AN ANALYSIS OF COASTAL TEMPERATE OLD FOREST RESIDUAL CARBON, STRUCTURE AND UNDERSTORY PLANT FLORISTICS AFTER WILDFIRE

by

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Abstract

I assessed some coastal temperature old-growth forests of southwestern British Columbia to understand their post-wildfire structure, carbon storage and biodiversity values. I used a remotely sensed relativized burn ratio and a composite burn index to compare measures of aboveground carbon, structural complexity and floristic diversity between burned and unburned reference plots years after four large wildfires. The unburned reference plots represented the natural range of variation of old growth values. In burned plots, 21 of 60 retained carbon values and 10 plots retained structural values similar to unburned old growth plots. There was an average of 12% floristic similarity between burned and unburned understory plant communities. For land managers, this method offers a way to compare old-growth values after wildfire in order to prioritize protection, salvage, and restoration options.

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List of Acronyms

AGC	Above-ground carbon
BEC	Biogeoclimatic ecosystem classification
CBI	Composite burn index
CWD	Coarse woody debris (also known as large woody material, or dead and down logs)
CWHms1	Southern moist submaritime Coastal Western Hemlock (BEC variant)
DBH	Diameter at breast height, 1.3 m above ground level
dNBR	Delta normalized burn ratio
GHG	Greenhouse gas
Mg ha^{-1}	Megagrams (tonnes) per hectare
NBR	Normalized burn ratio
NIR	Near-infrared wavelengths of electromagnetic radiation
OGMA	Old Growth Management Area
PNW	Pacific Northwest of North America
RBR	Relativized burn ratio
SWIR	Shortwave infrared wavelengths of electromagnetic radiation

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Chapter One – Literature Review

1.1. Introduction

Lower elevation coniferous coastal forests of British Columbia have been called temperate rainforests (MacKinnon 2003), the rainy climate responsible for historically infrequent fires, with fire return intervals from hundreds of years to over a thousand years between major fire disturbances (Daniels and Gray 2006, Parminter et al. 2014, Daniels et al. 2017). These long periods between stand initiating events allow stands to achieve a climax successional status, a distinct ecosystem of rich biodiversity and structure, known by many names, most commonly called old growth.

Globally, forests with no clear signs of human disturbance (primary forests) cover 1.11 billion hectares of land, however, because of multiple simultaneous and continuous pressures from human and natural disturbances, these forests have decreased by 81 million hectares since 1990 (FAO 2020). If these trends continue, there will be a net loss of forest and forest values into the future. Anthropogenic activity can directly remove whole forests from a landscape through historic and current logging practices for wood products, clearing land for agriculture and urbanization at rates well outside the limits of natural disturbance regimes (Lindenmayer et al. 2012). These pressures have already significantly decreased the amount of coastal forest that can be classified as "old-growth forest" along the west coast of Canada. (Trofymow et al. 2003, Gray et al. 2009, Price et al. 2020). The often disconnected and smaller patches of old growth remaining have more isolated and smaller old-growth interior habitat and more exposure to outside edge effects (Harper et al. 2005) which in turn makes otherwise resilient forests more susceptible to natural disturbance agents such as insects and fire (MacKinnon 1998). Overarching all this is the increasing incidence of drought caused by anthropogenic climate change which exacerbates the frequency and severity of events such as fire (Haughian et al. 2012, Halofsky et al. 2020).

As old forest patches become scarcer and more disconnected in coastal BC, unusually large wildfires may 'reset' stand development of so many patches such that no old-growth forest may remain in a landscape. However, managers may wish to distinguish areas of old forest that have experienced fires but may still retain sufficient old-growth value from areas that are so highly disturbed that they no longer meaningfully provide old forest values and need to be replaced with the next suitable candidate area.

1.2. Research Goal

To help forest managers triage where restoration funds will go to maintain old forest values after severe fires, the goal of this research is to use remotely sensed imagery and field observation to estimate burn severity thresholds at which selected old growth forest values are so greatly compromised that they could no longer fulfill management objectives for old growth within British Columbia's Coastal Western Hemlock moist submaritime (CWHms1) biogeoclimatic (BEC) variant. Chapter one reviews the literature to better understand the history and current state of old growth in the Pacific Northwest of North America. Chapters two and three outline the research methodology and results, respectively. Chapters four and five discuss some of the significance of these findings and potential applications of this research.

1.3. Literature Review

1.3.1. What is Old Growth?

The question of what is or is not old growth is a topic too broad to be fully covered here, but scholars have tried for decades to develop definitions in the Pacific Northwest of North America and elsewhere (Hilbert and Wiensczyk 2007). Early attempts at defining old growth included measuring the density of large old trees (greater than 200 years of age) with a threshold amount of snags and coarse woody debris (Franklin et al. 1986). In 1991, a British Columbia (BC) provincial government report concluded there was insufficient information to even define old growth in a practical way, based on the sheer number of attributes to measure (Hamilton and Nicholson 1991).

Old forest ecosystems can be referred to with terms such as “climax”, “ancient”, “primeval”, “primary”, “over-mature” and “decadent”, each with its own historical context and interpretation; however, they are not synonymous (Kneeshaw and Burton 1998, Spies 2004, Hilbert and Wiensczyk 2007). Primary forest is a “naturally regenerated forest of native tree species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed” (Mackey et al. 2021). This definition makes no mention of age, as it can refer to an old ecosystem in an early state of succession due to a recent natural disturbance such as an insect outbreak. This is different than a climax forest which refers to a successional stage of development where a group of species establishes and predominates so that the community replaces itself, rather than being replaced by other species (Oliver and Larson 1996). Climax forests refer to an ecosystem in a late successional stage, therefore climax forests tend to be primary forests, but not all primary forests are climax.

In the late 1990s researchers in the Pacific Northwest started to define old growth not only by measured properties (e.g., age or diameter), but by ecological processes. Wells et al. (1998) suggest that the dynamics of gap formation in the canopy (formed by fallen large trees) is the predominant process defining old growth. Others suggest that the old-growth stage of stand development is achieved when trees that dominate the stand are not those that seeded in after a major disturbance, but rather had grown up under the canopy of older trees (Oliver and Larson 1996). Kneeshaw and Burton (1998) expanded on this idea by differentiating "mature" forests from "old growth" by considering a ratio of the basal area (or dominance) of trees that established directly after a major disturbance and their replacements. Some models of forest development specifically use the term "old growth", while others do not, instead referring to ecological processes such as "pioneer cohort loss" or "shifting-gap phase" (Franklin et al. 2002)

What is clear is the recognition that old-growth forests include a continuum of ages, sizes, structures and stages (Wells et al. 1998). There is no clear line between one stage and the next, so land managers in the Pacific Northwest have adopted flexible definitions where old growth can be defined through indices of structural or biological attributes, simple age classes or dynamic processes (e.g., gap formation and fire disturbance) (Franklin et al. 2002, Holt et al. 2008, Moyer et al. 2010). The objectives of the project, (e.g., with respect to carbon sequestration, economic, or biodiversity values) guides the definition into categories of ecological, functional and structurally based definitions (Hilbert and Wiensczyk 2007, Holt et al. 2008). For example, the Great Bear Rainforest of coastal BC is a 6.4-million-hectare land management area with a flexible definition of old forest:

"(a) a stand of trees 250 years or older;

(b) an old, structurally complex stand comprised mainly of climax species where older seral remnants may still be present in the upper canopy and typically have:

(i) standing snags;

(ii) rotting logs on the ground; and

(iii) patchy understories; or

(c) a stand of trees that has reached the climax state for the ecosystem it is found in where trees naturally cycle at an age less than 250 years ” (B.C. Ministry of Forests, Lands and Natural Resource Operations 2016)

1.3.2. Natural Variability

The native species and ecological processes of an old North American forest today are the result of climate, soils, biogeographic history, and the cumulative effects of natural disturbances over time (Wong 2004, Hessburg et al. 2019). A disturbance regime is made up of many individual disturbance events, each with its own footprint, while the range in frequency, severity or extent of these footprints can be described by its Natural Range of Variability (NRV, also known as the range of natural variability, or the historical range of variability). The premise is that variability in disturbances is fundamental to most ecosystems and that past ecological conditions and processes (prior to European settlement) should guide management decisions in the present (Landres et al. 1999, Wong 2004). An individual disturbance in the present is a snapshot in time; its impact could be within the NRV (e.g., frequency of fires within a valley since the last ice age), or the impact could be outside of historic limits (e.g., severity of fires within a watershed in the last 400 years) (Landres et al. 1999, Wong 2004). Therefore, land managers need to set the scope of NRV for comparison, to describe what their management end goal is and where the current ecosystem state is in relation to that.

1.3.3. Old-Growth Values

In interviewing forestry leaders within Canada about old growth, Moyer et al. (2010) found that "…the natural state, or the absence of human disturbance, not the specific age of the forest, seemed to be an important part of what made it unique and valuable." Coastal temperate old-growth forests, which in some cases have gone over 6000 years without major fire disturbance (Lertzman et al. 2002), are an example of this natural state, reflecting a unique set of ecological processes developed over long periods free of industrial colonial forestry. These ecological processes provided the large standing timber that drove human economic activity in British Columbia from the 1800s to today, but recent decades have seen a push to recognize old-growth forests for their other values (LePage and Banner 2015). Old forests are an important part of human culture through recreation, tourism, traditional medicines, spiritual practices and even acknowledgement of the intrinsic value that they simply exist (Moyer et al. 2010, Sutherland et al. 2016, Gilhen-Baker et al. 2022). However, this review will focus on three commonly studied and important ecological values of temperate coastal old-growth forests, specifically their ability to store carbon, which is related to their structural development, which in turn influences old-forest biodiversity.

1.3.4. Old-Growth Carbon Storage

Forests are an important part of the global carbon cycle, absorbing carbon from the atmosphere via photosynthesis and releasing carbon via respiration and natural disturbances such as fire (Trofymow et al. 2008). Some of the difference between absorption and emission is the amount of carbon held in various terrestrial pools such as tree biomass, forest floor and soils (Luyssaert et al. 2008). Trees use the captured carbon to build the wood found in stems and branches, which eventually dies off and contributes to the forest floor and soil as decaying matter (Thompson 2009). The time and amount of carbon that transitions from one pool to the next varies with plant community composition, stand developmental stage and climate (Black et al. 2008) . Temperate forests, for example, capture 15-20% of annual human carbon emissions (Case et al. 2021). Therefore one way to mitigate the effects of anthropogenic climate change is through removing the most abundant greenhouse gas, carbon dioxide (CO_2), from the atmosphere through conservation of old forests and through forest management practices that maintain the inherent genetic, structural and biodiverse properties of forests (Thompson 2009).

From a management perspective then, it is important to acknowledge the balance between the rate of carbon sequestration, the rate of forest carbon emissions and the total amount of carbon stored. After a severe disturbance such as logging or fire in the PNW, decomposition of residual organic material releases ecosystem carbon to the atmosphere, such that the area becomes a net source of carbon for several years (Black et al. 2008). Surviving trees or new seedlings, with the overstory removed, begin to increase stem size and overall amount of leaf area, increasing their mass at an accelerating rate (Stephenson et al. 2014). The annual rate of carbon sequestration starts to exceed the amount of ecosystem carbon released from decaying vegetation and the forest becomes a carbon sink (Black et al. 2008). The rate of sequestration increases further until the tree canopy begins to close and access to direct sun decreases. The rate of development (and carbon sequestration) of individual trees starts to slow as trees approach middle age and continues to decline with age. Most trees will still continue to accumulate mass (carbon storage) continuously throughout their life, though that varies by species and climate (Stephenson et al. 2014). Because of mortality with age and inter-tree competition for resources, carbon begins to be released to the forest floor and soil and eventually back into the atmosphere, thus the overall ecosystem carbon sequestration rate of older forests is slower than younger forests at the landscape level (Stephenson et al. 2014, Gray et al. 2016).

At the same time, the total ecosystem carbon storage of temperate forests is much greater in older stands than in younger ones (McKinley et al. 2011, Matsuzaki et al. 2013, Gray et al. 2016). In a developing forest, many trees may die due to small-scale disturbance such as wind or pests, but some individuals will survive so that eventually there are a mix of large and old trees interspersed with younger trees. Given a long enough period free of disturbance, the largest 1 to 3% of trees can store as much as 42 to 50% of the biomass in older forests (Lutz et al. 2018, Mildrexler et al. 2020). Those that attain a size of 100 cm DBH or more in diameter can build three times as much aboveground dry biomass per year as compared to a tree of the same species of only 50 cm in diameter (Stephenson et al. 2014). In other words, relatively uncommon large-diameter trees have a disproportionate importance as individuals for carbon sequestration while living and carbon storage when dead.

There is debate as to whether old-growth forests *per unit area*, are a net sink, source or are carbon neutral due to the complexity of estimating landscape carbon and of modelling methodologies (Luyssaert et al. 2008, Gray et al. 2016, Gundersen et al. 2021). Regardless, overall carbon storage is greatest in ecosystems with large trees (Lutz et al. 2018, Mildrexler et al. 2020), particularly in the Pacific Northwest (Case et al. 2021). Conservation of ecosystems containing large trees for the purposes of carbon storage therefore is an important value of old-growth forests.

1.3.5. Old-Growth Structure

Structure is the spatial arrangement and diversity of physical features within an ecosystem, including living and dead trees (snags), coarse woody debris (CWD), heights and number of canopy layers and spacing between trees (Spies and Franklin 1991, Franklin et al. 2002, McElhinny et al. 2005). Structure has vertical (height) and horizontal (distance) components (Ishii et al. 2004, Latifi 2012), but can also refer to aspects of the age class structure (Mulverhill et al. 2019) such as the pattern of gap formation in the canopy over time (Wells et al. 1998, Frazer et al. 2000), or the decay of living trees to snags and CWD over time (Oliver and Larson 1996, Donato et al. 2016).

Structure is an important metric with which to study old growth because it is the basis for habitat that gives rise to biological diversity and it can often be more easily measured and manipulated compared to age or composition (Franklin et al. 2002, Bauhus et al. 2009). Latifi (2012) adapted a table from McElhinny et al. (2005) to describe commonly measured structural attributes grouped by forest stand component (Table *1*). The large number of attributes that may be studied reflects the complexity of old-growth ecosystems.

Table 1 - Commonly measured structural attributes grouped by theme; from (Latifi 2012) adapted from (McElhinny et al. 2005).

Forest stand element	**Structural attribute**
Foliage	Foliage height diversity
	Number of strata
	Foliage density within different strata
Canopy cover	Canopy cover
	Gap size classes
	Average gap size and the proportion of canopy in gaps
	Proportion of crowns with dead and broken tops
Tree diameter	Diameter at breast height (DBH)
	Standard deviation of DBH
	Diameter distribution
	Number of large trees
Tree height	Height of overstorey
	Standard deviation of tree height
	Height classes richness
Tree spacing	Clark - Evans and Cox indices, percentage of trees in clusters
	Stem count per ha
Stand biomass	Stand basal area
	Standing volume
	Biomass
Tree species	Species diversity and/or richness
	Relative abundance of key species
Overstorey vegetation	Shrub height
	Shrub cover
	Total understorey cover
	Understorey richness
	Saplings (shade tolerant) per ha
Dead wood	Number, volume or basal area of snags
	Volume of coarse woody debris
	Log volume by decay or diameter classes
	Coefficient of variation of log density

Structural complexity can enhance forest ecosystem productivity because three-dimensional complexity creates more habitat niches (Ishii et al. 2004), which in turn creates more opportunities for increased biodiversity (Spies and Franklin 1991, Arsenault and Bradfield

1995, Franklin et al. 2002, MacKinnon 2003). For example, downed trees create root pits and mounds, opening soil patches for plant and fungi colonization, while CWD provides ground cover for organisms, nutrients and a platform for new tree growth (Franklin et al. 2002). The gap left in the canopy from the fallen tree alters he amount of light that infiltrates to the forest floor, which then affects the diversity of flora due to varying shade tolerance (Lertzman et al. 1996, Wells et al. 1998, Ishii et al. 2004). Over time this results in an intermixing of young and old trees with large live and dead standing trees and multiple canopy layers (Oliver and Larson 1996).

The spatial distribution of structure, particularly of deadwood, affects how nutrients (Donato et al. 2016), wildfire fuel (Peterson et al. 2019) and carbon storage (Sutherland et al. 2016) are spread over an ecosystem and play various roles in decay, new growth and then decay again (Spies et al. 1988, Donato et al. 2016). Structure and time have been described as providing the foundation for building the values associated with old-growth forests (Parish and Antos 2004, Gerzon et al. 2011).

Large trees typically contribute to old-growth structure. Large is a subjective term, but a general definition would be that large trees are of a reproductive height, within the upper canopy layer of the forest with a larger diameter relative to the rest of the stand (Lutz et al. 2018). In British Columbia, some trees have attained diameters up to 6m DBH or heights of 85m (BC BigTree Registry, 2022). Trees that reach these sizes are generally very old and on productive sites; they are also relatively rare (Lindenmayer et al. 2012). For research purposes, it is more common for researchers to use smaller DBH thresholds such as 40 cm (Storch et al. 2018) or 50 cm (Spies and Franklin 1991, Gerzon et al. 2011) as the definition of large trees in the PNW. Large-diameter or very tall live trees are an important structural feature of temperate old-growth forests because of the unique structural properties that come with size (Lindenmayer and Laurance 2017). The presence of large branches can be platforms for epiphytic plants, wide boles higher up in the canopy provide opportunities for large cavity nesters (e.g., pileated woodpecker) and deeply fissured bark provide microclimates amenable for distinctive insects or fungi (Spies et al. 2018, Storch et al. 2018).

The marbled murrelet (*Brachyramphus marmoratus*), a small seabird, typically builds nesting platforms in trees greater than 30 m tall on branches between 15 cm to 74 cm in diameter, a diameter that is unlikely to be found in younger stands that high in the canopy. Because of its specific habitat requirements for coastal old growth, this species is threatened due to habitat loss in Canada (Committee on the Status of Endangered Wildlife in Canada 2012). Understanding and preserving old forest structure is therefore an essential part of conserving marbled murrelet populations or any other organism that is associated with old-growth forests.

While much is known about old growth and its structure in the Pacific Northwest, the research has not been evenly distributed among ecosystem types. Daniels et al. (2017) found there were knowledge gaps about the structure of forests among different biogeoclimatic zones (Green and Klinka 1994). Specifically, it was found that in the south coast of BC, the most well understood BEC zones were the wettest and driest along the climatic gradient from the ocean towards the interior (Daniels et al. 2017). Knowledge gaps also exist in the understanding of the long-term effects of fire in these same ecosystems (Hoffman et al. 2018). This provides an opportunity to study post-fire old-growth structure in the mid-moisture regime coastal BEC zones.

1.3.6. Old-Growth Plant Community

Biodiversity is the variety of genetic, species and ecosystem diversity between or across landscapes or ecosystems. Forests with more biodiversity usually support more ecosystem services and are more resilient to natural disturbance than less diverse communities (Thompson 2009, Roach et al. 2021). Some communities of vascular plants, bryophytes, lichens and fungi are more strongly associated with old-growth forests than younger forests, partly due to the long period of undisturbed development, the presence of large live and dead trees and large coarse woody debris as habitat (Pollock and Beechie 2014). For example, cyanolichens, a functional group of epiphytic canopy dwelling lichens that are highly associated with old forests, are important nitrogen fixers. Some species are damaged when they receive too much light from recent disturbance such a logging, while young even-aged stands have relatively darker canopies that have a negative effect on the lichen establishment and survival (Radies and Coxson 2004). For this group of organisms, canopy gap processes of old forests provide the optimal light levels (Gauslaa et al. 2019).

