generally less than 3 kg/ha/yr of N and range from 1-10 kg/ha/yr N (Johnson et al. 1982; Gessel and Cole 1973; Grier and McColl 1971). Plants with N fixing symbiants, most notably red alder, also contribute to annual inputs in the region. Red alder stands have been estimated to accumulate 41 to 321 kg/ha/yr of N (Tarrent and Miller, 1963; Newton et al., 1968; Johnson et al., 1982). Additional N is fixed by free-living N fixers and ranges from 0.01-12 kg/ha/yr of N in temperate forests (Reed et al., 2011). The only major loss of N besides harvest export on these sites is leaching. Gessel and Cole (1973) found annual leaching of N below the rooting zone was 0.6 kg/ha/yr, and Strahm et al. (2005) found annual leaching rates were as high as 4.5 kg/ha in an intact Douglas-fir stand on a higher quality site.

The largest percentage of species in any stand that is not Douglas-fir in this study is 18%, and if we assume that all trees that are not Douglas-fir are red alder and assume N accumulation by red alder is 181 kg/ha/yr, the middle of the range reported in the literature, then the maximum accumulation of N by red alder in any of these stand would be 33 kg/ha/yr. If we add the highest estimate of atmospheric input (10kg/ha/yr) and free living N fixation (12 kg/ha/yr), the total annual N input would be 55 kg/ha. With this high input rate we also assume a high leaching rate and subtract 4.5 kg/ha/yr of N for a total of 50.5 kg/ha/yr which multiplied by a 55 year rotation equals a total rotation length accumulation of 2700 kg/ha of N. Conversely, if we use the conservative estimate of N accumulation proposed by Gessel and Cole (1973) of 1.1 kg/ha/yr and subtract the low leaching rate of 0.6 kg/ha/yr of N and multiply by a 55 year rotation, it yields a low estimate of about 30 kg/ha of N accumulating during one rotation. Therefore, a reasonable estimated range of N accumulation during a 55 year rotation for the region based on published studies would be 30 to 2700 kg/ha. This large range demonstrates the difficulty of applying nutrient balance methods to a regional scale model.

The range of N accumulation estimated above using nutrient balance methods and N input and leaching values from previous studies does support the SR risk assessment used in this study. If ten percent of total site N stores are calculated for all sites in this study the range of those values is 38 to 2364 kg/ha which is similar to the range of N accumulation in the region estimated above to be 30 to 2700 kg/ha. This suggests that it is reasonable to expect the equivalent of 10% of a site's N store in this region to accumulate in 55 years meaning that harvest systems with an SR of 0.1 or less would be sustainable indefinitely, at least in terms of site N stores. There is, however a large amount of uncertainty and variability in N accumulation. A useful and conservative approach may be to consider further nutrient inventory on sites with SR greater than 0.1 before planning WTH for multiple rotations.

There are other sources of N loss following harvesting besides the removal of N in trees. Nitrification and leaching can increase following clear-cut harvesting in the region and the amount of N lost from the site depends largely on harvest methods and site preparation (Johnson and Curtis, 2001; Grigal, 2000). For example, losses of site N can also occur through soil erosion, which can increase following harvests. If slash is burned after SOH, a large portion of the N conserved using SOH over WTH can be lost. Likewise, broadcast burning can increase N loss if it is hot enough to volatilize N in the forest floor or mineral soil (Nambiar, 1996).

Adding fertilizer has been suggested as a means of preventing site degradation from increased nutrient export associated with WTH (Fox, 2000; Malkönen, 1976). In the coastal PNW, N fertilization with urea to increase growth is a widely established practice (Footen et al., 2009). Microbial uptake, leaching loss and volatilization results in fertilizer N uptake of 6-30% for Douglas-fir in the region (Mead, 2008). The low uptake efficiency of Douglas-fir suggests that fertilizer N addition may need to be greater than the N removed by harvest to maintain productivity on sites experiencing N depletion. Some researchers have concluded fertilizer N is transient and has no effects after 5-10 years (Binkley, 1986; Miller, 1988), but a recent study in western Washington Douglas-fir shows that some heavily fertilized stands showed increased productivity in the following rotation (Footen et al., 2009). Additions of N fertilizer will likely make it possible to maintain production on a site that has experienced N depletion, but whether it is an effective means of restoring or augmenting site N pools (and the necessary rates of fertilizer application) remains uncertain.

