Soil Nutrient and Vegetation Response to Ecological Restoration in a Coastal Douglas-fir Plantation on Galiano Island, BC

by

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Supervisory Committee

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Abstract

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Much emphasis has been placed on the recovery and maintenance of biodiversity and ecosystem services. Although a number of studies have focused on the relationship between carbon sequestration and ecosystem dynamics, few have focused on the effects of management activities oriented towards biodiversity values on soil carbon and nitrogen pools. The dual goals of restoration for ecosystem structure and function versus restoration for soil carbon sequestration may not be mutually exclusive. This research evaluates the ability of restoration work to meet both of these goals using the restoration work done by the Galiano Conservancy Association in a Coastal Douglas-fir forest on Galiano Island, British Columbia as a case study. The restoration in District Lot 63 was successful in terms of increasing both floristic diversity and stand structure heterogeneity. Significant changes in soil carbon were observed in the forest floor, and significant changes in both soil carbon and nitrogen were observed in the top 15 cm of the mineral soil. As time from treatment increased, soil carbon and nitrogen approached, and in some cases surpassed, reference area levels. The results from this study indicate that the restoration on Galiano Island was successful in terms of increasing the biodiversity values of the stand and may have no large long-term effects on soil carbon or nitrogen pools.

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Chapter 1: Introduction

The practice of ecological restoration is growing. Ecological restoration is generally concerned with assisting in the recovery of healthy natural ecosystems. Much emphasis has been placed on the recovery and maintenance of biodiversity and ecosystem services. Although a number of studies have focused on the relationship between carbon sequestration and ecosystem dynamics, few have focused on the effects of management activities oriented towards biodiversity values on soil carbon and nitrogen pools. The dual goals of restoration for ecosystem structure and function vs. restoration for soil carbon sequestration may not be mutually exclusive. This research evaluates the ability of restoration work to meet both of these goals using the restoration work done by the Galiano Conservancy Association on Galiano Island, British Columbia as a case study.

The purpose of this report is:

a.) To review the current state of knowledge regarding carbon and nitrogen dynamics in forest ecosystems;

b.) To review relevant ecological concepts and theories as they apply to forest biodiversity;

c.) To assess how the restoration of second-growth forests contribute to the maintenance of biodiversity values; and

d.) To assess how the restoration of second-growth forests effects soil carbon and nitrogen pools.

The current state of knowledge regarding carbon and nitrogen dynamics in forest ecosystems is discussed in Chapter 2. Relevant ecological concepts and theories as they apply to forest biodiversity conservation and management, and potential uses of second‐growth forests for the conservation of biodiversity are discussed in Chapter 3. Important forest structural attributes are highlights in Chapter 4, and management strategies for recruiting said attributes are discussed in Chapter 5. Chapter 6 describes the case study site and Chapters 7 through 9 details this studies methods, analysis, and results respectively. Chapter 10 discusses the results of the study in the context of the literature review presented in Chapters 2 through 5.

Chapter 2: Terrestrial Carbon and Nitrogen

2.1 Climate Change

International scientists participating in the UN-sponsored Intergovernmental Panel on Climate Change (IPCC) have confirmed that the earth is warming (IPCC 2007). The primary agent of global warming is an increase in the global atmospheric concentration of greenhouse gases (GHGs), particularly carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_20). For example, the atmospheric concentration of CO_2 has risen by approximately 30 percent from 270 ppm to 382 ppm since 1750, which far exceeds the natural range of this gas over the past 650 000 years (IPCC 2007). The increase in atmospheric CO_2 is largely the result of human-induced land cover change and the burning of fossil fuels (Millard et al. 2007). Given that between 50% (Wilson and Hebda 2008) and 75% of the carbon in terrestrial ecosystems is stored in forests (Schlesinger 1997), carbon sequestration in forest ecosystems will play an important role in the mitigation of global warming.

2.2 Terrestrial Carbon in Canada's Forests

Terrestrial systems absorb, cycle, store and release carbon through photosynthesis, respiration, decomposition, and burning. As a result of these processes, forests naturally remove carbon dioxide from the atmosphere and store large quantities of carbon in there biomass and soils. Forests in British Columbia (BC) have some of the highest carbon stores in Canada averaging 311 tons per hectare; coastal forests have the highest carbon

stores in BC containing between 600 and 1300 tonnes per hectare (Wilson and Hebda 2008). The National Forest Inventory has also confirmed that the coastal temperate forests of BC have the highest biomass C density in Canada (Power and Gillis 2006). That said, managed harvest rotation cycles in the Pacific Northwest are typically much shorter then the pre-logging natural disturbance return interval, creating a net release of carbon to the atmosphere when old-growth forests are transitioned into managed forests (Harmon et al. 1990). Moreover, much of the old-growth coastal temperate forests in the Pacific Northwest have been logged. In fact, all old-growth forest types that are either dominated or co-dominated by Douglas-fir within the CDF are on the province's list of rare and endangered ecosystems (Flynn 1999). Approximately 48% of the Coastal Douglas-fir biogeoclimatic zone (CDF) in BC has been converted from forests to other land cover types since settlement (Wilson and Hebda 2008). Less then 1% of the CDF zone remains as mature or old growth stands, all intact remnants are in small fragments, and there are very few high quality stands of these types left (Pojar et al. 2004) The appropriate stewardship of British Columbia's remaining forests is a key component for addressing climate change.

One climate change impact model projects a rapid expansion of the CDF zone, with a 336 % increase by 2085 (Hamann and Wang 2006). Furthermore, paleoecological studies confirm that the CDF zone was much larger under warmer, drier historical climactic conditions (Brown and Hebda 2002). These studies imply that the CDF zone is particularly resilient to the effects of climate change and will likely play an important role as ecosystems adapt to climate change over time. What little remains of the old-growth

forests in the CDF should be protected to provide genetic pools from which these ecosystems can expand (Wilson and Hebda 2008). However, given their current limited distribution, it is also imperative to consider the ecological restoration of degraded forested areas within this zone as a mechanism to increase the resiliency of BC's ecosystems to climate change as well as to mitigate GHG emissions.

2.3 Forest Carbon Stewardship

Forest carbon stewardship requires tracking carbon pools and quantifying changes in carbon stores resulting from management activities. Indicators of stand level carbon dynamics are the C content of biomass, soil C pools, growth and decomposition rates, and C transfers between biomass and soil pools (Kurz and Apps 1999). When a mature forest is logged, carbon is rapidly released to the atmosphere as organic material decomposes (Wilson and Hebda 2008). There is a net release of carbon to the atmosphere in forest ecosystems until trees grow large enough to take up more carbon in there living material than is released from the soil. The amount of carbon removed from the atmosphere by a forest has a complex relationship with the successional stage or stand age structure (Law et al. 2001). The transition between a forest acting as a carbon source or a carbon sink occurs ~20-30 years after regeneration (Chen et al. 2004). Also, forests that contain nitrogen-fixing trees typically accumulate more carbon in soils than similar forests that do not have nitrogen-fixing trees (Resh et al. 2002). This is attributed to either greater accumulation of recently fixed carbon or reduced decomposition of older soil carbon (Resh et al. 2002). However, the amount of carbon that exists in any pool at a given point in time is dependant on the disturbance history of the site (Trofymow et al.

2008). Different harvesting techniques have different impacts on the various carbon pools.

2.4 Terrestrial Carbon Pools

The IPCC (2003) breaks down terrestrial carbon into five pools: above-ground biomass, below-ground biomass, dead wood, litter, and soil organic matter. The Carbon Budget Model for the Canadian Forest Sector (CBM-CFS3) has broken down the terrestrial carbon pools in a different way (Figure 1). The finer resolution of terrestrial carbon pools in CBM-CFS3 is better suited for ecological studies because it allows for the improved representation of important ecological processes and the comparison of predictions with field measurements (Kurz et al. 2008).

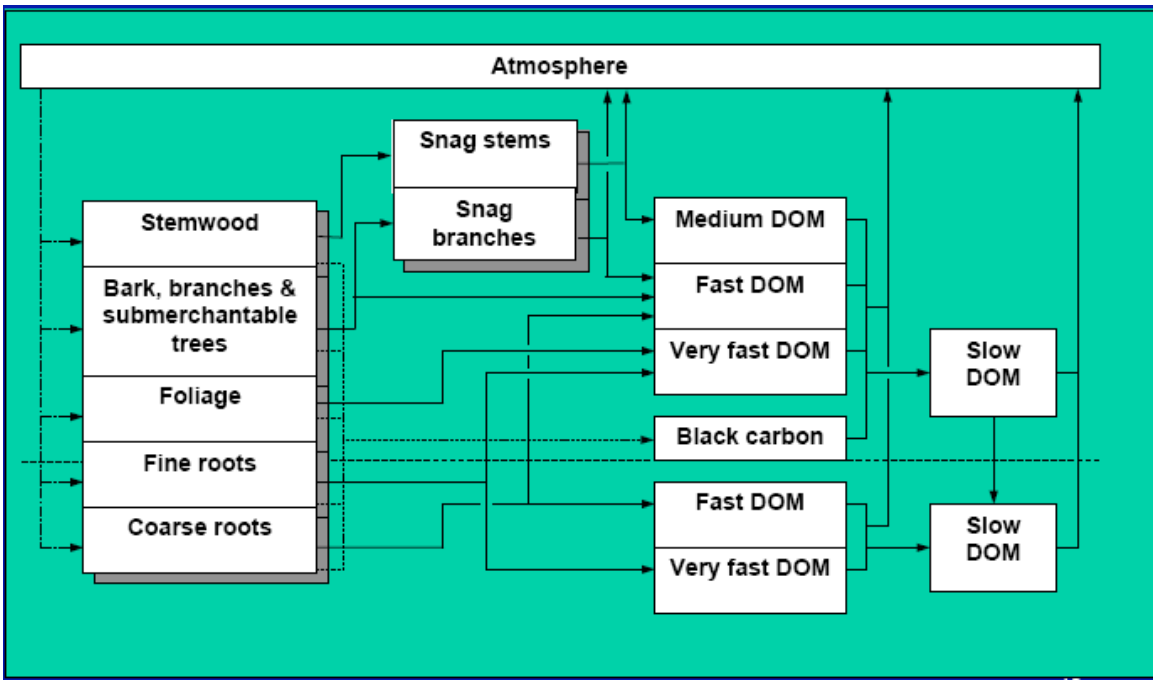

Figure 1: Carbon pool structure of the CBM-CFS3 (Kurz 2005)

The pools on the left hand side of Figure 1 represent biomass, the pools in the middle represent dead organic matter (DOM) that is variable in nature, and the pools on the far right represent stable DOM (Kurz 2005). The DOM pools are categorized according to the type of material they contain and their anticipated rate of decay. The C pools of a stand change due to growth, biomass turnover, litterfall, transfer, and decomposition. Within CBM-CFS3, simulation of turnover and disturbance processes causes the transfer of C from biomass pools to DOM pools and the loss of C from the ecosystem as gaseous emissions (Kurz et al. 2008). C is transferred between DOM pools and from DOM pools to the atmosphere through decay, transfer, and disturbance (Kurz et al. 2008). Carbon that remains in the ecosystem eventually ends up in the below-ground slow DOM pool (Kurz et al. 2008). Therefore, long-term C sequestration is dependant on the amount of soil organic matter (SOM) stored in the mineral soil horizon.

2.5 Above Ground Biomass, Litterfall, and Decomposition

The shrub and herbaceous layers are the primary component of floristic species diversity in the coastal forests of British Columbia (He and Barclay 2000). Understory species are an important component of these forests because they influence seedling establishment and growth, provide habitat and food for wildlife, and affect nutrient cycling and regeneration (He and Barclay 2000). The composition and development of understory species is strongly influenced by the overstory species growth stage. Above-ground productivity in the understory is greatest prior to crown closure. For example, salal (*Gaultheria shallon*), a dominant understory species in coastal Douglas-fir forests,

experiences a decline in productivity that corresponds with the increase in stand age and associated increase in canopy biomass (Long and Turner 1974). According to a study done by Turner and Long (1975), in a coastal Douglas-fir forest that is 22 years of age the understory is 5.5% of the above-ground biomass, while at 73 years of age the understory is less then 1% of the above-ground biomass. This change in biomass distribution is a function of the increasing quantity of wood (stem biomass) in the stand with increasing age.

However, there is also a qualitative change in the understory in relation to the forest stand maturing; the vascular understory decreases and the moss understory increases. In one study, a forest at 22 years of age contained mosses that made up 4% of the understory biomass and 0.4% of the understory above ground productivity, whereas at 73 years of age mosses represented 55% of the understory biomass and 82% of the understory above ground productivity (Turner and Long 1975). Furthermore, the understory is more important in terms of total stand productivity than total stand biomass; the understory of the 22-year-old stand accounted for 17% of the total productivity and the understory of the 73 year old stand provided 10% of the total productivity but the understory never made up more than 5.5% of the total above ground biomass (Turner and Long 1975).

Another consideration is that the changing quality of the understory litter from woody perennials to mosses likely affects decomposition rates. One must not only consider organic debris that falls on the forest floor, but also the portion of the moss mat that is already partially incorporated into the forest floor (Turner and Long 1975). This

discussion highlights an important point: the understory is more important in terms of productivity and organic matter inputs into the soil than the distribution of biomass indicates (Turner and Long 1975).

The rate of organic matter decomposition in forest ecosystems is affected by both the constituents of the litter and the location of the residues. As a stand ages, the quantity and quality of organic matter from both trees and the understory changes. For example, young trees produce more leaf litter then older trees, yet total tree litter increases with age as a result of increased inputs of wood in the form of twigs, branches and stems (Edmonds 1978). The addition of more wood to the forest floor may retard decomposition because it has a higher C/N ratio, and a higher concentration of lignin. In fact, an increase in lignin is often associated with an increase in the C/N ratio. There are two primary reasons why the C/N ratio is important (Brady and Weil 2002). First, intense competition among soil microorganisms for available soil nitrogen occurs when residues having a high C/N ratio are added to the soil; if the C/N ratio exceeds 25:1, soil microbes have to scavenge the soil solution which depletes the supply of soluble nitrogen. This is because most soil organisms need approximately 1 g of N for every 24 g of C in soil organic matter (Brady and Weil 2002). Secondly, the C/N ratio helps determine decay rate and the rate at which nitrogen is available for higher plants; the decay of organic matter can be slowed if nitrogen is absent or unavailable because this situation dictates lower levels of microbial activity and microbial activity plays a large role in nutrient availability for primary productivity. In short, mineralization and immobilization of nitrogen occur simultaneously in soil, and the net effect of these processes depends primarily on the C/N

ratio in the organic matter undergoing decomposition (Brady and Weil 2002). Nitrogen in and of itself is important because it influences ecosystems more then any other element and it is often the limiting element in natural ecosystems (Brady and Weil 2002).

The location of residues is important because surface litter decomposition is generally slower and more variable than residues incorporated into the soil via root deposition and faunal activity. This is because surface litter is more susceptible to extremes in temperature and moisture, nutrient elements that have been mineralized are more vulnerable to runoff or volatization, and surface litter is less accessible to most soil organisms except larger fauna such as earthworms and fungal mycelium (Brady and Weil 2002). Also, if the surface litter is low in nitrogen, fungi may transfer nitrogen from the soil through their hyphae to narrow the C/N ratio of the litter thereby depleting the nitrogen in the soil.

The quantity and quality of organic matter from the over- and understories also changes after thinning. Decomposition thinned stems and detritus in younger forests is higher than stems in mature forests because the sapwood volume is relatively greater in woody detritus from young trees than from old trees (Harmon et al. 1990), and leaf litter decay is greater in younger stands where temperature and moisture conditions are more favorable (Edmonds 1978). That said, the decay rate of any individual piece of dead wood is a function of substrate quality, microbial activity, air temperature, and available moisture (Yin 1999).

There are also large differences in biomass production and tissue nutrient concentrations between different tree species, which affects soil properties such as pH, nutrient cycling, and soil biota (Binkley and Giardina 1998). For instance, red alder (*Alnus rubra*), which is a common early successional species in BC, is a nitrogen fixer. Red alder are associated with *Frankia,* which are a genus of Actinomycetes that convert inert atmospheric dinitrogen gas to nitrogen-containing organic compounds. This symbiotic relationship lets alder colonize infertile soils and or highly disturbed soils which are often inhospitable to other plant species because of poor nutrient (mainly N) conditions that limit plant growth. Over time, alder builds the nitrogen capital of the soil through leaf litter and root exudates which in turn makes the site more hospitable to other species (Brady and Weil 2002). Furthermore, consistent large effects of nitrogen-fixing trees on soil carbon storage have been documented. However, it is not known if the higher carbon storage is a result of greater carbon inputs or reduced carbon outputs (Resh et al. 2002).

2.6 Soil Carbon Dynamics

Globally, soils contain twice as much carbon as vegetation or the atmosphere, and changes in soil carbon content can have significant impacts on the global carbon budget (Bellamy et al. 2005). Soil organic matter (SOM) contains approximately 1500 Pg C to a depth of 1 m (Eswaran et al. 2000). Soil organic matter is composed of accumulated, decaying plant and animal matter on or in the soil; this includes everything from carcasses from recently deceased soil invertebrates to millennia-old humified plant material (Janzen 2006). The principle source of soil organic matter is plant tissue; approximately 42% of dried plant tissue is carbon (Brady and Weil 2002). Soil carbon is

part of a dynamic cycle; the carbon content of a soil at any given time is a function of the rates of addition from photosynthetic C plant growth versus the rates of removal from decomposition, leaching, and other soil processes. Soil c sequestration may be managed to help slow the rise of atmospheric CO_2. This is just one of SOM benefits though because soil containing more organic matter is more productive and has persistent benefits through its physical effects on soil structure and moisture retention, and chemical effects such as ion exchange (Janzen 2006).

There are three mechanisms by which biomass are transferred to DOM: litterfall, mortality associated with stand breakup in the 'overmature' stand growth phase, and disturbance (Kurz and Apps 1999). Litterfall is composed of all annual transfers of biomass to dead organic matter (DOM) C pools. There are five main pathways through which C enters the soil; from litter, by transfer from roots to mycorrhizal fungi, directly into the soil as mycorrhizodeposits or secreted enzymes, and through grazing by soil fauna (Millard et al. 2007). Notably, the speed at which organic matter moves through these pathways is variable. For example, foliage has a turnover rate of around once per year, whereas fine root turnover occurs about three times per year (Lukac et al. 2003).

Soil fungi in coniferous forests consist mainly of molds and mushroom fungi. Molds are a filamentous fungi that play an important role in soil organic matter decomposition and are widely distributed (Brady and Weil 2002). Mushroom fungi are associates with trees where there is lots of moisture and organic residue and are extremely important in woody tissue decomposition (Brady and Weil 2002). Below ground fungal inputs of

biomass are considerable; particularly the fine root and mycorrhizal fungal component of the biomass pool (McDowell et al. 2001). In a study of Douglas-fir forests in western Oregon (Fogel and Hunt 1983), total mycorrhizal and saprotrophic hyphal biomass was estimated to be ca. 660 g m^{-2}. Furthermore, the C in extrametrical mycelium and associated bacteria form a carbon pool with a fast turnover rate (Godbold et al. 2006). The combination of high turnover rates and large biomass associated with mycorrhizal hyphae may prove to be a fundamental mechanism for the transfer of root derived C to soil C (Godbold et al. 2006). Furthermore, both ectomycorrhizal and arbuscular mycorrhizal fungi contain relatively recalcitrant compounds, chitin and glomalin respectively, which remain in the soil following fungal senescence (Treseder et al. 2007). Therefore, an increase in mycorrhizal hyphal biomass should increase C sequestration in forest soils

Rhizodeposition occurs when trees release labile C through the sloughing of root cells, and the release of low molecular mass exudates and organic secretions from their roots to adjacent soil (Phillips and Fahey 2006). This release causes alterations of the physical, chemical, and biological characteristics of the soil around roots and is known as the rhizosphere effect (Phillips and Fahey 2006). For most free living soil microbes, C substrates such as sugars, organic acids and amino acids are limiting factors for growth which explains why generally there is greater microbial activity in the rhizosphere in comparison to bulk soil (Millard et al. 2007). Furthermore, rhizodeposits can also act as primers for the degradation of existing SOM (Millard et al. 2007, Dijkstra and Cheng 2007). Thus an increase in C inputs into the soil in the rhizosphere will not necessarily lead to increased soil C storage, instead, enhanced soil organic carbon deposition may

result in a net soil carbon loss (Dijkstra and Cheng 2007). However, the rhizosphere effect is only applicable to the area immediately surrounding the roots.

Humus is a key component of the forest floor in that it provides nutrients, and contributes to soil structure and moisture retention (Prescott et al. 2000a). During the early stages of the transition from litter to humus, there is a rapid loss of solubles and cellulose, carbon is relatively available whereas nutrients are limiting, and there is immobilization of the limiting nutrient which is usually nitrogen (Prescott et al. 2000a). Once the content has stabilized and decay has slowed it can be considered humus. Humus is composed of the recalcitrant products of decomposition and is chemically stabilized (Prescott et al. 2000). Humus formation is thought to involve the microbial modification of lignin, the condensation of proteins into humus precursors, and the complexing into humus molecules of complex structures (Prescott et al. 2000a). “Relative to the original plant material, humus is low in carbohydrates (cellulose, hemicellulose), high in large polyphenolic molecules (usually measured as the acid-insoluble fraction or lignin component), and high in N. Most of the N in humus is bound in complex molecules of undetermined composition, and so can be considered to be immobilized and essentially unavailable to plants and most microorganism” (Prescott et al. 2000a).

There are three primary factors that control the rate of humus formation; climate (temperature and moisture conditions), chemical and physical characteristics of the litter (particularly lignin and phenolics), and the abundance and composition of soil microbial and faunal communities (Prescott et al. 2000a). The chemical characteristic of the litter is

the most significant factor influencing the proportion that becomes humus. Most of the C from the leaf litter component of the forest floor is rapidly respired by soil microbes and only the recalcitrant compounds are eventually stored as soil organic matter (Godbold et al. 2006). Thus it is the water soluble C content (i.e. sugars) that likely determine the initial litter decomposition rate whereas it is likely the secondary compounds (i.e. lignin, polyphenols, tannins) that determine later decomposition rates (Millard et al. 2007). Litter, such as conifer needles, with high levels of acid insoluble material (AIS) or 'lignin' inhibit decomposition through their resistance to enzymatic decomposition as well as their contribution to toughness which limits the accessibility of microbes to potential substrates (Prescott et al. 2000a). As the decay process continues there is a net loss of lignin and net N mineralization (Prescott et al. 2000a). The availability of carbon is thought to be more limiting during the later stages of decay after the readily metabolized C in the litter has been used up and the remaining C has been transformed into recalcitrant forms (Prescott et al. 2000a). The amount of humus that accumulates is dependant on the amount of the original litter mass that remains at the point at which the material becomes humus and decomposition slows (Prescott et al. 2000a). Other phenolic compounds such as phenolic acids, tannins, quinines, and humic and fluvic acids, contribute materials that are used in aromatic structures which make up the bulk of soil organic matter (Gallet and Lebreton 1995).

There are two main types of humus. Mor humus consists of surface accumulations and can be further subdivided into three distinct layers; the fresh litter layer (L), the partially decomposed but distinguishable Formultningsskiktet (F), and the relatively homogeneous

transformed humus (H). Mor humus forms are primarily the result of fungal decomposition which results in incomplete decomposition and nutrient immobilization; in other words, organic matter is not completely mineralized into CO_2 and nutrients (Prescott et al. 2000a). Mull humus is composed of organic matter that has been changed through soil fauna activity, bacterial decomposition, and mixed with mineral soil. Mull humus is the result of more complete decomposition and is characterized by greater nutrient availability (Prescott et al. 2000a). A third type of humus is moder. Moders are an intermediate form between mulls and mors, and are distinguishable because they exhibit characteristics of both main humus types (Prescott et al. 2000a). The type of humus that is formed depends on a combination of ecological factors, primarily climate, vegetation, and parent material.

Decomposition rates, leaching, and other soil processes are sensitive to changes in land use, climate, and other variables (Bellamy et al. 2005). For example, climate mediated variables such as soil temperature and soil moisture are limiting factors for soil microbes and thus influence rates of organic matter decomposition in soils. In the context of global warming, changes in soil moisture associated with changing precipitation and evapo-transpiration patterns, as well as changes in atmospheric CO_2 and nitrogen deposition, are likely to interact with changes in soil temperature in complicated ways, and the magnitude of these changes is unknown (Bellamy et al. 2005)

Other important considerations include the presence or absence of earthworms, soil pH, and soil texture. Earthworms have the ability to dramatically alter C content in forest

soils; soils with high earthworm activity have low soil organic matter and high pH relative to soils with low earthworm activity (Phillips and Fahey 2006). Earthworms are important macro-animals because they eat large amounts of detrus, soil organic matter, and microorganisms, they enhance fertility, and they create macropores. pH, a measure of soil acidity, is considered a master variable; pH affects a wide range of biological, chemical, and physical soil properties. For example, pH can affect the availability of nutrients, the formation and stabilization of aggregate structures, and microbial populations (Brady and Weil 2002). That said, pH varies across space due to drainage, erosion, fertilization, and acidifying processes near the soil surface, and through time due to season, organic matter decomposition, and spring plant growth (Brady and Weil 2002). And although plant reactions to pH are variable, most coniferous forest species grow well in acid soils. All else being equal, differences in soil texture can have a large impact on the amount of organic matter stored in the soil. Soils high in clay and silt generally have higher quantities of organic matter. This is because finer textured soils tend to produce more biomass, they are less well aerated, and organic matter is protected from decomposition by being bound to clay humus complexes or sequestered in soil aggregates (Brady and Weil 2002).

2.7 Restoration and Carbon sequestration in Douglas-fir Plantations

Plantation forests are established through the planting of one or more tree species in the process of reforestation (following a disturbance) or afforestation. Stands typically are even-aged with even spacing of trees (FAO 2006). A stand is defined as a community of trees that are homogeneous enough to be treated as a unit (Kurz et al. 2008). Historically, the primary objective of creating a plantation was the production of timber or fuel wood.

Recently, plantations have become a mechanism for fixing carbon (Brockerhoff et al. 2008). In terms of silviculture practises within plantations, the three primary factors influencing optimum C storage are rotation length, the amount of live mass harvested, and the amount of detritus removed from the forest through slash burning; carbon stores increase as rotation length increases but decrease as the amount of biomass and detritus removed increases (Harmon and Marks 2002). Understanding decomposition processes and the influence of forest management practises on them, and thus the carbon cycle, is crucial to maintaining the long-term productivity and carbon sequestration potential of managed forests. However, there is a lack of strong relationships of carbon pools with individual variables across different ecosystem types. This suggests that there is a complex interplay between climate, species composition, stand age, and soil properties such as texture (Homann et al. 2005). Any restoration plan that aims to maximize carbon sequestration while maintaining biodiversity values must be ecosystem specific.