Biodiversity increases the likelihood that some aspect(s) of the pre-disturbed state will continue as a 'biological legacy' after a disturbance (Franklin 1990). Surviving vegetation after a fire, for example, can provide restorative genets (e.g. seeds) and ramets (e.g. rhizomes) to 'jump-start' biological succession and facilitate growth of the early seral community and general forest regeneration and development (Swanson et al. 2011, Seidl et al. 2014, Pulsford et al. 2016). Ecological succession is a complex and dynamic process by which species dominance changes over time, dependent on the site conditions, level of disturbance, spatial patterns of vegetation and access to resources (Chang and Turner 2019). Early seral species are thought to influence mid-seral development which eventually gives way to species that are associated with late seral or climax stages (Finegan 1984). The diversity of plant species found in old-growth stages likely stems from the conditions that created diversity in earlier seral stages (Mackinnon 1998, Spies 2004), though the relationship is not clear (Gill et al. 2017).

Some species that are associated with old-growth forests provide human-use values. For example, First Nations of North America have long histories of traditional use of understory plants for medicinal, edible, cultural and trade purposes (Clason et al. 2008, Sutherland et al. 2016). Therefore, the conservation of old-growth areas that support these species is essential for these human use values in addition to their ecological values.

1.3.7. Old Growth After Fire

The short-term weather and long-term climate of an ecosystem are in part reflected in its range and extremes of temperature, wind and precipitation, which in turn impact their predisposition to disturbance such as wildfire when extreme droughts and heat events arise. Ecologically speaking, burn severity describes the impact of fire on vegetation and soils due to heating and combustion (Key and Benson 2006). Fire intensity, the energy released during a fire, is affected by factors such as topography and existing fuels, while fire severity, its ecological impact, may be affected by weather prior to the fire and presence of fire adapted plant species (Keeley 2009, Whitman et al. 2018). Fire intensity and burn severity are related, however more intense fires do not necessarily mean more severe burns if they move fast and die out quickly; the relationship is dynamic and complex. Fire behaviour is thus highly variable resulting in a heterogenous landscape of unburned, lightly burned, and severely burned patches (Burton et al. 2008, Guindon et al. 2021).

Fire severity is usually described with terms such as low, moderate and high and refers to the level of changes to vegetation structure and composition as well as physical and chemical changes to soil. Low-severity fires move along the surface of the forest floor, removing accumulated fine woody debris and charring or consuming understory vegetation but leaving the organic layer mostly intact with low amounts of tree mortality. Moderate-severity fire consumes all understory vegetation, large CWD is consumed or charred, organic soils are mostly consumed, tree mortality is substantial but some living canopy can still remain. High-severity fires consume almost all surface litter, and organic soil layers, the tree canopy is consumed, and all or almost all trees are killed (Key and Benson 2006, Keeley 2009, Whitman et al. 2018). Fires that consume the forest canopy are the most severe because removal of the overstory significantly changes the ecosystem through increased sun exposure, more extreme temperatures on the ground and in the air, higher wind velocity, and lower levels of relative humidity and moisture in litter and surface soil. (Swanson et al. 2011). These changes affect the plant community according to the varying shade tolerances and microclimate adaptations of individual species.

Burn severity measurement depends on the objectives of the project, which would determine the ecosystem elements to measure and the scale of study. The Composite Burn Index (CBI; Key and Benson 2006), is a field-based method that measures impacts to post-fire substrate and four vegetation strata to produce a single scaled score ranging from 0 (unburned) to 3 (high-severity burn) (Guindon et al. 2021). Having a scaled value allows different plots to be compared to each other within and among fires. This score can also be used to validate remotely sensed indices of burn severity over the entire footprint of one or more wildfire events.

In the case of coastal temperate old-growth forests, stand-initiating wildfires have historically not been an important natural disturbance (Lertzman et al. 2002, Gavin et al. 2003, Daniels et al. 2017). The moist climate facilitates highly productive ecosystems, generating plentiful amounts of woody debris as potential fuels, but the high levels of precipitation and stand structure can keep the understory cool and moist long into drought periods (Daniels and Gray 2006, Hessburg et al. 2019).

Structure in coastal old growth is more well developed than in younger stands partly because of those long periods between major stand-initiating disturbances (MacKinnon 2003, Daniels and Gray 2006, Daniels et al. 2017). Daniels et al. (2017) found that fires in BC's coastal forests often occurred in a patchwork of high, moderate and low burn severities (Daniels and Gray 2006, Daniels et al. 2017). As a result of differing burn severities, there is natural variation in the remaining structure or "biological legacies" left after a fire (DeLong and Kessler 2000, Franklin et al. 2002, Seidl et al. 2014). These legacies could be survivor trees that may act as seed sources for regeneration (Seidl et al. 2014) while standing or fallen dead wood acts as habitat (e.g., for wood-excavating insects and birds) or nutrients for future vegetation and fungi (Donato et al. 2016). The largest trees are more likely to survive fire, while saplings are more likely to perish (Franklin et al. 2002). As a result, the effect that post-fire legacy structures have on forest development in temperate coastal rain forests can be seen at least 120 years after fire (Hoffman et al. 2018).

Structure left after a fire therefore plays a role in the "starting point" for forest regeneration (DeLong and Kessler 2000, Franklin et al. 2002). Multiple fires in a landscape over multiple years typically exhibit variation in fire behaviour (e.g., surface fire vs. crown fire) within and among wildfire events; this variation in burn severity then determines the ecosystem recovery trajectory (Bartels et al. 2016). Over time, a mosaic of early successional (e.g., young trees) and late successional (e.g., surviving old trees) vegetation and legacy structures build up within an ecosystem, creating the various habitat niches, seed sources and nutrient cycles that characterize an old-growth forest (Seidl et al. 2014).

1.3.8. Remote Sensing of Forest Fire Impacts

Remote sensing involves the observation of the Earth from satellite and airborne sensors to monitor environmental changes. Satellites are equipped with sensors that receive light reflected off the Earth's surface or atmosphere and translates that information into digital data that can be analyzed quantitatively or portrayed as an image (Figure 1 – Left is imagery from 2013 before fire and right imagery from 2018 after fire in the south coastal mountains of British Columbia, approximately 22 km north of Harrison Lake. Imagery is a composite created by ESRI from imagery provided by Maxar and Earthstar Geographics. Fire perimeter polygon provided by the Province of British Columbia.

). Different sensors can read different wavelengths of light, which allows researchers to manipulate the information to understand patterns that otherwise wouldn't be visible to the human eye (NASA 2020). In this case, variations in the reflected light in fire-disturbed areas are highly correlated with ground-based measurements of burn severity such as CBI (Key and Benson 2006).

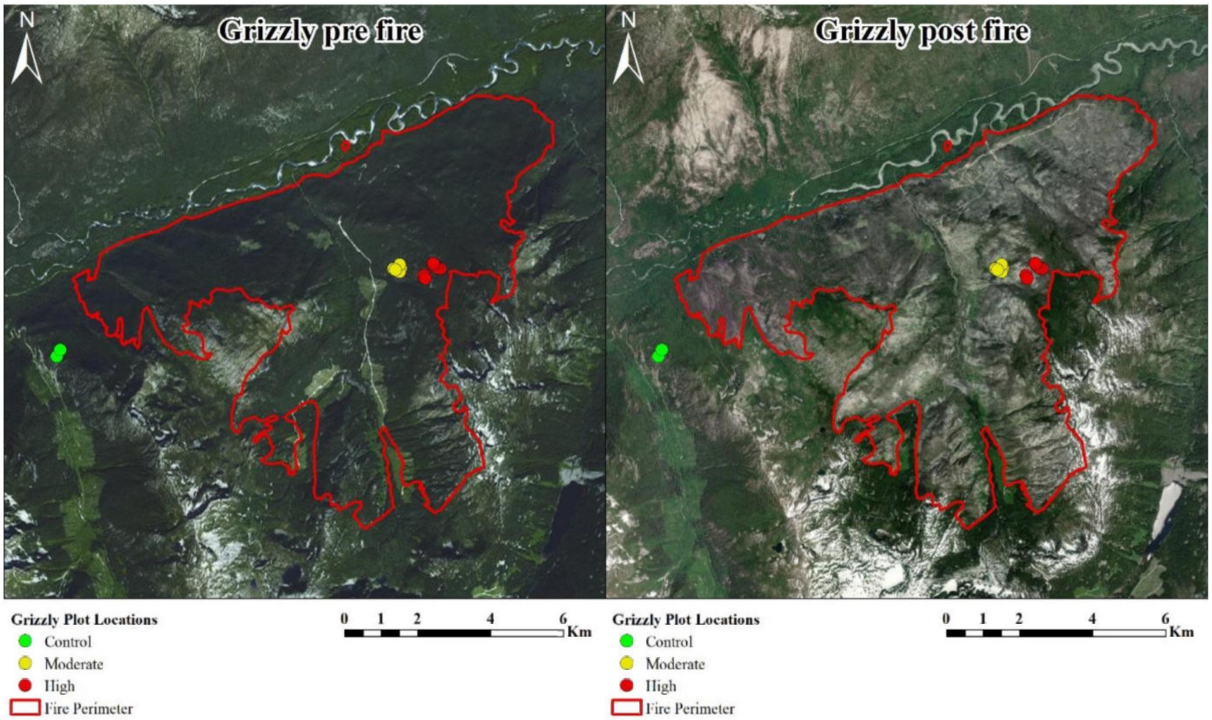

Figure 1 – Left is imagery from 2013 before fire and right imagery from 2018 after fire in the south coastal mountains of British Columbia, approximately 22 km north of Harrison Lake. Imagery is a composite created by ESRI from imagery provided by Maxar and Earthstar Geographics. Fire perimeter polygon provided by the Province of British Columbia.

In an undisturbed forest, healthy vegetation reflects near-infrared light (NIR, ~0.851 - 0.879 µm) to a much greater extent than shortwave-infrared (SWIR, ~1.566 – 2.294 µm) light (NASA 2020). In burned areas and areas with unhealthy vegetation, SWIR wavelengths reflect much more than NIR. Extracting the differences between relative reflectivity of NIR and SWIR wavelengths of light, a ratio of healthy to burned vegetation is created, which is then scaled to create the Normalized Burn Ratio (NBR; Equation 1). By differencing pre-and post-fire NBR images, a delta-NBR (dNBR; Equation 2) value is created, where areas that have experienced relatively little ecological change have low values, and areas that have experienced great change have high values (Key and Benson 2006) (Figure 2).

Equation 1 : $$\mathrm{NBR} = \left(\frac{\mathrm{NIR} - \mathrm{SWIR}}{\mathrm{NIR} + \mathrm{SWIR}} \right)$$

Equation 2 : $$\mathrm{dNBR} = \left(\mathrm{NBR}_{\textit{pre-fire}} - \mathrm{NBR}_{\textit{post-fire}} \right) \times 1000$$

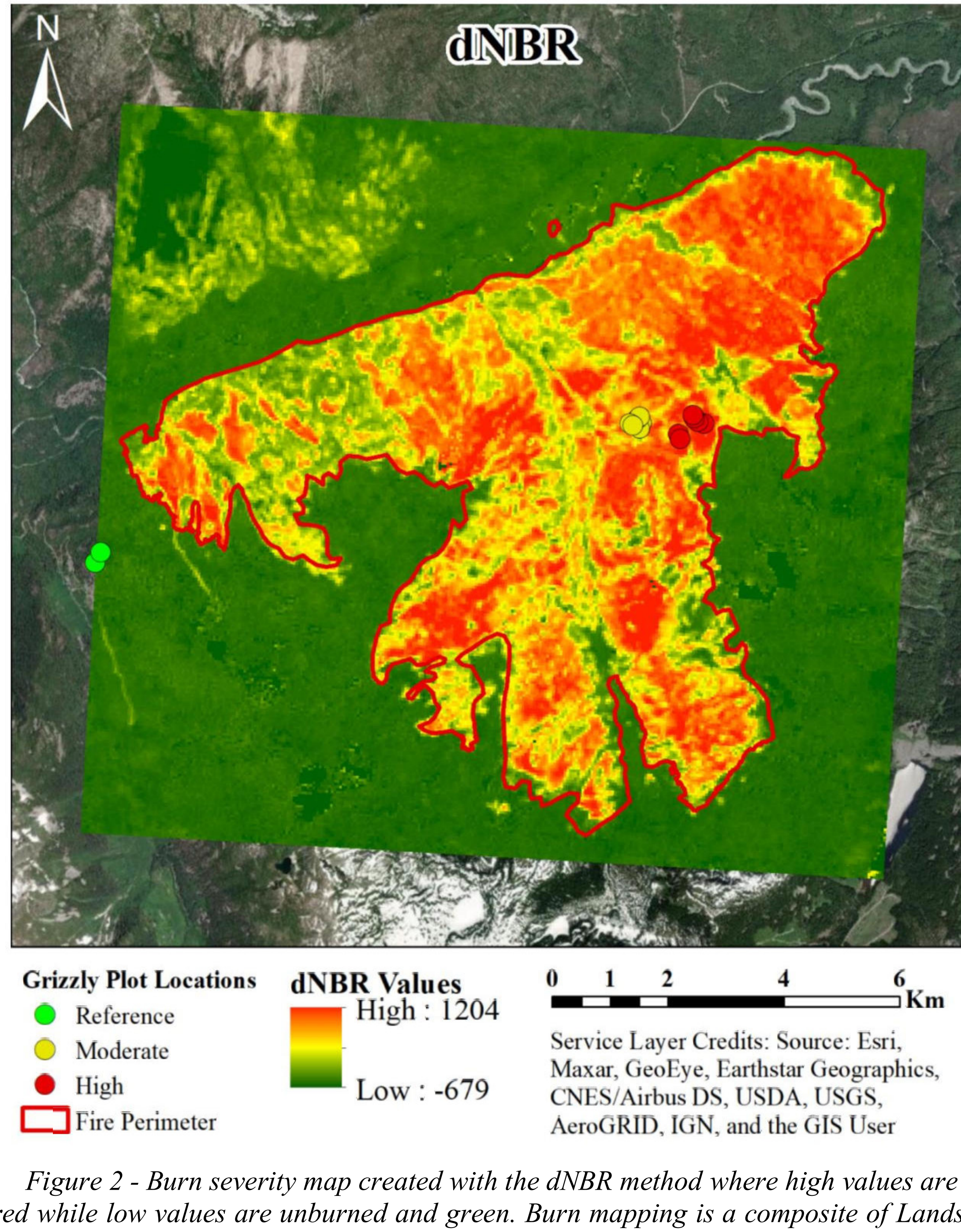

Figure 2 - Burn severity map created with the dNBR method where high values are severe and red while low values are unburned and green. Burn mapping is a composite of Landsat and Sentinel imagery of varying dates, provided courtesy of the Province of British Columbia. Base map imagery provided by ESRI.

The dNBR method is a measure of fire-induced change between pre- and post-fire pixels, but if the pre-fire vegetative cover is low (e.g., sparsely vegetated tundra) the resulting change in dNBR value will also be low, even though fire induced mortality may be high (Parks et al. 2014). In other words, if a pixel is sparsely treed and all trees are killed by fire, the resulting dNBR value may register only a low or moderate change, while a densely treed pixel experiencing the same mortality will register as a high amount of change (Miller and Thode 2007). In this case, both pixels went through stand-replacing disturbance, but only the second registered as high severity.

To minimize these types of errors in classifying high-severity burns in areas with mixed vegetation cover, equations such as the Relativized Burn Ratio (RBR, Equation 3, Figure 3) have been developed (Parks et al. 2014). In principle, relativized equations may also allow more consistent burn severity classification among multiple wildfires across entire landscapes as they would all be calculated on the same scale, though there is debate about which relative equations provide the highest correspondence with ground-based measurements of burn severity (Miller and Thode 2007, Parks et al. 2018, Whitman et al. 2020).

Equation 3 :

$$\mathrm{RBR} = \left(\frac{\mathrm{dNBR}}{(\mathrm{NBR}_{\mathrm{prefire}} + 1.001)} \right)$$

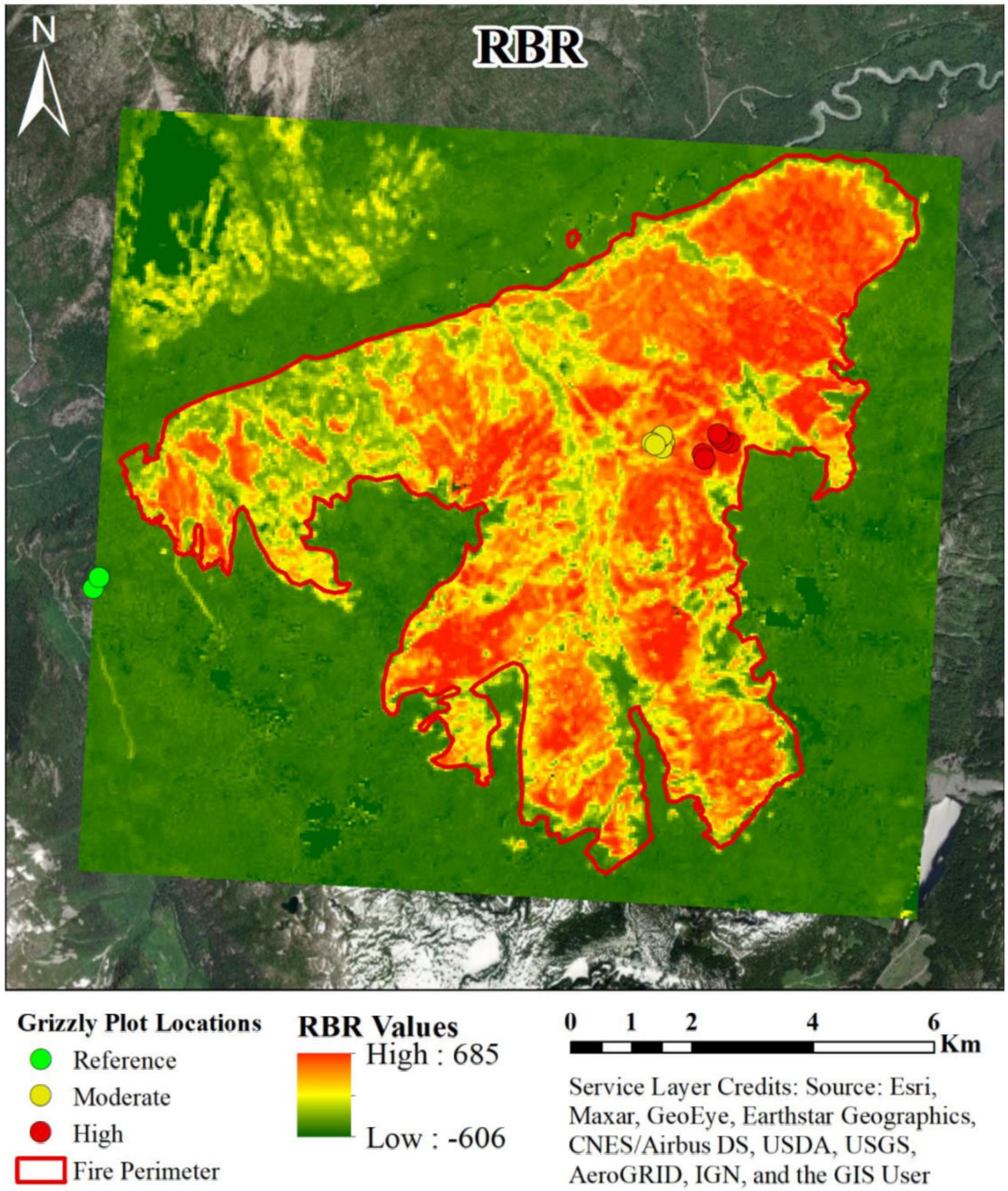

Figure 3 – RBR Map of the Grizzly Fire. It may look identical, but there are subtle differences, in particular notice the difference in values between high on this figure and the previous. Burn mapping is a composite of Landsat and Sentinel imagery of varying dates, provided courtesy of the Province of British Columbia. Base map imagery provided by ESRI.