Conclusions

Using the ORGANON growth and yield simulator in combination with biomass and nutrient content equations we developed a method of estimating N export under different harvest intensities. This method is easily replicated for a specific site with basic stand information that most land managers already have or can acquire inexpensively. The method was applied to 68 Douglas-fir stands throughout the coastal PNW to provide a regional analysis of risk based on site stability ratios for N depletion and productivity loss associated with WTH and SOH. Conclusions from this study include:

- There is generally a low risk of N depletion or productivity loss with SOH, but nearly half the plots had some risk of N depletion or loss of productivity with WTH as indicated by an SR greater than 0.1.
- Stands with elevated risk of N depletion or productivity loss tend to be concentrated on young glacial soils of B.C and the Puget Sound region.
- Based on the strong relationship of total site N to SR, it is predicted that sites with greater than about 9000 and 4000 kg/ha N are at low risk of productivity loss or N depletion throughout the region with WTH and SOH respectively.
- Rotation length, harvesting methods, and site preparation can have effects on N losses and should be considered in conjunction with harvest intensity.
- Fertilization may be a viable means of maintaining site productivity and nutrient stores under WTH regimes, but it may necessary to fertilize at a higher rate or more often than currently practiced.

NARA OUTPUTS

Peer-Reviewed Journal Articles

Callesen, Ingeborg, Robert Harrison, Inge Stupak, Jeff Hatten, Karsten Raulund-Rasmussen, James Boyle, Nicholas Clarke andDarlene Zabowski. 2015. Carbon storage and nutrient mobilization from soil minerals by deep roots and rhizospheres. Forest Ecology and Management 10.1016/j.foreco.2015.08.019. Available at: http://soilslab.cfr.washington.edu/publications/Callesen-etal-2015.pdf

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Harrison, Rob, Kim Littke, Austin Himes, Erika Knight, Jason James, Christiana Dietzen, Eric Turnblom, Marcia Ciol, Scott Holub, Greg Ettl and Tom Terry. Nutrient Limitations on Intensive Biomass Production in PNW Douglas-fir. Presentation at Soil Science Society of America Annual Meeting. November 5th, 2014. Long Beach, CA. Oral Presentation.

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Michelsen-Correa, S. and R. Harrison. 2014. "Uptake efficiency and transport of applied nitrogen fertilizers in Douglas-fir forests of the Pacific Northwest". Soil Science Society of America Annual Meeting. Long Beach, CA. November.

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Michelsen-Correa, Stephani. 2014. Applications of Stables Isotope Biogeochemistry in Pollution Tracing Studies. Presented at Chang'an University. July, 25, 2014. Xi'an, China.

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Norton, M. 2014. "Douglas Fir Stump Decomposition: Modeling Carbon Residence Times". Northwest Advance Renewables Alliance Annual Meeting. Museum of Flight, Boeing Field, Seattle WA. September.

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Norton, Steven M. and James, J. Carbon and Nitrogen in Tree Farms of Washington and Oregon. Presented to Geologic Information Systems Class December, 9 2014. University of Washington, Seattle, WA

Norton, Steven M., Harrison, R and Zabowski, D. Douglas Fir Stump Decomposition: Modeling Carbon Residence Times. Presented at Northwest Advance Renewables Alliance Annual Meeting, September 16, 2014. Museum of Flight, Boeing Field, Seattle WA.