In terms of total C storage, there is between 2.2 and 2.3 times as much storage in 450 year old natural stand of Douglas-fir-Western Hemlock (*Tsuga heterophylla*) then in a 60 year old Douglas-fir plantation (Harmon et al. 1990). However, the process of stand biomass development and C accumulation is accelerated in plantations compared to natural stands due to the higher initial density of stems in plantations and subsequent earlier crown closure (Long and Turner 1974). For example, total tree biomass in a 73 year old natural stand is comparable to 42 year old plantation (Long and Turner 1974). Thus stand age and structure play an important role in determining the magnitudes and patterns of C cycling processes within forested ecosystems (Humphreys et al. 2006). In

the context of soil carbon, there does not appear to be a trade-off between restoring a coastal Douglas-fir plantation for biodiversity through thinning versus carbon sequestration if the measures used to increase biodiversity do not reduce organic matter accumulation on the forest floor. If the amount of C input into the soil was fixed, decay would need to be suppressed to increase the carbon stores. However, if the restoration measures used on site do not remove any organic matter, then all thinned trees and leaf litter are left on site and add to the detrital component of carbon storage.

Importantly, although forest plantations sequester some carbon while alive, due to the simplified nature of these forests, it is more likely that at some point they will succumb to disease, insect outbreaks and/ or fire and therefore may not be considered reliable carbon offsets (Wilson and Hebda 2008). Alternatively, healthy, functioning, diverse ecosystems tend to be more resilient and therefore less vulnerable. There is the potential to restore forest plantations to a healthy functioning state. The question is, how do we restore a plantation forest so that it will simultaneously optimize carbon sequestration and increase function and biodiversity?

Chapter 3: Biodiversity

Increasing biodiversity values in plantation forests serves primarily two purposes. First, increasing the structural and species diversity of a stand leads to increased resistance and resilience of the stand in the face of both deterministic and stochastic threats. Second, by employing restoration it is possible to change the successional trajectory of a stand from a monoculture composed of one age class towards a more diverse stand which increases the utility of the stand in terms of wildlife use. The purpose of this chapter is to: a.) define biodiversity; b.) discuss important ecological concepts and theories as they apply to biodiversity; c.) give an overview on causes of declines in forest biodiversity; and d.) outline how second growth plantation forests can be used by forest species. Each of these four points will be referenced in either the study site, chapter 6, or the discussion, chapter 10.

3.1 Defining Biodiversity

According to Article 2 of the Convention on Biological Diversity (2008), "biodiversity" is defined as "the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems, and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems." Within forested ecosystems, biodiversity can be considered at different levels including ecosystems, landscapes, populations, species, and genetics. Complex interactions occur within and between these levels; this complexity allows organisms to adapt to changing environmental conditions and to maintain ecosystem functions (CBD 2008). The evaluation of biodiversity requires measurable parameters

that act as correlates or surrogates for biodiversity. Features of biodiversity can be clearly defined by specific attributes (e.g., species at risk), and measures (e.g., number of individuals) (The Royal Society 2003).

3.2 Forest Ecology: Concepts and Theories Applied to Biodiversity

Landscape ecology provides the theoretical scientific basis for reconnecting and restoring fragmented habitats (Forman and Godron 1986). In terrestrial ecosystems, habitat fragmentation can occur when changes in land use or land cover transform a contiguous habitat patch into disjunct patches. Particular landscape patterns, such as the size, shape, connectivity and configuration of habitat remnants, have certain implications for both biotic and abiotic processes. Patch dynamics focus on the creation of spatial heterogeneity within landscapes and how that heterogeneity influences the flow of energy, matter, species, and information across the landscape (Zipperer et al. 2000).

Fragmentation is significant because it reduces the available area of forest habitat, increases the isolation of forest patches, and increases the edge effects in the remaining patches. There are two key theoretical frameworks in community and population ecology that have been used to study habitat fragmentation; the theories of island biogeography (MacArthur and Wilson 1967), and metapopulation dynamics (Levins 1969). The former has been used as a guide to assess the influence of patch size and isolation on species composition, whereas the latter has focused attention on connectivity and interchange between spatially distributed populations (Collinge 1996).

3.2.1 Island Biogeography Theory

According to the island biogeography theory, the number of species in a remnant patch of forest is not only controlled by the habitats and resources present on site, but also by the balance of immigration and local extinction (MacArthur and Wilson 1967). Patterns of immigration are primarily determined by distance from other sources of potential colonists. Habitat patches that are relatively close to other patches are more likely to be occupied then more isolated patches because they are likely to be recolonized after a local extinction event (Pulliam and Johnson 2002). Larger patches can support larger populations, which are less vulnerable to extinction (MacArthur and Wilson 1967). All other things being equal, smaller remnant patches support fewer native species then larger patches (Bellamy et al. 1996). This is not only because larger forest remnants are likely to have a higher ratio of colonisations to extinctions, but also because they are more likely to have undisturbed components necessary to some species (Harris 1984), and are more likely to contain a range of habitats for different species (Fox 1983).

3.2.2 Metapopulation Theory

Metapopulation theory can be thought of as an extension of island biogeography from habitat patches to population patches. Patchy populations are true metapopulations only if movement between sub‐populations is neither very common nor very uncommon (Hanski and Simberloff 1997). Clusters of populations may interact over time through the exchange of individuals or genetic material, and individual populations may frequently go extinct and the same area recolonized at a later time by immigrants from extant

populations (Pulliam and Johnson 2002). The dynamic nature of local extinctions and recolonization dictate that any particular patch of habitat may or may not be occupied at a given point in time, however, the metapopulation as a whole persists because some patches are always populated (Pulliam and Johnson 2002). In addition, large populations are less likely to go extinct than small populations, and large habitat patches, which are more likely to support large populations, are more likely to be occupied than small patches.

3.2.3 Patch-Matrix-Corridor Model

An extension of the aforementioned fragmentation theories is the patch-matrix-corridor model (Forman 1995). Within large patch and matrix landscapes, disturbances create a diverse, shifting mosaic of successional stages and physical settings of different origin and size (Bormann and Likens 1979). The patch-matrix-corridor model is significant because it recognizes that the ability of a species to reach remnant forest patches depends on how inhospitable, or permeable, the landscape matrix surrounding the patch is (Forman 1995). With this model we move away from the often misleading conceptualization of landscapes as areas of forest/ habitat or non-forest/ non-habitat, toward the idea that the landscape matrix surrounding remnant forest patches may be neither uniformly unsuitable as habitat nor serve as a complete barrier to the dispersal of forest taxa (Kupfer et al. 2006). Thus, the extent to which fragmentation affects a given species depends on how the landscape has been modified, what constitutes suitable habitat for the species, mode and scale of movement, and dispersal behaviour (Fischer and Lindenmayer 2007). Furthermore, the rate of recovery of an ecosystem or species at any scale following a disturbance is not only strongly influenced by the availability of

nearby organisms or propagules, but also by biological legacies, such as seed banks, for recolonization (Holling 1973).

3.2.4 Habitat Heterogeneity Hypothesis

The habitat heterogeneity hypothesis suggests that structurally complex patches may provide more niches and different opportunities to exploit environmental resource and thus species diversity will increase with patch complexity (Bazzaz 1975). This concept is supported by the research done by Tilman et al. (1997a). However, there are discrepancies in the literature regarding the relationship between habitat heterogeneity and fauna diversity. This is because the relationship between patch heterogeneity in terms of vegetation architecture and animal species diversity depends on:

a) How habitat heterogeneity is perceived by the animal guild studies;

b) How species diversity is measured;

c) How habitat heterogeneity is defined;

d) How vegetation structure is measured; and,

e) The spacio-temporal scale of the study (Tews et al. 2004).

Furthermore, species diversity patterns show year-to-year and season-to-season variations, which have important implications for across-study comparisons (Tews et al. 2004).

3.2.5 Conservation Area Functionality

One of the most important tasks in conserving biodiversity is determining what areas should be set aside for each species and or ecosystem. This decision should be based, in part, on the area's functionality or ecological integrity. Four attributes that can be examined to assess a potential conservation area's functionality have been suggested: composition and structure of the focal ecosystems and species; dominant environmental regimes, including natural disturbance; minimum dynamic area; and connectivity (Poiani et al. 2000). Key compositional and structural components for a given species may include age structure, evidence of reproduction, population size or abundance, genetic diversity, and minimum viable population (Poiani et al. 2000).

Other important compositional and structural components for ecosystems may include abundance of invasive species, presence of species that indicate unaltered ecological processes, abundance of important prey species, evidence of reproduction of dominant species, existence of characteristic species diversity, and evidence of vertical or strata layering (Poiani et al. 2000). Important dominant environmental regimes may include grazing or herbivory, hydrologic and water chemistry regimes (e.g. surface and groundwater), geomorphic processes, climatic regimes (e.g. temperature and precipitation), fire regimes, and many types of natural disturbances (Poiani et al. 2000). The area required to ensure survival or recolonization of a given species has been termed the minimum dynamic area (Picket and Thompson 1978). An important consideration in the creation of minimum dynamic areas is disturbance size. For example, Baker (1992) suggests that conservation areas should be large relative to maximum disturbance size to minimize their vulnerability to fatal loss of organisms, to reduce the chance of

disturbance spreading into surrounding developed lands, and to minimize the influences of adjacent lands on the size and spread of the disturbance. Developing scientific estimates of minimum dynamic area and metapopulation structure for biodiversity at different scales is one of the critical frontiers of applied conservation biology (Poiani et al. 2000). Given the limited resources available for restoration, it is pertinent that those resources are used in areas that maximize their benefits.

3.2.6 Hierarchal Structures and Models for Understanding Complex Systems

Ultimately, a comprehensive plan for the protection of biodiversity must include all elements of biodiversity from genes to landscapes and is thus hierarchical both in spatial scales and biological levels of organization (Noss and Cooperrider 1994). Several different conceptual hierarchies have been developed to facilitate the understanding of complex systems such as forest ecosystems. A hierarchy explains relationships within a system by ranking levels of organization. Levels may be defined by a variety of attributes including physical or spatial structure, or interaction rates. Furthermore, two types of hierarchies can be distinguished: structural and control.

Control hierarchies exist when components at one level exert control on components at a lower level that may not be a subsystem of the controlling unit and thus are considered non-nested (O'Neill 1989). This type of hierarchy might be employed in the study of relationships between plants, herbivores, and carnivores.

A structural hierarchy, on the other hand, focuses on subsystems within systems and thus is nested (O'Neill 1989). An example of a structural hierarchy is the relationships

between genes, organisms, and populations. The most useful hierarchy to employ in classifying and analyzing the ecology of forest remnants within landscapes is the structural hierarchy because it allows for the examination of faster processes at a fine scale in site-specific environments, and the clustering of detail to expose more general slower processes at the coarser landscape scale.

The concept of a structural hierarchy has been built on by Holling (2001), who presents the concept of "panarchy" to describe complex adaptive socioecological systems. Within this framework, systems are interlinked in infinite four-phase adaptive cycles of growth, accumulation, restructuring, and renewal. These transformational cycles occur in nested sets among variables that share similar speeds and spatial attributes at various scales in both space and time. This hierarchy lends itself well to the examination of succession in forest ecosystems. This concept is important because it helps define the role of biological legacies. That is, cycles are interlinked because at each level in the hierarchy, small amounts of information and or material are communicated to the next higher level. In the case of forests, this material is in the form of biological legacies such as snags and seed banks, and nutrient capital such as soil organic matter and total nitrogen. Importantly, the sustainability of an adaptive forest ecosystem is determined by the functioning of cycles, as well as the communication between them. For example, as an ecosystem goes through the process of succession biomass accumulates both above ground and in the soil; restructuring by wind-throw has a very different impact on the renewal phase of the adaptive cycle then restructuring from forest harvesting, or in the case of this study, variable density thinning. This is because these disturbances result in different remnant structures and nutrient pools which then influence how renewal occurs.

3.2.7 Natural Variability

Landscapes are often characterized by spatial heterogeneity that is the result of physical setting, biological agents, processes of disturbance, and stress (Pickett and Rogers 1997), as well as human agency. The original physical template of the landscape reflects the geology, geomorphology, and soils of the site. Organisms affect landscape heterogeneity through their growth, interactions, and legacies or ecological memory. Disturbance affects the structure of a site through physical force, and humans affect the site in a number of both direct and indirect ways. Alternatively, spatial heterogeneity helps control biodiversity by both creating and closing opportunities for organisms, and influences nutrient cycling by generating barriers or pathways to the flow of energy and materials (Pickett et al. 1997).

Natural disturbances such as fire, wind throw, insect and disease outbreaks, mass wasting, surface erosion, and catastrophic events are particularly important because they influence forest succession and affect soil productivity (Maynard 2002). Ecosystem attributes vary naturally in response to fluctuations in natural disturbance and climate. For example, the dominant natural disturbance agent in Canada's boreal forest is fire, and variable fire rates over time and space create a dynamic patchwork of forest types that provide habitat for a diverse assemblage of species. From the perspective of biodiversity management, it is important to understand that within certain limits, changes in the population patterns of plant and animal life are normal. Management activities must recognize this and attempt to mimic this natural range of variability.

Natural ranges of variability can be determined by reconstructing historic patterns and processes. Methods include simulation modeling, historic accounts such as early land surveys, interpretation of historical air photos, and paleoecological studies of sediments, charcoal, tree rings and pollen (Poiani et al. 2000). Additionally, thermographs, rainfall hyetographs, hydrographs, and outputs from simulation models can be statistically summarized to describe natural ranges of variability (Baker 1992; Morgan et al. 1994; Richter et al. 1996). However, when data are not available regarding historical patterns and processes, deductions with respect to cause and effect relationships may be drawn from reference ecosystems and similar organisms (Arcese and Sinclair 1997).

Within coastal Douglas-fir forests, fire is the historically dominant disturbance agent with an approximate average fire return interval of 230 years (Agee 1991). However, in the context of plantation forests, this estimate may no longer apply due to changes in tree density, species composition, and fuel loads. And for the same reasons, the severity of fires that may occur has likely changed. Densely packed trees that have not dropped there lower branches present a significant risk in terms of ladder fuel and subsequent high intensity crown fires vs. low to moderate intensity ground fires which are generally not stand replacing events in Coastal Douglas-fir forests because Douglas-fir bark is fire resistant (Hosie 1969).

3.2.8 Ecosystem Composition, Structure and Function

An ecosystem can be defined as a physical environment and suite of organisms in a specific area that are functionally linked (Pickett and Rogers 1997). Forest ecosystems are typically quantified according to compositional, functional, and structural attributes.

Forests as habitat may be broadly defined as the range of environments suitable for a given species (Fischer and Lindenmayer 2007). Each species responds individualistically to a range of processes connected to its needs for food, shelter, space, climactic conditions, and interspecific processes such as competition, predation, and mutualism (Fischer and Lindenmayer 2006). Thus, responses to landscape features are often species-specific.

Composition refers to the variety and proportion of various species present and represents a major component of biodiversity (Franklin et al. 2002). An example of its influence is clear in how the species and density of plants interact to drive local evapotranspiration rates (Eviner 2002). The importance of composition will be discussed in more detail in Section 3.2.8.

Function refers to the work carried out by an ecosystem and is a general term used to describe a suite of processes such as primary production, ecosystem respiration, biogeochemical transformations, information transfer, and material transport that occur within ecosystems and link the structural components (Grimm et al. 2000). The function of an ecosystem, or rates of key ecosystem processes, is limited by the structure of the ecosystem. For example, primary production is limited by soil nutrients, temperature, and soil water availability, and these factors are mediated by local climate and weather conditions. Function can be thought of as an integrated measure of what a unit (ecosystem or part of ecosystem) does in the context of its surroundings (Grimm et al. 2000). Ecosystem function is generally quantified by measuring the magnitude and

dynamics of ecosystem processes or rates and directions of energy transfer. For example, primary production can be measured through biomass accumulation.

Structure refers to the component parts of the system including both the variety of individual structures such as trees, snags, and coarse woody debris of various sizes and conditions, as well as the spatial arrangement of these structures, such as whether they are evenly spaced or clustered (Franklin et al. 2002). An alternative perspective is that forest structure is the physical stage on which ecological variables interact. Forest structure can mediate communities within the stand by providing resources for and influencing interactions between organisms; stand structure can also influence ecosystem processes through its modification of environmental conditions and resource availability (Byrne 2007). For example, it is known that above ground structure mediates soil temperature through interception and absorption of solar radiation and its ability to transfer heat energy into the soil (Geiger et al. 2003). It is hypothesized that forest stand structure affects resource pools both directly and indirectly. A direct relationship would be when the vegetation provides resources in the form of litter and roots. An indirect relationship would be when the vegetation influences the abundance of detrivores, reducing the amount of litter (Byrne 2007). Furthermore, parameters of vegetation structural complexity such as vegetation type, height, and coverage may not be significant by themselves, but how they interact may be significant. For example, in carabid beetle diversity study (Brose 2003), beetle diversity was not correlated with any one of these variables, yet the correlation with the multivariate structural gradient was highly

significant. Types of structures particularly important in the context of the restoration of forest ecosystems will be discussed in Chapter 4.

Importantly, all three ecosystem attributes (composition, structure, and function) change during the successional development of a forest stand. It is necessary to keep in mind that patches are not static; they are dynamic entities that change through vegetation succession, plant and animal dispersal, physical disturbance etc. (Pickett et al. 1997). Ecosystem structure is dictated by ecosystem processes that control and limit the transformation of material, energy, organisms, and information in and across ecosystems (Dale et al. 2000). And ecological processes function at many different time scales, for example, decomposition occurs over hours to decades, whereas soil formation occurs on a scale from decades to centuries (Dale et al. 2000).

3.2.9 Functional Diversity vs. Species Diversity

There are three mechanisms by which species diversity, and by extension a significant component of composition, contributes to ecosystem function (Petchey 2000). First, communities with many species are more likely to contain species with particularly unique traits. Second, communities with many species contain a greater range of species traits and therefore use resources more completely. Lastly, communities with many species are likely to have a higher frequency of facilitative interactions between species.

The biotic component of an ecosystem is often broken down into functional trophic groups; plant producers, consumers that feed on plants and each other, and decomposers.

Functional groups are guilds of species that are classified on the basis of intrinsic physiological and morphological differences, resource requirement differences, seasonality of growth, or life history (Tilman et al. 1997b). If, for example, species are classified according to trophic groups, these groups may be further subdivided based on life history. This is because differences in life forms affect ecosystem properties and processes such as nutrient flow; perennials maintain storage pools of energy and nutrients for subsequent growing seasons, while annuals only have seed storage and thus are wholly dependent on photosynthesis and nutrient uptake (Vitousek 1990).

Recent studies by Tilman et al. (1997b) address the influence of functional diversity on ecosystem processes. They found that: a.) the functional group component of diversity is a greater determinant of ecosystem function then the species component of diversity; b.) factors that alter ecosystem composition are likely to impact ecosystem function; c.) all species do not contribute to ecosystem function equally; and d.) different ecosystem processes are likely to be affected by different functional groups and species. Petchey (2000) confirmed this, and found that communities containing species from different functional groups have higher levels of ecosystem function, as well as higher variation in ecosystem function. However, the research by Tilman et al. (1997b) shows that species diversity and functional diversity are correlated; each is significant by itself, as is species diversity within functional groups. The results of this research indicate that species losses from managed ecosystems may have significant effects on the productivity and sustainability of those ecosystems (Tilman et al. 1997a). The same may be said for species additions to forest ecosystems as a result of restoration activities.

The importance of both functional diversity and species diversity is evident in forest ecosystems. For example, tree species diversity contributes to ecosystem structure and function when species with different life forms and autecology are included, such as species of both evergreen and deciduous behaviours, and shade-tolerant and shade-intolerant habits (Franklin et al. 2002). Many coastal Douglas-fir forests have a lower tree stratum composed of species with limited height potential such as Pacific yew (*Taxus brevifolia)*, and this lower stratum may make unique contributions to ecosystem function (Franklin et al. 2002). Furthermore, a variety of tree species produce snags and logs that differ widely in decomposition rates and patterns resulting in higher structural diversity (Harmon et al. 1986). In short, forests that contain more tree species are more likely to function better and provide more resources for other organisms. In this context, releasing volunteer tree species that differ from planted stock through thinning planted stock will change the composition and increase the function and diversity of a plantation.

3.3 Causes of Declining Forest Biodiversity

Forest biological diversity is the result of evolutionary processes over thousands and even millions of years. An important point to remember during this discussion is that ecosystems are complex networks of interconnected organisms, the loss of any one component of an ecosystem can affect all remaining species, and this often occurs in ways we do not yet understand (Backhouse 2000). Naturally, some existing species become extinct; the ‘background’ extinction rate is about one species per million per year (Wilson 1992). The issue is that human activity has increased extinction rates to between

1000 and 10000 times this background level in places such as rainforests by reduction in area alone (Wilson 1992).

3.3.1 Deforestation, Fragmentation, and Degradation of Forests

The three major anthropogenic categories of disturbance associated with declines in biological diversity in forest ecosystems are deforestation, fragmentation, and degradation. The mechanisms by which these disturbances occur include the conversion of forests to agricultural land, overgrazing, unmitigated shifting cultivation, unsustainable forest management, the introduction of invasive plant and animal species, infrastructure development, mining and oil exploration, anthropogenic forest fires, pollution, and climate change (CBD 2008). The concepts and theories pertaining to deforestation and fragmentation have been discussed in section 3.2, the remainder of this section will focus on habitat degradation with an emphasis on soil processes.

Habitat degradation may be generally defined as the gradual deterioration of habitat quality for a given species (Fischer and Lindenmayer 2007). Habitat degradation is significant because, for example, it may cause species to occur at lower densities (Felton et al. 2003), or may make them unable to breed (Battin 2004). In forest ecosystems, habitat degradation may be difficult to detect because it can take a long time to manifest. For example, it can take decades, even centuries, for a tree to complete the cycle of germination, maturation and decay after a clear-cut (Thompson et al. 2005), and this has powerful implications for both snag and coarse woody debris recruitment.

Habitat degradation that is the result of forest harvesting is different than that caused by other events such as acid rain. Generally, native vegetation is cleared first in areas that are characterized by high primary productivity (Norton et al. 1995). The influence of harvesting on productivity varies greatly among species or forest type, time of harvest, and inherent soil properties (Maynard 2002). Nutrient losses due to biomass removal may be substantial after harvesting. The magnitude of the loss depends on the type of harvest (whole tree vs. boles only), harvest system used (selective logging vs. clear cut), forest type, and time of year (Maynard 2002). In general, the influence of forestry practices on nutrient loss is long-term, and harvesting rotation times that are less than the time it takes to replace lost nutrients will eventually reduce soil productivity (Maynard 2002). Other harvesting impacts on soil include soil compaction, displacement, and organic matter loss associated with roads, skid trails, and landings. Compaction is particularly important because it negatively affects soil structure, aeration, water infiltration, runoff, and surface erosion, all of which reduce soil productivity (Maynard 2002). Furthermore, recovery after compaction has been found to vary from several years to several decades in boreal, temperate, and tropical forests (Grigal 2000; Kozlowski 1999). This is significant because soils are the medium on which most vegetation grows.

3.3.2 Climate Change

Climate change can have a significant effect on biodiversity because when environmental regimes and natural disturbances are pushed outside of their natural ranges of variability, changes in ecosystems will follow (Poiani et al. 2000). In terrestrial systems, enabling local and regional scale migrations will be critical for the survival of

species in the face of climate change (Poiani et al. 2000); however, it has been suggested that the current trend of forest fragmentation may negatively affect the ability of species to undertake large-scale movements such as seasonal migrations and climate change-induced range shifts (Soule et al. 2004). Furthermore, the effects of climate change on forest biodiversity can act indirectly, cumulatively with other disturbances, or possess feedbacks or time lags.

3.3.3 Invasive Species

Invasive alien species are considered the second largest threat to native species biodiversity next to habitat loss (Wilson 1992). Ecosystem processes are likely to be affected by the invasion of novel organisms (Tilman et al. 1997b). Non-native invasive species are able to alter ecosystem properties if: a) they have considerably different capacities to acquire and use resources than native species, b) they change the trophic organization at the invasion site, or c) they modify disturbance frequency and or intensity (Vitousek 1990). For example, adding novel generalist herbivores to a system can depress producer populations and/or net primary productivity (Vitousek 1990). Even if introduced species do not become invasive, novel species have the potential to alter community structure and ecosystem processes through predation, competition, their potential role as a pathogen, as vectors of diseases, and through their effects on water balance, productivity, and habitat structure (Drake et al. 1989). For example, soil nutrient availability may be diminished through the introduction of plants that produce low‐quality acid litter (Vitousek 1990). Furthermore, fire intensity can be heightened in

the presence of the invasive species Scotch broom because it is highly flammable due to its high oil content.

3.4 Contributions of Second growth Forests to Biodiversity

Biodiversity varies considerably with stand age, but not all species need old‐growth forests to fulfill their habitat requirements. At different points in successional development, younger stands can provide sufficient structural complexity, species diversity, and ecological function to provide habitat for a variety of different species. Although initial regrowth may be relatively homogenous, heterogeneous forest ecosystems emerge when underlying edaphic and microclimatic gradients (Samuels and Drake 1997), combined with differential success of colonizing species in microsites, eventually cause local divergence in species composition and distribution (Harrelson and Matlack 2006). The precise time scale of compositional divergence is unclear (Harrelson and Matlack 2006); though, as time from disturbance increases, second-growth communities follow a trajectory of increasing richness and changing composition that approaches nearby primary forests (Flinn and Marks 2004). It is often the goal of ecological restoration to speed up this transition. Specific management tools can be applied to second-growth forests to recruit various habitat characteristics, such as wildlife trees, in a much shorter time frame than would occur with natural regeneration alone. Habitat recruitment will be discussed in more detail in Chapter 5.

At a landscape scale, second-growth forests can contribute to biodiversity within an environmental matrix of different land use types through habitat supplementation or

complementation, by improving connectivity between remnant forests, and by buffering remnant forests from edge effects (Brockerhoff et al. 2008).