Landsat 9, the most recent satellite of the Landsat series, has a resolution of 15m, 30m and 100m depending upon the spectral band being analyzed (NASA 2020), while the Sentinel-2 satellites have resolutions of 10m, 20m and 60m (European Space Agency 2020). At these moderate resolutions, issues can arise in detecting change less than the size of a pixel and change beneath the canopy. Sub-pixel heterogeneity can occur where burned and unburned vegetation exist in the same pixel, meaning that a mixed signal is often interpreted as unburned when in reality it is burned (Kolden et al. 2012). Since imagery generally captures the forest canopy rather than the forest floor, sub-canopy burning in a low severity fire would also not be detectable (Meddens et al. 2016), therefore remotely sensed indices need to be validated with ground surveys if more than canopy-level assessment is required (Guindon et al. 2021).

Regardless, remote sensing measures of burn severity are a continuum of values which have been correlated with on the ground measures of burn severity. In nature, there is no exact value at which an ecosystem is deemed to have crossed a threshold (e.g., severely vs moderately burned). For practical interpretation, the continuous values can be grouped or divided into categories such as low, moderate or high severity or even more nuanced categories (Table 2) depending on the objectives of the project.

Table 2 - Examples of multiple burn severity categories with their corresponding dNBR ranges (from Key and Benson 2006).

Severity level	**dNBR range**
Unburned	– 600 to + 99
Low severity	+ 100 to + 269
Moderate-low severity	+ 270 to + 439
Moderate-high severity	+ 440 to + 659
High severity	+ 660 to + 1300

1.4. Research Objectives

My research aim was to understand how fire severity differs between old growth sites within the CWHms1 and compare the residual carbon, structure and floristic similarity to the range of natural variation of those variables in unburned reference plots. To do this, I needed to relate a remotely sensed measure of burned severity to the NRV of unburned stands in order to understand if there is a burn severity threshold at which selected old growth forest values are compromised (Section 1.2). I approached this goal through the following objectives:

1) Using ground-based burn severity measures to validate remotely-sensed relativized burn ratio (RBR) values.
2) Comparing the estimated aboveground carbon stocks in burned and unburned stands to identify sites that are outside the natural range of variation for unburned old growth.
3) Measuring physical stand structure from ground-based measurements to compare with RBR values to identify sites that are outside the natural range of variation for unburned old growth.
4) Comparing plant species richness and floristic similarity between burned and unburned stands to identify sites that are outside the natural range of variation for unburned old growth.

Chapter Two – Methods

2.1. Study Area

This study was conducted in British Columbia's Coastal Western Hemlock (CWH) Biogeoclimatic (BEC) zone (Figure 4). The CWH occurs from sea level to middle elevations, its climate moderated by moist air coming from the Pacific Ocean heading east over the Coast Mountains. Due to orographic uplift, the windward edges of the range experiences relatively large amounts of precipitation, tapering off in the rain shadow inland. In general, the climate has cool summers and mild winters. As its name implies, western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) is the most common late-successional tree species, with Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), western redcedar (*Thuja plicata* Donn ex D.Don) and amabilis fir (*Abies amabilis* Douglas ex J.Forbes) being the other important tree species (Meidinger and Pojar 1991).

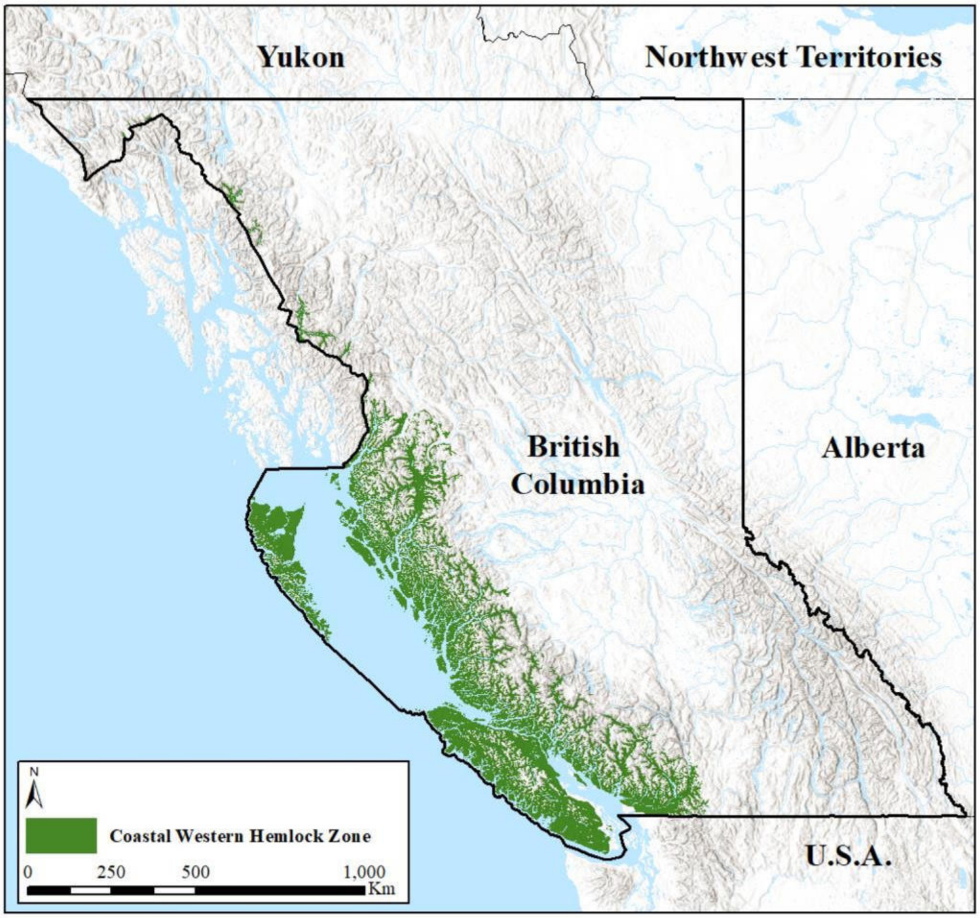

Figure 4 – Overview map of British Columbia, Canada, showing the range of the Coastal Western Hemlock (CWH) biogeoclimatic zone.

Within an elevational band between 650 m to 1350 m of the CWH is the southern moist submaritime variant (CWHms1, Figure 5). The mean annual temperature historically has been 5.7 °C, with a seasonal average ranging from -4.4 °C to 15.3 °C. The historic total annual precipitation has been 1415 mm, with approximately 81% of the precipitation occurring between October and April, and about 46% of that precipitation falling as snow (Green and Klinka 1994). Climate models predict that between 2041 and 2070, total annual precipitation along coastal British Columbia could marginally increase, but mean annual temperatures at 1000 m elevation however could rise to 7.3 °C, with seasonal averages ranging from -2.5°C to 17.8°C. These models predict that though precipitation may remain steady, only 23% would fall as snow (data derived from ClimateBC based upon general circulation models for an intermediate GHG emission scenario (Wang et al. 2016).

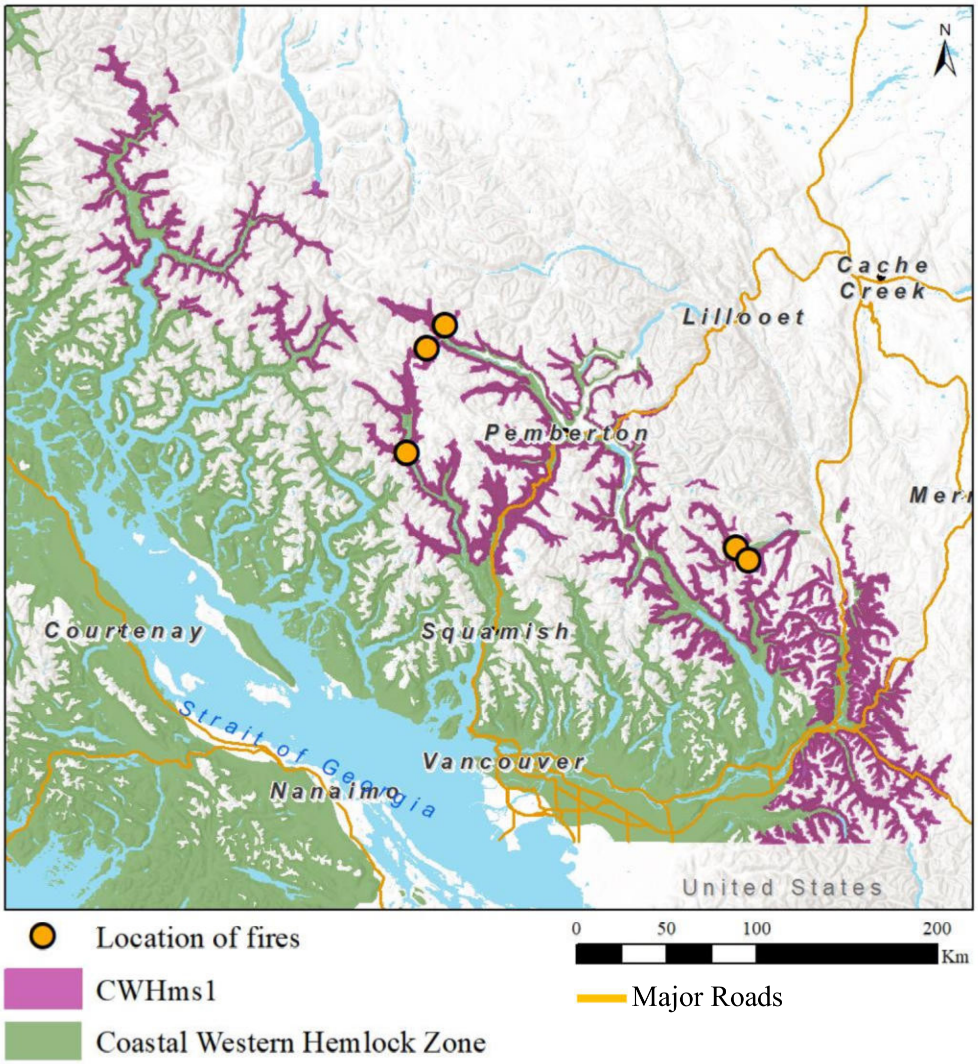

Figure 5 - The entire distribution of the CWH moist submaritime southern variant (CWHms1) in coastal British Columbia, showing the approximate locations of field sampling.

Historical data show that summers in the CWHms1 tend to be drier than nearby maritime CWH subzones, which increases the opportunity for major disturbance such as fire (Green and Klinka 1994). Current Provincial guidelines estimate that the mean return interval for stand-initiating disturbances (e.g., fire) in the CWHms1 is around 200 years. Fires may have been of moderate size (1000 ha), with occasional larger fires (B.C. Ministry of Forests and B.C. Ministry of Environment 1995). Current research shows that historically, a series of more frequent smaller mixed-severity fires with less frequent (return interval greater than 200 years) stand-initiating moderate fires seeming to have prevailed (Daniels et al. 2017).

The summer of 2014 saw the third-highest amount of burned area across BC up to that point, followed by 2015 with an above-average number of wildfires (B.C. Wildfire Service 2020). Unusually hot and dry weather conditions in 2015 led to several large fires in BC's coastal forests (B.C. Wildfire Service 2022) including the Boulder, Meager, Elaho, Nahatlatch and Grizzly wildfires (Figure 6). These fires were all relatively large for the coastal climate and all occurred within one year of each other. While forestry activities occurred in each area, patches of undisturbed primary forest still existed nearby, making them ideal study areas for understanding fire impacts on old forest values in temperate coastal areas.

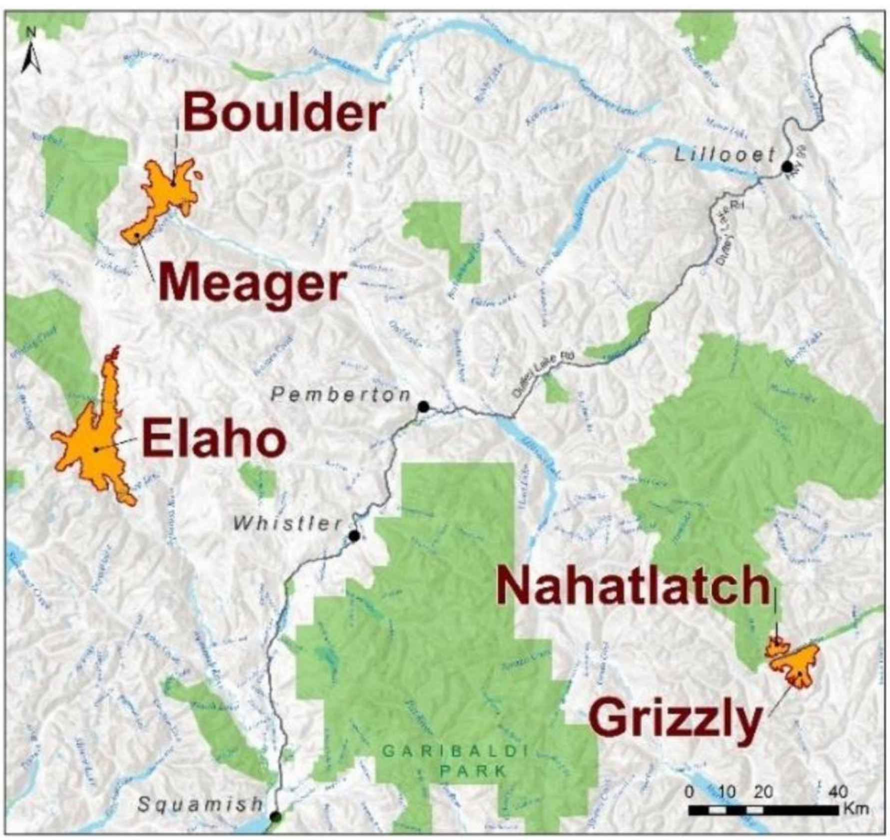

Figure 6 – The five 2014 and 2015 wildfires examined in this study. Research sites were spread out up to 200 km apart to capture the variation in conditions within the CWHms1 BEC variant.

2.2. Site Selection and Field Data Collection

Initial site selection was based on a GIS overlay analysis, combining spatial layers of vegetation cover (e.g., primary forest, having no known harvesting), topographic information (mid-elevation topography (650 to 1200 m)) and biogeoclimatic mapping (in or adjacent to the CWHms1). This was then combined with burn severity spatial layers produced by the Province of BC which use the delta normalized burn ratio (dNBR) method (Key and Benson 2006) to compare multi-spectral satellite imagery before and after a fire to map burn severity. After all layers were combined, candidate areas were investigated with helicopter and ground surveys to find final plot locations that met the above criteria as well as avoiding areas close to logged or burned edges.

To collect a broad picture of pre- and post-burn conditions in the CWHms1, five general locations were selected: Boulder, Meager, Elaho, Nahatlatch and Grizzly within the perimeter of four wildfires as mapped by the BC Wildfire Service. Note that Boulder and Meager sites were both within the final Boulder fire perimeter (Table 3). Sample plots were located far enough apart from each other to capture a sample of the unique conditions found within the CWHms1, while avoiding confounding factors such edge effects from nearby logging activity. Each site contained 14 plots: 6 at sites mapped as high burn severity, 6 moderately burned, and 2 unburned reference sites for a total of 70 plots for this project. Each site was surveyed once, but this process took two seasons, in the summers of 2018 and 2019, three and four years after wildfire.

Table 3 - Summary information on the four coastal wildfires assessed in this study and their approximate centre locations and identification

Site	Latitude °N	Longitude °W	Fire Discovery Date	Cause	Fire Size (Hectares)

Boulder	50.65058	-123.3954	June 30, 2015	Lightning	6684
Meager	50.27362	-123.6143			
Grizzly	49.92458	-121.8815	June 30, 2015	Lightning	2912
Nahatlatch	50.57847	-123.4881	July 14, 2014	Lightning	793
Elaho	49.94459	-121.9613	May 21, 2015	Lightning	12495

Field methods for plot description (Figure 7) followed the methods of the BC field manual for Describing Terrestrial Ecosystems (B.C. Ministry of Forests and Range and B.C. Ministry of Environment 2010). While approximate field sampling areas were estimated through the above mentioned GIS overlay analysis, final plot location was determined in the field based on accessibility (proximity to a helicopter landing site) and safety (e.g., not on a cliff). Within an 11.28 m radius (400 m^2 area) of plot centre, general site description was recorded with GPS elevation, dominant slope (handheld clinometer), aspect (magnetic compass calibrated to true north) and slope position. Canopy and tree heights were measured with a laser rangefinder, while crown closure and percent cover by vegetation strata and substrate was estimated by eye, referencing the foliage cover examples found in the Land Management Handbook (Figure 3.2 in B.C. Ministry of Forests and Range and B.C. Ministry of Environment 2010). All trees greater than 12.5 cm DBH were recorded for diameter, species, tree or snag class, height, char height, crown scorch and presence of any insect damage post-fire. Within a 5.64 m radius (100m^2 area) of plot centre, all understory plant species were identified and their cover visually estimated to the nearest 0.1% cover (i.e., to approximately 0.4m^2 of plot area, as per table 3.3 in the BC land management handbook #25 (B.C. Ministry of Forests and Range and B.C. Ministry of Environment 2010). If only a single small specimen was located, it would receive a percent cover such as 0.001% just to denote its presence. Identification was to species level, except for several herb species such as sedges that were identified to genus only. Small trees (between 5.0

and 12.5 cm DBH) were measured for the same variables as large trees; trees with DBH <5.0 cm were just recorded for species and percent cover.