Norton, Steven M., Harrison, R and Zabowski, D. Residence Time of Carbon and Decomposition of Douglas fir Stumps. Presented to Seattle Central Visiting Class, February 18, 2015. University of Washington,

Norton, Steven M., Harrison, R. and Burt, E. Creating a Tree Farm Lifecycle Model: Using a Resistograph to Determine Carbon Pools in Forest Stumps. Presented at UW Soils and Soils Research overview to Seattle University Class, February 12, 2014. University of Washington, Seattle, WA

Norton, Steven M., Harrison, R. and Burt, E. Tree Farm Stump Lifecycle: How Stumps Relate to Northwest Renewables. Presented at Northwest Advanced Renewables Alliance Research at UW overview to the McCall Outdoor Science School – NARA Journal Class, March 6th, 2014. University of Washington, Seattle, WA

Norton, Steven M., Harrison, R. and Littke, K. Residence Time of Carbon and Decomposition of Douglas fir Stumps. Presented at Stand Management Cooperative Fall Meeting and Field Trip, September 10, 2014. Oregon State University, LaSells Stewart Center, Corvallis, OR.

Norton, Steven M., Harrison, R., Zabowski, D. and Littke, K. Douglas-Fir Stump Decomposition: Modeling Carbon Residence Times. Presented at University of Washington Graduate Student Symposium, March 6, 2015. University of Washington, Seattle, WA

Norton, Steven M., Harrison, R., Zabowski, D. and Littke, K. Residence Time of Carbon and Decomposition of Douglas fir Analysis of Presented at 509:

NARA OUTCOMES

Training

Name	Affiliation	Role	Contribution
Jason James	Univ of Washington	MS student	Field and lab, presentations, publications
Matt Norton	Univ of Washington	MS student	Field and lab, presentations, MS thesis
Erika Knight	Univ of Washington	MS student	Field and lab, presentations, MS thesis
Jean Vance	Univ of Washington	MS student	Field and lab, presentations, MS thesis
Paul Footen	Univ of Washington	MS student	Field and lab, presentations, MS thesis
Jason James (also MS student)	Univ of Washington	PhD student	Field and lab, presentations
Chrisitiana Dietzen	Univ of Washington	PhD student	Field and lab, presentations
Marcella Menegale	Univ of Washington	PhD student	Field and lab, presentations
Stephani Michelsel-Correa	Univ of Washington	PhD student	Field and lab, presentations
Dr. Kim Littke	Univ of Washington	Post-doc	Field and lab, presentations, publications
Dr. Warren Devine	Univ of Washington	Post-doc	Field and lab, presentations, publications
Dr. Marcia Ciol	Univ of Washington	Post-doc	Field and lab, presentations, publications
Luis Souza	Univ. Sao Paulo	(Brazilian Intern)	Field and lab
Rodolfo Bernardi	Univ. Sao Paulo	(Brazilian Intern)	Field and lab
Thiago Bonassi	Univ. Sao Paulo	(Brazilian Intern)	Field and lab
Eduardo Marques	Univ. Sao Paulo	(Brazilian Intern)	Field and lab
Nikolas Mateus	Univ. Sao Paulo	(Brazilian Intern)	Field and lab
Dr. Luiz Ribas	Univ Sao Paulo	(Brazil visiting Faculty)	Field and lab
Dr. Ling Jiang	Chang'An Univ	(China visiting Faculty)	Field and lab
Dr. Youfu Zhang	Chang'An Univ	(China visiting Faculty)	Field and lab

Research Problems Class – Research Presentations Stumps March, 13, 2015. University of Washington, Seattle

I have been working directly with Industry on intensively-managed production forestry for over 30 years, particularly through the PNW Stand Management Coopertative, which helps with research questions on sustainably and profitably managing forests for production. We have taken results from our work on the NARA project directly to industry partners, including nearly all of the major companies that are likely to provide feedstock for any bioenergy, biofuels and biochemical plants that might be developed in the region.

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