3.4.1 Habitat Supplementation or Complementation

Some species that survive in remnant forest patches may compensate for habitat loss by using resources in the surrounding landscape matrix (Brockerhoff et al. 2008). This is because, as stated by the patch-matrix-corridor model (Section 3.2.3), fragmented landscapes are composed of gradients in habitat conditions for forest dwelling species. If, for example, there is both mature second-growth forest and pasture land adjacent to a remnant patch of old-growth forest, forest fauna are more likely to hunt and/or breed in the second-growth forest than the pasture, since the second-growth forest has more structural similarity to primary forest (Gascon et al. 1999). In a study done by Gascon et al. (1999), a substantial number (40 - 80%) of nominally primary forest species were observed using matrix habitats; however, most of these observations were made in areas close to large forest tracts which provide potential sources of immigration. Furthermore, the capacity of matrix habitat to support forest dwelling species is largely determined by the history and intensity of land use (Lawton et al. 1998).

3.4.2 Connectivity

Connectivity refers to the degree to which a site or landscape is connected for a specific species. Forest habitat value and threshold of connectivity for any given species depends on the abundance and spatial arrangement of their habitat, and the dispersal capabilities of the species (Dale at al. 2000); the species-specific ability of organisms to move, disperse, migrate, or recolonize is related to both life history characteristics and

ecological processes (Hansen and Urban 1992). Connectivity is a threshold dynamic; incremental losses of habitat will result in incremental effects on the presence and abundance of species; however, at some point, a threshold is passed and the effects on species presence and abundance are dramatic (Dale et al. 2000). The presence of second-growth forests, both plantations and those that have naturally regenerated, can enhance indigenous biodiversity in fragmented landscapes by improving connectivity between indigenous forest remnants (Hampson and Peterken 1998).

3.4.3 Buffering

Buffers mitigate potential negative impacts from incompatible land uses that occur adjacent to wildlife conservation areas. In addition, beach, estuary, and forest buffers can provide connectivity between watersheds, and riparian buffers can provide elevational linkages within watersheds for wildlife. Second-growth forests adjacent to old-growth remnants, sensitive ecosystems, and/or critical wildlife habitat can also be used to help maintain the attributes that characterize these areas (Brockerhoff et al. 2008). Forest buffers reduce edge effects, extend the effective size of core areas, reduce the potential for invasion by organisms adapted to edge environments, enhance forest interior habitat, reduce the likelihood of blowdown within core areas, and reduce disturbance of important sites such as nest and breeding areas (Brockerhoff et al. 2008). Other potential benefits may include a temporary refuge for plants and animals after natural disturbances within core areas, and a source of replacement species for old-growth core areas lost to catastrophic disturbances (Brockerhoff et al. 2008). Furthermore, if one assumes that many areas on crown land adjacent to critical wildlife habitat will eventually be cut, we

can also assume there will be greater edge effects on those areas in the future than occur at present (Miadenoff et al. 1994). Forest buffering is a tool that can be employed to guard against this problem.

Chapter 4: Forest Structure

4.1 Structural Attributes

Structural attributes of forest stands are both theoretically and practically important for understanding and managing forest ecosystems. This is because structure is: a.) the attribute most often manipulated to achieve management objectives following stand establishment; b.) a measurable surrogate for functions such as productivity, or organisms such as cavity dwelling birds that are difficult to measure directly; and c.) directly valuable in terms of products such as timber and in providing services such as carbon sequestration (Franklin et al. 2002). Some structural features of forest stands, including individual structural elements and spatial patterns of structural elements, are presented in Table 1.

Table 1: Important structural features of forest stands (Adapted from Franklin et al. 2002)

Individual Structures	Important Attributes
Live trees	Species, density, mean diameter, range in diameter, height, canopy depth
Large-diameter live trees	Species, density, decadence including presence of decay columns, crown condition, bark characteristics
Large-diameter branches	Species, density, size, individual or arrays, presence of arboreal 'soil'.
Lower-canopy tree community	Composition, density, height
Ground community	Composition, density, deciduous or evergreen
Standing dead trees (snags)	Species, size, decay state, density
Coarse woody debris (logs)	Species, density, decay state, volume, mass
Root wads and root holes	Density, size, age
Organic soil layers	Depth, chemical and physical properties, biota

Spatial Patterns	
Vertical distribution of foliage/ canopy	Depth, continuity, cumulative distribution
Horizontal distribution of structures	Spatial pattern: random, dispersed, or aggregated
Gaps and anti-gaps	Size, shape, density

4.2 Old-growth Forests

In order to understand the contributions of forests to biodiversity, it is pertinent to understand the development processes associated with old-growth forests; if we are going to attempt to mimic the structure and function of old-growth forests we must recognize how they form.

Old-growth forests have been defined in terms of stand structure (e.g., Franklin et al. 1981), stand development processes (e.g., Oliver and Larson 1996), and tree age (e.g., MacKinnon and Vold 1998). For the purposes of this discussion, old-growth forest will be defined according to the MacKinnon and Vold (1998) B.C. inventory: old-growth forests are those that are greater than 250 years old on the coast and greater than 140 years old in the interior, except for lodgepole pine forests, which are considered to be old-growth at greater than 120 years old. These ages reflect the point at which structural and biological characteristics associated with old-growth forests become established.

There are many pathways to natural forest development. However, many old-growth forests have developed under the influence of moderate to severe natural disturbances such as wildfires, windstorms, droughts, floods, and insect or disease outbreaks. After a

natural disturbance, patches of living and many recently fallen trees remain, often preventing the regeneration of dense stands of uniform young trees. In old-growth forests, this means that the dominant trees vary considerably in age. This is because they started growing anywhere from several years before to decades after the disturbance that either killed the stand or created canopy gaps.

During the development of old-growth forests, young trees that were previously part of the understory generally grow rapidly after a disturbance due to wide spacing and the consequent lack of competition from other trees. This is supported by tree ring studies that show that in many old-growth forests the dominant trees frequently underwent rapid diameter and height growth in their first 50 to 80 years (Rapp 2002). Furthermore, due to the open space around the tree crowns, these trees kept more branches then they would have if other trees had been growing close to them (Rapp 2002). The result of this process is the development of crowns that are both wide and deep.

As these stands aged, the canopies began to close again. When old trees die, they created gaps in the canopy and, in some cases, deep crown redeveloped through epicormic branching (Rapp 2002). For example, older Douglas-firs develop branches from dormant buds on their trunks when they are exposed to light (Rapp 2002). These branches, along with shade tolerant saplings, contribute to the creation of a bottom-loaded canopy. Thus, the complex structure of old-growth forests is the result of variability. Variability in spacing allows some trees to grow rapidly and keep more live branches, and patchy mortality makes holes in the developing forest that allows other

trees to grow. The result of these processes is a forest with many tree species, ages, and sizes (Rapp 2002). Table 2 shows some of the pertinent structural processes that are operational during the successional development of forest stands in the approximate order of their first appearance.

Table 2: Structural processes during the successional development of forest stands (Adapted from Franklin et al., 2002)

Structural Process	Additional Description	Typical Stand Age
Disturbance and legacy creation		0
Establishment of new cohort of trees and plants		Between 0-10 years
Canopy closure of tree layer		Between 20-35 years
Competitive exclusion of ground flora	Shading	
Lower tree canopy loss	Death and pruning of lower branch systems	
Biomass accumulation		Between 40-100 years
Density dependant tree mortality	Thinning mortality due to competition among tree life forms	Between 40-100 years
Density independent tree mortality	Mortality due to agents such as wind, disease or insects	
Canopy gap initiation and expansion		
Generation of coarse woody debris and snags		
Uprooting of trees	Toppled trees cause soil disruption as well as the creation of structures such as root wades	
Understory re-development	Shrub and herb layers	
Establishment of shade tolerant tree species	Assuming the pioneer cohort are shade-intolerant species	
Shade patch development	Anti-gap development	
Maturation of pioneer tree cohort	Achievement of maximum crown height and crown spread	Between 100-200 years
Canopy elaboration	Development of multi layered or continuous	Between 150-350 years

	canopy through growth of shade tolerant species into co-dominant canopy position or re-establishment of lower branch systems on shade-intolerant dominants	
Development of live tree decadence	Multiple tops, dead tops, bole and top rots, cavities, brooms	
Development of large branches and large branch systems		
Associated development of rich epiphytic communities on large branches		
Pioneer cohort loss		Between 800-1200 years

Older stands generally provide better habitat than younger stands because of their increased spatial and vertical heterogeneity, well-developed soil organic layers and associated fungal floras, increased large woody debris, better light environment, and inter-specific facilitation (Brockerhoff et al. 2008). Old-growth structural attributes include a diverse tree community, large dominant trees with substantial crowns, smaller shade tolerant trees, multilayered canopies, snags or wildlife trees, canopy gaps, and abundance of coarse woody debris, and understory patchiness with some shrub and herb development (Franklin et al. 1981**).**

4.3 Wildlife Trees

The most significant contribution of snags in forest ecosystems is their function as wildlife trees. “A wildlife tree is any standing dead or alive tree with special characteristics that provide valuable habitat for the conservation or enhancement of wildlife” (Thompson et al. 2005). In fact, more than 80 species of vertebrates and

countless invertebrates depend on dead or deteriorating trees at some point in their life cycle (Thompson et al. 2005); in coastal Douglas-fir forests 16% of all vertebrates in BC require wildlife trees to some extent in their life cycle (Machmer & Steeger 1995). Some of their uses of wildlife trees are nesting, feeding, communicating through drumming and marking, roosting, shelter, and over-wintering (Thompson et al. 2005).

Within a natural stand, there are often multiple species of trees in many different sizes and conditions. It usually requires 100 or more years to recruit trees of sufficient size and condition to function as useful wildlife trees when relying on natural processes (Manning 2006). Large old trees with features such as multiple or dead tops, bole or top decays and cavities contribute to the structural diversity within the stand. Examples of habitat features found in wildlife trees include natural cavities in bigleaf maples (*Acer macrophyllum*), the "chimney effect" in western redcedars (*Thuja plicata*), live hardwood with primary cavity excavation and feeding holes, heart rot and loose bark in grand fir (*Abies grandis*), and nesting and perching sites on broken topped Douglas-fir snags (Thompson et al. 2005). In other words, specific features such as decay cavities, large diameter branches, and distinctive bark features might be explicitly recognized because of unique functional and habitat roles (Franklin et al. 2002).

Wildlife trees are generally created through mortality agents such as fire, disease, insects, windthrow, snowpress, lightning, and wildlife excavation. Death by fire produces a very different type of wildlife tree than gradual death by insects or disease, and tree species and local climate also influence the way a tree deteriorates and decays

(Thompson et al. 2005). The most significant indicators of wildlife tree quality are height, diameter, decay stage, location, distribution, and cause of death; but, the value of any particular tree for a given species depends on a variety of attributes, the most important of which are structure, age, condition, abundance, tree species, geographic location, and surrounding habitat features (Thompson et al. 2005).

4.4 Coarse Woody Debris

An important structural feature of a wide variety of old-growth forests in North America is a relatively large quantity of downed big logs (Sturtevant et al. 1997). Piece size, species (Harmon et al. 1986), and disturbance history (Spies and Franklin 1988) all affect the speed and type of decay of any CWD piece; while abundance, size, state of decay, and spatial distribution are all factors affecting the use of coarse woody debris (CWD) by wildlife species (Keisker 2000).

CWD is an important nutrient source and growing substrate for numerous species of bacteria, fungi, saprophytic plants, lichens, and mosses that are essential in decay, nitrogen movement, and other nutrient and moisture cycling processes (Thompson et al. 2005). For example, in one study of 6 Biogeoclimatic Subzones in British Columbia, 70% of the 243 plant species recorded grew on CWD, with 23% of those species being restricted to CWD (Song 1997). Other important functions of CWD include carbon storage, erosion control, buffering microclimates suitable for seedling establishment, cover from predators, shelter and access routes for small mammals during heavy snow cover, and contributions to stream stability, complexity, and geomorphology (Thompson et al. 2005).

Coarse woody debris also provides feeding, breeding, and shelter substrates for many invertebrates, small mammals, and amphibians (Thompson et al. 2005). Dupuis et al. (1995) found three to six times more *Plethodon vehiculum* salamanders in old-growth forests than in younger second-growth forests in coastal B.C.; this was partly attributed to the availability of CWD. Also, Davis (1998) found that the two salamander species observed in coastal B.C. used different decay classes of CWD. This suggests that old-growth forests are more likely to provide suitable habitat for both species then single-aged managed forests because old-growth forests generally provide a range of decay classes.

4.5 Mycorrhizal Fungi

Mycorrhizal fungi form symbiotic relationships with many forest plants. These fungi grow on plant roots and absorb sugars the plant produces by photosynthesis. In return, the plant gains access to water and nutrients that the fungi absorb from the soil through a network of filaments (Flynn 1999). The importance of mycorrhizae cannot be overstated – plant growth is greatly increased in the presence of these fungi, and some plants cannot grow without their fungal partner. Mycorrhizal fungi are considered to be *the* keystone of coastal Douglas-fir forests (Flynn 1999).

4.6 Second-growth Plantation Forests

Many second-growth forests are plantations. Logging and subsequent planting imitate natural forest development processes in some ways, but are not the same as natural processes (Rapp 2002). This is in part because plantations are planned to maximize

timber production. Plantation forests are created through the planting of one or more tree species and are typically of an even aged structure with an even spacing of trees. Hand-planted second-growth stands are typically much denser than stands natural processes would have created, and plantation forests generally contain few, if any, biological legacies such as snags and fallen trees or CWD (Rapp 2002). This generally translates into less habitat diversity and structural complexity then more natural forests. Plantation forests can be expected to be more like natural forests if they are composed of locally occurring native tree species, however, even plantations of exotic trees may have an understory composed of native flora and fauna.

Many plantations are intensively managed; they may have undergone site preparation such as ploughing, harrowing, use of fertilizers and herbicides, and may be subjected to silviculture practices such as thinning and the elimination of competitive woody understory regeneration (Brockerhoff et al. 2008). For example, forest fertilization is often employed because it accelerates overall stand development by increasing bole diameters and canopy closure, and by accelerating understory brush dieback and self-pruning below the canopy (Manning et al. 2006).

In the absence of management activities aimed at decreasing competition between the planted trees and natural regeneration, mono-specific plantations are often replaced by mixed forests comprised of both planted species and other floristic elements from the surrounding forest areas (Brockerhoff et al. 2008). Although natural forests generally offer superior habitat for native forest species than plantation forests, the degree to which

plantation forests can provide suitable habitat for native species is dependent on management intensity and plantation species composition and structure in relation to surrounding natural forests (Brockerhoff et al. 2008).

Plantation management practices that generally increase biodiversity values within a stand include:

a) Selecting tree species that provide resources and structures that favour native species;

b) Avoiding intensive site preparation that destroys herbaceous vegetation and coarse woody debris;

c.) Implementing wider tree spacing and heavy pre-commercial thinning to help maintain understory vegetation;

d) Increasing rotation length; and

e.) Maintaining some structural attributes such as old trees or snags (Brockerhoff et al. 2008).

Chapter 5: Management Options to Promote Biodiversity in Second-growth Forests

5.1 Habitat Recruitment

Forest managers can use the data on density, growth rates, and ages of old-growth stands to design silvicultural options that could put second-growth forests on different pathways likely to lead to greater forest complexity and habitat diversity (Rapp 2002). Management strategies employed in young forest to increase biodiversity values are based on the idea that affinity of those wildlife species occurring in late-seral forests is more likely attributable to ecological characteristics such as structure than the age of the forest (Hayes et al. 1997).

Second-growth stands will likely develop old-growth characteristics as a result of natural events; however, this process can take considerable time. Disturbances such as windstorms, ice storms, root rot, insect infestations, and fire will likely occur in unmanaged plantations over the course of a century; trees in very dense stands, like plantations, are often not very sturdy and are prone to blowdown (Rapp 2002), while root rot and wind throw could create small openings leading to structural complexity if seeds of shade tolerant trees are present in the seed bank.

These somewhat unpredictable disturbances could either put these plantations on a pathway that leads to structural diversity, or kill enough trees that new stands regenerate naturally; specific stand response depends on many factors including the plantation size

and the characteristics of the stands around it (Rapp 2002). If the trees over a large area are destroyed, it can be a major setback in forest succession, thus no management is an option for forest managers that may have negative consequences for biodiversity.

5.2 Partial Cut Harvesting Systems

Partial cut harvesting systems are designed to retain individual trees or groups of trees. Examples of partial cut harvesting systems include variable retention, sheltered, seed tree, and clear-cut with reserve systems. In areas where these systems have been employed in the past, both patches and individual leave trees may be considered for long-term retention to enhance recruitment of large diameter wildlife trees.

5.3 Thinning for Specific Structural Attributes

It is possible to speed up the recruitment of late seral structural features and complexity through silviculture strategies such as pre-commercial thinning (PCT) and variable-density thinning (VDT) (Carey and Wilson 2001). For example, in dense, uniform conifer plantations, one or more VDTs are likely to accelerate the development of some old-growth characteristics, perhaps by decades, in comparison to stands where no action is taken (Rapp 2002). This is supported by computer simulations which suggest that heavy thinning of young Douglas-fir stands at age 15 or 30, accompanied by under planting, can accelerate the development of aspects of stand structure found in late-seral stage forests such as tree diameter, crown depth, and limb diameter (Barbour et al. 1997). Furthermore, By delaying thinning until a stand is 30 years of age and increasing residual stand density from 75 to 150 trees per hectare, it may be possible to grow stands with

multi-storied structure, abundant large snag candidate trees (in excess of 85 cm DBH at 100 years), and sufficient residual stems to carry out a second thinning to recover moderate quality wood (Barbour et al. 1997).

Thinning to develop old-growth characteristics is different than thinning to maximize timber production. This is because timber management uses evenly spaced thinning to produce uniform stands and removes just enough trees to maximize the growth in volume of the stand as a whole, rather than maximize the growth in volume per tree (Rapp 2002). That said, stand tending such as pre-commercial thinning and pruning can have beneficial effects on wildlife habitat by affecting the volume and diameter of snags and CWD recruited into the stand. That is, stand tending decreases the future volume of CWD but increases its average future size, and spacing or thinning increases tree incremental growth, thus recruiting trees to become larger snags at an earlier age (Manning et al. 2006). Importantly, dense plantation trees are most responsive to thinning when they are less than 80 years old; options for accelerating forest structure development may diminish substantially if stands are not thinned when they are young (Rapp 2002).

5.3.1 Pre-commercial Thinning

PCT reduces tree competition and increases the amount of growing space per tree, which concentrates the growth potential of the site on the remaining trees (Sullivan et al. 2006). PCT produces high vigour trees with deep crowns and relatively rapid individual tree growth (Sullivan et al. 2006). Furthermore, PCT leads to an increase in abundance of understory tree classes, shrubs, and herbs, which produce an enhanced array of habitats

and microhabitats in the understory of these stands (Sullivan et al. 2006). This is evident in a study in the Oregon Coast Range where two out of three stands between 25 to 30 years of age were thinned to different densities. A few years after thinning, the stands exhibited major differences in tree size, range of variability, and understory development (Rapp 2002). The unthinned stand tree density was 199 trees per ha, and the average diameter at breast height (DBH) was 20 cm. In comparison, the most heavily thinned stand tree density was 42 trees per ha, and the average DBH was 38 cm: almost double the average in the untreated stand (Rapp 2002). Additionally, both of the thinned stands had a greater range of variability in tree size then the unthinned stand, which was generally uniform in size (Rapp 2002). The mixture of large and small trees in the treated stands is important because it will provide increased structural diversity as the stand ages. Finally, understory development in the unthinned stand was almost non-existent, with essentially no green plants on the ground. Alternatively, the moderately thinned stand (75 trees/ ha) had a few plants on the ground, whereas the heavily thinned stand had many herbs, shrubs, and saplings growing (Rapp 2002). Thus, it is the response of crop trees to silviculture practices, in terms of diameter and height growth as well as crown architecture, which drives the development of habitat attributes such as understory composition, abundance, and stand level structural diversity (Sullivan et al. 2006). The influence of PCT on successional development may be temporary; however, the temporal scale depends on the range of thinning intensity (Sullivan et al. 2006).

5.3.2 Variable Density Thinning

Given that old-growth forests are the product of variability, spacing should vary in thinning treatments intended to accelerate the development of old-growth structural characteristics. When it comes to VDT, there is no standardized procedure because thinning designs are site-specific and depend on the characteristics and landscape context of each stand. However, some suggestions have been made: VDT thinning can be done by thinning to different densities in 0.1- to 0.4-ha patches, by leaving small 0.1-to 0.2-ha un-thinned patches, and in other areas creating very small gaps, up to a 0.1 ha in size at most (Rapp 2002). The reason for the small gap size is to prevent susceptibility to wind throw while simultaneously encouraging understory development and large, open-grown trees. In addition, the un-thinned and lightly thinned areas are suitable for shade-tolerant seedlings such as hemlock and western red cedar.

5.4 Snag/ Wildlife Tree Recruitment

As discussed in Section 4.3, snags, particularly those with heart rot and/or cavities, are an integral structural element for many wildlife species; and yet, this valuable forest attribute is often lost from stands when short-rotation forestry is practiced. Operationally, there are three types of wildlife tree management strategies that can be employed to maintain snags across a landscape: wildlife tree patches (WTPs), individual live tree retention, and artificially created wildlife trees (Stone et al. 2002). Regardless of which wildlife tree retention and or recruitment technique is used, there are several variables

that need to be examined on a species-by-species basis in order to ensure that the habitat goals for each species are reached. These variables include size (area), composition (tree species, decay class, basal area), distribution and density (stems/ha), and condition (age class, decay class) (Manning et al. 2006). Furthermore, wildlife tree retention areas should be large enough to buffer key wildlife trees (those containing nest cavities, broken tops, stem scars, hollows or cracks) from adjacent harvesting areas, and provide some undisturbed habitat and interior forest-like conditions (Manning et al. 2006). In other words, a patch should be centered on a well-used wildlife tree or group of wildlife trees (Stone et al. 2002), and, where ecologically appropriate (i.e. the patch contains the desired habitat features), a roughly circular patch shape will optimize forest interior habitat.

Management tools that might be employed to create snags include topping trees with chainsaws or explosives, girdling trees, cavity creation using chainsaws, and inoculating trees with native fungus.

Fungal inoculation is a relatively new technique and is still under development. The ecological and operational feasibility of fungal inoculation as a habitat enhancement tool is still being evaluated; however, as of 2006, interim results indicate that it is an efficient and effective means for creating a tree that contains heart rot (Manning 2006). Furthermore, it is predicted that when these trees are excavated for use by wildlife, they will eventually become hollow trees, which further increases their habitat value (Manning 2006).

Two inoculation procedures were investigated to determine which produces the best results (Manning 2006). These include: a) injecting the native heart‐rot fungus, *Phellinus pini,* by climbing a tree, drilling a hole, and inserting a wooden dowel that has been cultured with a locally collected fungal strain; and b) shooting the tree trunk with a bullet that contains a smaller wooden dowel cultured with the same fungus (Manning 2006). Both techniques have resulted in the spread of fungal decay within the tree both above and below the point of inoculation. Four years after the treatment, no wildlife activity had been seen in the treated trees; however, it is expected that usable heartwood decay columns, sufficient for nest cavity construction by primary cavity-excavating bird species, will be created between 5-15 years post-treatment (Manning 2006). Importantly, there appears to be little risk of spread to non-targeted trees using this technique because of the reproductive history of the fungi. Recommended species for inoculation include Douglas-fir and spruce *(Picea* spp) (Manning 2006).

Fungal inoculation is used on live, healthy trees. Results indicate that the fungus does not usually kill the tree; rather, a compartmentalized decay column is produced within the trunk within three to six years. Inoculated trees generally maintain their foliage and growth form, and it is assumed they continue to put on new incremental growth and function as seed sources. This is significant because trees in this condition provide habitat for a longer period of time then dead snags, are likely to provide fewer worker safety or operational concerns, and are less likely to be felled by firewood cutters (Manning 2006).

That said, you would expect their longevity to be reduced in comparison to live healthy trees.

5.5 Coarse Woody Debris Retention and Recruitment

Eventually, snags will break apart and become coarse woody debris. Alternatively, CWD can be recruited by leaving a mixture of coniferous and deciduous thinned trees on the forest floor. The reason for using a mixture is that coniferous trees decay more slowly than deciduous trees (Manning et al. 2006). This means that deciduous CWD can provide important short‐term ecological benefits whereas coniferous CWD provides ecological benefits for a greater period of time. Ideally, larger CWD pieces will be either recruited or maintained because larger pieces decay more slowly, hold more moisture, present less of a fire hazard, and provide more habitat value to a greater number of wildlife species (Manning et al. 2006). Additionally, when recruiting CWD, it should be left on site in a way that mimics its natural distribution of randomness and connectivity, with some clumping and layering (Manning et al. 2006).

Regardless of which CWD retention and/or recruitment technique is used, there are several variables that need to be examined on a species-by-species basis in order to ensure that the habitat goals for each species are reached for CWD. Variables to consider include the amount (volume), condition (species and decay class), and distribution of CWD. Furthermore, forest health variables such as insects and fuel loading must be considered in the context of this evaluation (Manning et al. 2006).

5.6 Additional Forest Restoration Techniques

Some additional common forest restoration techniques include prescribed burning, under-planting, and mycorrhizal fungus recruitment. In some cases, low intensity under-burning might be used to help diversify the stand structure; and under-planting with several tree species, particularly shade tolerant conifers when they are absent, may be employed to increase vertical heterogeneity where required (Rapp 2002). Although these methods are not within the scope of this document, additional information can be found in *The Once and Future Forests: a Guide to Forest Restoration Strategies* (Sauer 1998), *Silviculture for Structural Diversity: a New Look at Multiaged Systems* (O'Hara 1998), and *Restoring Fire-Dependant Ponderosa Pine Forests in Western Montana* (Arno et al. 1995).

Chapter 6: Study Site

6.1 The Study Site

District Lot 63 (DL63) is a 65 ha Douglas-fir plantation within the Pebble Beach Nature Reserve on Galiano Island (Figure 2) and is owned by the Galiano Conservancy Association. The land is bound by a section 219 covenant held by the Islands Trust Fund and the Province of British Columbia. Section 219 covenants are voluntary agreements to conserve land or protect features relating to it; they are agreements between private land owners and designated organizations registered on the land title and are legally binding on the future owners of the property. In the context of DL63 this is important because it means that this parcel of land will never be developed.