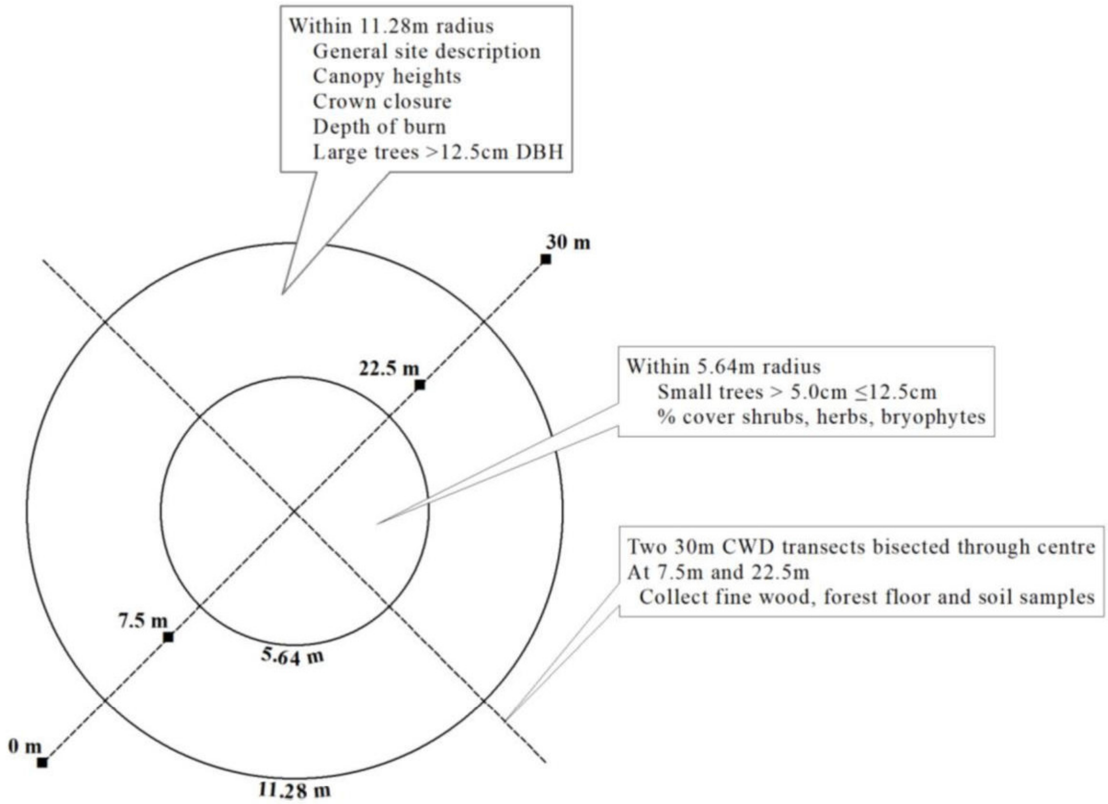

Figure 7 – Diagram of data collection plot summarizing data collected within 11.28 m, 5.64 m radii and along 30m bisecting transects. Note that transects were not adjusted for slope, so that the projected (map) area of each plot may vary depending on slope steepness.

A random number generator was used to pick a direction to establish two perpendicular 30 m transects bisected through plot centre. Along each transect small woody material (1.1-2.5 cm diameter) was tallied, while wood greater than 2.6 cm was recorded for its position along the transect, while species, diameter, length and decay class were also recorded (following decay class definitions from Maser et al. 1979). At 7.5 m and 22.5 m along each transect, small woody material was collected within a 10 cm by 10 cm frame, bagged and labelled for weighing in the lab. Forest floor to the depth of mineral soil within the frame was destructively sampled and the depth of duff measured. Mineral soil was sampled to a maximum depth of 15 cm. All forest floor samples were kept separate, while mineral soil samples were bulked (combined and mixed) into a single bag to give a site average for chemical analysis.

Old-growth structure and composition measures included: the numbers and percent cover by species of live trees, shrubs, and herbs; the numbers, heights, diameters, decay class and burn class by species of standing dead structure; and the lengths, diameters, decay class and burn class by species of downed woody material. Carbon measures included: biomass of live and dead aboveground pools (with no estimates of foliage) and chemical analysis of duff/forest floor and mineral soils for below ground carbon pools, but no estimate of live or dead root pools.

In total, 70 plots in 5 sites were established in burned stands. Of these, 56 were within the CWHms1 BEC boundary, with the other 14 sites just outside the CHWms1 within 350 m of the nominally mapped boundary. Boulder reference sites were ~300 m uphill, nominally in the Mountain Hemlock Leeward Moist Maritime (MHmm2) variant, Grizzly high-severity sites were ~350 m uphill in the Engelmann Spruce Subalpine Fir Stein Moist Warm (ESSFmw2) variant, and the Nahatlatch reference and low/moderate sites were within the Coastal Western Hemlock Southern Dry Submaritime (CWHds1) variant, ~200 m downhill of the nominal boundary to CWHms1. The BEC boundaries are thresholds modelled from many field samples of vegetation, soils and topography, combined with air photo interpretation, expert opinion and finally a GIS analysis that predicts the extent of the BEC unit (Ministry of Forests 2022). This means that the location of any given boundary is estimated, and more truly represents a broad transition zone; thus, this nominal difference in BEC zones among sites is considered negligible.

Boulder, Meager and Elaho sites were located west of the town of Pemberton, within 80 km of each other, while Nahatlatch and Grizzly sites were located east of Pemberton within 11 km of each other. The approximate distance between the Nahatlatch/Grizzly fires and

Boulder/Elaho fires was 200 km. Being in different valleys, the terrain varied as would be expected. For example, Boulder sites faced in a southerly slope direction while Elaho faced north-easterly, Grizzly faced westerly, Meager faced south-easterly and Nahatlatch southerly. At this latitude, northerly aspect slopes receive significantly less sun exposure than south-facing slopes, which adds variability in species composition due to differences in moisture retention and growing conditions. Other examples of natural variation included the amount of exposed rock prior to fire (Figure 8) or dominance by a particular herbaceous species (e.g., fireweed, *Epilobium angustifolium*, in middle photo in Figure 9).

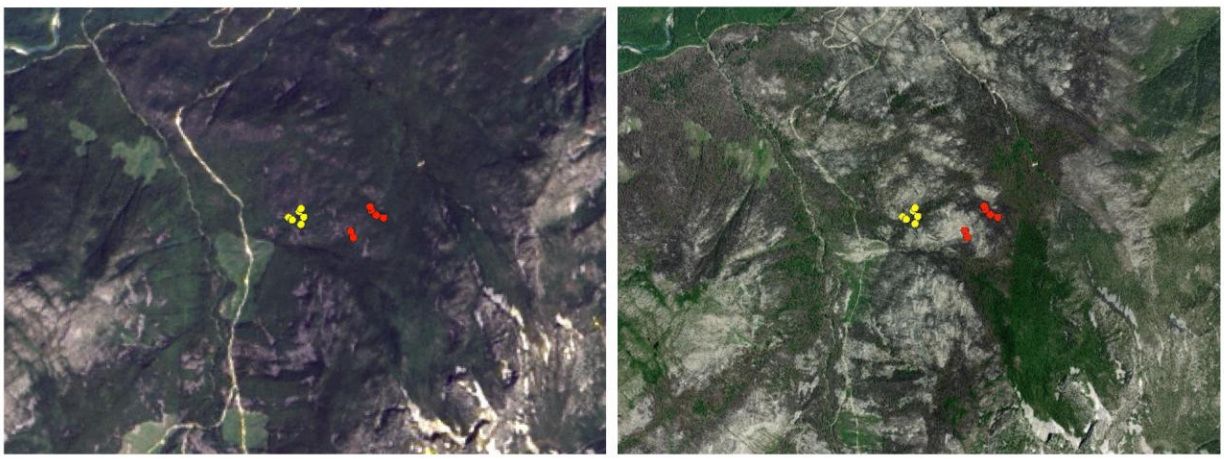

Figure 8 – Satellite imagery of Grizzly fire area captured on July 26th, 2013 (left, Landsat 8 OLI imagery) before fire in 2015 and on July 16, 2018 (right, Imagery © 2018 Maxar) 3 years post-fire. Moderate severity (yellow) and high severity (red) points represent plot locations. Though the plot areas were undisturbed prior to fire, there was still a fair amount of exposed rock, explaining partially the increase in exposed rock post-fire.

Reference Moderate Severity High Severity

Figure 9 – Photographs of typical plots to show differences in aboveground structure, living trees and understory vegetation among reference (unburned), moderate and high severity burn sites. Left is from a reference site in the Boulder area, middle is a moderate burn site from Elaho, right is a high severity burn site within the Meager area. These photos were taken 4 years post-fire.

2.3. Description of the Burned Stands

The wide range of conditions found in burned and unburned stands in the CWHms1 is described in *Table 4*.

Table 4 – Site descriptions comparing burned and unburned sites. An xx in the tree code denotes trees were so burned or decayed that species identification was only possible to genus. Ba=Abies amabilis, Cw = Thuja plicata, Fd = Pseudotsuga menziesii, Hm = Tsuga mertensiana, Hw = Tsuga heterophylla , Hxm = Tsuga mertensiana x heterophylla hybrid, *Pl = Pinus contorta, Pw = Pinus monticola , Sxx* = Unknown *Picea, Yc = Chamaecyparis nootkatensis, Xc* = Unknown conifer

	Reference Sites				
	Boulder	**Elaho**	**Grizzly**	**Meager**	**Nahatlatch**
Number of plots	2	2	2	2	2
Elevation (m)	1146	860	1107	1167	759
Aspect	270	195	334	205	229
Slope	70%	22%	22%	50%	17%
Latitude	50.7143	50.3933	49.9117	50.5722	49.9411
Longitude	-123.4608	-123.5307	-121.9655	-123.5238	-121.9694
Dominant Species	Ba	Ba	Hw	Ba	Fd
Secondary Species	Hw	Hm, Yc, Hw, Hxm	Ba, Hm, Fd	Hw, Fd, Cw	Hw, Cw, Pl
% Dead Trees*	18%	4%	16%	14%	12%
Total Basal Area (m^2 ha^{-1})	88	52	79	89	55

Table 4, continued.

	Burn Sites				
Number of plots	12	12	12	12	12
Elevation (m)	1107	823	1121	1177	770
Aspect	138	67	276	138	168
Slope	58%	33%	45%	38%	40%
Latitude	50.6519	50.2831	49.9251	50.5848	49.9439
Longitude	-123.391	-123.6112	-121.8777	-123.4775	-121.9594
Dominant Species	Cxx	Ba	Xc	Ba	Fd
Secondary Species	Xc, Hw, Bxx, Ba, Fd, Hxx	Hxx, Xc, Hw, Cy, Cw, Hm, Cxx	Hw, Pl, Fd, Cw, Sxx, Ba	Hm, Hw, Xc, Hxx	Xc, Pl, Cxx, Cw, Cy, Bxx
% Dead Trees*	94%	96%	98%	93%	99%
Total Basal Area (m^2 ha^{-1})	53	57	41	33	22

*Trees defined as having DBH $\geq$ 12.5 cm

2.4. Burn Severity Assessment

Burn severity was assessed after field work was completed using a modified burn severity index modelled after the composite burn index (CBI) developed by (Key and Benson 2006). In their index, data is mostly collected by ocular estimation of changes to various biological strata, from tree canopy crown loss down to depth of soil charring. The index is essentially an average of many different plot-level estimations. One benefit of this approach is that it is acceptable to omit factors as long as enough overall factors have been collected. However, as field data collection occurred up to four years after the fire, short-term assessment variables such as consumption of fuels were not considered as this was a measure of remnant dead wood after fire, while woody material could have fallen from dead trees in the intervening years.

Since the same variables were recorded in reference and burn plots, I created site specific comparisons between a plot and its average reference plot (e.g,, Boulder burned only compared to Boulder reference). In the CBI methodology, comparisons are made with field forms putting observations into categorical bins, but I used R to create a continuous score that subtracted observed values between a plot and its average reference site (e.g., % cover of shrubs in burned – unburned = % cover difference). I then divided the value by the unburned value to produce a percent relative change (Equation 4). The CBI variables I substituted included the percent green in the intermediate and big tree strata for percent living trees, as needles had fallen off most dead trees and not estimating % cover by colonizing species.

Equation 4: Relative Change in strata = $\left(\frac{burned - unburned}{unburned} * 100\%\right)$.

After all variables were differenced, each was assigned to a stratum (substrates, herbs, etc.) and assigned a score based on the burn severity thresholds from Key and Benson (Table 5). For example, if the change in percent cover of exposed rock was between 25 and 40%, it received a score of 2.0. The final CBI score sheet is included in Appendix C.

Table 5 – The composite burn index (CBI) scale as developed by Key and Benson (2006). The modified CBI uses the same 3.0 scale as CBI

No Effect	Low	Moderate	High
≤ 0.1	> 0.1 ≤ 1.5	> 1.5 ≤ 2.25	>2.25

After all field data were collected, remotely sensed burn severity raster values were calculated, assembled, and provided by the Province of BC, using the relativized burn ratio methods of Parks et al. (2018). The method uses the Google Earth™ Engine platform to automatically collect and stack several satellite images before and after a wildfire to average the cloud-free pixel values over all the stacked images. A benefit of this method is the analyst only needs to input a fire perimeter and dates which the script then uses to collect multiple sets of imagery and automatically perform the calculations for dNBR and RBR. As an added quality control measure, a $dNBR_{offset}$ dataset is created that averages unburned pixels outside of a fire to account for differences in phenology (e.g., season) between sets of images. This offset is then subtracted from each dNBR raster, which in turn is used to create the RBR rasters.

RBR was chosen as the basis for comparison instead of dNBR because it is a relativized index, as opposed to an absolute index like dNBR. Relative methods provide a more reliable measure of severity when comparing multiple fires over a landscape (Miller and Thode 2007), and in heterogenous landscapes such as this study area, the $dNBR_{offset}$ accounts for differences among fires due to phenological differences in imagery captured at different seasons (Parks et al. 2018).

The RBR rasters covered all burned sample locations and unburned reference sites except for reference sites B-C-1 and B-C-2. In their case, RBR values were set to 0 as it was assumed there were no major differences in the before and after images. At other reference sites, RBR values ranged from -36 to 59, indicating that there is some variation in values even in unburned reference sites.

2.5. Carbon

Forest floor samples consisted of organic leaf litter and decomposing vegetation and were sampled to the top of the mineral soil layer. Forest floor percent carbon was estimated by measuring the depth of each sample and multiplying it by 100 cm^2 (the area of the sample transect) to produce a volume of biomass. This was multiplied by 0.13 g/cm^3 (bulk density estimate of forest litter from Little and Ohmann (1988)) to produce an estimate of dry biomass weight in grams. This value was then multiplied by the percent carbon for that plot from lab chemical analysis to give an estimate of g carbon per 100 cm^2 sample and multiplied by 1 (e.g., $\frac{Xg}{100cm^2} * \frac{10{,}000cm^2}{1m^2} * \frac{1\ Mg}{1{,}000{,}000g} * \frac{10{,}000m^2}{1\ ha}$) to convert to Mg ha^{-1}.

Fine wood samples were collected in the same 10 cm x 10 cm quadrat at four points within each plot area and were combined into a single bag. This was returned to the lab and dried in an oven at 70° C for 48 hours to remove moisture, then weighed to give a dry biomass weight (grams per $0.04m^2$ per plot). This was multiplied by 100 to convert to Mg ha^{-1}.

Stem carbon allometric equations are from Canadian national biomass equations for British Columbia (Ung et al. 2008) except for yellow-cedar (*Callitropsis nootkatensis (D. Don) Oerst. Ex D.P. Little*) and mountain hemlock (*Tsuga mertensiana (Bong.) Carrière*) which had no regionally specific parameter estimates; the estimated biomass for those species was calculated from equations from Ter-Mikaelian and Korzukhin (1997). Both sets of equations use DBH and height in metres to estimate dry mass in kg. Values for foliage, branches, bark and stem were calculated for all trees, however foliage biomass estimates were removed from burned trees as more than 93% of trees in burned areas were dead and mostly defoliated. In addition, the amount of residual bark and branches was variable in dead trees, with some snags having both, while others had neither. As bark and branch retention had not been recorded in the field, biomass estimates of standing burned trees are an average of minimum estimates (stem only) and maximum estimates (stem, branches and bark). Final carbon was calculated by multiplying stem biomass estimates by 50%. Finally, because all plots were measured in the field with the same radius for plot areas, a correction factor was applied to compensate for differences in slope ($r' = \frac{r}{(\cos\,\alpha)^{0.5}}$, r = new plot radius, α = slope, Kleinn et al. 2002).

Coarse woody debris biomass and carbon estimates were calculated using Huber's formula, $V=LA_m$, where volume equals the length of wood multiplied by the cross-sectional area of the piece at the point where it crossed the transect (Fraver et al. 2007). Volume was then multiplied by species-specific bulk density (Gonzalez 1990) and then a decay class constant was applied (Woodall and Monleon 2008) to provide an estimated biomass of the CWD. As with stem carbon, biomass was multiplied by 50% to produce a final estimate of carbon in Mg ha^{-1}.

It should be noted that site visits were 3 and 4 years after the fire occurrence. Finewood and CWD could have accumulated from decaying trees killed in the fire and were not considered a good metric to measure immediate post-fire effects (e.g., consumption of surface fuels) on the various carbon pools. In other words, carbon was estimated by plot totals as opposed to specific carbon pools because pools may have shifted in the intervening years between fire and sampling.

2.6. Old Growth Structure

A structural index was created to quantify comparisons within burned sites and to unburned reference sites. This structural index is a modified version based on suggested variables from Storch et al. (2018) and McElhinny et al. (2005) using measurements of DBH, length of CWD, height of trees, number of decay classes and species richness of trees

Table 6). Some variables listed in Storch et al. (2018) were not chosen because they were not measured in the field (e.g., bark types) or did not apply (e.g., diversity of flowering trees because no flowering tree species were found).

Values for each variable were standardized to a score between zero and one, where a is the low score value (0), b is the high score value (1) and then averaged by plot (Equation 3).

Equation 5 : $$x_{normalized} = (b - a)\frac{x - min(x)}{max(x) - min(x)} + a$$

Each plot was then assigned an overall structure score by averaging all eight of the normalized variable scores.

Table 6 – Variables used in the structural index and an explanation of their importance; from Storch et al. (2018)

Variable	***Explanation***
Quadratic mean DBH of trees	Common variable to describe stand structure; higher DBHq implies older and taller stands with high biomass, typical forest microclimate, and more presence of habitat attributes of mature forests
Standard Deviation of DBH	High standard deviation of DBH implies a diverse stand structure with patches of different densities and tree dimensions; many niches are provided for different taxa; relates to canopy layering
Standard deviation of mean height	Standard deviation of stand height describes the vertical heterogeneity of stands directly; relates to canopy layering
Mean DBH of CWD	Important structural element for many taxa of xylobiotic species (habitat, food source, regeneration niche); surrogate for deadwood types and number/ha of dead wood pieces
Mean DBH of standing deadwood	Important structural element for many taxa of xylobiotic species (habitat and food source); more suitable than volume/ha because of strong extrapolation effects when sampled on small plots
Number of decay classes	Important for many taxonomic groups; many decay classes indicate a continuous recruitment of deadwood; indicator for natural forest conditions
Volume of large live trees (≥ 40cm DBH)	Large trees have a special function as habitat or source of food for many taxa; they have a greater probability to provide microhabitat structures such as hollows, crown dead wood, etc.
Species richness of trees of live trees ≥ 7cm DBH	Live trees continue to grow and develop structural complexity as habitat and for carbon sequestration. In addition, live trees are an important source for future deadwood to replenish current deadwood as it decays. (McElhinny et al. 2005). Important for diversity of dependent species, in particular host-specific herbivores, detritivores, symbionts and pathogens

2.7. Plant Community

All analyses were performed using the Vegan (Oksanen et al. 2020) package in R studio version 2021.09.0+351 (R Core Team 2021). Species richness was calculated for descriptive purposes using the Vegan specnumber (number of species) function, defined as the number of species of trees and understory plants occurring within a 5.64 m radius of plot centre (100 m^2).