Figure 2: Galiano Island District Lot 63

6.2 Landscape Context

DL63 is at a narrow point of Galiano island and covers approximately 80% of the width of the island at that location. Two of the parcels adjacent to DL63 are regional parks. Given that the Identified Wildlife Management Strategy for the CDF recommends restoring natural conditions and maximizing connectivity, and two of their prioritization tools for creating wildlife habitat areas include protecting communities that could become part of a network and protecting communities adjacent to natural occurrences of other communities, this is important (Pojar et al. 2004). The Pebble Beach Reserve, which is close to the same size as DL63, is classified as an older second growth ecosystem under the sensitive ecosystem inventory. This is important in terms of the permeability of the landscape matrix surrounding DL63; given the location of DL63 in the context of other wildlife habitat patches there are readily available sources of propagules from neighboring patches and DL63 is more likely to be used by native fauna then if it were more isolated from other forested areas. Furthermore, as restoration of DL63 progresses and the composition and structure of the stand more closely resembles that of the Pebble Beach Reserve, it is more likely to be used by a myriad of wildlife elements. Finally, given the forested state of adjacent land parcels, there is relatively low structural contrast between DL63 and neighboring areas indicating that edge effects are minimized. That said, there are several persistent non-native invasive species along paths, a hydro right-of-way, and the road leading to the forest lot. Theories supporting the importance of landscape context can be found in section 3.2.

6.3 Climate

The climate of Galiano Island is dominated by the influence of the rainshadow effect of the Olympic and Vancouver Island Mountains and the moderating effects of the ocean. Galiano Island has a cool Mediterranean climate which is characterized by warm dry summers and mild wet winters.

Climate data is not available for Galiano Island; Climate data for Mayne Island is used as a proxy in this document. The months of January and February produce the coldest mean temperatures, with an average minimum temperature of 1.8° Celsius, and an average maximum temperature of 7.6° Celsius (Environment Canada 2010). July and August are the warmest months, with an average minimum temperature of 11.3° Celsius and an average maximum temperature of 23.0° Celsius in July (Environment Canada 2010). Annual rainfall ranges from 597.3 mm to 1152.6 mm (Harrison 1994). Over 75% of the total annual precipitation falls during the winter months (November through February), with less than 10% falling as snow.

From mid-June to early October the combined effects of low precipitation, warm temperatures, and high sunshine hours often results in a moisture deficit on Galiano Island (Harrison 1994). During these months drought conditions are common in areas of recent clearcuts, such as DL63, and can result in an extreme forest fire hazard (Gaylor et al. 2002).

6.4 Soils

The soils of DL63 are composed of three major types, Saturna soils, Brigatine soils, and Metchosin soils (Green et al. 1989); however, only Saturna and Brigatine soils were included in this study. Saturna soils cover approximately 62% of Galiano Island and are the most common soils found in DL63. Brigatine soils cover approximately 6.8% of Galiano Island and are a very small component of DL63. Both Saturna and Brigatine soils are Brunisolic soils and occur where the climate is characterized by warm, dry summers with high moisture deficits and relatively low total precipitation (Jungen 1985).

Saturna soils are classified as Orthic Dystric Brunisol and are formed on top of sandstone. Saturna soils are moist throughout the late fall to spring, and are droughty during the summer. Saturna soils are characterized by well drained loamy sand to sandy loam soils which occur at depths of less then 125 cm with an underlying restrictive layer of consolidated bedrock. Effective rooting depth is 45 cm. Coarse fragment content is highly variable, ranging from 20 to 80%, and consists of both colluvial and glacial material. Natural vegetation on Saturna soils includes Douglas-fir, scattered arbutus, some grand fir, salal, western bracken, and dull Oregon-grape. Saturna soils are noted for their stoniness, shallow soils over bedrock, doughtiness, low fertility, and frequency of rock outcrops (Green et al. 1989).

Brigatine soils are classified as Gleyed Dystric Brunisol and are formed on top of fine textured subsoils; these subsoils often experience seasonally fluctuating water tables and are saturated to approximately 60cm below the forest floor during the winter season.

Drought conditions may occur during the summer when the water table drops to below 75 cm from the forest floor. Perched water table conditions may occur above the fine-textured subsoil. Brigatine soils are characterized by imperfectly drained soils that have a layer of loamy sand to sandy loam of marine or fluvial origin overlaying deep silty clay loam to silty clay marine deposits that are usually stone free. Effective rooting depth is 80 cm. Coarse fragment content ranges from 0 to 10%, and consists of gravel to angular gravel. Natural vegetation on Brigatine soils consists of western red cedar, red alder, Douglas-fir, sword fern, salal, and western bracken. Brigatine soils are noted for their high acidity (pH 5.1-5.5), the low moisture-holding capacity of the upper horizons, and low inherent fertility (Green et al. 1989).

Although soil maps show the presence of both Brigatine and Saturna soils within the study area, no significant differences in coarse fragment content, humus form, or soil texture were observed in either the forest floor, first or second mineral soil layers. The coarse fragments within the soil profile tended to be sub-rounded to sub-angular and the volume of said fragments was relatively high ranging from 25 to 60%. The humus form in all plots was a Mor. Hand texturing indicated that most soils were loamy sands to sandy loams; this was confirmed in the lab. All soils in the first and second mineral layers were in the coarse to medium textural groups. Furthermore, no mottling or gleying was observed in any soil layer; this suggests there is not a fluxuating water table within the study area. As a result of these observations, no differentiation was made between Brigatine and Saturna soils during the analysis.

6.5 History

DL63 was clearcut in two phases; once in 1967 when ~20% of the vegetation was removed, and again in 1978 when ~76% of the vegetation was removed. Post harvest treatments were different for each of these phases. After the 1967 cut, the harvested area was burned removing all slash and remnant vegetation. After the 1978 cut, slash and topsoil were bulldozed into windrows, and a broadcast burn was attempted but failed to take. Today, DL63 primarily consists of a 43 year old and a 32 year old evenly spaced closed canopy Douglas-fir plantation.

6.6 Biophysical Description: Reference Area

The 1978 clear cut is the focus of this study because logistically it made the most sense. In general, this area is on the cusp of the stem exclusion stage moving from pole/sapling to young forest and is characterized by an even aged, single story Douglas-fir canopy, with stems uniformly spaced along a north-east facing slope. In several areas within this site, site and soil characteristics such as steep slopes, soil compaction, and soil depth resulted in restrained canopy growth and favourable conditions for understory development. There are also several moisture receiving polygons which have a considerable broad leaf content; these polygons consist largely of red alder which is a common floristic element from the surrounding landscape. In other areas, long parallel windrows have maintained canopy gaps since stand initiation and thus are currently centers of understory diversity. However, these areas are quickly being shaded out by the surrounding conifers. Furthermore, these windrows are the only major gaps within the stand outside of the broadleaf polygons, and the size, shape, and density of the windrows

is not characteristic of gaps found in naturally occurring forests. The remaining areas within the 1978 cut area are characterized by dense canopies and understory exclusion. The understory community is extremely sparse and consists of primarily bryophytes with the odd sword fern and salal. There are no large diameter trees or snags except for several stumps all of which are less then 1 meter tall. Aside from the stumps, there is almost no CWD on the forest floor outside of the windrows. That said, there are many small Douglas-fir snags (dbh <10cm) throughout the stand as a result of competitive exclusion. However, given there size, these snags have limited value as wildlife trees. There are very few lower canopy trees or tall shrubs which translate into a lack of vertical canopy depth and continuity. That said, there is an abundance of ladder fuel branches. The stand is at a stage where the lower branches have died but have not yet dropped; given the density of the stand this can be a significant fire hazard where historically infrequent non-catastrophic ground fires were the dominant natural disturbance. There are also no root wads and holes which are a natural component of forest floor structural complexity. Finally, the forest floor is very shallow (~4-6cm) and consists primarily of fine twigs and Douglas-fir needles.

6.7 Restoration Treatments

The Galiano Conservancy Association began restoration in District Lot 63 in 2001. In order to promote local genetic integrity and species diversity, native naturally regenerated tree species were given precedence over planted tree stalk during thinning. In other words, only planted Douglas-fir were thinned. Thinning techniques included topping and girdling trees, and pulling over trees. These actions mimicked both density dependant

(natural self thinning) and density independent (windthrow) mortality (Section 4). The purpose of these different thinning approaches was manifold and included:

a.) increasing structural diversity through the creation of snags, short term CWD, and root wad structures;

b.) growth promotion in existing large trees;

c.) canopy gap creation to encourage understory growth;

d.) a reduction in ladder fuels;

e.) increasing genetic diversity through the release of native Galiano tree seedlings;

f.) increasing species diversity through the release of under represented species such as arbutus (*Arbutus menziesii*); and the release of rare species such as western yew (*Texus brevifolia*).

Additionally, some of the larger pieces of CWD from the windrows were erected to provide immediate large snag complexity. The large quantities of wood piled in the windrows were redistributed across the forest floor to increase forest floor complexity and nutrient cycling. Furthermore, all downed thinned trees were left on site to contribute to CWD volume. Some under planting with Western redcedar (*Thuja plicata*) was also done.

6.5 Hypothesis and Predictions

The first objective of this study was to determine if the restoration was successful in terms of changes in structural attributes. Indicators of successful restoration were defined

as a significant increase in the size and number of snags, the amount of CWD, understory species richness and abundance, and a significant reduction in the density of planted Douglas-fir or canopy cover. The null hypothesis was that there would be no difference in these structural attributes between treatment and reference areas indicating that the restoration was unsuccessful. However, I predicted that all of these attributes had changed as a result of treatment.

The second objective of this study was to determine how the restoration affected soil carbon and nitrogen at different depths. Because the treatments took place over different seasons and years, soil is highly variable over space and time, and there is a myriad of direct and indirect pathways related to changes in soil carbon and nitrogen, this was a more complicated endeavour. It was predicted that soil carbon and nitrogen would be most affected in the forest floor and the first mineral layer, and any effects would become less pronounced with depth. Predicted mechanisms driving these changes include an increase in Douglas fir needle litter from thinned trees, a change in the quality and quantity of understory litter, an increase in dead tree root biomass below the forest floor as a consequence of thinning, an increase in carbon volitization from the soil associated with soil disturbance related to pulling over trees, and an increase in carbon leaching into the soil from an increase in the quantity of decaying wood and leaf litter on the soil surface. In the scope of this study it was not possible to measure some of these potential mechanisms directly and it was assumed that tree thinning could be used as a proxy for increases in leaf litter and dead root biomass, and species richness for quality of understory litter. The first null hypothesis was that there would be no difference in soil

carbon and nitrogen means between treatment and reference areas at each soil depth. The second null hypothesis was that soil carbon and nitrogen would show no relationship between the numbers or volume of trees thinned, the number of understory species present, or the amount of CWD on the forest floor.

Chapter 7: Methods

7.1 Plot Locations

I used ARC-GIS maps created by the Galiano Conservancy Association prior to treatment to determine polygon boundaries outlining the location of the two plantation age cohorts and different treatment years and seasons within each cohort. Polygons excluded from the study include all areas treated after 2006, areas identified by the conservancy as having a high degree of natural variation as compared with the rest of the study site prior to restoration treatments, areas noted as having high mycelial abundance, and the windrows. These polygons were excluded to reduce the amount of 'noise' in the data. Furthermore, windrows were excluded from the study because it was assumed that the soil below the windrows would have a much higher carbon content then the soil throughout the rest of district lot 63, and because it would be extremely difficult to safely move the piles of decaying wood to booth accurately measure the wood and access the soil for sampling purposes. The sampling pool was further reduced to the Douglas-fir Salal site series, and areas with similar slope, aspect, slope position, drainage patterns, and elevation.

A stratified random sampling strategy was used in this study. The study area was stratified into six different polygon groups according to the year and season when restoration treatments where performed (Table 3). This was done to capture changes in

vegetation related to the timing of the treatment relative to the timing of the observation. The area within district lot 63 that had had no treatment was used as a reference group.

A minimum of three 10 x 10 meter plots were then randomly created within each polygon group. This plot size was chosen because in some areas it was just below the minimum distance between windrows. These fixed area plots were used for all sampling approaches in order to effectively integrate the methods for assessing measured forest variables and tease apart their interactions. Several different sampling protocols were used. For example, although the soil samples were collected in accordance with Canada's national forest inventory methods, their ground sampling guidelines for site characteristics and vegetation could not be used because this would have involved destructive sampling which was considered inappropriate for this restoration site. And in the case of coarse woody debris, fixed area plots represent a deviation from provincial standards. However, BC's use of line intersect sampling is not a function of increased accuracy or precision, but rather a reflection of efficiency and because of its historical use in fuel loading assessments (Marshall and Davis 2002). Furthermore, using fixed-area plots to measure this attribute is not without precedent (Pesonen et al. 2009; Woldendorp et al. 2004; Woodall et al. 2009; Marshall and Davis 2002).

Randomization occurred in the office prior to field work, and the locations of the plots were found using gps equipment (Thales Mobile Mapper). Originally, the total number of sample plots was 24, the total number of reference plots was 8. This sample size was determined based on the expected effect size of the treatments on the physiognomic

characteristics of the stand and the resources available for the study and soil sample analysis. However, two of the treatment plots were later found to have been placed over dismantled windrows, and one of the reference plots was too close to the district lot boundary; these plots were discarded. Each plot was laid out with a measuring tape and compass using cardinal compass points from the pre-determined center of the plot.

Table 3: Stratified random sampling strategy

Year	Season	Plot #s
2004	Growing Season	09, 10, 11, 17, 18
2004	Dormant Season	19, 21, 22
2005	Growing Season	05, 13, 14
2005	Dormant Season	15, 16, 20
2006	Growing Season	01, 02, 08
2006	Dormant Season	03, 04, 06, 07

7.2 Vegetation and Stand Structure Measurements

Within each sample plot the following measurements were taken: a.) understory species richness and abundance; b.) CWD fragment number, volume and decay class; d.) tree species richness, cover, diameter at breast height (dbh), height; and e.) snag counts, dbh, height, decay class, and treatment where applicable.

Bryophytes, which are ubiquitous throughout lot 63, were excluded from the study due to time restrictions. All vascular understory species were identified (Pojar and Mackinnon 1994), recorded, and abundance estimated using the guidelines for describing vegetation outlined in the Field Manual for Describing Terrestrial Ecosystems (BC MOF 1998).

Specimens of unknown species were collected for verification, put in a plant press, and either brought to the herbarium at the University of Victoria for identification or consulted on with an expert. Each plot was visited in both the spring and late summer of 2008 to capture the full range of species on site. The grass family in BC is very large and complex, and in several cases it was not possible to identify grasses and sedges to the species level because they were not flowering at the time of collection. For this reason, the abundance of graminoids was recorded for each plot but not species richness.

All trees in all plots were recorded and measured. This included species, treatment where applicable, if it was alive or dead, decay class where applicable, wildlife use, diameter at breast height (DBH), crown cover, height to live crown, and tree height. For all snags, decay class was determined using a combination of the visual appearance codes and crown condition codes for wildlife trees (BC MOF 1998). All trees were observed for wildlife use including open nesting, cavity nesting, denning, roosting, and feeding (BC MOF 1998). DBH, i.e. 1.3 meters above ground, was measured for all trees, and on slopes breast height was measured from the high side of the tree. Crown cover was measured using the point-intercept method through the center of each plot at 2 meter intervals for a total of six points. Height to live crown, which was defined as the height on the stem at which live branches occupy about three-quarters of the stem circumference (BC MOF 1998), and tree height measurements were taken with either a clinometer and tape or a hypsometer depending on equipment availability. Tree volume for each species was calculated using the Interactive Tree Compiler program (BC MOFR, 2010).

Coarse woody debris (CWD) has been defined as all non-living tree biomass and may include whole fallen trees, downed branches, fragments of wood, stumps, and standing dead trees (Woldendorp et al. 2004; Woodall et al. 2009). For the purposes of this study, snags have been separated out into a different component to reflect booth stand level changes associated with the thinning treatments and for the assessment of wildlife habitat. Stumps that were <1m tall were classified as CWD, whereas stumps that were >1m tall were classified as snags (McGee et al. 1999). Approaches for measuring CWD in relation to minimum diameter, multiple segments, broken ends, crooked and curved pieces, branched and forked pieces, buried pieces, and windfall were done according to the Field Manual for Describing Terrestrial Ecosystems (BC MOF 1998). For example, only the portion of downed dead wood above a 7.5 cm diameter threshold and above the forest floor was included in this study (BC MOF 1998). Within each plot, each piece of CWD was measured using a tape measure and dbh tape; this included the length, diameter at each end, and decay class of each piece. For pieces that fell on the plot boundary, only the sections of the piece that was within the plot were measured.

7.3 Soil Sampling Procedure

Soil samples were collected in the center of each plot in the fall of 2008 prior to the rainy season. Soil samples were collected in accordance with the detailed procedures for soil bulk density sampling outlined in Canada's National Forest Inventory Ground Sampling Guidelines Version 4.1.

In the center of each plot, all living vegetation, woody debris, green moss, and lichens were removed from a 20 cm x 20 cm sampling frame. The forest floor was cut along the inner surface of the frame and separated from the surrounding soil. The entire volume of the forest floor within the confines of the sampling frame was collected. Rocks and pebbles collected with the forest floor material were discarded. The depth of the forest floor organic sample within the excavation was measured at four different locations, recorded and averaged. This measurement was used to calculate the volume of the excavation and in turn the bulk density of the forest floor organic layer.

Multiple soil samples were collected at multiple depths to determine the variation in soil carbon and nitrogen content by depth. It was assumed that the restoration treatments would have a larger impact on the upper soil horizons then the lower soil horizons. Furthermore, the national protocol calls for more samples to be taken from the upper horizons then the lower horizons. For these reasons, forest floor and mineral layer one samples were taken for all plots, whereas mineral layer two samples were taken for 2-3 plots per polygon group, and mineral layer three samples were taken from one plot per polygon group.

7.4 Soil Sample Lab Analysis

Soil samples were analyzed according to the Canadian Forest Service Pacific Forestry Centre Chemical Services Lab Procedures for the National Forest Inventory (Ministry of Sustainable Resource Management 2005).

7.4.1 Forest Floor Total Carbon, Total Nitrogen, and Total Sulfur

Forest floor soil samples were air dried for 14 days and separated into the following components: cobbles and stones; live roots; > 8 mm/ non forest floor; < 8 mm. Each component of each sample was then weighed and the mass recorded. For each sample, the < 8 mm sample fraction was homogenized and a representative portion was selected. From this selection a subsample was ground in a Wiley Mill to pass 1 mm. The 1 mm sample was then re-dried at 70 degrees C for 24 hours. Total Carbon, Total Nitrogen and Total Sulfur were then determined on the 1 mm material by the LECO CNS 2000 Elemental Analyzer and the values were recorded.

7.4.2 Mineral Soil Total Carbon, Total Nitrogen, and Total Sulfur

Mineral soil samples were air dried for 14 days and separated into the following components: < 2 mm; 2 mm- 75 mm (gravel); > 75 mm (cobbles); organic matter (root mass). Each component of each sample was then weighed and the mass recorded. In none of the samples did organic material account for more then 1% of the total mass of the sample, therefore this fraction was discarded. A sub-sample of the homogenized < 2 mm fraction was removed from each sample and ground to 100 mesh in the Siebtechnic Mill. Total Carbon, Total Nitrogen and Total Sulfur were then determined on the 1 mm material by the LECO CNS 2000 Elemental Analyzer and the values were recorded.

7.4.3 Soil Bulk Density, Carbon Density, and Nitrogen Density

Bulk density is defined as the mass of dry soil per unit of bulk volume, including the air space (Brady and Weil, 2002). It is necessary to calculate bulk density in order to

determine the relative carbon and nitrogen densities of each soil sample. For forest floor bulk density, a 20 cm x 20 cm template was used to delineate the boundary of the sample, and the forest floor including litter and humus was cut, removed, and bagged taking care not to compress the soil sample underneath. The volume of each sample was calculated by averaging the depth of the four sides of the sample and multiplying this average by the dimensions of the frame (Ministry of Sustainable Resource Management 2005). For each mineral soil sample, a 15 cm diameter template was used to delineate the boundary of the sample and mineral soil was extracted to a minimum depth of 15 cm and a maximum depth of 20 cm. Extreme care was taken not to compact the sides of the hole to avoid affecting bulk density. The volume of mineral samples was calculated by filling the hole with glass beads, then measuring the volume of the beads in each hole (Ministry of Sustainable Resource Management 2005). The volume of all soil samples was necessary to calculate bulk density. The calculations for bulk density and soil nutrients in tons per hectare are as follows:

Forest Floor Bulk Density < 8 mm = (Mass total – Mass gravel – Mass live roots – Mass forest floor > 8 mm) / (Volume forest floor sample – (Mass gravel / 2.65) – ((Mass forest floor > 8 mm + Mass live roots) / 0.5))

Mineral Soil Bulk Density < 2 mm = Mass of oven dried < 2 mm soil / (Volume mineral soil sample – ((Mass cobbles + Mass gravel) / 2.65) – (Mass dried roots / 0.5))

Converting % C & % N into tons/ha = % C * Bulk Density * Soil layer depth

7.4.4 Mineral Soil Texture

Soil texture was determined through the Particle Size Analysis by Bouyoucos Hydrometer Method (Kalra and Maynard 1992). A sodium hexametaphosphate dispersion solution was made to break up any residual soil aggregates and the ASTM 152H soil hydrometers were calibrated. A portion of each of the 2 mm air dried mineral soil samples were mixed with the dispersion solution and distilled water, mixed with a motor mixer, transferred into sedimentation cylinders. Two replicates and a control were included in each batch. The liquid solution was left to equilibrate and reach room temperature over night. The contents of each cylinder was then mixed with a plunger, the hydrometers lowered into the suspension, and readings recorded at 40 seconds, and 120 minutes, and the temperature of the control was recorded. Corrections for temperatures were made. Calculations were as follows:

- A = Silt + Clay (%)

  = (corrected hydrometer reading @ 40 seconds/ sample weight)*100
- B = Clay (%)

  = (corrected hydrometer reading @ 120 minutes/ sample weight)*100
- Silt (%) = (A) – (B)
- Sand (%) = 100 – (A)

Texture class of each sample was then determined by referencing the calculated values for % Sand and % Clay on the texture triangle. The point of intersection of these two values determined the textural class.

7.4.5 Forest Floor and Mineral Soil pH

The pH meter was calibrated using buffers pH 4 and pH 7. A 0.01 m calcium chloride solution was created; this solution was mixed with both organic and organic soil samples of a pre-determined weight. Two replicates were included in each batch. These suspensions were stirred 2-3 times over a 30 minute period, and then they were left to stand for an hour to enable the suspended clay to settle. The pH of the soil was then measured by immersing the combination electrode into the supernatant solution, and the values were recorded.

Chapter 8: Statistical Analysis

8.1 Structural Analysis Overview

The purpose of this analysis was to determine if there was a significant difference in structural attributes between treatment and reference plots. In other words, was the restoration successful at increasing the number of snags, the dbh of snags, the volume and fragment number of coarse woody debris, and species richness. Figure 3 is a schematic diagram outlining the statistical analysis process used in this study.

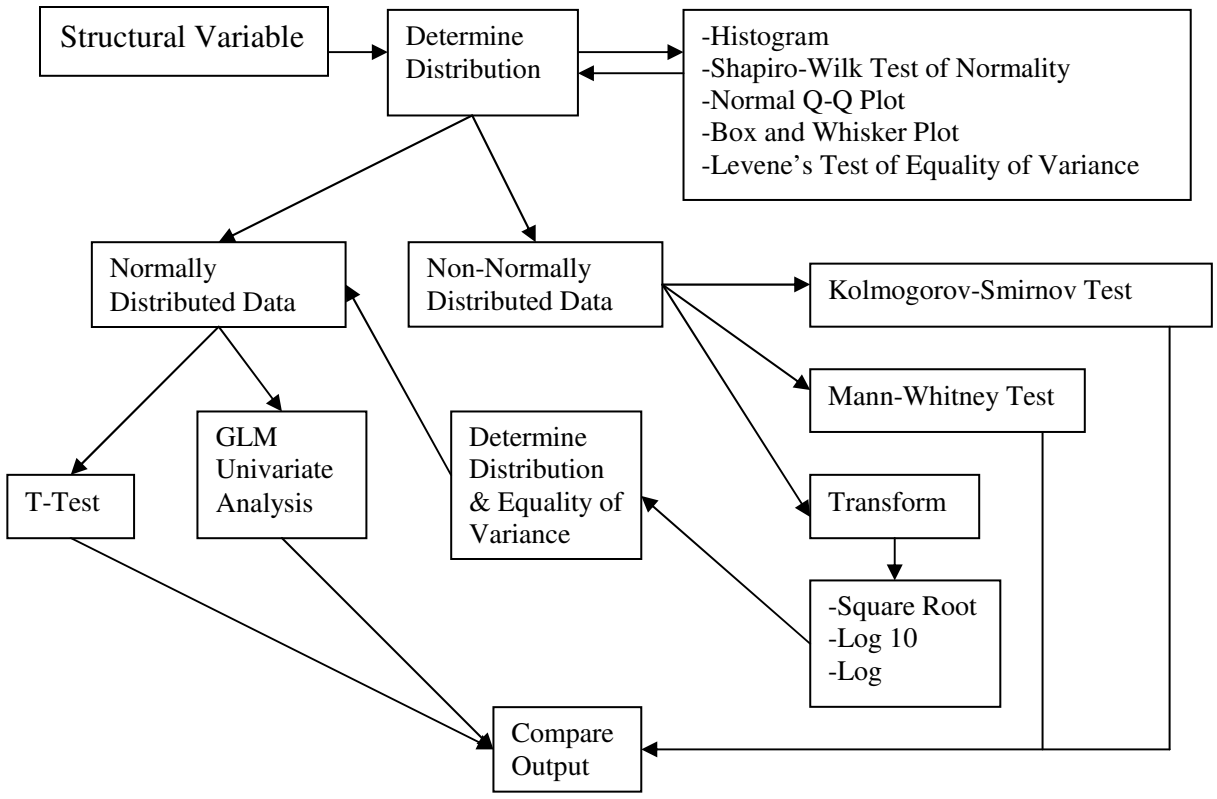

Figure 3: Stand structure statistical analysis overview

It was not necessary to differentiate between year and season of treatment because these factors did not influence the number or spatial arraignment of the structural variables. For each structural variable, the distributions of the data for both the reference plots and treatment plots were determined. If the data showed a normal distribution, both a t-test and a General Linear Model Univariate Analysis were performed and the results were compared. If however, the data was non-normal, both a Kolmogorov-Smirnov and Mann-Whitney test was performed and the results compared. Furthermore, all non-normal data was transformed using square root, log 10, and log; if the transformations resulted in normal data the above parametric tests were also used and compared with the non-parametric results.

The decision to use multiple test statistics was based on the idea that different tests have different strengths. The un-paired t-test was used because it is the most commonly used method to evaluate the differences in means between two groups, and can be used even if the sample sizes are relatively small providing the data are normally distributed within each group and the variation of values in the two groups is not reliably different. The General Linear Model Univariate Analysis was used to conduct an analysis of variance and was included because the GLM has been designed to accommodate unbalanced designs and calculates the post-hoc Observed Power of the test statistic. Notably, there is some debate in the literature pertaining to the validity of post-hoc power analysis; it has been suggested that the logic underlying post-hoc power analysis is fundamentally flawed (Levine and Ensome, 2001), and because power is mathematically directly related to p-values, calculating power once you know the p-value associated with

a statistic adds no new information (Hoenig and Heisey 2001). The two-sample Kolmogorov-Smirnov Test (KS-test) and the Mann-Whitney test (MW-test) are both alternative non-parametric methods of the two-sample t-test. The KS-test is used to test whether two independent samples come from the same sample, or can be considered to be significantly different; The KS-test tests if the maximum absolute difference in cumulative distributions of the two groups is large enough to be significant, in which case the two groups are found not to be from the same distribution. The MW-test is the most often used non-parametric significance test for comparing two independent samples; The MW-test takes the difference between mean ranks of these two samples as the statistic. In short, the KS-test tests differences in the shapes of the distributions of the two groups, where as the MW-test tests the locations of the ranks of the two groups. Appendix A shows a condensed version of the analysis. In the cases of natural snag density mean dbh, CWD volume, red alder volume and red alder density, none of the transformations resulted in a normal distribution so only non-parametric statistics were used in these cases. Furthermore, although the Shapiro-Wilk test for both natural snag density and total snag density after transformations indicates that these variables have non-normal distributions, a visual examination of Q-Q plots and histograms shows that they are very close to normal. For this reason parametric as well as non-parametric statistics were applied to these variables. Levene's test of Equality of Variances was used to verify the assumption of equal variances across samples; in the case of residual Douglas-fir volume and residual Douglas-fir density equal variances can not be assumed. The results of this analysis will be discussed in the next chapter.

8.2 Soil Nutrient Analysis Overview

The purpose of this analysis was threefold. First, I wanted to determine if there was a significant difference in soil carbon and nitrogen between treatment and reference plots, years of treatment, and season of treatment; In other words, did the restoration treatment significantly affect soil carbon, nitrogen, the C:N ratio, and pH? Second, I wanted to determine if the significant differences that were found were correlated with specific changes in stand structure and composition that resulted from the treatments; In other words, were the changes in soil carbon and nitrogen more closely related to one structural change then another. Third, I wanted to rule out the possibility that the significant differences observed were a result of natural variation; In other words, were the differences in soil carbon and nitrogen related to differences in alder density or soil texture. Figure 4 is a schematic diagram outlining the statistical analysis process used in this portion of the study. The right side of the diagram shows how the analysis for differences in soil variables between treatment year, season, and reference sites were addressed, where as the left side of the diagram shows the correlation procedure which pertains to purpose two and three.

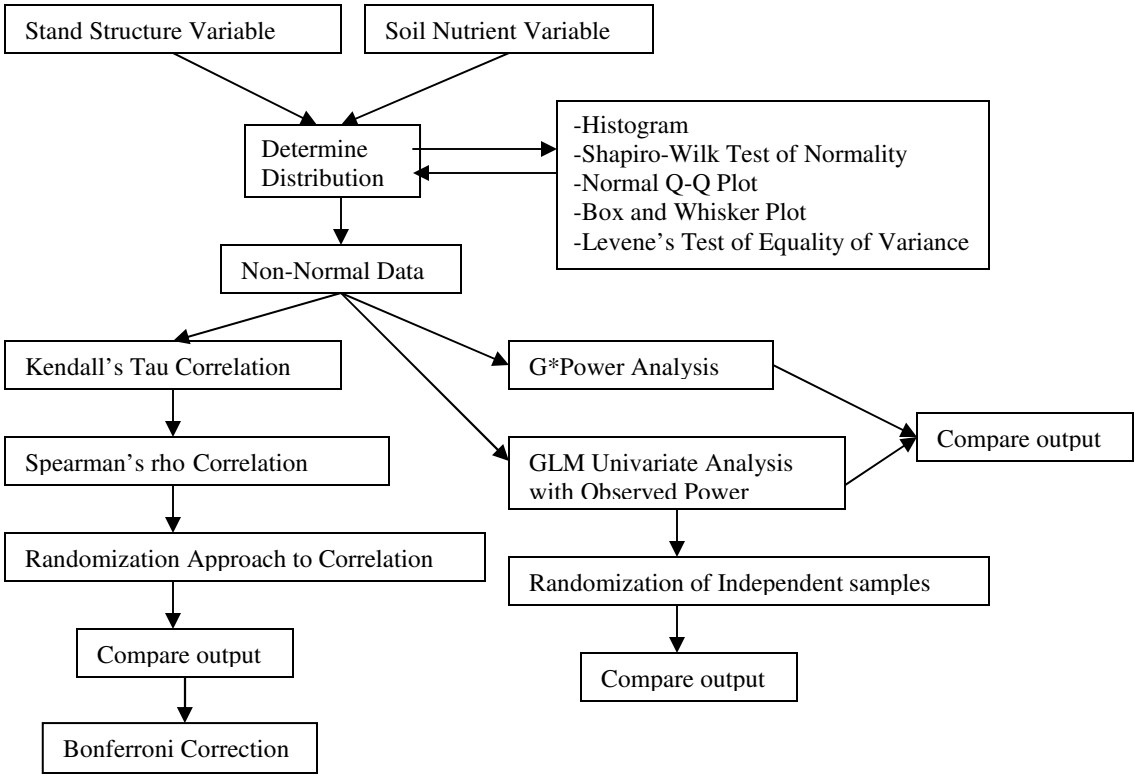

Figure 4: Soil nutrient statistical analysis overview

Originally I was going to conduct an ANOVA for my analysis; however, I decided that this approach was not adequate because after conducting exploratory statistics to determine the distribution of the data, variances etc. I determined that the data did not meet the requirements for that particular statistical test. In other words, my previous assumptions about the amount of variability in the soil and the effect sizes of the treatments were incorrect. I used G*Power software (Erdfelder et al. 1996) to conduct a power analysis of all of the soil chemical parameters including main factors such as year of treatment and potential interactions associated with season of treatment. Power is the probability of detecting an effect given that the effect is actually there. Effect size is the difference of two group means divided by the pooled standard deviation. A very general

rule of thumb is that a small effect is 1% of the variance (d=0.25), a medium effect is 6% of the variance (d=0.5), and a large effect is 15% of the variance (d=0.8) (Cohen 1988). However, it is important to note that just because an effect is statistically significant, it does not necessarily mean that the effect is biologically significant. In other words, it is possible to statistically detect small effect sizes but they are generally not considered biologically relevant. The reason I did this analysis was to determine power, given the magnitude of the effect sizes observed and the number of samples taken. I used the post hoc option first to determine the power values given my sample sizes, effect sizes, and alpha level. I used the a priori option second to determine how much larger the sample sizes would need to be given effect sizes, the alpha level and desired power value. The general trend of the analysis indicated that, given the small effect sizes, relatively small sample sizes, variability within the data, unbalanced design, and non-normal distributions, in some cases I would need hundreds of more samples in order to detect differences with a power of 0.8 or higher. Furthermore, in some cases there was reasonable power to detect a main effects model (year of treatment), but not enough power to when the model included interaction (year and season of treatment). Given the cost of soil sample analysis and time required to collect said samples, it was determined that an additional field season was not feasible. For this reason I conducted independent samples randomization tests to detect significant differences in changes in soil variables.

Randomization was chosen because it can be applied to t-tests and f-tests through data permutation without any parametric assumptions being fulfilled (Hooton 1992). In other words, small sample sizes, unknown error distributions, unequal variances, unbalanced

designs, and missing values can be easily accommodated (Hooton 1992). Randomization permutations are not concerned with what kinds of populations or parameters lie behind the data, only how likely it is that, if variable x is independent of treatment, I would get the particular numbers that I have distributed across groups in the way that they are (Howell 2007). Many people are now promoting the use of randomization tests even when parametric and non-parametric tests are applicable (Palmer 2010). Randomization tests answer the question "How likely is it that if the null hypothesis were true, I would observe a value this extreme just due to chance?" The procedure for a randomization test is as follows (Palmer 2010):

1. Formulate a test statistic
2. Define the null hypothesis
3. Create a new data set consisting of the original data but randomly re-arranged
4. Calculate the test statistic for the re-arranged data set and compare it to the original 'true' data set value
5. Repeat steps 3 and 4 hundreds of times
6. If the 'true' test statistic is greater then 95% of the random values, then you can reject the null hypothesis at $p<0.05$ for a one tailed test (for a two tailed test the cutoff should be 97.5%)

For the randomization test on two independent samples in this study I used a t value; instead of just taking the difference between the means, I also divided this value by the standard error of the difference. I used this because it represents the difference relative to

the variability of the differences (Howell 2007). However, the difference between the means will lead to the same conclusion. I used 1000 iterations, I did this because an increase in the number of iterations gives us a better estimate of p, but does not increase the likelihood of significance (Palmer 2010). This randomization process results in 1000 random reallocations of the original observations to two groups, and because the data was randomly shuffled into two groups, the distribution of these permutations shows what t values we would be likely to obtain when the null hypothesis is true. The distribution of the permutations can also be used to identify the presence of outliers; outliers would turn up in every permutation, sometimes in one group and sometimes in another, and would cause the resulting t value to swing back and forth between positive and negative, creating an unusual sampling distribution. There were no bimodal sampling distributions in my permutations. I then went back to the original data and asked, "would the results that I obtained be likely to arise if the null hypothesis were true?" If the probability of obtaining a t values as extreme as the original data was less then $\alpha = .05$, I rejected the null hypothesis and concluded that there was a difference between the groups under consideration.

Correlations were used to assess the presence and strength of relationships between structural variables and soil variables for all soil layers exhibiting significant differences in the previous component of the analysis, namely the forest floor and the first mineral soil layer. The number of samples for this component of the analysis was higher (n=29) than for the previous analysis which used a range of sample sizes depending on the soil layer being analysed. It is expected that 'noise' in the data resulting from year of

treatment may impede the ability to detect relationships. However, it is not possible to perform these correlations according to year of treatment because the sample sizes would be much smaller then what is recommended for correlations. That said, scatter plots presented with the results do differentiate year of treatment and can be used as a tool to examine and explain the noise in the data. And because parametric assumptions regarding normality and heterogeneity were violated, Kendall's tau and Spearman's rho were used. Spearman's rank correlation is more widely used then Kendall's tau because it is easier to compute. However, the distribution of Kendall's tau has slightly better statistical properties and there is a direct interpretation of Kendall's tau in terms of probabilities of observing concordant and discordant pairs (Conover 1980). Spearman's rho is interpreted the same way a parametric Pearson's correlation coefficient is, i.e., the proportion of variability accounted for. Kendall's Tau, on the other hand, represents a probability, i.e., the difference between the probability that the observed data are in the same order versus the probability that the data are not in the same order.

The randomization approach to correlation was used in all cases where either Spearman's rho or Kendall's tau resulted in a significant result. When we use the randomization approach to correlation coefficients, we permute the Y values, while holding the X values constant. Units are drawn, without replacement, from the rearranged sample values, and a correlation coefficient is computed. After this process has been repeated 1000 times, the resulting correlation coefficients form the sampling distribution of r. This means that the expected value of the correlation between X and Y will be 0.00, not the correlation in the original data (Howell 2001). We then look at the correlation on

the original data, and count how many of the 1000 randomizations exceeded the ± value from the original data. By dividing the number of randomized values that exceed the original correlation by the number of permutation we get the probability under the null hypothesis of an r value exceeding the original "true" r value; if the probability is <.05 we can reject the null hypothesis that there is no correlation (Howell 2001).

Finally, a Bonferroni correction was applied to all statistically significant correlations to account for spurious correlations or type one errors. Scheffé's method was used; this is a very conservative approach. As a result of the Bonferroni correction, the following correlations are not considered statistically significant: within the forest floor- c and cwd fragment #, c and total snag density, c and Douglas-fir snag volume, pH and Douglas-fir density; within the first mineral soil layer- pH and alder volume, and pH and alder density. However, given that these relationships are biologically significant, correlation values prior to the Bonferroni adjustments are reported in the text. Arguments supporting biological significance are outlined in the discussion (Section 10).

Chapter 9: Results and Interpretation

9.1 Structural Analysis Results and Interpretation

The results of the structural analysis are presented in Appendix B. For each of the structural variables, each statistical tests showed slightly different levels of significance, yet all tests with the exception of CWD volume resulted in the same conclusion. For the purposes of this thesis only the Mann-Whitney test statistic results will be reported in the text. When interpreting the t-test results in Appendix B, equal variances should not be assumed for residual Douglas-fir volume because according to Levene's test of Equality of Variances in this case this assumption was violated.

9.1.1 Snag Analysis and Interpretation

There was no statistically significant difference in mean natural snag densities ($z = -0.859$, $p = 0.409$) or natural snag mean dbh ($z = 0.154$, $p = 0.901$) between treatment and reference plots. The number and size of naturally occurring snags was similar throughout the stand irrespective of treatment. The slightly lower number of naturally occurring snags in the treatment areas is likely due to small snags inadvertently being knocked down during the treatment process. The implication is that any changes in total snag density in the treatment areas are a reflection of the treatment. There was no significant difference ($z = -1.715$, $p = 0.088$) in total snag density. Given that a large component of

the treatment involved creating snags through topping and girdling trees, with an average of 5 treated trees per treatment plot, this result was unexpected.

Total snag density takes into consideration all stems above 7.5 cm dbh for all snag species. This includes a substantial number of naturally occurring alder snag stems that are the result of stand self thinning. However, the number of stems in a plot is not synonymous with the number of trees in a plot; species such as red alder (*Alnus rubra*) often have multiple stems per tree, and those plots with many stems are generally plots with high alder content. Furthermore, alder stems generally have a much smaller basal area than Douglas-fir stems. For example, although there is more then three times the number of alder stems than Douglas-fir stems in plot number 9, the total basal area for each species within this plot is very similar. For this reason it may be more useful to look at tree and snag volume. Furthermore, because variable density thinning was employed in the treatment regime, not all treatment plots contained girdled or topped trees. These two factors may explain the noise in the data and the inability to detect significant differences in total snag density between treatment and reference plots.

There was a significant difference in Douglas-fir snag volume ($z = -2.858$, $p = 0.003$) between treatment and reference sites (Figure 5). All created snags were Douglas-fir and there was an average of 5.6 Douglas-fir snags in the treatment plots and 2 Douglas-fir snags in the reference plots. Additionally, the created snags had a significantly larger mean dbh ($z = 4.087$, $p = 0.000$) then the naturally occurring snags (Figure 6). This is biologically significant and will be discussed in more detail in the following chapter. In

summary, there are more Douglas-fir snags in the treatment area, and these snags are larger then those in the reference area.

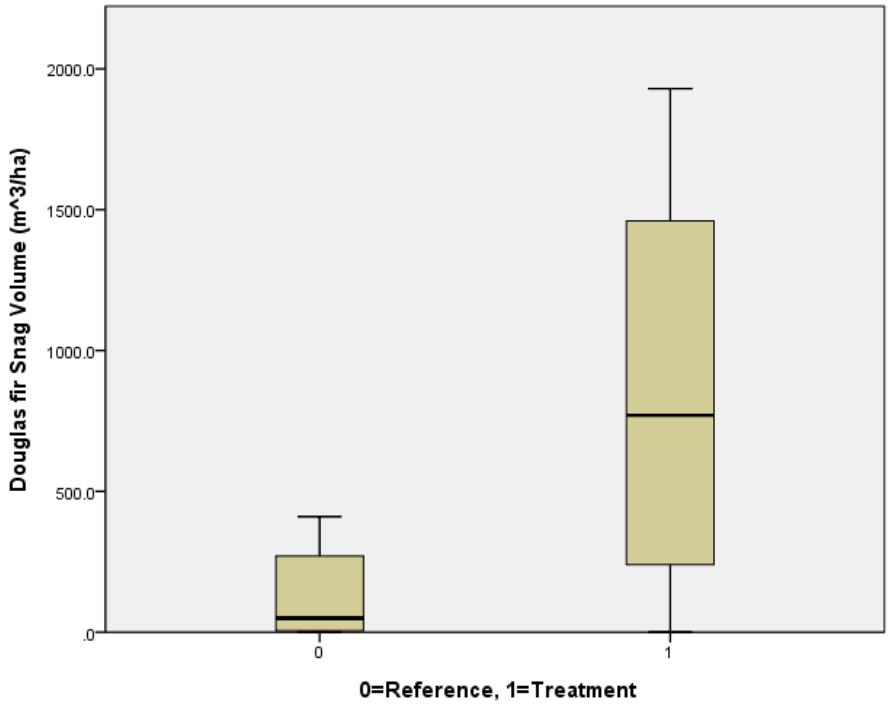

Figure 5: Reference vs. treatment Douglas-fir snag volume

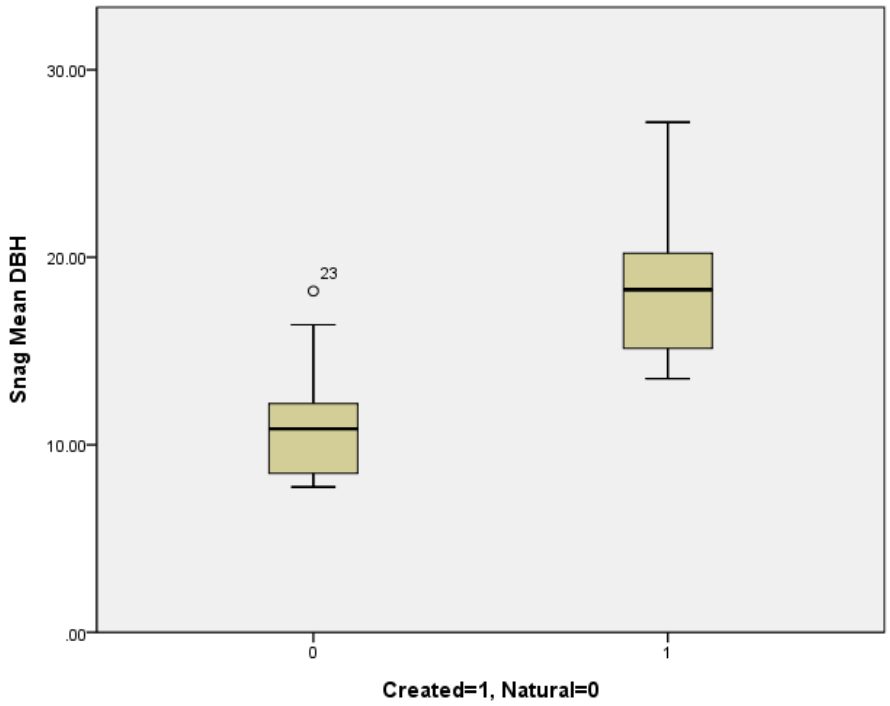

Figure 6: Created snag vs. natural snag mean DBH, points in figure identified by number refer to outliers

9.1.2 Tree analysis and Interpretation

Two species of trees were examined in the analysis, Douglas-fir and red alder. The dominant tree in the stand was Douglas-fir which represented 67% of the trees in the treatment area and 77% of the trees in the reference area. The sub-dominant tree was red alder which represented 23% of the trees in the treatment area and 9% of the trees in the reference area. All other tree species combined represented only 10% of the trees in the treatment area and 12% of the trees in the reference area and thus were considered negligible. These percentages were calculated separately for treatment and reference areas, the only tree species transferred from the living biomass pool to the dead biomass pool was Douglas-fir, so changes in proportions are to be expected. Another reason why

red alder has been included in this analysis is it's well documented influence on soil nitrogen and carbon pools. This will be discussed in detail in the soil analysis and interpretation section. However, from a structural standpoint it is important to note that there was a significant difference in Douglas-fir volume (z = -2.803, p = 0.004) and density (z = -3.598, p = 0.000) between treatment and reference areas (Figures 7 and 8 respectively), but there was no significant difference in red alder volume (z = -0.393, p = 0.709) or density (z = -0.474, p = 0.672) between treatment and reference areas. In other words, there were significantly fewer Douglas-fir in the treatment area then in the reference area, and approximately the same number of red alder in the treatment and reference areas.

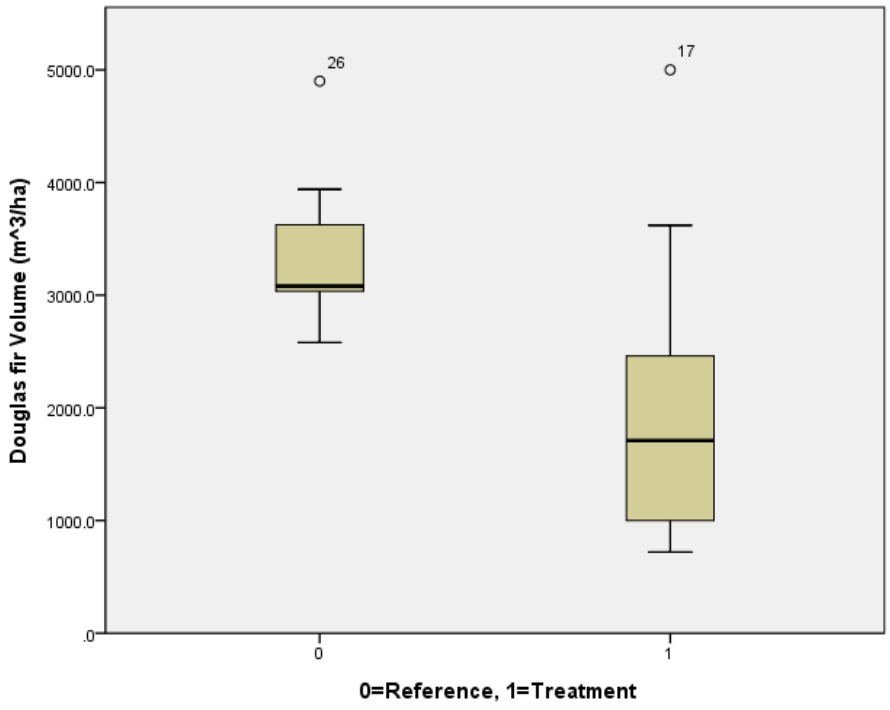

Figure 7: Reference vs. treatment Douglas-fir volume, points in figure identified by number refer to outliers

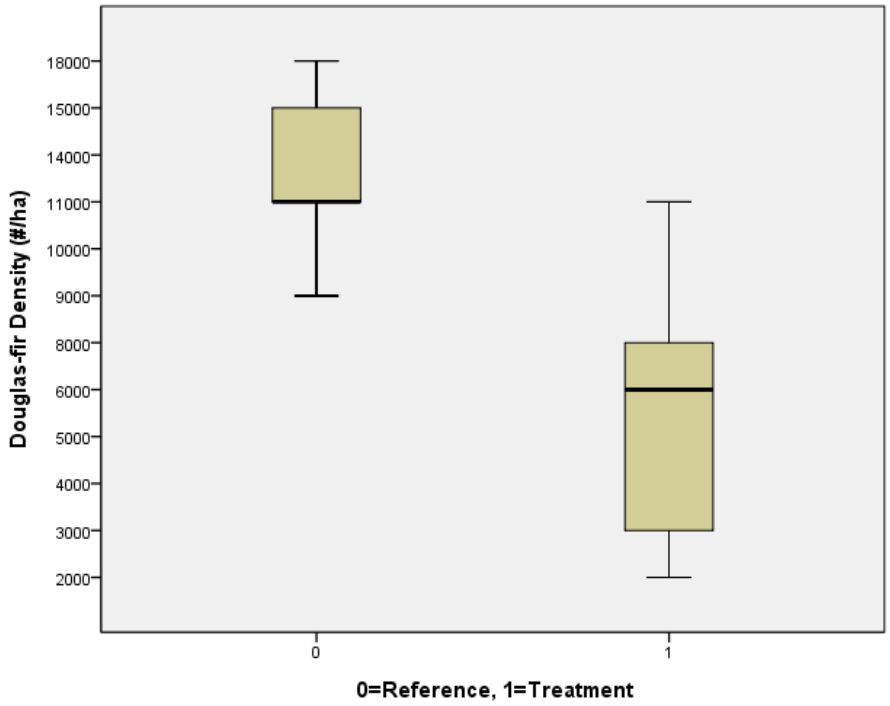

Figure 8: Reference vs. treatment Douglas-fir density

9.1.3 Coarse Woody Debris Analysis and Interpretation

In terms of coarse woody debris, both the number of fragments ($z = 3.628$, $p = 0.000$) and volume ($z = 2.319$, $p = 0.020$) was significantly higher in the treatment area then in the reference area (Figures 9 and 10 respectively). The mean number of pieces across treatment area was 9909 per hectare, whereas the mean number of pieces across the reference area was 2285 per hectare. The mean volume across the treatment area was 954.55 m^3/ha where as the mean volume across the reference area was 288.57 m^3/ha. The interpretation of this difference is that the restoration treatments resulted in a significant increase in the amount of CWD in the treatment area.