Floristic similarity (species list) between reference sites and burn sites was analyzed using the Jaccard similarity index. This index considers how many species are shared between burned *and* unburned plots, as well as how many species are present in burned *or* unburned plots, but not both. The Jaccard Index is calculated using the Jaccard Distance formula which requires presence/absence data, thus plant community data were transformed for use with the vegdist function in the Vegan package. Jaccard distance is defined as $Jd = \frac{2B}{1+B}$ where B is the Bray-Curtis dissimilarity index, $B = \frac{X+Y-2Z}{X+Y}$, X and Y are the number of species found only at each of the two sites being compared and Z is the number of species that occur at both sites (Oksanen et al. 2020). This was converted to the Jaccard similarity index by subtracting the Jaccard distance from one ($J_i = 1 - Jd$).

Values were calculated for every combination between the 70 plots (n=4900), which was pared down to compare each plot with the mean Jaccard index of the nearby reference plots. In other words, each Jaccard index value represents the degree of floristic commonality, on a scale of 0 to 1, between a plot and its corresponding average reference site, where 0 means no similarity in species presence and 1 means 100% similarity in species presence. For example, the mean similarity between Grizzly high severity burn plots and the two Grizzly reference plots was 0.13, meaning ~13% of species were found in both the high severity burned and unburned plots.

Chapter Three – Results

3.1. Modified Composite Burn Index

The average modified CBI score (and standard deviation in brackets) for sites mapped as high severity sites was 2.7 (0.21), while moderate-severity sites received an average score of 2.3 (0.25), which by CBI guidelines is still considered high severity. The lowest total score by area was 1.9 in Elaho (Figure 10A), which indicates that of the 60 burn plots, none could be considered low burn severity as described by site selection criteria. Tree mortality was greater than 93% at all burned sites (Table 4) which was an important factor in determining why all sites are classified as having experienced moderate or greater burn severity by the modified CBI score.

When looking within individual fires, the threshold between high and moderate severity was more distinct in some fires (e.g., Boulder, Elaho), but overlapped in others (e.g., Grizzly, Meager) which shows that strictly based upon the modified CBI score, there is no obvious threshold across all sites (Figure 10A). No significant differences in modified CBI score were found among all five wildfires as a whole (Kruskal-Wallis, $H=9.18$, $p=0.06$).

To test for differences in grouped burn severity, the underlying distribution of the modified CBI scores were found to be non-normal (Shapiro-Wilk, $W=0.95$, $p=0.01$), thus a Wilcoxon signed rank test was performed which showed a significant difference ($W = 809$, $p<0.001$) between the medians of high and moderate burn severity sites (Figure 10B).

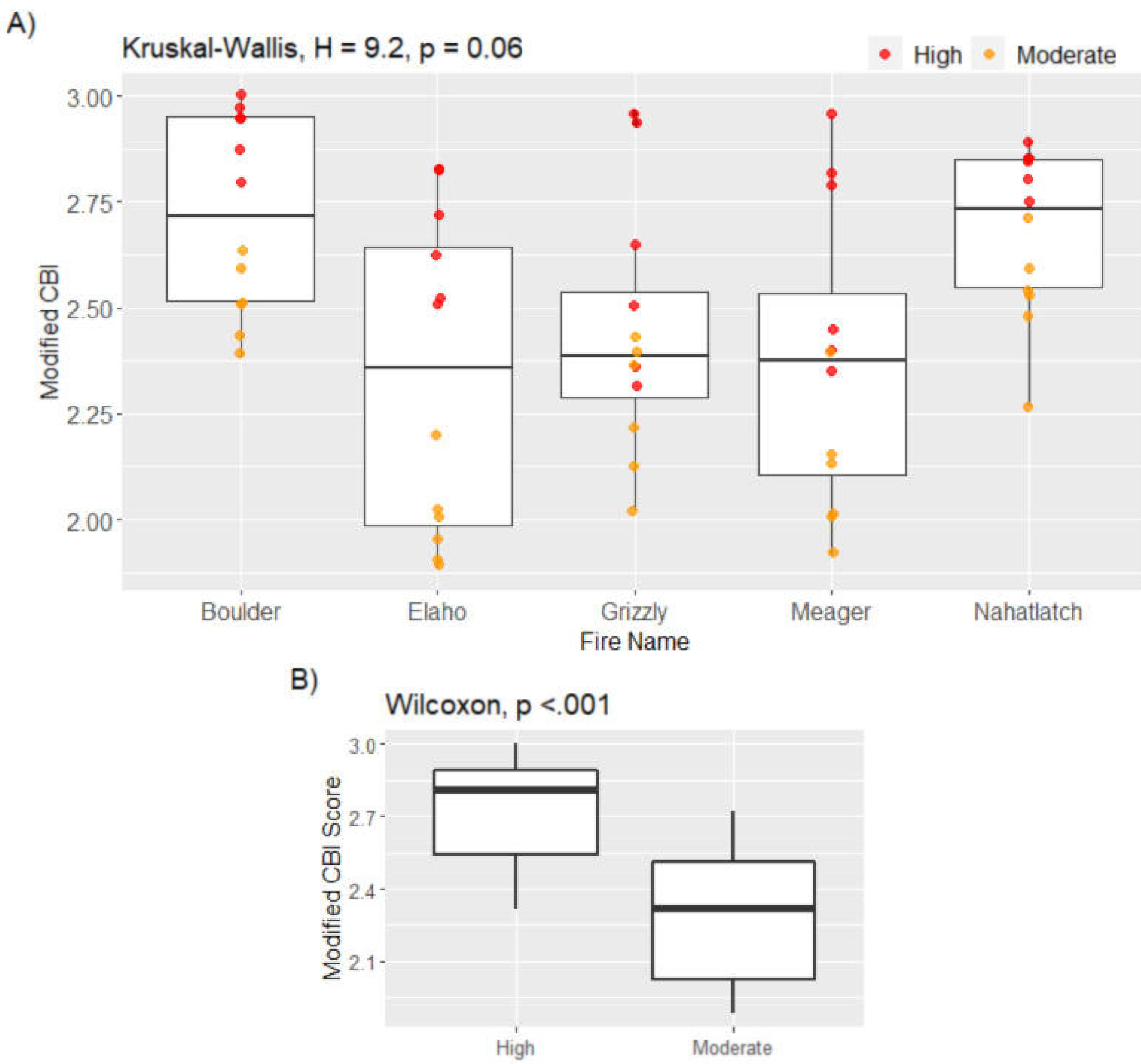

Figure 10 – Box and whisker charts of Modified CBI score by site and by burn severity. Each point represents a plot within a particular fire (x axis) showing the range of values of the modified CBI scores (y axis). Orange points represent moderately burned plots, red points represent high severity plots. All points are >1.8, but Nahatlatch and Boulder fires have higher mean scores, suggesting generally more severe fires. B) Boxplot comparing modified CBI values grouped by fire severity. All box and whisker plots show median values (thick horizontal line), 25% and 75% quartiles (lower and upper box limits, respectively), and 5% and 95% percentiles (lower and upper whiskers, respectively) of the data categories being compared. Despite all scores being above 1.8, there is still a significant difference between high severity and moderate severity plots

Parks et al. (2014) established that the relationship between RBR and CBI was non-linear, thus it was assumed that the relationship between the modified CBI and RBR was also non-linear, which was supported when analysis found linear regression of modified CBI vs RBR failed assumptions of normality (W=0.71, p <0.001) and homoscedasticity. The Spearman's rank correlation between CBI and RBR (ρ = 0.75) showed a clear association within this dataset. To describe how field measurements relate to remote sensing measurements, the data then were fitted to an asymptotic non-linear least square regression model, $y = 4.03 * \frac{x}{251.7+x}$ (68 df, S=0.32; Figure 11). Since R^2 is not appropriate for non-linear models (Spiess and Neumeyer 2010), the standard error of regression (S) was used to measure the goodness of fit, where smaller values of S indicate better fit.

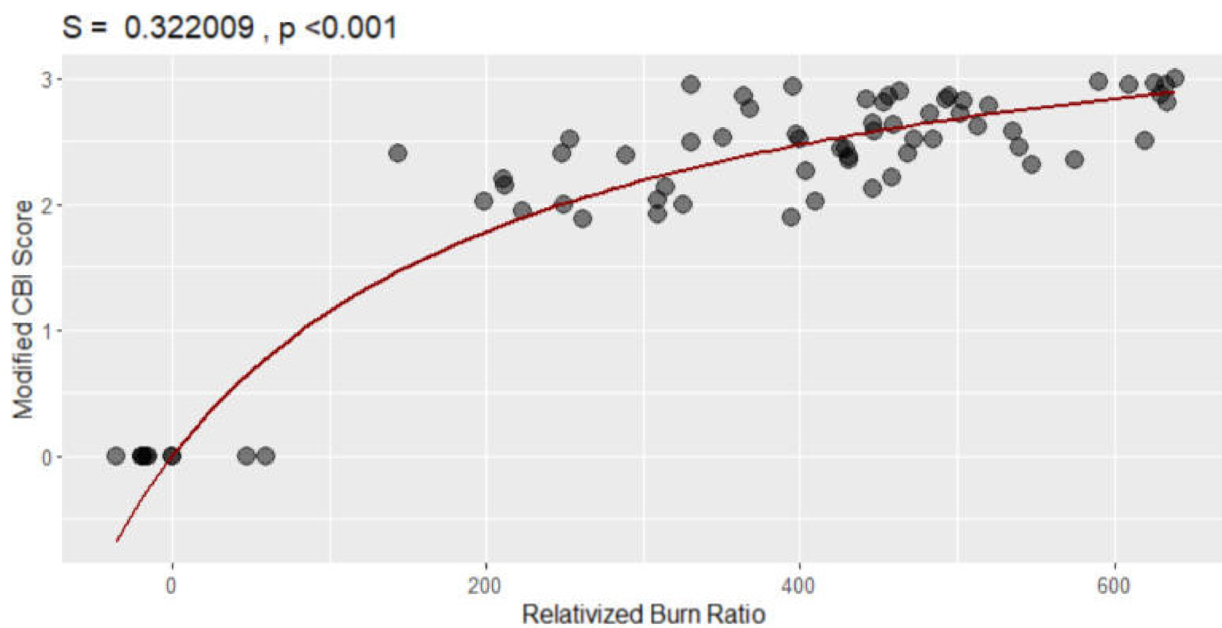

Figure 11 – Scatterplot of modified CBI vs RBR. As RBR values increase, the modified CBI scores approach 3.0 in an asymptotic way. No low-severity burn sites were sampled between 60 and 144 RBR, thus this regression line may not be appropriate for predictions in that range.

3.2. Carbon

Totals for aboveground carbon for each plot are shown in Appendix B. Average aboveground carbon, consisting of the average sum of forest floor, fine wood, coarse woody debris and stem carbon, varied from 157 to 348 Mg ha^{-1} across all fires (Table 7). The minimum total AGC for a single plot was 14 Mg ha^{-1} in a high severity Meager plot and the maximum total of 565 Mg ha^{-1} in an unburned Nahatlatch plot (Figure 12A & Appendix B).

Table 7 – Average aboveground carbon by site with standard deviation

Site	Burned Mean AGC (Mg ha^{-1})	SD	Reference Mean AGC (Mg ha^{-1})	SD
Boulder	156	50	380	125
Elaho	322	106	370	133
Grizzly	118	58	240	93
Meager	113	79	312	104
Nahatlatch	77	58	439	179
Across all sites	157	112	348	120

Quantile plots and Shapiro-Wilk tests for normality indicated that the AGC pools were normally distributed (W=0.974, p = 0.148), however residuals were unequal in a fitted vs residuals plot. A Kruskal-Wallis and Dunn's test with Bonferroni corrections tested differences in the total amount of C between fires, showing that Elaho contained significantly more C than Grizzly (p=0.0002), Meager (p=0.0025) and Nahatlatch (p=0.0001). The same procedure tested for AGC differences between burn severity classes across all sites revealed significant differences in average AGC between all combinations of plots: high severity and unburned plots (p<0.0001), high and moderate severity plots (p=0.0137) and moderate severity and unburned plots (p=0.0261) (Figure 12B).

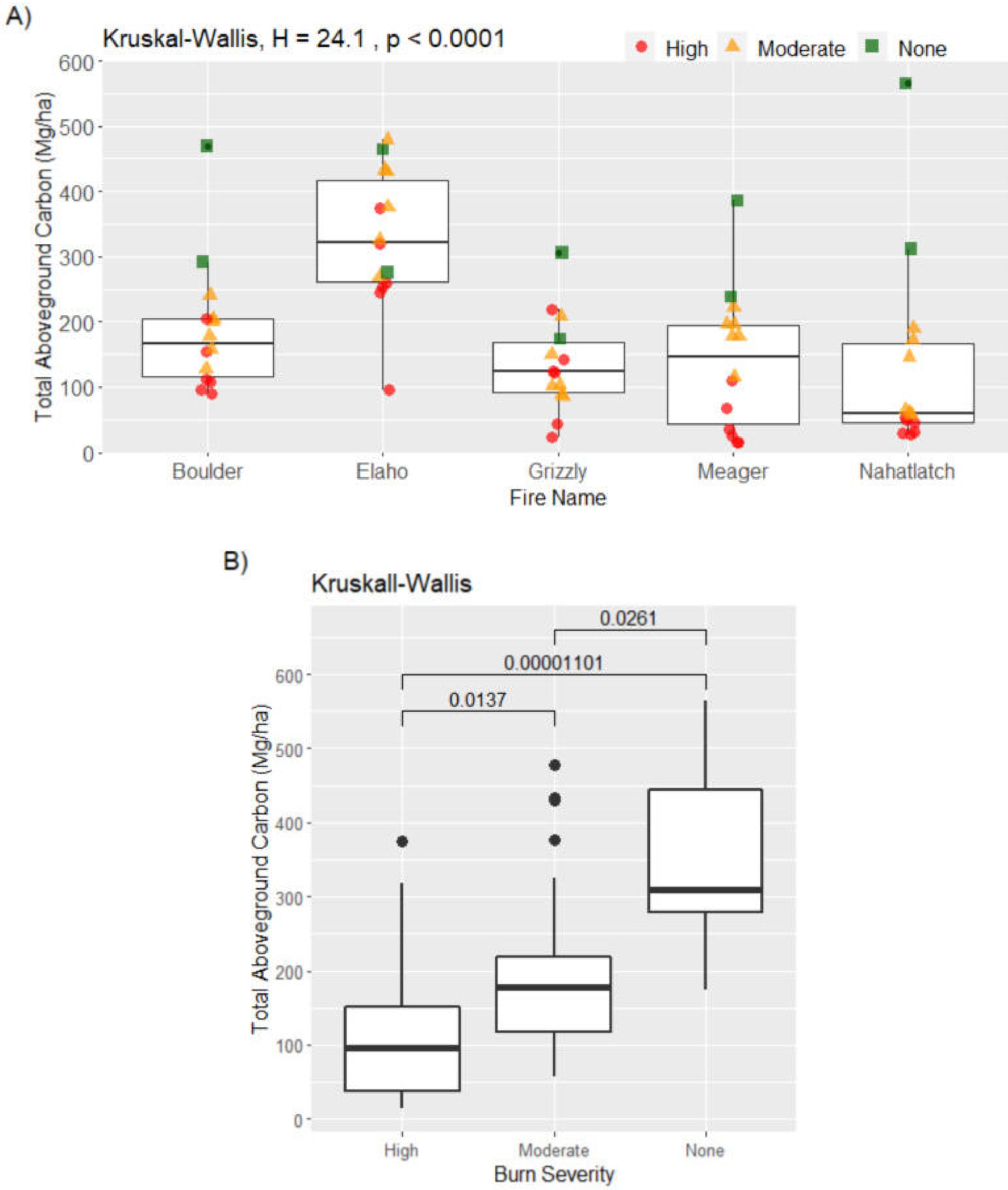

Figure 12 – Box and whisker charts of AGC by site and by burn severity. Elaho's mean AGC is significantly different than other sites, indicating the presence of larger diameter trees and deeper forest floor. B) Boxplot comparing AGC by burn severity class when grouping all sites. Significant differences (indicated by p values <0.05 for paired comparisons) were found between high/moderate and high/no burn severity

A spearman rank test between AGC and RBR showed a negative association ($\rho = -0.42$, $p = 0.0003$) between RBR and the amount of carbon (Figure 13). Linear modelling failed assumptions of normality and heteroskedasticity when trying to describe the relationship between RBR and AGC, which suggests RBR does not appear to be a good linear predictor of AGC for modelling purposes.

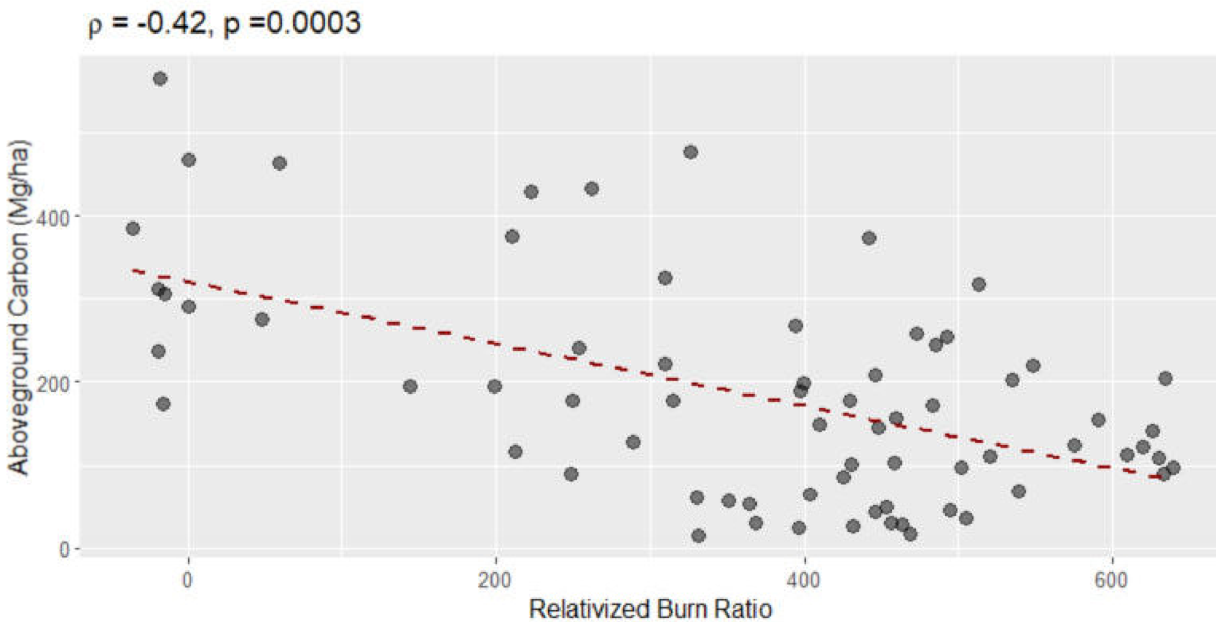

Figure 13 – Scatterplot of AGC vs RBR with dashed trend line. AGC is negatively associated with RBR, but AGC can remain quite high even at higher RBR values found for some severely burned sites.

A severity-themed scatterplot of AGC vs. RBR highlighted the upper and lower bounds of the natural range of variability in C found in unburned stands (Figure 14). Burned sites that fall within this band (21 of 60) therefore have similar amounts of AGC to unburned reference sites. Sites below the band (39 of 60) are all burned sites that are below the limits of natural variation for unburned old-growth forests in the study area. Of the plots found in the band, most (11 of 21) come from Elaho plots, which after inspection showed that 35% of all standing trees over 60 cm DBH and 82% of coarse woody debris over 60 cm DBH were found in Elaho plots.

In other words, Elaho plots contained the highest number of large trees in both burned and unburned plots.