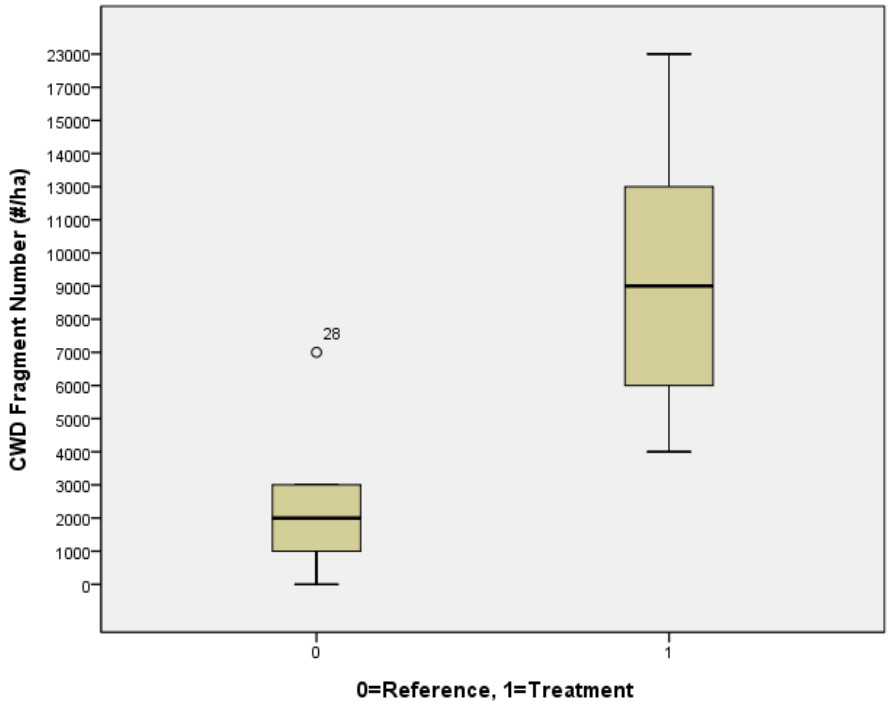

Figure 9: Treatment vs. reference CWD fragment number, points in figure identified by number refer to outliers

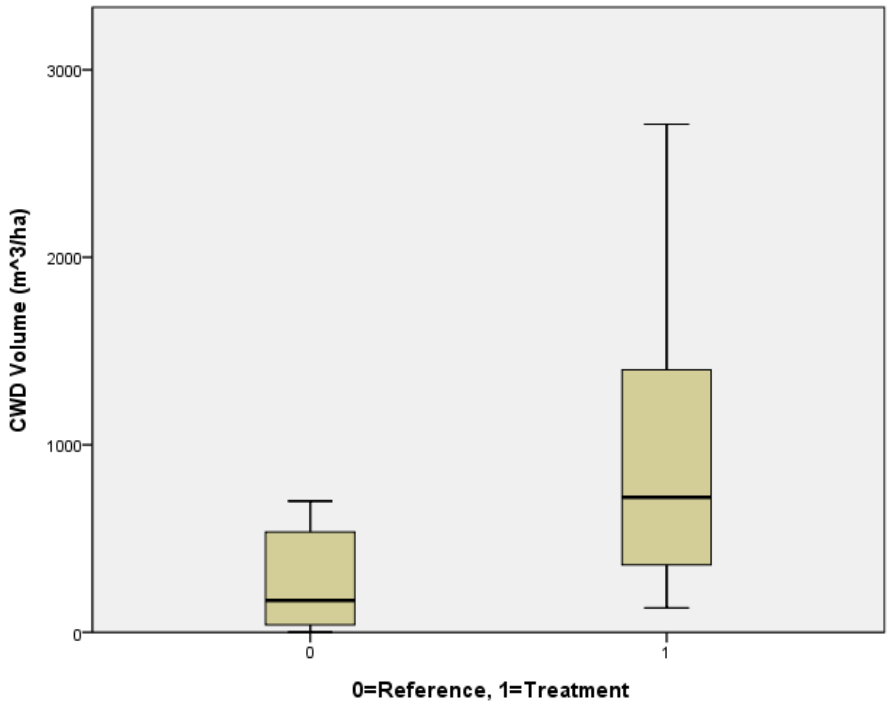

Figure 10: Treatment vs. reference CWD volume

9.1.4 Understory Analysis and Interpretation

The number of understory species in the treatment plots ranges from 6 to 25, with an average of 16 species. In comparison, the number of understory species in the reference plots ranges from 1 to 9, with an average of 4 species. A statistical analysis of species richness showed a significant difference ($z = -3.778$, $p = 0.000$) between treatment and reference areas (Figure 11). There was also an obvious difference in the percent cover of vascular understory species between treatment and reference plots. The percent cover of understory species in the treatment plots ranges from 10 to 100%, with an average of 45% cover. In comparison, the percent cover of understory species in the reference plots ranges from 1 to 20%, with an average of 7% cover. Given the fact that only one of the

control plots contained a percent cover for vascular understory species exceeding 10%, it was deemed that the difference between reference and treatment areas was so evident that statistics were not necessary.

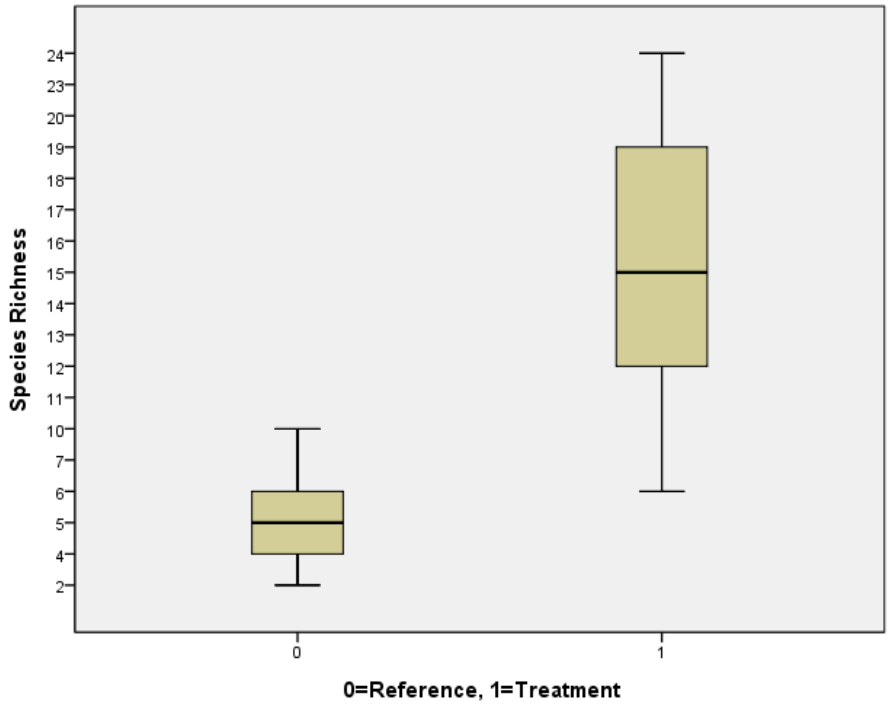

Figure 11: Treatment vs. reference species richness

9.1.5 Nitrogen Indicator Plant Analysis

During the understory vegetation analysis, it was observed that a number of vascular species that had come into the stand were known as indicators of nitrogen rich-medium sites (Klinka et al. 1995). Given that nitrogen is the major growth limiting nutrient element in coastal Douglas-fir forests, this was unexpected. Table 4 shows a summary of the nitrogen indicator plant analysis for the treatment area; no indicator plant analysis was conducted for the reference site because there were not enough species in most plots

to conduct said analysis. The first row in the table represents the plot numbers for the treatment area.

Table 4: Nitrogen indicator plant analysis

Nitrogen Rich Indicators	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**
Acer macrophyllum						x				x	x			x				x		x		
Achlys triphylla																					x	
Alnus rubra	x				x	x	x	x	x	x	x	x		x	x		x				x	x
Claytonia sibirica					x		x	x		x	x		x			x				x		
Galium aparine	x	x	x	x	x		x	x	x		x	x	x	x	x	x		x	x	x	x	
Galium triflorum	x	x				x		x			x								x		x	x
Lathyrus nevadensis		x	x	x	x	x		x	x	x		x				x	x	x	x			
Mycelis muralis	x	x	x			x	x			x	x		x	x		x		x	x	x	x	x
Polystichum munitum	x	x	x	x	x	x	x	x	x	x	x	x			x	x	x	x	x	x	x	x
Rubus parviflous						x																
Sambucus racemosa			x																			
Stellaria crispa	x		x			x													x			
Urtica lyallii	x	x	x			x	x	x			x	x	x	x		x	x	x	x	x	x	x
Nitrogen Medium Indicators	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**
Amelanchier alnifolia			x	x											x							
Cytisus scoparius		x	x		x								x	x	x					x		
Holodiscus discolor						x	x		x				x	x	x	x				x		
Lonicera hispidula	x	x	x		x	x				x	x	x	x	x	x	x	x	x	x	x	x	x
Loncira		x	x								x			x		x					x	

ciliosa																						
Mahonia nervosa	x		x	x	x							x		x	x				x	x		x
Ranunculus occidentalis	x	x	x			x				x		x	x		x	x			x	x	x	
Rosa gymnocarpa		x	x				x							x								
Rubus ursinus	x	x	x	x	x	x	x	x						x	x	x		x	x	x	x	x
Trientalis latifolia			x		x										x	x						
Nitrogen Poor Indicators	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**
Gaultheria shallon	x	x	x	x	x	x	x		x			x		x	x	x		x	x	x	x	x
Goodyera oblongifolia		x																				
Vaccinium parvifolium			x	x		x		x				x		x		x		x	x			x

9.2 Soil Analysis Results and Interpretation

The number of factors related to restoration treatment and influencing soil carbon and soil nitrogen in DL63 was potentially large. These included treatment year, tree species composition , tree density, above and below ground litter inputs as a function of snag density, CWD volume, CWD fragment number, soil nitrogen, soil depth, soil texture, and pH.

A summary of the overall F test which demonstrates if at least one difference exists and significant results of the multiple comparisons which addresses differences in soil carbon, soil nitrogen, C:N ratio, and pH between restoration work done in 2004, 2005, 2006 and the reference site can be found in Appendix C . The fourth column shows the results from the GLM analysis, and the fifth column shows the results from the randomization

analysis. The results of the GLM analysis should be viewed with some scepticism, although the Kolmogorov-Smirnov and Shapiro-Wilk tests of normality showed no significant deviations from normality, examination of box plots, histograms, and q-q plots did not always support this conclusion. Tamhane's T2 was used for the multiple comparisons to take into account unequal cell sizes and the violation of the homogeneity of variances assumption. The randomization results on the other hand, are not subject to these distributional assumptions and are valid. For this reason, all reported significant levels with respect to this component of the analysis are randomization results.

9.2.1 Soil Carbon

Results from randomization analysis show significant differences in soil carbon in both the forest floor and the first 15 cm of the mineral soil. There was a significant difference in forest floor soil carbon between the treatments done in 2004 and 2006 ($p = 0.008$), and the reference site and 2006 ($p = 0.002$). There were significant differences in mineral layer 1 soil carbon between treatments done in 2004 and 2005 ($p = 0.011$), 2004 and 2006 ($p = 0.016$), 2005 and the reference site ($p = 0.007$), and 2006 and the reference site ($p = 0.045$). The only comparisons that did not show a significant difference in soil carbon in the first mineral soil layer was the treatments done in 2004 compared to the reference site, and the treatments done in 2005 compared to those done in 2006 . There were no significant differences in soil carbon between treatment years and the reference site for either the second mineral soil layer (15-35 cm) or the third mineral soil layer (35-55 cm). Figure 12 shows a trend of decreasing soil carbon with time from treatment for the forest

floor; Treatment areas treated in 2006 have higher forest floor soil carbon levels then treatment areas treated in 2004. Figure 12 indicates that over time treatment area forest floor soil carbon can be expected to approach reference area forest floor soil carbon levels. Figure 13 shows a trend of increasing soil carbon with time from treatment for the first mineral soil layer indicating that over time treatment area mineral layer 1 soil carbon can be expected to approach, and potentially exceed, reference area mineral layer 1 soil carbon levels. Potential drivers for these trends include the accumulation of organic matter on the forest floor associated with killing trees and subsequent increased litter inputs, the decay of accumulated organic matter on the forest floor over time, and leaching of carbon from the forest floor to the first mineral layer.

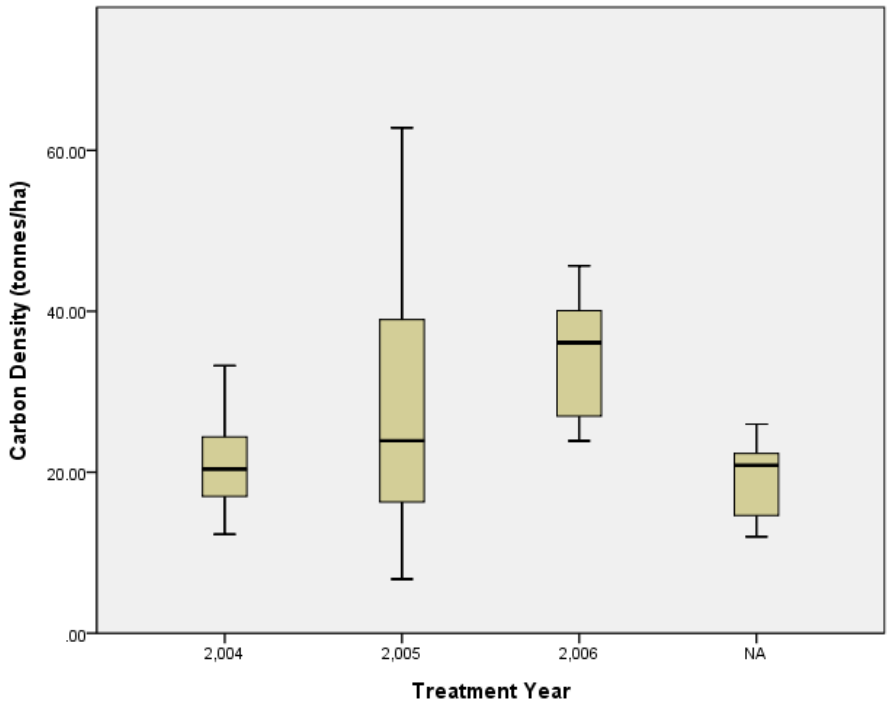

Figure 12: Forest floor soil carbon across treatment years and the reference site

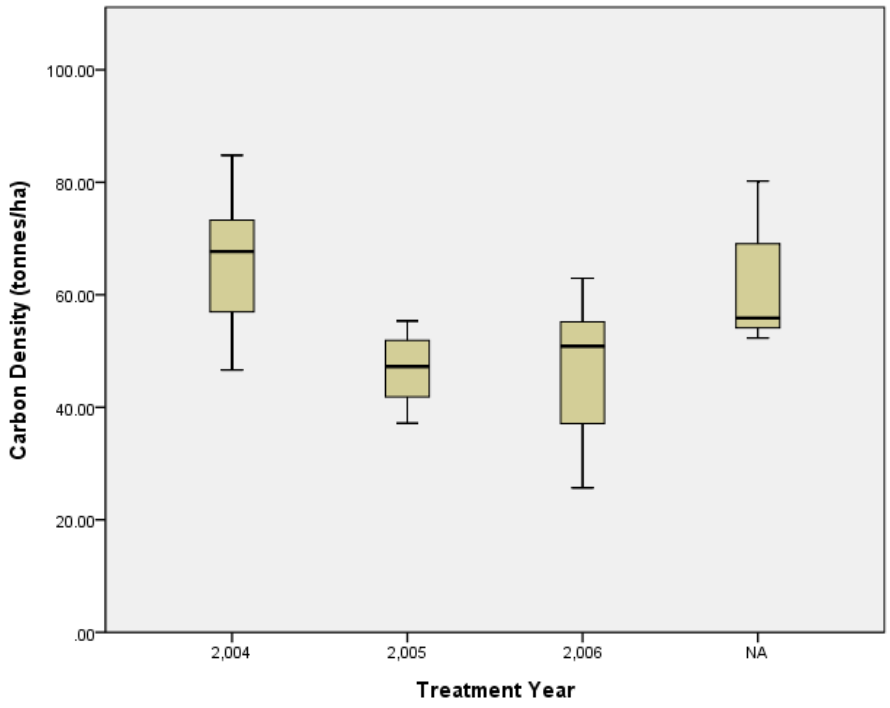

Figure 13: Mineral layer 1 soil carbon across treatment years and the reference site

9.2.2 Soil Nitrogen

The analysis of soil nitrogen showed several significant differences between the reference site and treatments done in different years for both the forest floor and the mineral soil to a depth of 15 cm. Within the forest floor there was a significant difference in soil nitrogen between the reference site and treatments done in 2006 ($p = 0.003$). Within the first mineral layer there were significant differences in soil nitrogen between 2004 and 2006 ($p = 0.019$), and 2004 and 2005 ($p = 0.013$). There were no significant differences in soil nitrogen between treatment years and the reference site for either the second mineral soil layer or the third mineral soil layer. Neither figure 14 or 15 shows a

definable trend for forest floor soil nitrogen or mineral layer 1 soil nitrogen across treatment years and the reference site.

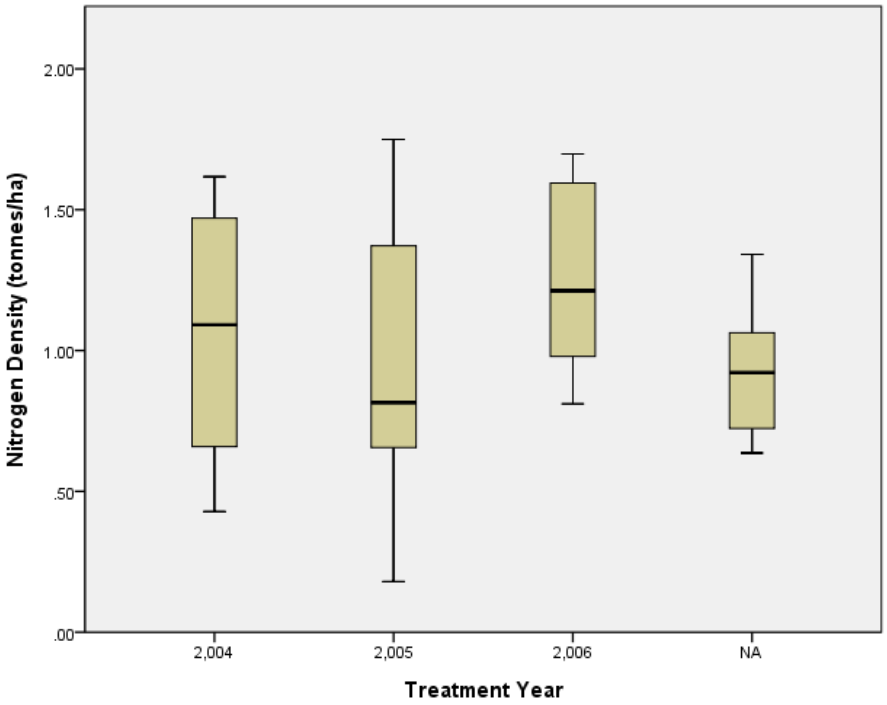

Figure 14: Forest floor nitrogen across treatment years and the reference site

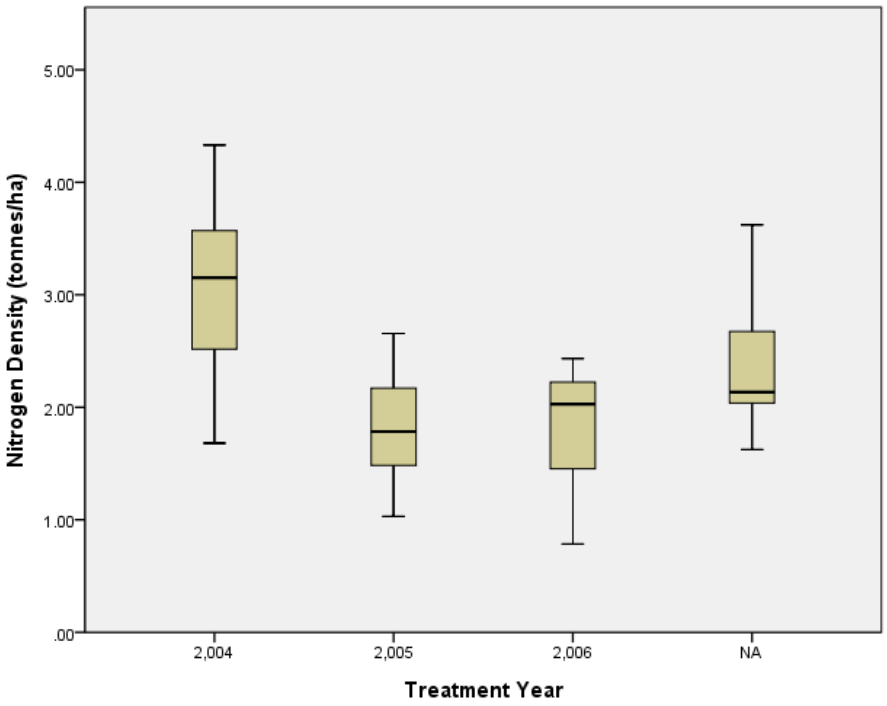

Figure 15: Mineral soil layer 1 nitrogen across treatment years and the reference site

9.2.3 Soil C:N Ratio

The randomization analysis of the soil C:N ratio showed several significant differences between treatment years and the reference site for the forest floor. Within the forest floor there was a significant difference between 2004 and 2005 ($p = 0.001$), 2004 and 2006 ($p = 0.014$), the reference site and 2005 ($p = 0.001$), and the reference site and 2006 ($p = 0.002$). Notably, there were no significant differences between treatments done in 2004 and the reference site for the forest floor or treatment years and the reference site for any of the mineral soil layers. Figure 16 shows the results from this analysis, no trends are readily observable for the forest floor.

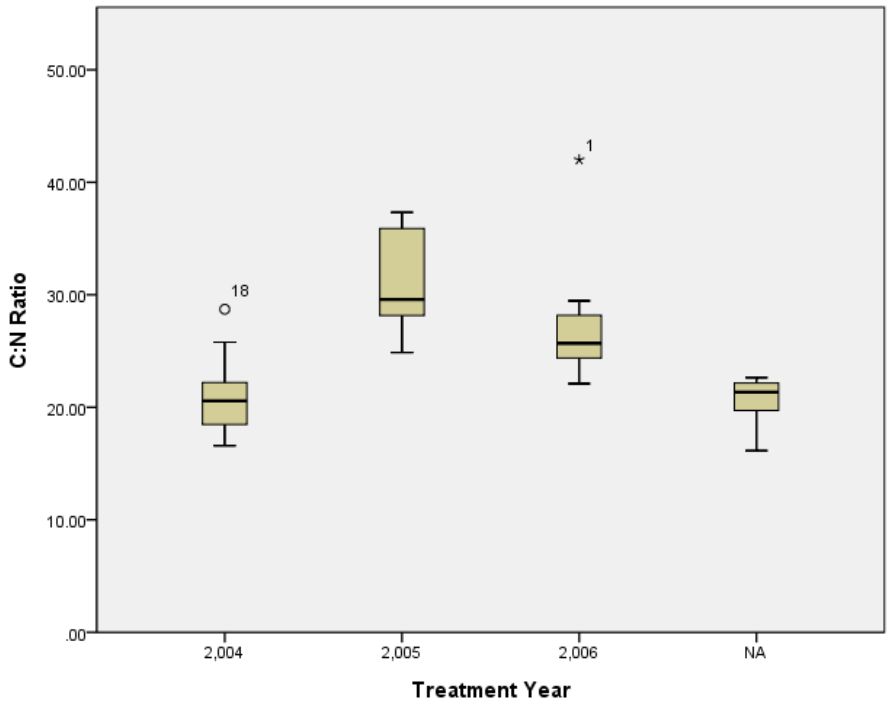

Figure 16: Forest floor C:N ratio across treatment years and the reference site, points in figure identified by number refer to outliers

9.2.4 Soil pH

The analysis of soil pH showed one significant difference between treatment year and the reference site for the forest floor, and two significant differences between treatment years and the reference site for the first mineral layer. For the forest floor there was a significant difference between the reference site and treatments done in 2004 ($p = 0.021$), and for the first mineral layer there was a significant difference between the reference site and treatments done in 2004 ($p = 0.009$), as well as a significant difference in treatments done in 2004 as compared to those done in 2005 ($p = 0.034$). Figures 17 and 18 illustrate these differences. Notably, this is the only soil parameter that shows any significant

difference between the treatments done in 2004 and the reference site; in all other cases areas treated in 2004 are very similar to the reference site.

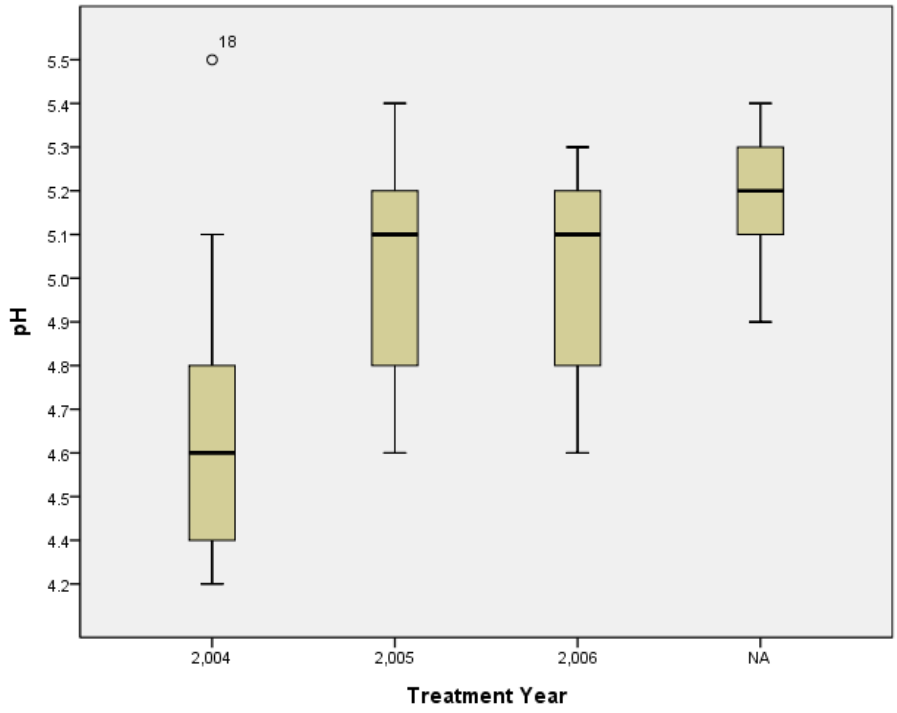

Figure 17: Forest floor pH across years of treatment and the reference site, points in figure identified by number refer to outliers

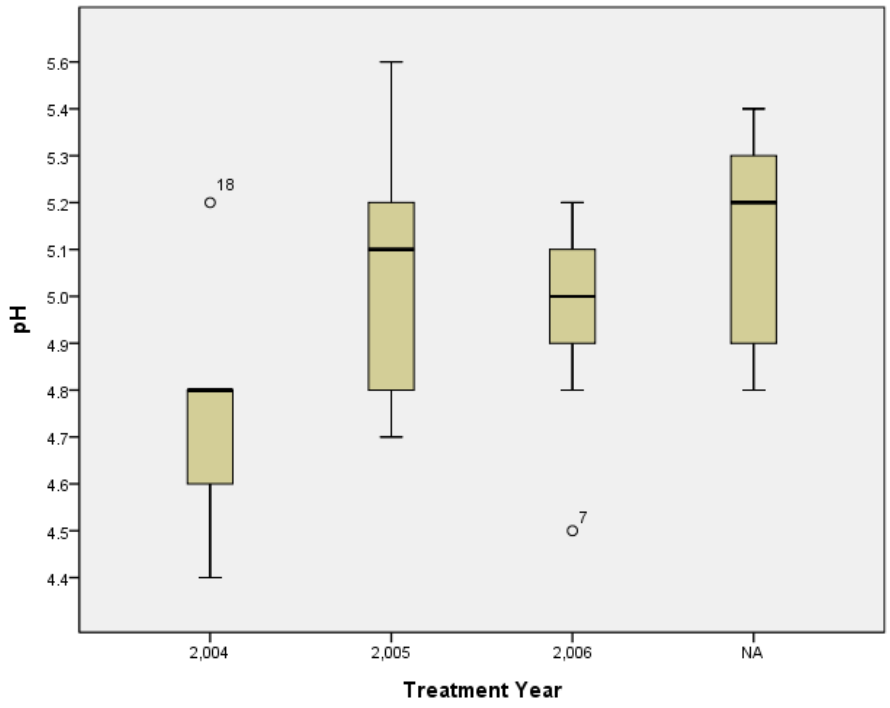

Figure 18: Mineral soil layer 1 pH across years of treatment and the reference site, points in figure identified by number refer to outliers

9.3 Correlations between Structural Variables and Soil Variables

There are some significant differences in soil variables between years of treatment and the reference site for the forest floor and the first mineral soil layer (Section 9.2.1). The current line of inquiry refers to how soil variables and structural variables are related and the strength or magnitude of those relationships as it pertains to the forest floor and the first mineral soil layer. A summary of the results from this component of the analysis can be found in Appendix D. Notably, there were no correlations between either soil carbon and soil texture or soil carbon and alder density. Nor were there any correlations between soil nitrogen and soil texture, or soil nitrogen and alder density. This was unexpected because in general, soil nutrients increase with increased clay content, and soils below

nitrogen fixers contain more nitrogen and carbon. The vast majority of the soil samples were either sandy loams or loamy sands. Furthermore, there were no samples with high clay contents in the first mineral layer, which is where all restoration related soil chemical changes took place.

In the majority of cases all three tests resulted in the same conclusion. However, there are some notable exceptions. Namely, within the forests floor the relationships between carbon and total snag density, carbon and Douglas-fir snag volume, and carbon and species richness and within the first mineral soil layer, the relationship between pH and alder density, and pH and alder volume. All values presented in the text are the Spearman's rho correlation coefficient and significance level.

9.3.1 Forest Floor

The results from this analysis suggest that there is a moderate relationship (rho = 0.473, p = 0.010) between the number of fragments of CWD and forest floor soil carbon. There also appears to be a moderate relationship (rho = 0.414, p = 0.026) between Douglas-fir snag volume and forest floor soil carbon. Figures 19 and 20 show these relationships. Potential reasons for these relationships will be discussed in chapter 10.

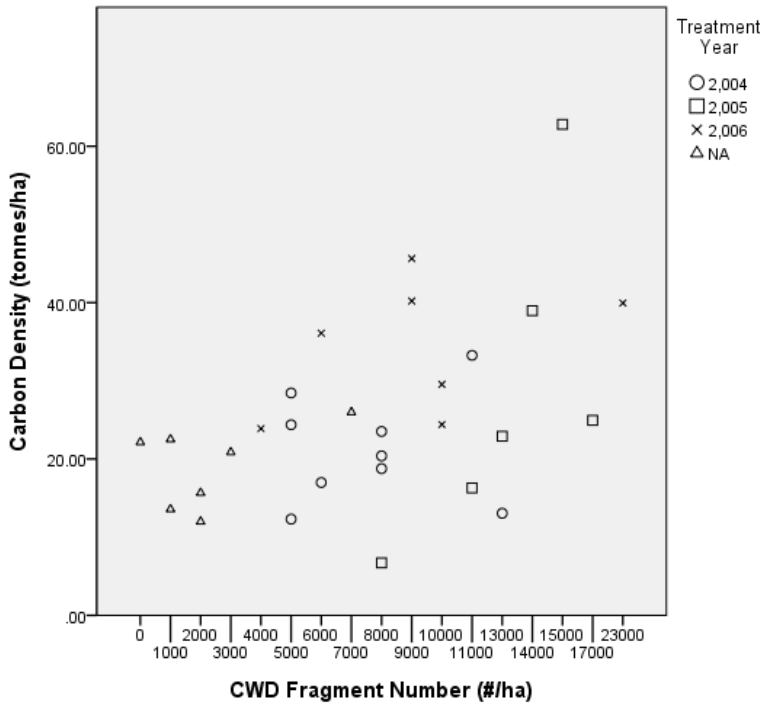

Figure 19: Correlation between forest floor soil carbon and CWD fragment number

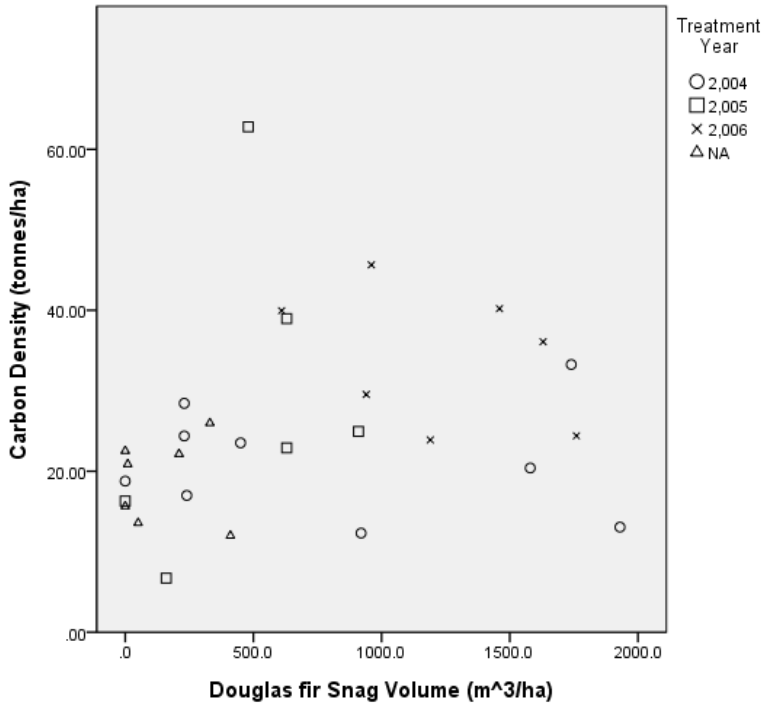

Figure 20: Correlation between forest floor soil carbon and Douglas-fir snag volume

There appears to be a weak relationship (rho = 0.371, p = 0.047) between Douglas-fir snag volume and forest floor pH, a strong negative relationship (rho = -0.616, p = 0.000) between alder volume and forest floor pH, a moderate relationship (rho = 0.487, p = 0.007) between Douglas-fir density and forest floor pH, and a moderate negative relationship (rho = -0.569, p = 0.001) between alder density and forest floor soil pH. Figures 21 through 24 depict these relationships and will be discussed in chapter 10.

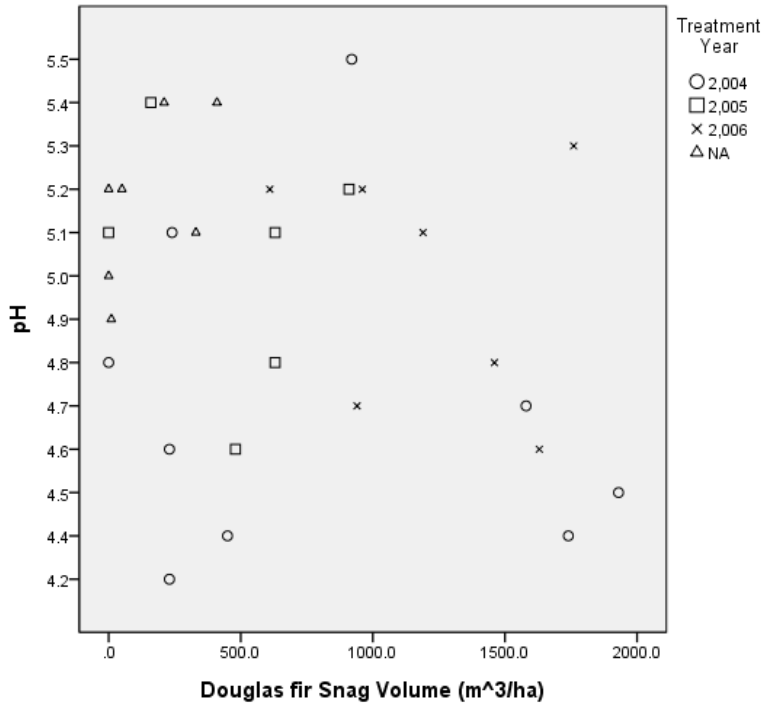

Figure 21: Correlation between forest floor pH and Douglas-fir snag volume

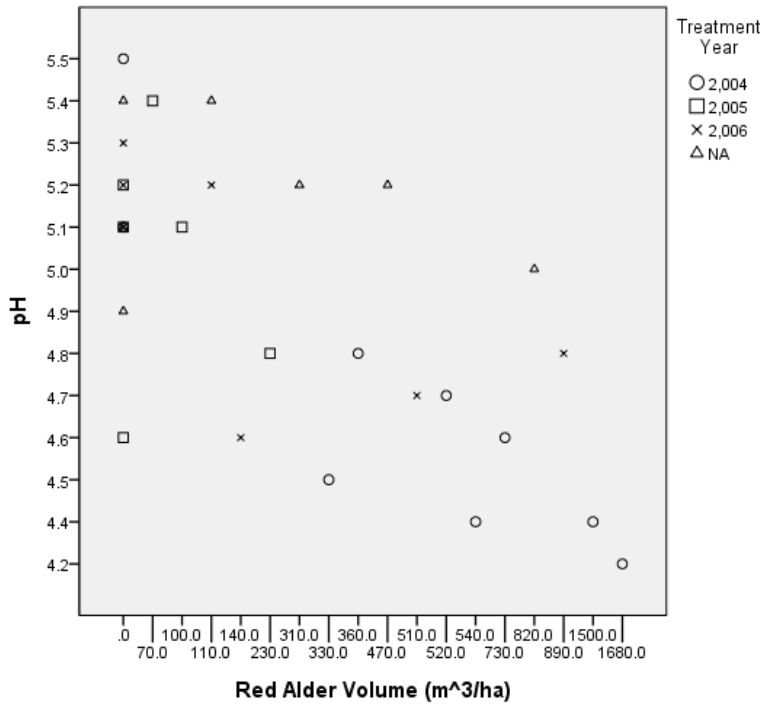

Figure 22: Correlation between forest floor pH and red alder volume

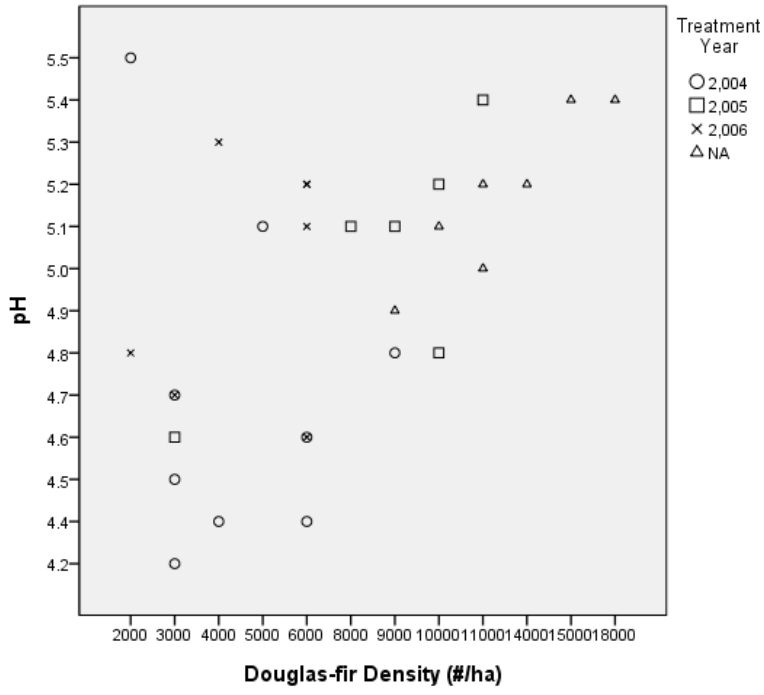

Figure 23: Correlation between forest floor pH and Douglas-fir density

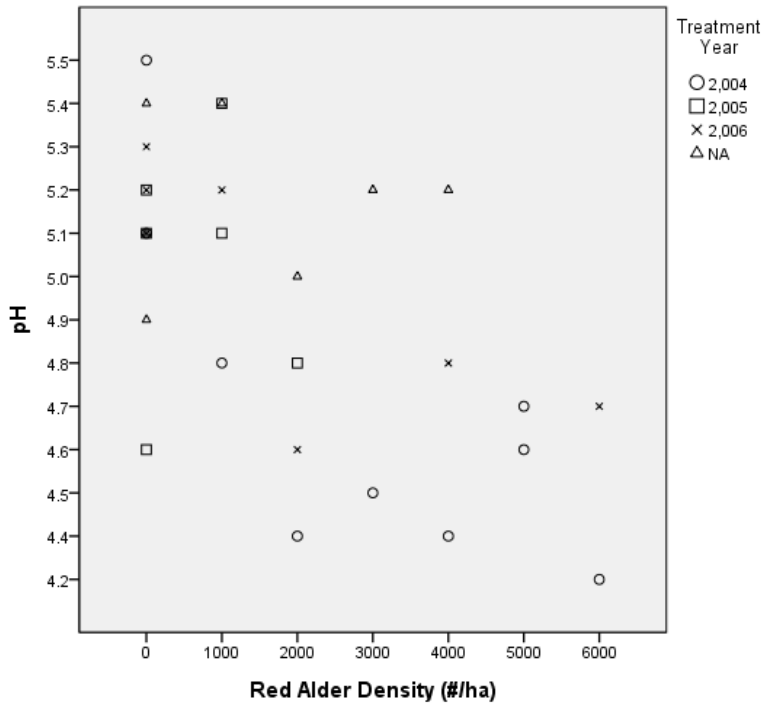

Figure 24: Correlation between forest floor pH and red alder density

There is also evidence to support a very strong relationship (rho = 0.864, p = 0.000) between forest floor soil carbon and forest floor soil nitrogen (Figure 25). Additionally, there appears to be a moderate relationship (rho = 0.461, p = 0.012) between forest floor soil nitrogen and forest floor pH (Figure 30). These relationships will be explored in chapter 10.

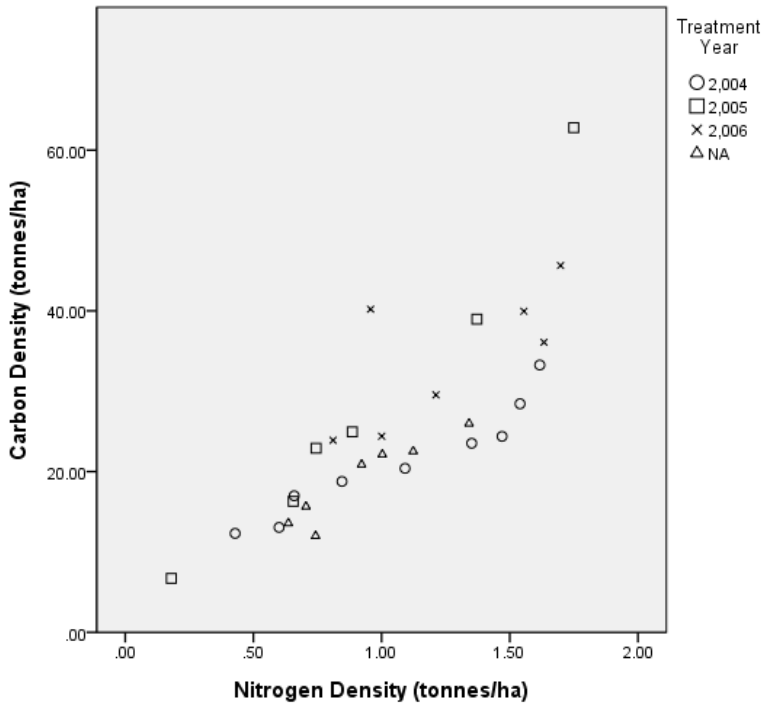

Figure 25: Correlation between forest floor soil carbon and nitrogen

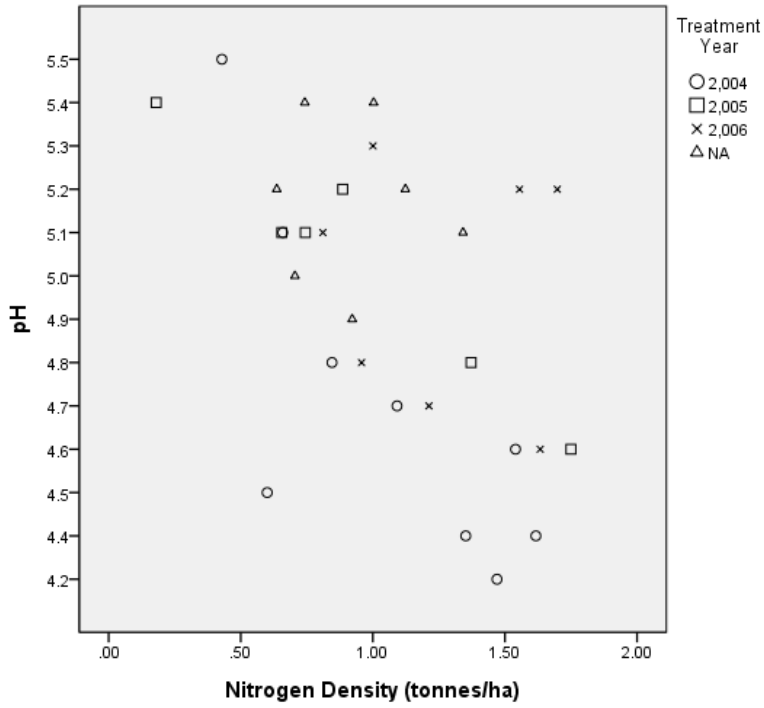

Figure 26: Correlation between forest floor soil nitrogen and pH

9.3.2 Mineral Soil Layer 1

There were only two statistically significant relationships found within this component of the analysis. There appears to be a weak negative relationship (rho = -0.395, p = 0.034) between alder volume and mineral soil layer 1 pH, and a strong relationship (rho = 0.738, p = 0.000) between mineral soil layer 1 carbon and mineral soil layer 1 nitrogen. Figures 27 and 28 depict these relationships and will be discussed in more detail in chapter 10.

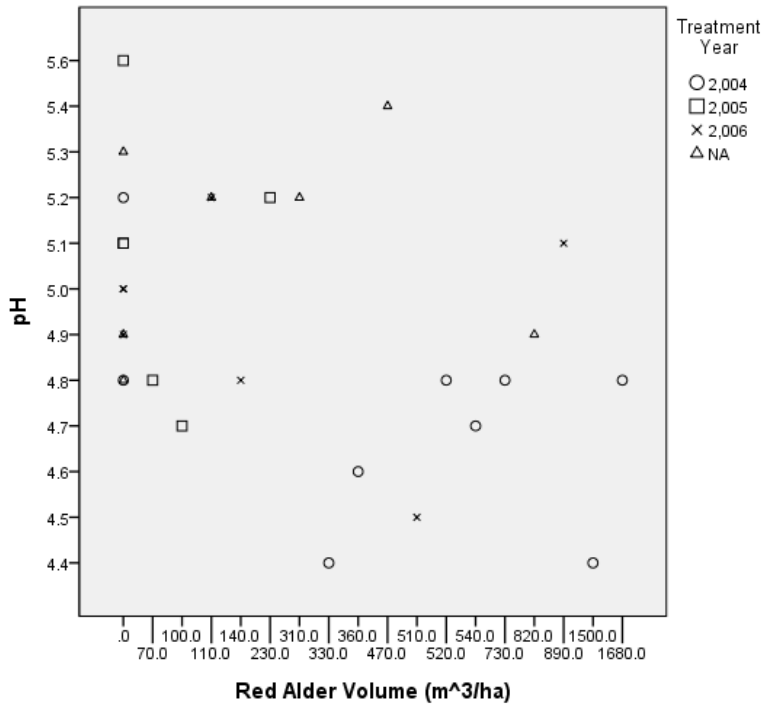

Figure 27: Correlation between mineral soil layer 1 pH and red alder volume

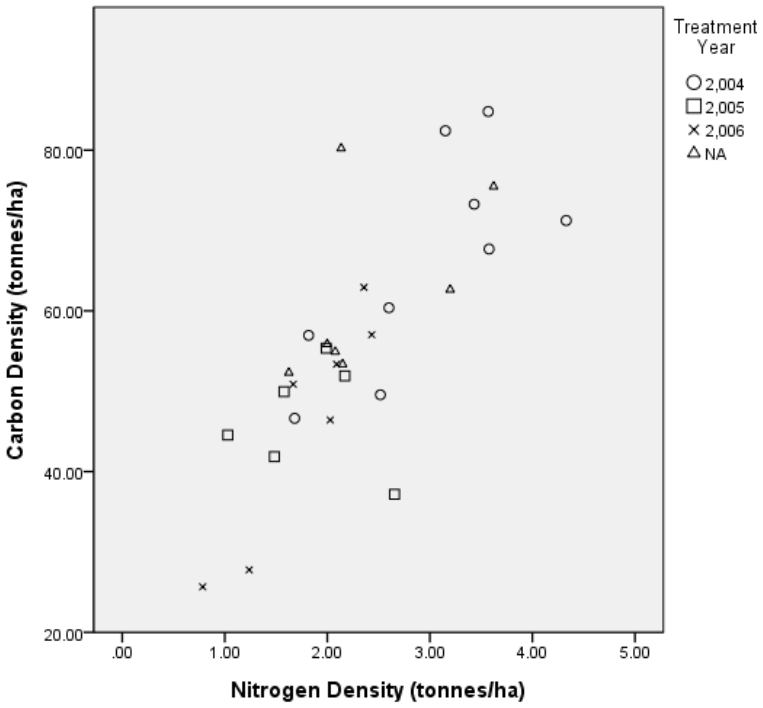

Figure 28: Correlation between mineral soil layer 1 carbon and nitrogen

10. Discussion

It is the goal of ecological restoration to emulate the natural composition, structure, and functioning of a degraded ecosystem in a manner that will leave it self sustaining and integrated with the ecological landscape in which it occurs (Higgs 1997). It is apparent that the restoration treatments have changed the structure and composition of the stand and that the treatments have influenced nutrient cycling processes within the forest floor and the mineral soil to a depth of 15 cm.

The restoration practices applied to DL63 generally increased the biodiversity values within the stand. Both Douglas-fir and released volunteer species such as red alder and big leaf maple provide resources and structures favourable to native flora and fauna; the implementation of wider tree spacing and canopy gaps increased understory vegetation; and the recruitment of structural attributes such as snags and CWD resulted in a significant increase of said attributes throughout the stand (Section 4.6). Following is a brief discussion of the most important measured variables highlighting the influence of the restoration.

10.1 Structure

10.1.1 CWD

The CWD that has been moved from the windrows to the forest floor seem to have been placed in a way that mimics its natural distribution of randomness and connectivity, with

some clumping and layering. These pieces could contribute to the structural diversity of the forest floor for quite some time providing they are of sufficient size, species, and decay stage. However, it was not possible to determine species for most of the CWD fragments due to there advanced decay status; very few CWD pieces pulled out of the windrows still had bark on them or other simple species identifying features. Along the same vein, the vast majority of the CWD was in the moderate to advanced decay classes three and four; the implication of this state of decay is a reduced longevity for these pieces. In other words, these pieces will likely not contribute to wildlife habitat for very long. On the other hand, because the majority of these CWD pieces are in full contact with the forest floor, have soft blocky pieces, and likely invading roots in the heartwood, leaching processes and nutrient cycling from the wood to the forest floor is likely much higher then if they were not as decomposed. Thus the positive relationship between the number of CWD fragments and soil carbon is not surprising. However, the observed relationship may be less direct. It is possible that when the wood was in the windrows it was colonized by fungi, which has potentially entered the soil and is increasing soil carbon.

It was originally thought that thinned trees that had been placed on the forest floor were less significant due to the small size of this wood and its associated rapid decay rates. In other words, it was thought that pulled over trees would only contribute to forest floor structure for a short period of time. However, those thinned trees that are suspended above the ground, like many of the pulled over trees, will decay at slower rates than those on the ground (Mattson et al. 1987). Furthermore, a study done by Edmonds and Eglitis

(1989) suggests that it would take approximately 115 years for 24 cm diameter Douglas-fir log to reach 95% decay where as it would take approximately 60 years for a 37 cm diameter Doulas-fir log to reach 95% decay. The reason for this seemingly illogical conclusion was that small logs were not as easily attacked by wood-boring insects which actually spread wood-rotting fungus. Thus there is the potential that pulled over trees will be part of the stand for quite some time.

This means that windrow derived CWD can provide important short-term ecological benefits whereas newly created CWD provides ecological benefits for a greater period of time. As of right now, the majority of the CWD pieces are relatively small. Ideally, larger CWD pieces will be recruited as the stand continues to age, which is likely due to the predicted increase in incremental tree growth resulting from thinning treatments. Larger pieces are of higher value than what is currently in the stand because they generally decay more slowly, hold more moisture, present less of a fire hazard, and provide more habitat value to a greater number of wildlife species (Manning et al. 2006).

10.1.2 Snags

Mortality rates are generally highest in young seral stages (Franklin et al. 1987), due to canopy closure and stem exclusion. High mortality rates are evident in DL63 where there are high densities of very small snags irrespective of treatment. However, it is predicted that snag production rates and density will generally fall with increasing stand age, but mean snag size and longevity will generally increase (Cline et al. 1980).

In terms of snag recruitment, it is apparent that there are more Douglas-fir snags in the treated areas then the control areas. However, the value of this increase must be discussed in the context of both wildlife use and nutrient cycling. Characteristics that affect the value of individual snags as habitat include cause of death, diameter, tree form, bark condition, tree species, and tree height (Lofroth 1998).