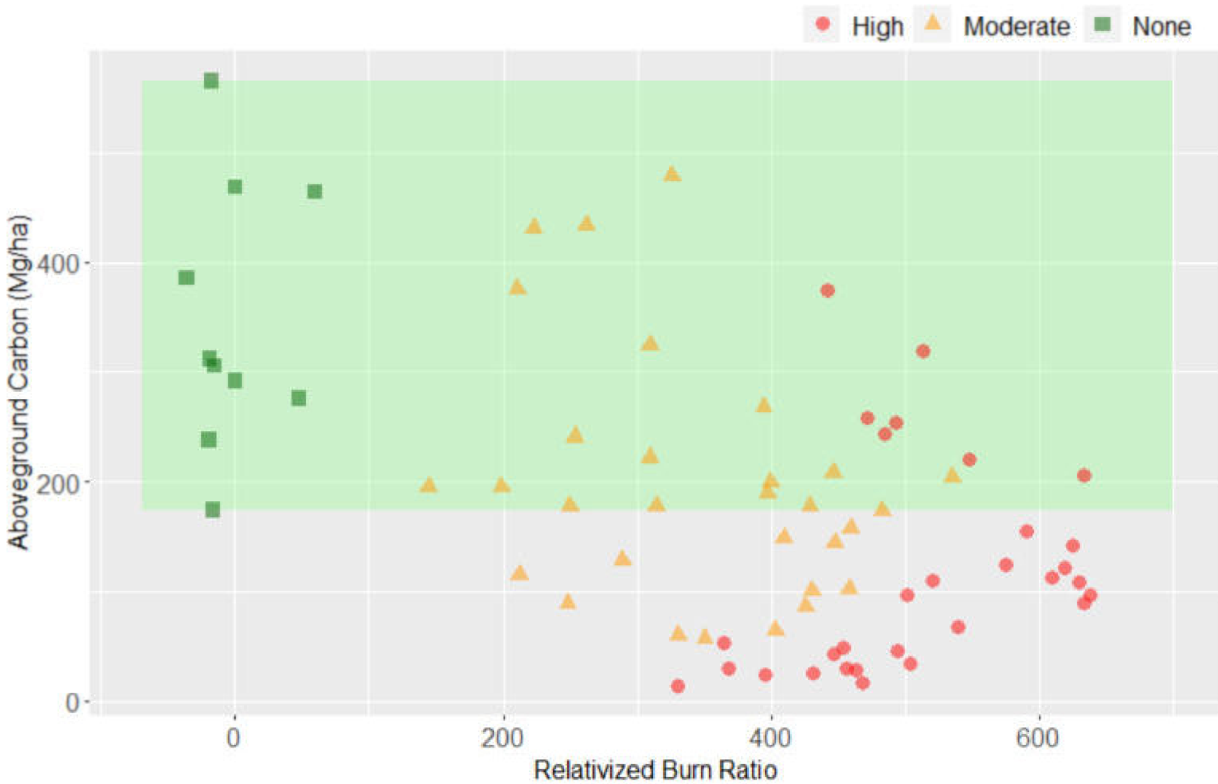

Figure 14 – Scatterplot of AGC vs RBR with natural range of variation (NRV) in unburned plots (green band). Of the 36 plots below the NRV, 14 are from are moderate burn severity plots. Interestingly, 7 high burn plots are within the unburned NRV.

A graph was constructed to show the relative difference in AGC between burned sites and their corresponding reference site (Figure 15). The dashed line is drawn at the lowest green point, which in this case is a plot that has 57% less AGC than its reference plots, but is still found within the green band in the preceding graph (Figure 14). Points on or near this line represent plots that have maintained AGC to be comparable to unburned reference plots. Points far below the line have the largest losses in AGC and would indicate the most severe fire damage to AGC values.

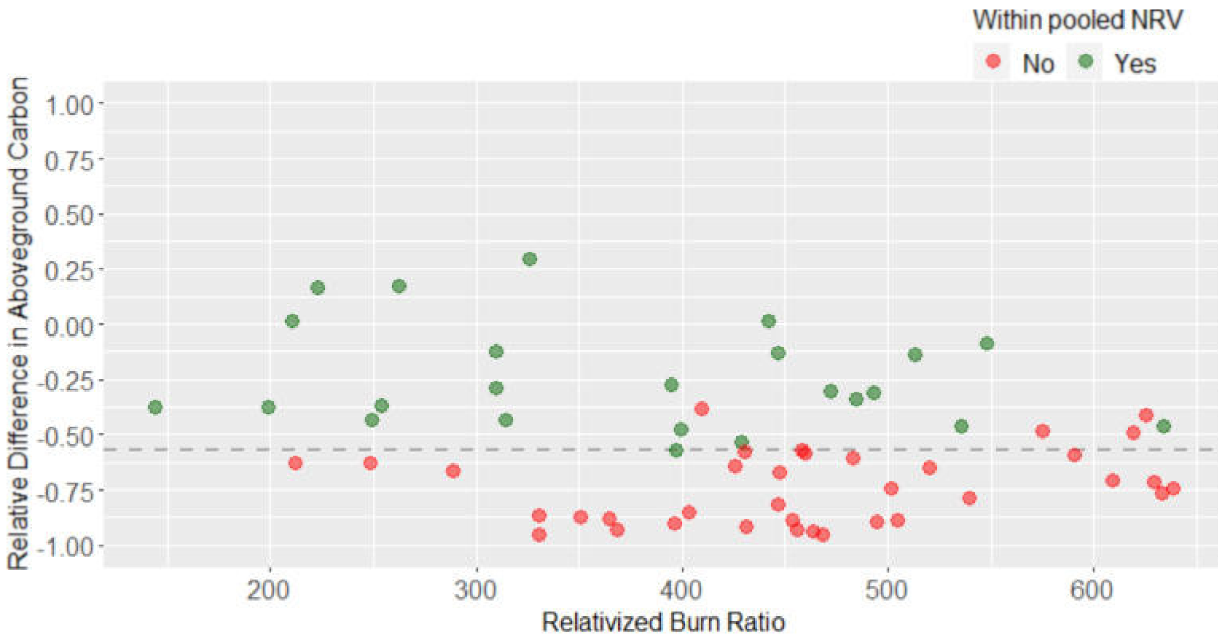

Figure 15 – Scatterplot of relative difference in AGC between burned plots and their corresponding reference site. The two red points at the top are Elaho plots that contained several large-diameter CWD and standing trees. Red points just above the line are sites that have up to ~50% less carbon than their local reference site, but across all sites may still maintain old-growth carbon value despite being outside of natural variation limits.

3.3. Old Growth Structure

The average structural index values for high, moderate and reference sites was 0.2, 0.3 and 0.7, respectively; all values for structural index are shown in Appendix A. The minimum structural index value was 0.05 at a Nahatlatch high burn severity site, while the maximum value was 0.89 at an Elaho reference site.

Quantile plots and Shapiro-Wilk test for normality indicated that the structural index scores were not normally distributed ($W=0.83$, $p<0.001$) and non-parametric methods were needed for further comparison. A Kruskal-Wallis test showed that structural index value differences between fires were not significant ($H=6.92$, $p=0.14$) (Figure 16A).

Significant differences in structural index values between burn severities were found using Kruskal-Wallis tests ($H=29.2$, $p<0.001$) and further pairwise comparisons using the Dunn's test with Bonferroni correction, which showed that significant differences existed between all three pairs (high/moderate p=0.0295, high/none <0.001, moderate/none p=0.001, Figure 16B).

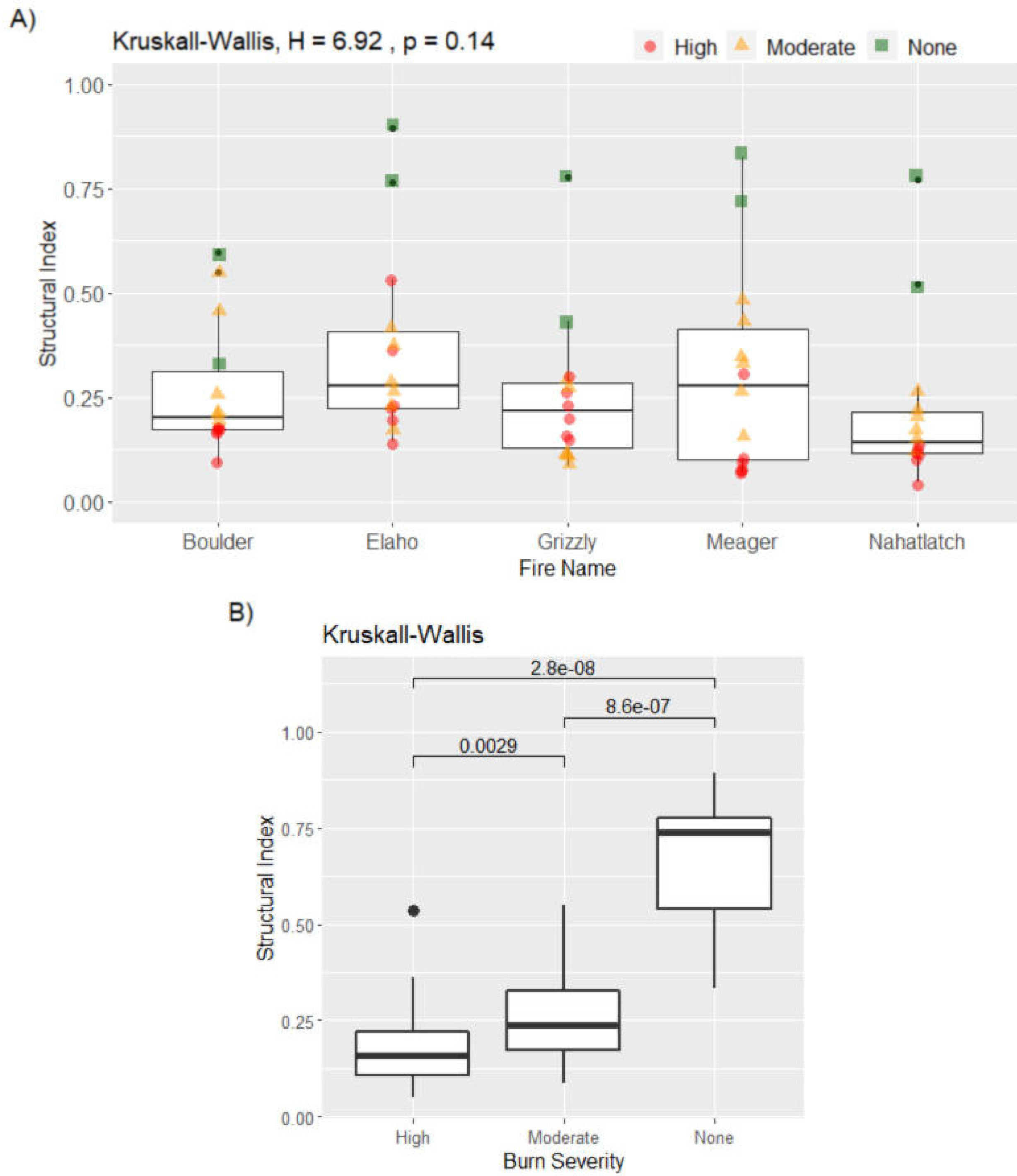

Figure 16 – Box and whiskers chart of structural index by site and by burn severity.Comparing structural values among fires, there appears to be no major differences in medians. However, Boulder has two burn sites particularly high in structural index score as

compared to other Boulder sites, but not when compared to Elaho or Meager. B) Significant differences exist between all burn severities (indicated by p values <0.05 for paired comparisons), but unburned reference sites were appreciably greater than both moderate and high burn sites for structural index.

Similarly, a spearman rank test between RBR and SI values returned ρ= -0.59, showing that as RBR values increased, structure score decreased. A polynomial regression model was chosen and, unlike the non-linear least squares method in the CBI vs RBR model, polynomial regression is still a type of linear regression, thus R^2 was chosen as an appropriate measure of goodness of fit (R^2=0.55, p<0.001) (Figure 17).

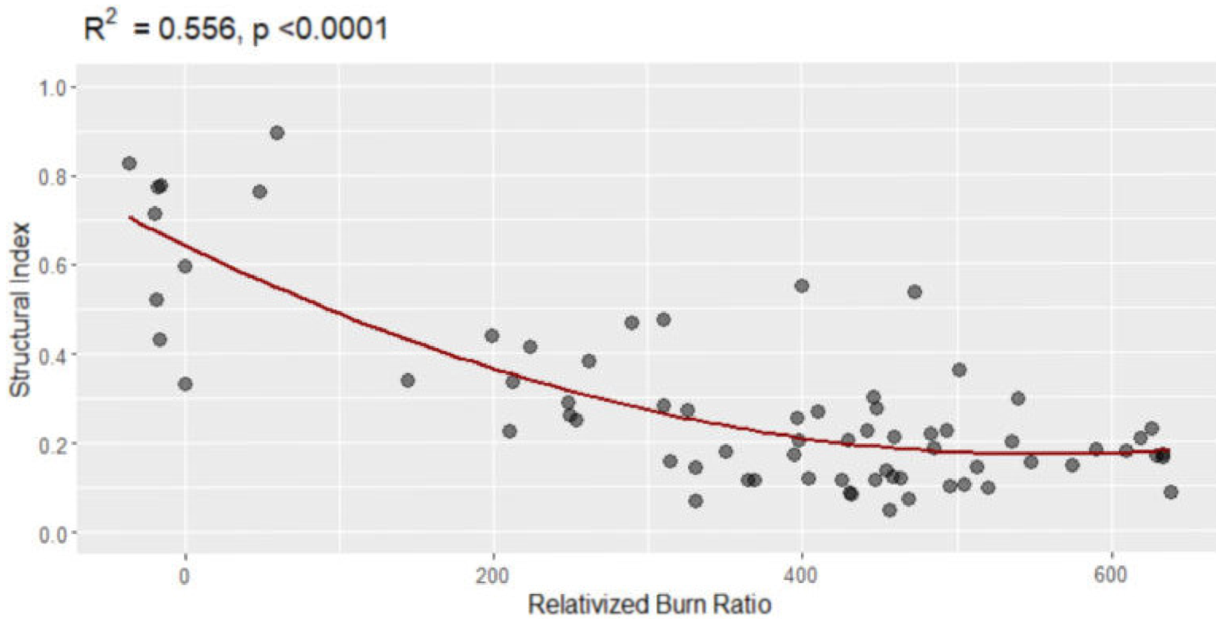

Figure 17 – Scatterplot of structural index vs RBR. This second-order polynomial regression line is illustrating the general trend that as RBR increases, structural index decreases until it starts to hit a floor where additional RBR values predict smaller losses of structure.

A subsequent scatterplot of the structural index vs RBR was created to highlight the natural range of variation (NRV) in the structural index (Figure 18). The green band is determined by the upper and lower scores of structural index from unburned reference sites.

Burn plots that fall within this band (10 of the 60 sampled) therefore have similar structural index scores to unburned reference sites. Plots below the band (50 of 60) are all burned sites that have lost enough structure that they are below the natural variation limits of unburned old growth.

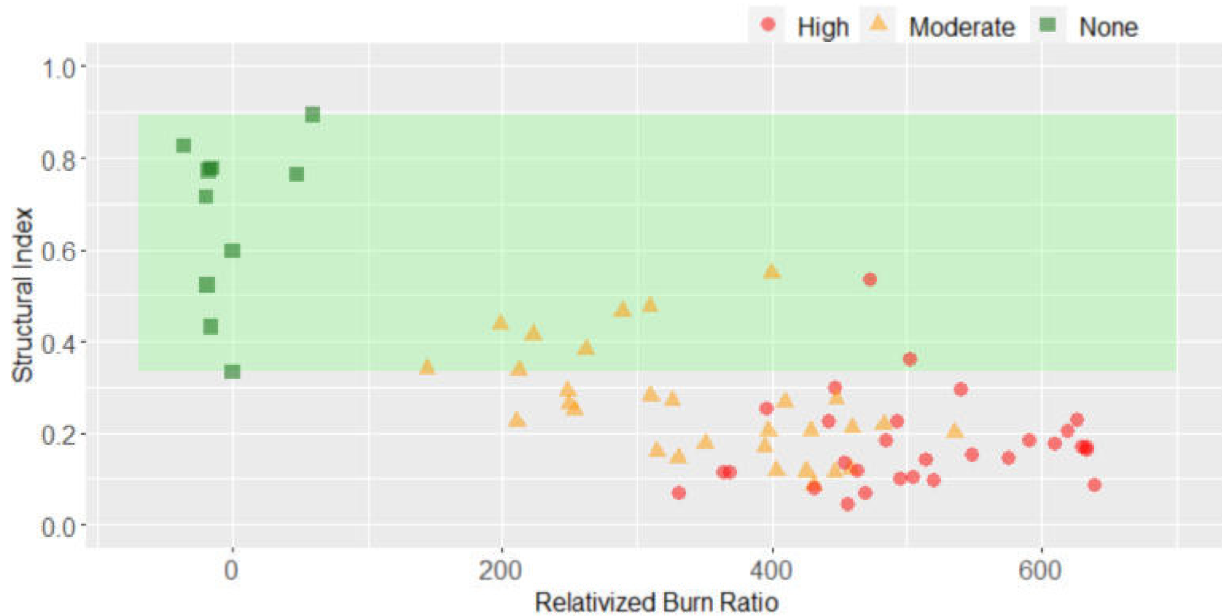

Figure 18 – Scatterplot of structural index vs RBR with NRV. The green band represents the natural range of variation in the unburned old-growth stands sampled, while each point's colour represents its original (mapped dNBR-based) burn severity class. Points within the band, despite their burn severity, represent plots that retain undisturbed levels of structural old growth values, while plots below the band represent plots that are interpreted as having lost some old growth structural value.

By averaging the structural index score for the two reference plots per fire an average unburned score was created. As an additional way to 'narrow the field' of plots that may have lost structural values, a relativized structural index score ($\frac{\text{burned}-\text{unburned}}{\text{unburned}}$) and graph (Figure 19) was created. This shows the relative change in structure score from a plot to its corresponding average reference site. Red points are plots that were below the green bar in Figure 18, green points are plots that were within it. The dashed line is drawn at the highest relative loss of points within the NRV (the lowest green point, -0.56). Points near this line therefore represent plots that are on the threshold of NRV. Points below the line (44 out of 60) have lost the most structure.