Cause of death is important because the ability of decay organisms such as fungi and insects to invade wildlife trees affects the ability of other wildlife elements to use said trees. For example, decay organisms further weaken or soften tree tissue which allows primary cavity nesters to excavate nests (Thomas et al. 1979). This is evident in DL63, where only the topped trees and erected snags showed any evidence of wildlife use, generally in the form of woodpecker excavations. This is likely due to the fact that topping a tree more closely mimics natural processes then girdling a tree; when the top of a tree is broken off it provides a location for water to pool and spores to land which may result in fungal rot from the inside out. Girdling a tree on the other hand, may disrupt the trees respiration process and ultimately kill the tree, however, where the tree is wounded it will exude sap effectively preventing fungi from entering the tree. The result is girdled trees likely rot from the outside -in vs. the inside-out which may have implications for wildlife use of said trees. The work done by Todd Manning with fungal inoculation may be a more effective alternative to girdling trees.

The size of the snag is important because it strongly influences which wildlife species will use them. The naturally occurring snags in DL63 have a mean dbh of 10.0 cm and

are likely of little use to most wildlife species. The created snags, on the other hand, have a mean dbh of 18.3 cm and may be of use to smaller cavity nesters such as nuthatches, chickadees, and mammalian species such as small bats and red squirrels (Lofroth 1998). Nuthatches and chickadees are more likely to use the snags in DL63 because of their affinity for Douglas-fir snags (Lundquist and Mariani 1991). Obviously the snag size classes in DL63 exclude the use of these trees by larger wildlife elements such as black bears.

The tree form, tree height, and bark condition of the girdled trees in DL63 is essentially the same throughout the treatment area. There were very few observed broken tree tops which can be used as nesting platforms, and the relatively even height of the stand is not conducive to the provision of perches for birds of prey (Lofroth 1998). Furthermore, a lack of loose bark at this stage in the decay process effectively excludes the use of these snags by herpetofauna that use the space between loose bark and the trunk. That said, thinning increases tree incremental growth, thus recruiting trees to become larger snags at an earlier age (Manning et al. 2006). And as the stand ages, tree form, tree height, and snag bark conditions will change.

An additional benefit of the restoration work has been a significant reduction in ladder fuels in the treatment area. In order to facilitate the girdling and topping of trees, live and dead branches were pruned to height no less then 4 m, and all pruned branches were piled. This process effectively reduced the fire hazard potential of the treatment area.

10.2 Vegetation

It appears that processes of understory succession and development are proceeding at rates that are characteristic of a maturing seral stand that has gone through initial natural thinning; there is essentially one age class in the overstory but the understory is exhibiting regeneration in a much younger age class and includes some latter seral species which are shade tolerant such as western red cedar (BC MOF 1998). Furthermore, species such as oceanspray (*Holodiscus discolour*) contribute to vertical / strata layering. Interestingly, although there is evidently an increase in the percent cover of vascular species in the sample plots as compared with the control plots, there is no immediately obvious change in bryophyte cover.

The increase in understory cover and species richness is beneficial for the stand not only in terms of species diversity, but also wildlife habitat/ microhabitats in the understory of the stand. The understory alpha diversity of the treatment and reference areas is 40 and 15 respectively. The understory beta diversity between the treatment and reference areas is 25. The understory gamma diversity of the entire stand is 40; the reference area did not contain any unique understory species i.e. all species found in the reference area were also found in the treatment area.

A range of tree species seedlings were found in both plot types indicating that there is a seedbank and natural tree regeneration is taking place regardless of treatment. How successful these seedlings are may be dependant on light availability. If this is the case one would expect a higher survival rate in the treated stands where thinning has increased

access to this resource. However, given the time frame of this project, seedling success has not been measured.

A notable observation within all plot types is the appearance of the non-indigenous invasive species, Scotch broom (*Cytisus scoparius*). It was anticipated that some invasive non-native plants would be observed in the treatment areas due to soil disturbance and changes in the light environment. It was, however, a surprise to find Scotch broom in control plots. Regardless, it is anticipated that Scotch broom will not be a problem over the long term due to crown closure over time.

Finally, although no records have been found that indicate DL63 underwent fertilization, it is a common forestry practise. Furthermore, the combination of the nitrogen indicator plant analysis and low carbon nitrogen ratios found through out the stand support the theory that DL63 was in fact fertilized at some point. Unfortunately MacMillan Bloedel could not be reached for comment.

10.3 Wildlife Observations

Within DL63 wildlife observations, including sightings and signs such as scat, were made and noted throughout the field season. Although a statistical analysis was not possible due to the disproportionate amount of time spent in the treatment area because of the complexity and number of plots in the treatment area, these observations reflect on the value of the restoration in terms of wildlife habitat creation and ultimately biodiversity. However, it is important to note that these are antidotal observations. Within the control

area, scat from Black tailed deer (*Odocoileous hemionus*) was found adjacent to two windrows, and there were several sightings of the Northern Flicker (*Colaptes auratus*), and Brown Creeper (*Certhia americana*). Within the treatment area, evidence of wildlife elements included scat from both the Black tailed deer and Raccoon (*Procyon lotor*), as well as tree cavities indicating the presence of the Pileated Woodpecker (*Dryocopus pileatus*), and potentially the Downy Woodpecker (*Dendrocopos pubescens*). Actual sightings in the treatment area included the Red legged frog (*Rana aurora*), Pacific tree frog (*Hyla regilla*), Douglas squirrel (*Tamiasciurus douglasii*), Black tailed deer, Pileated Woodpecker, Red-Breasted Sapsucker (*Sphyrapicus varius*), Dark-eyed Junco (*Junco caniceps*), Northern Flicker, Chestnut-backed Chickadee (*Parus rufescens*), and Brown Creeper. The implication of these observations is that there is potentially more suitable habitat for these species in the treatment area then in the reference area.

10.4 Soil

One of the challenging problems associated with estimating temporal and spatial changes in carbon and nitrogen pools is that the observed differences are often small compared to the pool sizes (Rothe et al. 2002). This is compounded by the inherently high spatial and temporal natural variability of these pools. The implication of this is that the author may not have detected changes in soil carbon and nitrogen with small effect sizes. On the other hand, the data shows that the only significant changes that have taken place are located in the forest floor and top 15 cm of mineral soil; soil carbon and nitrogen at depths greater then 15 cm appear to be unaffected by the restoration

treatments. This is important because as mentioned in section 2.4, long term soil C sequestration is dependant on the amount of SOM stored in the mineral soil; if the mineral soil below a depth of 15 cm is unaffected by the restoration, potential negative impacts on overall soil carbon are reduced. Furthermore, the potential trend observed for soil carbon in the first mineral layer across treatment years and the reference site suggests that over time soil carbon levels in the treatment area may match and even exceed those of the reference area. And it is possible that as the accumulated litter associated with killed treated trees decays, leaching processes will carry soluble carbon and nitrogen from the forest floor and upper mineral soil layer to the lower soil layers.

An important component of this study was the effect of snag creation on soil carbon and nitrogen. After a Douglas-fir has been killed, it takes several years for the tree to drop its needles and twigs. And it takes approximately 7-11 years for those needles to reach 95% decay (Edmonds 1980, Fogel and Cromack 1977). Given the previously mentioned litter dynamics, it was expected that forest floor soil carbon would increase after a post treatment lag time of several years, but would decrease once the large input of litter from the killed trees had reached an advanced state of decay. What was not expected was the observed time line associated with this process. The carbon density of the forest floor for the area treated in 2006 was significantly higher then the carbon density of the forest floor treated in 2004 indicating that leaf litter from trees killed in 2006 contributed a significant amount of carbon to the forest floor in just two years. Furthermore, the relatively similar forest floor carbon levels of the area treated in 2004 and the reference site indicate that decay and leaching processes may have occurred at a much faster rate

than the literature suggests. That said, studies have shown that the direct effects of high soil nitrogen supply generally increases initial decomposition rates of fresh litter but inhibits the decomposition of humified soil carbon (Resh et al. 2002).

It was also expected that initially mineral soil carbon would decrease as a result of soil disturbance associated with the treatments, and that this downward trend would be reversed once the foliage had dropped and foliage decay was underway. The interpretation of figure 13 supports this theory. In terms of soil nitrogen, it was expected that after a brief period of immobilization, nitrogen would be released from the foliage through decay processes and leached into the soil. There was no evidence to support this theory.

One would expect that mixed alder and Douglas-fir plots would show the bulk of nitrogen accumulation in the mineral soil to a depth of 15 cm because this is the mineral soil layer where higher concentrations of fine tree roots and red alder nodules should occur (Rothe et al. 2002). A cursory look at soil nitrogen across all soil layers irrespective of treatment implies that this is the case (Figure 29).

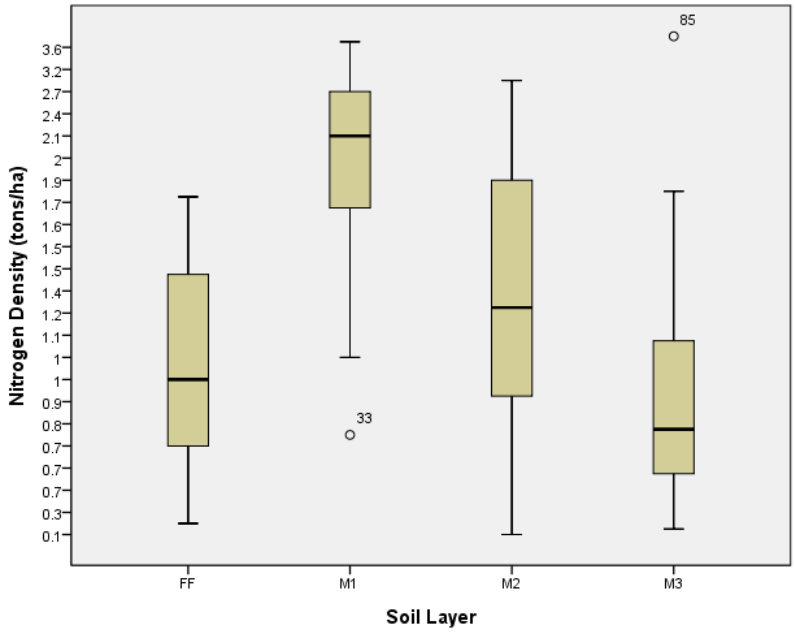

Figure 29: Soil nitrogen across all soil layers

Furthermore, the strong relationship observed between soil carbon and soil nitrogen is not surprising; several studies have shown that an increase in soil nitrogen under nitrogen fixers is concomitant with an increase in soil carbon (Rothe et al. 2002). One of the implications of this observation is that restoration practitioners aiming to both increase biodiversity values and increase soil carbon may find it advantageous to encourage mixed stands containing locally important tree species such as Douglas-fir and nitrogen fixers such as red alder. On the other hand, the relatively low C/N ratios found across DL63 in conjunction with the vegetation analysis and knowledge of common forestry practices suggest that DL63 may have been fertilized. The low C/N ratio of the soil means that there is potentially an abundant supply of soluble nitrogen for soil microbes and higher plants. However, runoff containing large quantities of nitrogen may have deleterious effects downstream. In this case, leaving thinned stems on site could be beneficial by

providing a high C:N substrate for nitrogen immobilization thus reducing nitrate leaching (Perakis et al. 2006). That said, it is generally considered that nitrogen cycling is more rapid on sites with high availability of nitrogen (Prescott et al. 2000b).

The negative relationship between N-fixing trees and soil pH is well documented (Van Miegroet and Cole 1984; Binkley and Sollins 1990; Rhoades et al. 2001). Van Miegroet and Cole (1984) link the decrease in soil pH to decreased base saturation; they suggest that lower base saturation results from the production of H^+ in nitrification, where H^+ displace base cations, which are then leached from the soil with NO_3. Binkley and Sollins (1990) assert that base saturation is only one of the factors of the soil exchange complex that determine soil pH; pH of a soil solution is determined by the quantity of weak acids, the degree of dissociation of the acids, and the strength of the acids. Thus acid strength in soils derives from the composite contribution of many types of acids of varying strengths (Binkley and Sollins 1990). That said, the observed relationship between red alder and pH can be at least partially attributed to base saturation.

10.5 Conclusion

Although no increases in soil carbon were observed within the time frame of this project, the dual goals of restoration for ecosystem structure and function versus restoration for soil carbon sequestration do not appear to be mutually exclusive. The restoration in District Lot 63 was successful in terms of increasing both floristic diversity and stand structure heterogeneity. Furthermore, there were more wildlife species

observed in the treatment area then the reference area. Significant changes in soil carbon were observed in the forest floor, and significant changes in both soil carbon and nitrogen were observed in the top 15 cm of the mineral soil. As time from treatment increased, soil carbon and nitrogen approached, and in some cases surpassed, reference area levels. The results from this study indicate that the restoration on Galiano Island was successful in terms of increasing the biodiversity values of the stand and had no large short-term effects on soil carbon or nitrogen.

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Appendix A
Structural Analysis Overview

Variable	Treatment/ Control	Visual Distribution	Shapiro-Wilk	Transformation	Shapiro-Wilk	Levene's Test
Natural Snag Density	T	Non-normal	**0.012**	SQRT	**0.043**	0.092
Natural Snag Density	C	Non-normal	0.107	SQRT	0.376	
Total Snag Density	T	Normal	0.481	SQRT	**0.043**	0.545
Total Snag Density	C	Non-normal	0.107	SQRT	0.079	
Douglas-fir Snag Volume	T	Non-normal (Marginal)	0.122	LN	0.079	0.181
Douglas-fir Snag Volume	C	Non-normal	0.075	LN	0.305	
Natural Snag Mean DBH	T	Non-normal (Marginal)	0.127	LOG 10	0.264	
Natural Snag Mean DBH	C	Non-normal	0.121	NA		
Created Snag Mean DBH	T	Non-normal	**0.001**	LOG 10	0.663	0.369
CWD # of Fragments	T	Normal (Marginal)	0.056	SQRT	0.573	0.945
CWD # of Fragments	C	Non-normal (Marginal)	0.081	SQRT	0.783	
CWD Volume	T	Non-normal	**0.046**	NA		
CWD Volume	C	Non-normal	0.113	NA		
Residual Douglas-fir Volume	T	Non-normal	**0.017**	LOG 10	0.605	**0.019**
Residual Douglas-fir Volume	C	Non-normal (Marginal)	0.161	LOG 10	0.394	
Red Alder Volume	T	Non-normal	0	NA		
Red Alder Volume	C	Non-normal	0.084	NA		
Residual Douglas-fir Density	T	Non-normal (Marginal)	0.054	LN, SQRT	.113, .122	**0.03**
Residual Douglas-fir Density	C	Non-normal (Marginal)	0.491	LN, SQRT	.696, .604	0.33
Red Alder Density	T	Non-normal	**0.002**	NA		
Red Alder Density	C	Non-normal	0.163	NA		
Species Richness	T	Non-normal (Marginal)	0.795	LN	0.142	0.086
Species Richness	C	Non-normal (Marginal)	0.414	LN	0.392	

Appendix B
Structural Analysis Results Summary

Variable	Mann-Whitney	Kolmogorov-Smirnov	GLM Univariate Analysis	T-test
Natural Snag Density	0.409	0.779	Sig. 0.335; Observed Power 0.157	Equal Variance Assumed 0.335, Equal Variance Not Assumed 0.487
Natural Snag Mean DBH	0 .901	0.958	NA	NA
Total Snag Density	0.088	0.098	Sig. 0.068; Observed Power 0.451	Equal Variance Assumed 0.068, Equal Variance Not Assumed 0.121
Douglas-fir Snag Volume	0.003	0.007	Sig. 0.010; Observed Power 0.764	Equal Variance Assumed 0.010, Equal Variance Not Assumed 0.038
Natural vs. Created Snag Mean DBH	0 .000	0.000	Sig. 0.005; Observed Power 0.828	Equal Variance Assumed 0.005, Equal Variance Not Assumed 0.005
Residual Douglas-fir Volume	0.004	0.004	Sig. 0.003; Observed Power 0.875	Equal Variance Assumed 0.003, Equal Variance Not Assumed 0.000
Residual Douglas-fir Density	0.000	0.004	NA	NA
Red Alder Volume	0.709	0.997	NA	NA
Red Alder Density	0.672	0.995	NA	NA
CWD # of Fragments	0.000	0.001	Sig. 0.000; Observed Power 1.000	Equal Variance Assumed 0.000, Equal Variance Not Assumed 0.001
CWD Volume	0.020	0.141	NA	NA
Species Richness	0.000	0.000	Sig. 0.000; Observed Power 1.000	Equal Variance Assumed 0.000, Equal Variance Not Assumed 0.001

Appendix C
Treatment Year and Reference Site Soil Analysis Summary

Soil Variable	Soil Layer	Treatment Years	GLM Analysis*	Randomization**
Carbon	Forest Floor	All	0.048, 0.639	NA
Carbon	Forest Floor	2004, 2006	**0.039**, NA	**0.008**
Carbon	Forest Floor	Ref, 2006	**0.013**, NA	**0.002**
Carbon	Mineral 1	All	0.006, 0.879	NA
Carbon	Mineral 1	2004, 2005	**0.021**, NA	**0.011**
Carbon	Mineral 1	2004, 2006	-	**0.016**
Carbon	Mineral 1	Ref, 2005	-	**0.007**
Carbon	Mineral 1	Ref, 2006	-	**0.045**
Carbon	Mineral 2	All	0.705, 0.127	NA
Carbon	Mineral 3	All	0.513, 0.154	NA
Nitrogen	Forest Floor	All	0.403, 0.242	NA
Nitrogen	Forest Floor	Ref, 2006	-	**0.003**
Nitrogen	Mineral 1	All	0.011, 0.829	NA
Nitrogen	Mineral 1	2004, 2005	0.054, NA	**0.019**
Nitrogen	Mineral 1	2004, 2006	**0.044**, NA	**0.013**
Nitrogen	Mineral 1	All	0.011, 0.829	NA
Nitrogen	Mineral 2	All	0.207, 0.365	NA
Nitrogen	Mineral 3	All	0.558, 0.141	NA
C:N Ratio	Forest Floor	All	0.001, 0.981	NA
C:N Ratio	Forest Floor	2004, 2005	**0.014**, NA	**0.001**
C:N Ratio	Forest Floor	2004, 2006	-	**0.014**
C:N Ratio	Forest Floor	Ref, 2005	**0.012**, NA	**0.001**
C:N Ratio	Forest Floor	Ref, 2006	-	**0.002**
C:N Ratio	Mineral 1	All	0.442, 0.224	NA
C:N Ratio	Mineral 2	All	0.288, 0.299	NA
C:N Ratio	Mineral 3	All	0.359, 0.213	NA
pH	Forest Floor	All	0.030, 0.71	NA
pH	Forest Floor	Ref, 2004	**0.047**, NA	**0.021**
pH	Mineral 1	All	0.023, 0.744	NA
pH	Mineral 1	Ref, 2004	**0.042**, NA	**0.009**
pH	Mineral 1	2004, 2005	-	**0.034**
pH	Mineral 2	All	0.149, 0.430	NA
pH	Mineral 3	All	0.677, 0.113	NA

* Sig., Observed Power; ** Sig.

Appendix D
Stand Structure and Soil Nutrient Correlation Summary

Variables	Soil Layer	Kendall's Tau	Spearman's rho	Randomization
Carbon, CWD Volume	Forest Floor	0.054; 0.68	0.060; 0.758	; 0.829
Carbon , CWD Fragment Number	Forest Floor	0.337; **0.012**	0.473; **0.010**	0.479; **0.015**
Carbon, Total Snag Density	Forest Floor	0.267; **0.048**	0.365; 0.052	0.197; 0.304
Carbon, Douglas-fir Snag Volume	Forest Floor	0.279; **0.035**	0.414; **0.026**	0.243; 0.218
Carbon, Alder Volume	Forest Floor	0.096; 0.486	0.128; 0.509	-
Carbon, Douglas-fir Density	Forest Floor	-0.237; 0.081	-0.314; 0.097	-
Carbon, Alder Density	Forest Floor	0.079; 0.573	0.132; 0.496	-
Carbon, Species Richness	Forest Floor	0.258; 0.053	0.317; 0.094	0.365; **0.045**
Nitrogen, CWD Volume\	Forest Floor	-0.035; 0.793	-0.057; 0.770	-
Nitrogen, CWD Fragment Number	Forest Floor	0.171; 0.201	0.246; 0.198	-
Nitrogen, Created Snag Density	Forest Floor	0.179; 0.193	0.255; 0.182	-
Nitrogen, Total Snag Density	Forest Floor	0.150; 0.266	0.230; 0.231	-
Nitrogen; Douglas-fir Snag Volume	Forest Floor	0.184; 0.164	0.254; 0.184	-
Nitrogen, Douglas-fir Volume	Forest Floor	-0.222; 0.091	-0.322; 0.088	-
Nitrogen, Alder Volume	Forest Floor	0.191; 0.164	0.260; 0.173	-
Nitrogen, Douglas-fir Density	Forest Floor	-0.144; 0.289	-0.211; 0.273	-
Nitrogen, Alder Density	Forest Floor	0.178; 0.206	0.257; 0.178	-
Nitrogen, Species Richness	Forest Floor	0.108; 0.419	0.129; 0.505	-
pH, CWD Volume	Forest Floor	0.018; 0.895	0.025; 0.898	-
pH, CWD Fragment Number	Forest Floor	-0.167; 0.225	-0.241; 0.208	-
pH, Created Snag	Forest Floor	-0.200; 0.159	-0.289; 0.128	-

Density				
pH, Total Snag Density	Forest Floor	-0.188; 0.177	-0.258; 0.176	-
pH, Douglas-fir Snag Volume	Forest Floor	-0.142; 0.297	-0.209; 0.278	-
pH, Douglas-fir Volume	Forest Floor	0.274; **0.043**	0.371; **0.047**	0.398; **0.036**
pH, Alder Volume	Forest Floor	-0.469; **0.001**	-0.616; **0.000**	-0.680; **0.000**
pH, Douglas-fir Density	Forest Floor	0.381; **0.007**	0.487; **0.007**	0.528; **0.006**
pH, Alder Density	Forest Floor	-0.444; **0.002**	-0.569; **0.001**	-0.623; **0.000**
pH, Species Richness	Forest Floor	-0.031; 0.820	-0.071; 0.715	-
Carbon, Nitrogen	Forest Floor	0.714; **0.000**	0.864; **0.000**	0.817; **0.000**
Carbon, pH	Forest Floor	-0.259; 0.056	-0.364; 0.052	-0.343; 0.072
Nitrogen, pH	Forest Floor	-0.361; **0.008**	-0.461; **0.012**	-0.538; **0.005**
Carbon, CWD Volume	Mineral Layer 1	-0.158; 0.230	-0.238; 0.213	-
Carbon , CWD Fragment Number	Mineral Layer 1	-0.196; 0.142	-0.299; 0.115	-
Carbon, Created Snag Density	Mineral Layer 1	-0.195; 0.157	-0.279; 0.143	-
Carbon, Total Snag Density	Mineral Layer 1	-0.120; 0.375	-0.198; 0.302	-
Carbon, Douglas-fir Snag Volume	Mineral Layer 1	-0.159; 0.229	-0.209; 0.276	-
Carbon, Alder Volume	Mineral Layer 1	0.164; 0.231	0.259; 0.176	-
Carbon, Douglas-fir Volume	Mineral Layer 1	0.138; 0.294	0.182; 0.345	-
Carbon, Douglas-fir Density	Mineral Layer 1	0.021; 0.879	0.028; 0.886	-
Carbon, Alder Density	Mineral Layer 1	0.123; 0.382	0.177; 0.359	-
Carbon, Species Richness	Mineral Layer 1	-0.193; 0.148	-0.313; 0.098	-
Carbon, Soil Texture	Mineral Layer 1	0.142; 0.340	0.178; 0.356	
Nitrogen, CWD Volume\	Mineral Layer 1	-0.084; 0.523	-0.089; 0.645	-
Nitrogen, CWD Fragment Number	Mineral Layer 1	-0.101; 0.452	-0.137; 0.478	-
Nitrogen, Created Snag Density	Mineral Layer 1	-0.174; 0.207	-0.247; 0.197	-
Nitrogen, Total Snag Density	Mineral Layer 1	-0.125; 0.355	-0.158; 0.412	-
Nitrogen; Douglas-fir	Mineral Layer 1	-0.199; 0.367	-0.154; 0.425	-

Snag Volume				
Nitrogen, Douglas-fir Volume	Mineral Layer 1	-0.010; 0.940	-0.076; 0.694	-
Nitrogen, Alder Volume	Mineral Layer 1	0.186; 0.176	0.278; 0.144	-
Nitrogen, Douglas-fir Density	Mineral Layer 1	-0.129; 0.343	-0.201; 0.295	-
Nitrogen, Alder Density	Mineral Layer 1	0.150; 0.285	0.200; 0.299	-
Nitrogen, Species Richness	Mineral Layer 1	-0.128; 0.338	-0.212; 0.270	-
Nitrogen, Soil Texture	Mineral Layer 1	0.129; 0.385	0.169; 0.379	
pH, CWD Volume	Mineral Layer 1	-0.099; 0.470	-0.110; 0.570	-
pH, CWD Fragment Number	Mineral Layer 1	-0.209; 0.132	-0.205; 0.287	-
pH, Created Snag Density	Mineral Layer 1	-0.172; 0.230	-0.216; 0.261	-
pH, Total Snag Density	Mineral Layer 1	-0.171; 0.222	-0.236; 0.217	-
pH, Douglas-fir Snag Volume	Mineral Layer 1	-0.181; 0.189	-0.218; 0.256	-
pH, Douglas-fir Volume	Mineral Layer 1	0.052; 0.704	0.067; 0.729	-
pH, Alder Volume	Mineral Layer 1	-0.271; 0.058	-0.395; **0.034**	-0.386; **0.048**
pH, Douglas-fir Density	Mineral Layer 1	0.250; 0.078	0.308; 0.104	-
pH, Alder Density	Mineral Layer 1	-0.274; 0.062	-0.363; 0.053	-0.358; **0.049**
pH, Species Richness	Mineral Layer 1	-0.034; 0.804	-0.038; 0.846	-
pH, Soil Texture	Mineral Layer 1	0.129; 0.404	0.160; 0.407	
Carbon, Nitrogen	Mineral Layer 1	0.557; **0.000**	0.738; **0.000**	0.767; **0.000**
Carbon, pH	Mineral Layer 1	-0.042; 0.761	-0.045; 0.818	-
Nitrogen, pH	Mineral Layer 1	0.052; 0.704	0.064; 0.740	-

(Correlation Coefficient; Significance)