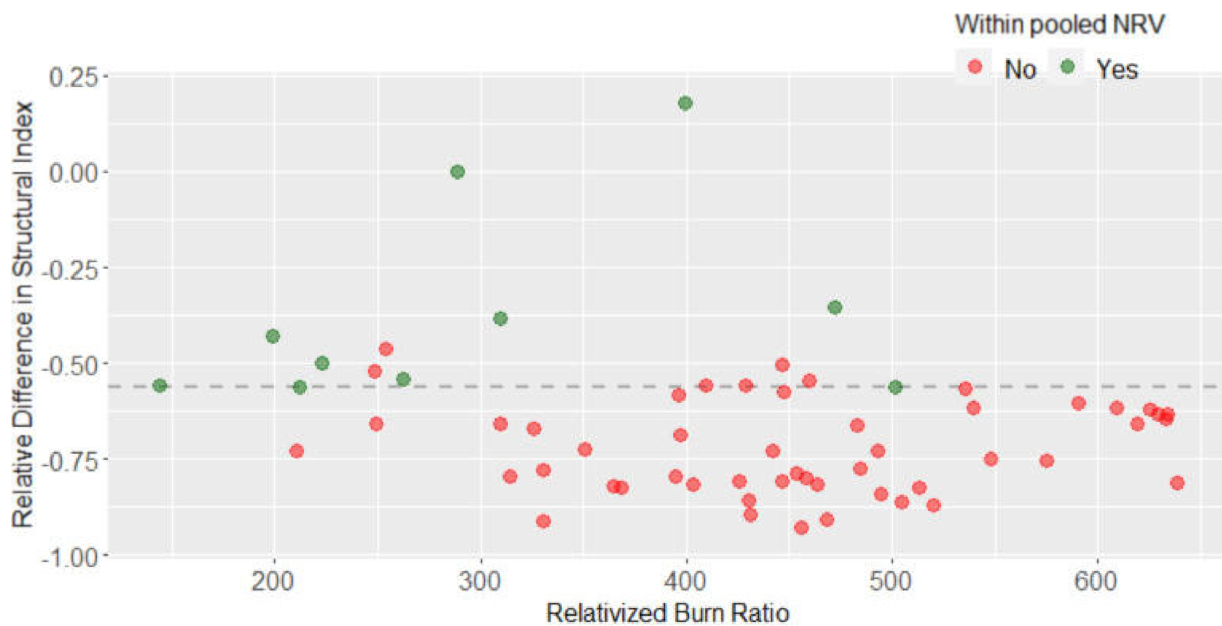

Figure 19 – Scatterplot of relative difference in structural index values vs RBR. Relative differences between plots and their corresponding average reference values. A value of -0.50 would mean a site has 50% less structural indicators than its reference plots. Values above 0 show that despite being burned, a site has relatively more structural value than its reference site due to the unique structure of that burned site (e.g., presence of large CWD).

3.4. Plant Community

In total, 114 species of plant were recorded across all plots. The number of species per plot varied from one unburned reference plot of 22 species, to two burned plots that contained only four plant species per plot (Table 8). On average, moderately burned plots contained the highest total number of species per plot (12.1 species per 400 m^2 plot). Mean Jaccard similarity values (a scale from 0 to 1) ranged from 0.0 similarity to 0.25 similarity between burned and unburned plots**Error! Reference source not found.** (Table 8).

Table 8 – Summary of understory plant diversity.

	High	Moderate	None	Pooled	Max	Min
Total Number of Plant Species	60	75	53	114	-	-
Total Unique Species	14	11	19	-	-	-
Mean Richness per Plot	9.0	12.1	11.8	10.7	22	4
Mean Jaccard Similarity Index*	0.10	0.14	0.47	0.17	0.25**	0.00

* Each Jaccard index value represents the similarity between a plot and its corresponding average reference site. Jaccard value for None is the mean similarity of among all reference plots.** the maximum Jaccard value was 0.70 which was between two reference plots, while 0.25 is the maximum similarity found between a burned plot and its reference plot.

Jaccard values were also grouped by site to inspect differences across the landscape (Figure 20). For example, the unburned set of plots, "None Meager" only had a Jaccard similarity index score of 0.17 when compared to "None Nahatlatch", showing that even within reference plots there was relatively few common species.

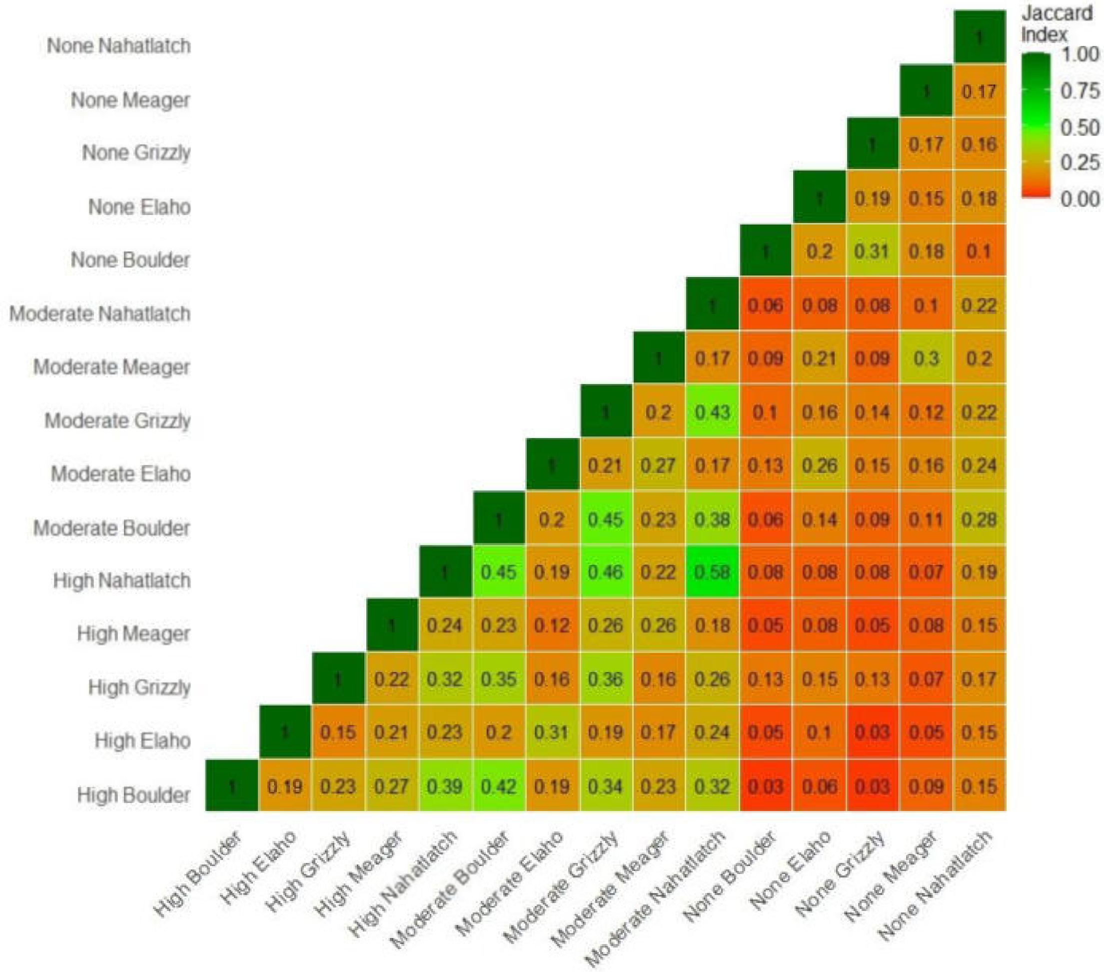

Figure 20 – Matrix of mean Jaccard index values by site. Values approaching 1 mean sites have similar understory species presence and are coloured shades of green. Pairs with low similarity in the understory have low values and are oranges and reds.

In terms of unique plants, those that were found only in a single category, reference plots contained the highest total number of species (n=19) that were not found in other plots. Plots mapped as moderately burned contained the lowest number of unique species (Table 9). Some plants could only be identified to genus, so if a genus was found in more than one category it was removed from the unique list.

Table 9 – List of unique understory species by burn severity

High	Moderate	None
Alnus sitchensis	*Berteroa incana*	*Acer glabrum*
Ceanothus velutinus	*Betula papyrifera*	*Alectoria sarmentosa*
Ceratodon purpureus	*Calamagrostis* sp.	*Balsamorhiza sagittata*
Hx (Hybrid *Tsuga*)	*Cornus stolonifera*	Boraginaceae family
Orthilia secunda	*Heuchera micanthra*	*Brachythecium frigidum*
Penstemon sp.	*Mahonia nervosa*	*Cladina arbuscula*
Polygonum sp.	*Oplopanax horridus*	*Cladonia bellidiflora*
Rhytidiadelphus triquetrus	*Pellia neesiana*	*Gaultheria shallon*
Ribes lacustre	*Phleum pratense*	*Hylocomium splendens*
Sambucus racemosa	*Rosa* sp.	*Hypogymnia inactiva*
Maianthemum racemosa	*Rubus arcticus*	*Kindbergia oregana*
Trillium sp.		*Listera caurina*
Viola sp.		*Lobaria* sp.
Asteraceae family*		*Mnium* sp.
		Nephroma resupinatum
		Pw (*Pinus monticola*)
		Rhytidiadelphus loreus
		Rubus pubescens
		Taxus brevifolia

** No reference photo was taken, identification is best field estimate.*

Quantile plots and a Shapiro-Wilk test for normality indicated that Jaccard similarity values were not normally distributed ($W = 0.74$, $p < 0.001$). A Kruskal-Wallis test (overall $H = 13.8$, $p = 0.0081$) and post-hoc Dunn's test with Bonferroni corrections tested differences in mean Jaccard values between fires, which showed significant differences between Boulder and Meager ($p = 0.0009$) and between Boulder and Nahatlatch ($p = 0.0027$) (Figure 21A). The same procedure tested for differences in burn severity across all plots ($H = 29.4$, $p < 0.0001$) revealed significant differences in average Jaccard similarity between high severity and unburned ($p < 0.0001$) and moderate severity and unburned ($p = 0.0002$) plots, while no differences were detected between high and moderate severity plots ($p = 0.1105$) (Figure 21B).

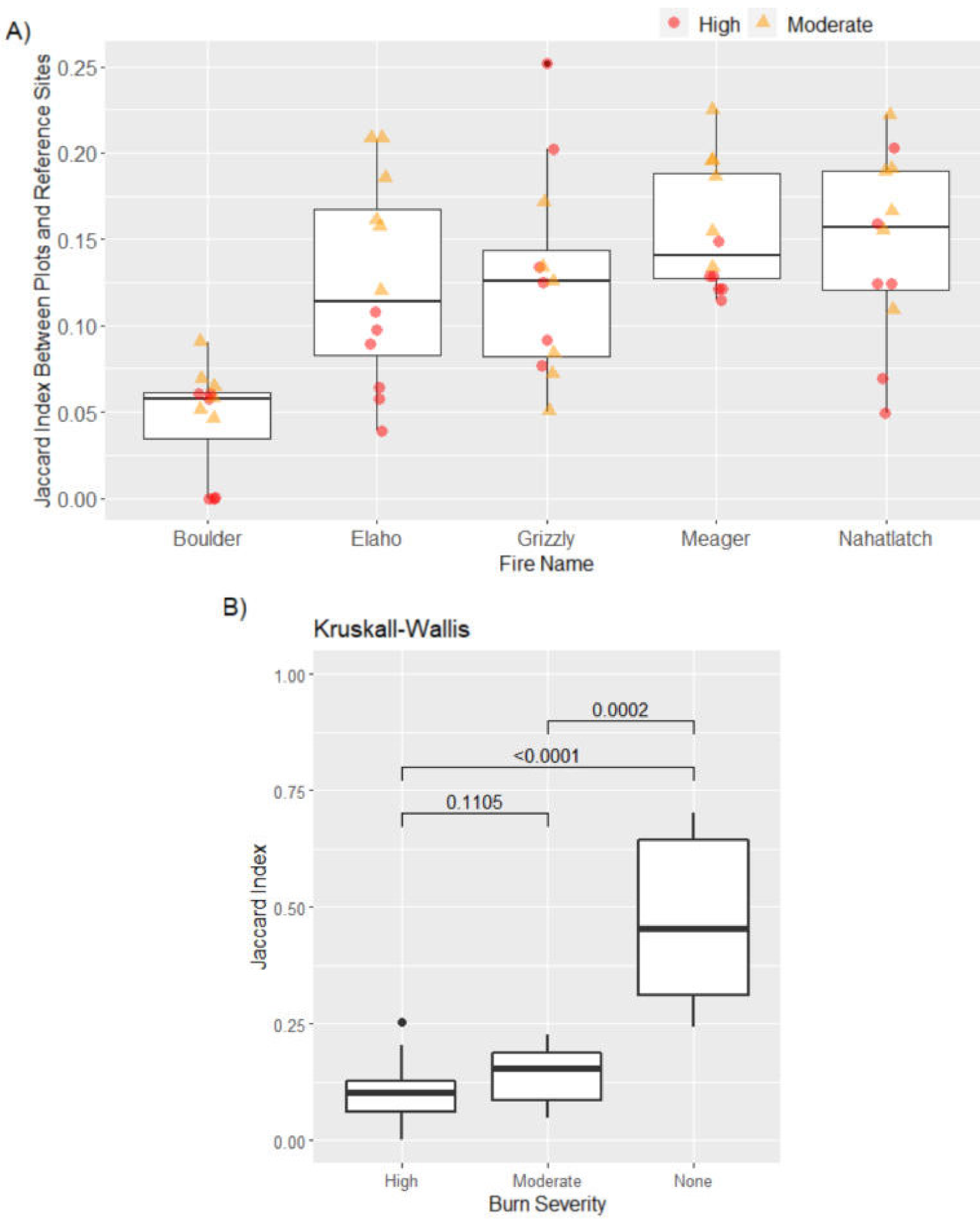

Figure 21 – Jaccard similarity index values by site and burn severity. A) Boulder Jaccard values were significantly lower than Meager and Nahatlatch plots B) Significant difference exist between burned and unburned plots, but not between burn severity classes. Note that Jaccard similarity values are a comparison of floristic similarity in a burn severity class to the average of all reference plots, but the unburned reference plots were also included to show the range of floristic similarity among reference plots.

The spearman rank test found a negative association between Jaccard similarity values and RBR ($\rho = -0.63$, $p < 0.0001$), in general showing that as RBR increased, the species similarity between burned and unburned plots decreased (Figure 22).

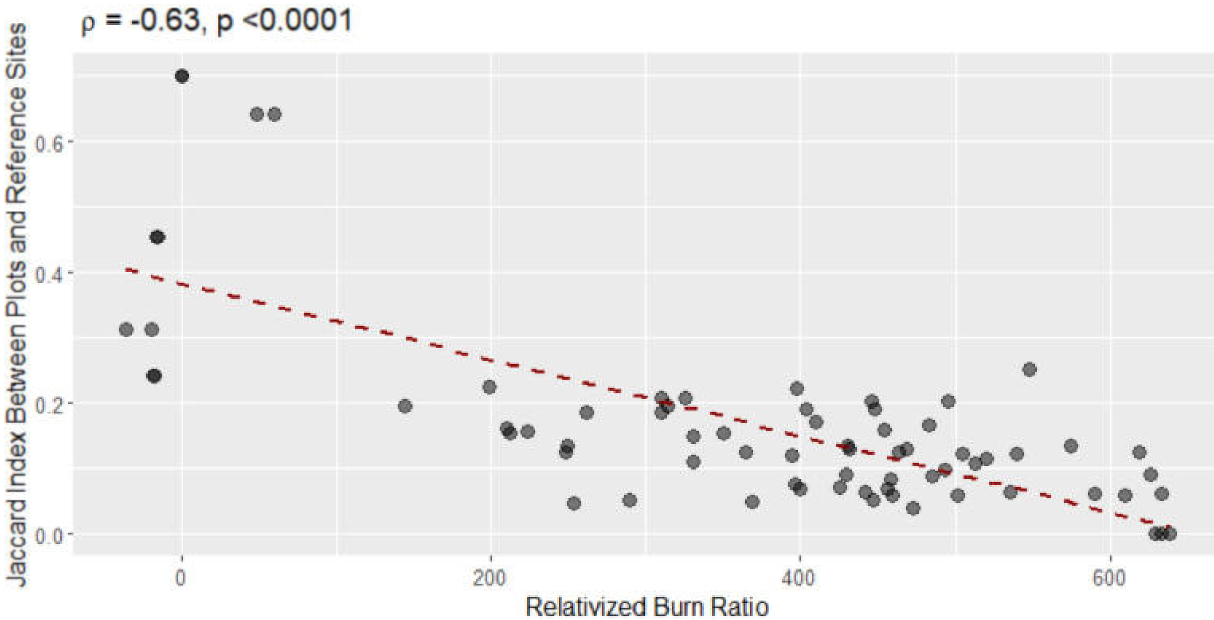

Figure 22 – Plot of Jaccard Similarity Index values vs RBR with dashed trend line.

A plot of Jaccard vs. RBR found that only a single high severity plot fell within the NRV for unburned reference plots; the rest of the burned plots had average Jaccard values below 0.25 (Figure 23), while the mean similarity was 0.12. Three plots had values of 0.0, meaning they had no floristic similarity with reference plots.

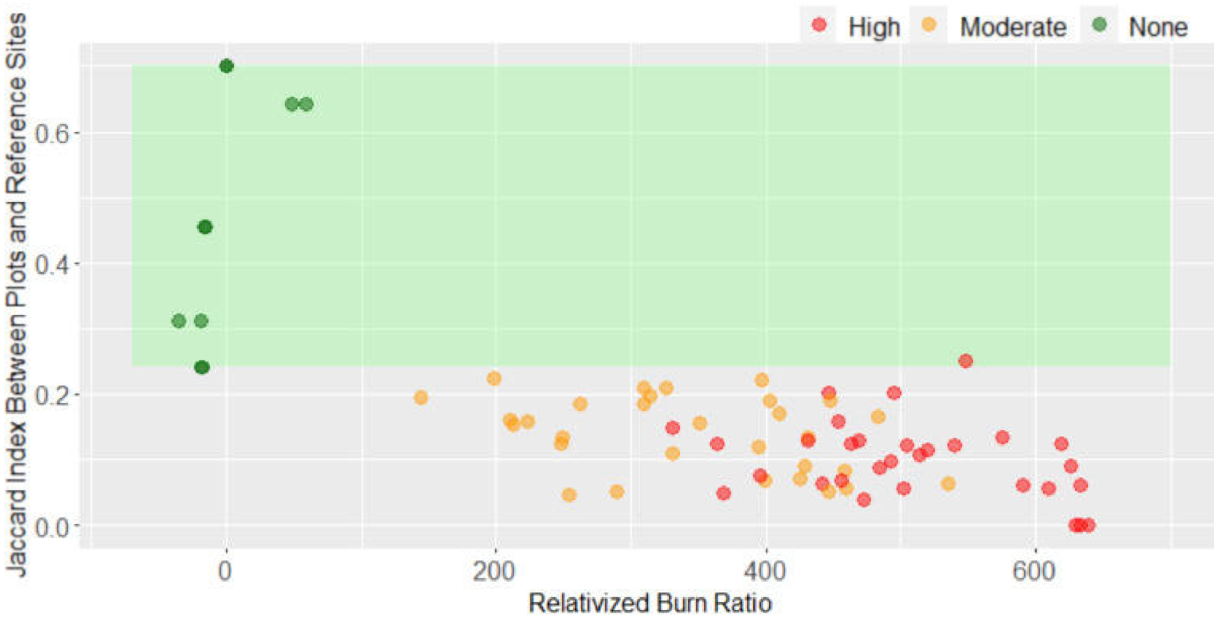

Figure 23 – Scatterplot of Jaccard similarity index values vs RBR. The top of the natural range of variation (NRV) envelope for unburned reference plots shows the greatest similarity between unburned plots, represented by a reference plot that has a similarity of 0.70 with its neighboring plot. The bottom of the green NRV envelope shows the lowest similarity between two reference plots (0.24), that were in this case geographically quite distant from each other. Overall, burned plots have less than 0.24 similarity in understory plant floristics with reference plots.

Comparing burned plots to their corresponding reference plots, rather than all reference plots, showed that 49 burned plots had relatively lower Jaccard similarity values than the single site that had been within the unburned NRV (Figure 24). In other words, when species lists were compared only to corresponding reference plots, 11 plots were more similar to their reference site than a site that fell within the unburned NRV of all plots.

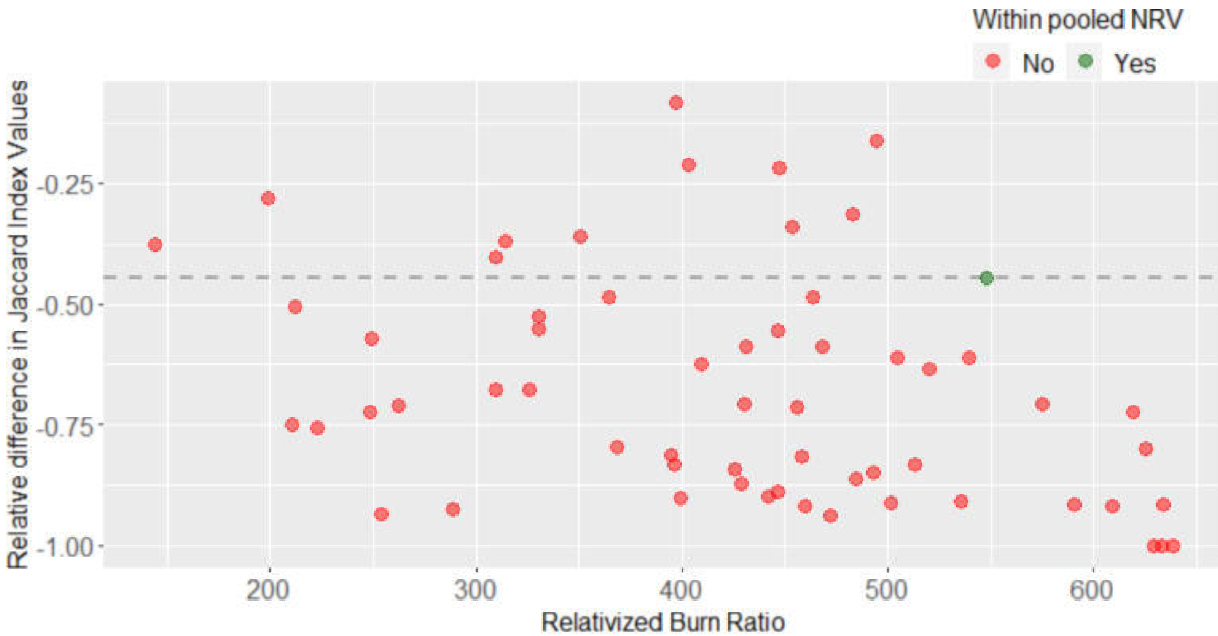

Figure 24 – Scatterplot of relative difference in Jaccard similarity values vs RBR. Points below the line are burned plots that are between 44% and 100% different in understory species presence when compared to old growth reference plots.

Chapter Four – Discussion

4.1. Burn Severity Assessment

The modified CBI scores had no values below 1.9, so all plots would be considered at least moderately burned by the thresholds suggested by Key and Benson (2006). Ground observations support this: generally there were signs of complete surface burning based on differences between reference plots and burned plots in forest floor depths, percent cover of exposed rock and soil, the presence of colonizing herb cover (e.g., fireweed), very high tree mortality and extensive bark char. Statistical analysis shows there was a significant difference in modified CBI values between moderate and high burn severity plots (Figure 10). Therefore, the modified CBI values were satisfactory in showing burn severity differences between high-severity, moderate-severity, and unburned reference plots and could be used to compare to remotely sensed measures.

The modified CBI had a high correlation with RBR (ρ=0.75), but because of the failure to meet assumptions of parametric statistics, no R^2 value could be calculated; however the standard error of regression was quite low (Figure 11), suggesting the model fit the data well. The resulting asymptotic trend is similar to that of Parks et. al. (2018), who compared 18 fires and 1681 CBI plots using non-linear regression methods. On average they found that the RBR method classified pixels correctly (as defined by CBI) around 74% of the time. The high correlation in the present research and the high accuracy of Parks et al. (2018) validates the use of RBR as a method of remotely determining burn severity. An important caveat is that detection of low severity burns in the sub-canopy using dNBR or RBR methods can be confounded by canopy obstruction and spectral mixing within a pixel (Latifi 2012, Kolden et al. 2012), therefore remote sensing is better at detecting change in moderate and high severity burns rather than low.

But as all burned plots described here were of moderate burn severity or higher, with low canopy cover (12% on average), the use here is acceptable.

4.2. Carbon

Medium resolution remotely sensed measures of burn severity were found not to be a good predictor of residual AGC in linear modelling. This wasn't unexpected, as the pixel resolution of the imagery is often greater than the size of an individual tree and is essentially a two-dimensional image from above of a three-dimensional object (Latifi 2012). Other remote sensing technologies such as Synthetic Aperture Radar or spaceborne LiDAR are much more effective at estimating aboveground biomass (Latifi 2012, Mitchell et al. 2017). However, RBR provides a good estimation of ecological change at a landscape level (Figure 3) and is a way to highlight vegetation and burn severity patterns, which in this case shows that RBR is correlated with AGC (Figure 13).

When comparing AGC estimates between remotely sensed burn severities (none, moderate, high), 24 burned plots were found to contain as much total AGC as at least one reference plot, seven of these from high burn severity plots (Figure 14). Thus, not all highly burned plots have necessarily lost their old growth carbon storage pool. Along the same line, fourteen plots were moderately burned, but contained less total carbon than all the reference plots, so moderately burned plots do not always retain their old growth carbon pools either. For decision makers trying to find areas for restoration or preservation of carbon storage, targeting high or moderate burn severity areas defined by the current remote sensing thresholds is not an appropriate strategy.

There was only a moderate difference in AGC between moderate severity and high severity plots, which was surprising because one would expect less carbon in highly burned ecosystems. However, this analysis grouped the measured carbon pools to produce a sum total where changes in individual carbon pools were not measured; there could be great differences in the form of AGC. In an unburned ecosystem, carbon is spread amongst standing live and dead trees, CWD, forest floor and mineral soil. After fire, carbon pools shift, from live to dead standing or from standing dead to CWD and from any pool to atmospheric carbon through combustion (Eskelson et al. 2016, Ping et al. 2022). An example by Lutz et al. (2020) found that delayed post-fire mortality of large standing trees contributed to the pool of large CWD as CWD was consumed by fire, so that there was little to no difference in CWD pre-fire and six years post-fire. It is possible that moderately burned plots had abnormally large amounts of pre-fire surface fuels that retained large amounts of carbon after fire, so that no apparent difference in AGC was detectable. Others have pointed out that in studies without pre-fire data from the same plots (as this one is), it is impossible to know if unique site conditions existed which would have impacted fire behaviour and post-fire residual carbon pools (Peterson et al. 2019). This could be the case in the Grizzly plots, as satellite imagery (Figure 8) showed that prior to fire there was a fair amount of exposed rock, so it is possible that no difference in AGC in moderately burned plots was detected because there was less vegetated area to burn in the first place, the rock acting as a fire buffer of sorts.

An important finding is that 24 of the burned plots contained AGC within the unburned NRV of this study (Figure 14). In these, the presence of dead wood (particularly large dead wood) has important implications for carbon storage. Large woody debris takes longer to decay than smaller debris (Yin 1999) and at the same time wood that has burned but not fully combusted is coated in charcoal, which makes the wood more resistant to decay (Maestrini et al. 2017). In one experiment, large-diameter dead wood accounted for 45% of AGC, 62 years after fire (Schaedel et al. 2017), which means that residual dead wood can continue to store carbon, slowly releasing it to soil and atmospheric pools over decades, potentially centuries (Maestrini et al. 2017, Lutz et al. 2020), while the surrounding ecosystem regenerates and starts to ramp up carbon sequestration. This means that disallowing the removal of large-diameter standing live and dead standing trees in post-fire areas is an important step that land managers can take to meeting landscape carbon storage objectives (Lindenmayer et al. 2008, Mildrexler et al. 2020).

As a method of detecting old growth carbon values, RBR analyzed in this way is not a useful tool. No obvious threshold values discerned a meaningful value.

4.3. Old Growth Structure

While it was established that RBR is not a good linear predictor of AGC, carbon is still related to forest structure (Figure 25) and RBR can linearly explain a certain amount of variability in structural index values (Figure 17). This makes sense, as fire has in varying amounts removed the tree canopy (directly or from post-fire mortality), exposing subcanopy conditions that were previously unseen in satellite imagery. Combining optical remotely sensed imagery with ground plots does improve models for understanding broad landcover changes like this, but ultimately cannot predict changes in other structural elements that require finer resolution measurements such as DBH, or that require oblique angles such as tree height determinations (Latifi 2012).

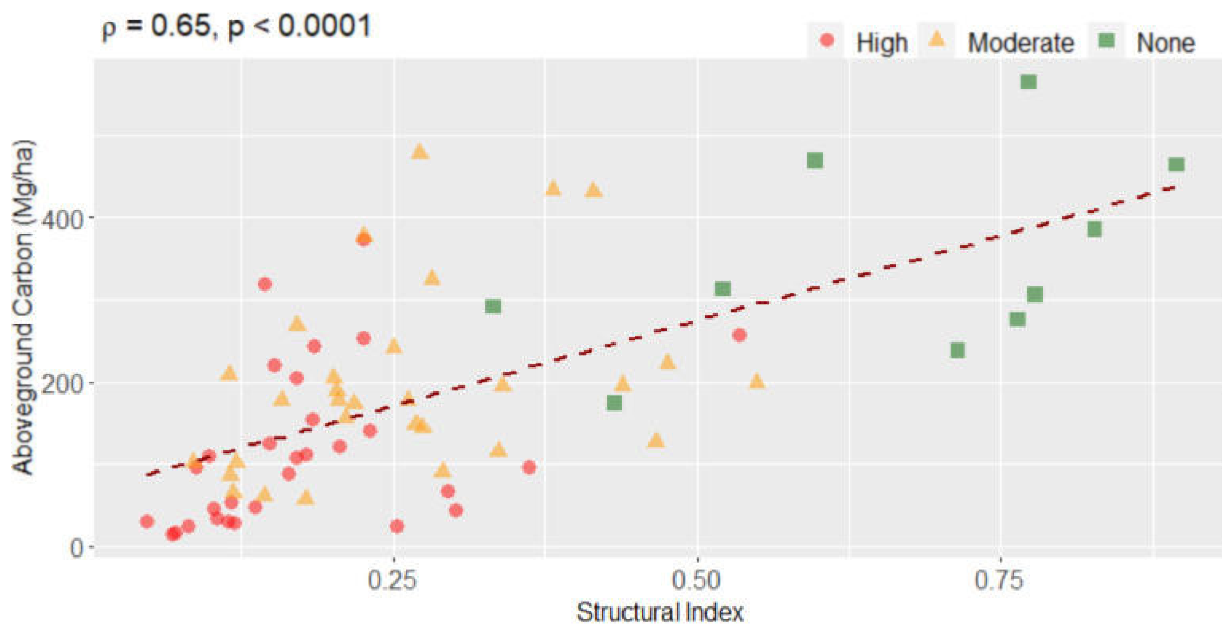

Figure 25 – Scatterplot of AGC vs structural index. Higher structural index values are significantly correlated with higher AGC as shown by the dashed trend line. This graph also shows that unburned reference plots had the highest structural values, but some burned plots had higher structural index values than some unburned plots.

There was a subtle, but significant difference in structural index values between the medians of high and moderately burned plots (Figure 16B). However, since the thresholds applied to burn severity mapping are fairly subjective (Key and Benson 2006, Lentile et al. 2007), it may be more beneficial to focus on the clear structural differences between burned plots as a whole compared to reference plots. Of the 60 burn plots, ten retained structural index values of equal or greater value as reference plots (Figure 18). The residual structure in these plots is important as potential habitat availability for fire survivors or colonists, which ultimately is expected to support increased forest species diversity (Swanson et al. 2011, Gerzon et al. 2011). It is also possible that differences in plot structure of these ten plots may have existed due to the unique development history and pre-burn structure of those plots (DeLong and Kessler 2000). Historical incidences of windthrow, erosion and disease, could have produced a particularly high level of structure of uneven-aged trees and CWD that made the stand more resilient to changes induced by fire.

A cursory examination of the underlying data table (Appendix A) shows that nine of the ten structurally complex plots have at least one live tree, whereas this is the case in only seven of the remaining 50 burned plots. Four of the ten plots contain at least one live tree over 40 cm DBH, while only two of the remaining 50 do as well. Survivor trees, particularly large ones, are important biological legacies that limit post-fire soil erosion and nutrient loss, act as seed sources for vegetation regeneration (Seidl et al. 2014, Meddens et al. 2016, Meigs and Krawchuk 2018), and can even bolster seedling growth (Simard et al. 1997). The proximity and abundance of live trees to wildfire-disturbed areas can accelerate canopy closure in forests (Larson and Franklin 2005), which would facilitate faster carbon stock recovery (Seidl et al. 2014). Land managers could therefore consider protecting areas around such living residual structure, as they may still contribute old growth structural values and then target restoration efforts to sites with the lowest structural index scores.

Consider the scatterplot showing the NRV (Figure 18) and the 15 burned plots to the left of approximately 330 on the RBR axis. Of these, 9 are in the NRV and 6 are below, which means values lower than 330 correctly identified 60% of the plots as having old growth structural values. In this case, this is the same for the relative graph (Figure 19), which also correctly identified 60% of plots as having old growth structural values.

4.4. Plant Community

Even though most plots had similar levels of diversity, it was found that as burn severity increased, floristic similarity between burned plots and their corresponding reference sites decreased (Figure 22). Boulder plots had significantly lower Jaccard similarity values with unburned sites than Meager and Nahatlatch (Figure 21A), showing that Boulder has less species commonality with its reference sites, which may indicate that the Boulder fire is in a much earlier state of succession than other sites, or the understory community is dominated by a particular colonizing species and thus furthest away from capturing old-growth plant diversity. Overall, burned sites share less than 25% similarity in floristics to unburned reference sites (Figure 23). These species could be surviving individuals, sprouted from surviving reproductive structures (seeds, rhizomes), or re-colonizers seeded from nearby unburned patches (Swanson et al. 2011).

If early successional trajectories can influence late successional development as suggested by biological legacy theory (Franklin et al. 2000) or the initial floristic composition hypothesis (Finegan 1984), then the plots with higher Jaccard values would sooner achieve old growth community composition than plots with lower scores. It can be argued that such theoretical models are too simplistic and don't account for stochastic factors, repeat disturbances or climate change (Gill et al. 2017). Secondary succession is a complex topic, with many potential mechanisms (Pulsford et al. 2016) and much is unknown (Chang and Turner 2019). However, the presence of at least some old-growth species three and four years post-fire suggests that these forests have the potential to recover old-growth biodiversity.

Considering Figures 23 and 24 and the plots left of 330 on the RBR axis, it is apparent that the absolute measure (Figure 23) fails to differentiate between any plots for floristic similarity. Figure 24 however does find 4 plots above the line, indicating that a threshold of 330 RBR correctly identified 27% of plots as having old growth floristic similarity. This suggests that while 330 may be a useful threshold for structure it is not a useful metric for Jaccard similarity indices.

Chapter Five – Conclusions and Applications

5.1 Scientific Conclusions

The purpose of this research was to compare three measures of old-growth forest values (aboveground carbon, structural complexity and understory plant diversity) between burned and unburned reference plots in order to understand what old-growth values may remain post-fire in the CWHms1 BEC zone. Reference sites were used to demonstrate the range of natural variation with which to compare the burned variables, with the understanding that there is no comparison to historical disturbance regime, this is just a 'snapshot' of natural variation found in old forests in this region.

The use of medium-resolution remotely sensed measures of forest wildfire burn severity is well established and valid for general broad analysis (Key and Benson 2006, Miller and Thode 2007, Parks et al. 2018). In this instance, relativized burn severity maps (RBR) together with 60 CBI plots were used to compare values across an entire landscape. The strength of this method is that such maps are easily made for land managers, but users need to be aware that this method is not appropriate for detecting change beneath a canopy or for measuring variables that require fine resolution (e.g., less than 30m pixel resolution) such as tree height, DBH or abundance by plant species. With this in mind, RBR as a remotely sensed indicator of burn severity, was significantly and negatively correlated with measures of AGC and structural index values.

Common remotely sensed thresholds of high and moderate burn severity were found to not be completely reliable indicators of residual old-growth values, as examples of high and moderate burn severity plots were found within unburned NRV limits as well as below. In this research, the lowest unburned reference value of carbon, structure or plant diversity was used as a new threshold to separate plots of low value from plots that may retain old-growth values.

Despite moderate- and high-severity fire disturbance, it was shown that 24 of 60 plots contained as much AGC as at least one unburned reference site. The implication is that because of charring and residual large CWD, these plots still maintain adequate carbon storage and will continue to do so for some time, barring future major disturbance such as fire or salvage logging.

For structural values, 10 plots were found within the NRV of unburned sites; these plots are particularly important for their role as habitat for surviving biota due to the presence of large-diameter snags, CWD or live trees. These biological legacies are also important in that they can limit post-fire erosion and nutrient loss and can promote forest regeneration, which in turn influences the rate of carbon sequestration, carbon storage and understory diversity in future.

Plant diversity was negatively correlated with RBR, but burned sites generally maintained a similar amount of understory plant diversity at any RBR level. Burned sites within the unburned NRV are not necessarily fulfilling old-growth diversity values, but this may indicate that enough resources exist that, through normal secondary succession over time, will lead to old-growth specific biodiversity. However, despite similar numbers of species, on average high burn severity plots had 10% and moderate burn severity plots had 14% mean floristic commonality with unburned plots (Table 8). If biological legacy theory is correct, then the presence of these common species should facilitate ecological succession towards old-growth status, but it is unclear whether plant diversity after disturbance influences ecosystem regeneration trajectories (Gill et al. 2017). The mechanisms for succession are not fully understood (Pulsford et al. 2016, Chang and Turner 2019), and more research into post-fire disturbance and recovery specific to coastal temperate rainforests is needed.

5.2 Applications

British Columbia uses data on the natural variation in the frequency, size and severity of stand-replacing natural disturbances as a guide to set landscape objectives for conservation of old forest values and function (BC Ministry of Forests and BC Ministry of Environment 1995, Kumi et al. 1999). The fundamental principle of this approach is that the overall native ecological function of an area will be maintained through conservation of a variety of ecosystems in a variety of successional stages that would exist prior to logging (BC Ministry of Forests and BC Ministry of Environment 1995, McElhinny et al. 2005). While old growth management areas (OGMAs) may be spatially delineated within a legally-binding forest stewardship plan (FSP), they may also exist as draft spatial units (not yet legalized within an old growth order), or as aspatial summaries of amounts of area to be managed for late successional values (Gorley 2012).

The nature of old-growth management is currently generating passionate discourse in BC (Gorley and Merkel 2020, Price et al. 2021) and new management strategies (e.g., Great Bear Rainforest Order) have been implemented in some areas to balance the many values of old-growth forests (B.C. Ministry of Forests, Lands and Natural Resource Operations and Province of British Columbia, Victoria, B.C. 2016). There are remaining questions about OGMA patch size and issues with edge effects (Bezzola and Coxson 2020), about representation and protection of high productivity old growth (Price et al. 2021), and managing old-growth forests in the uncertainty of fire regimes (Kopra and Feller 2007). This is especially true as the frequency and intensity of wildfires have been and are expected to continue to increase with climate change (Haughian et al. 2012) with ramifications to forest resilience (Daniels et al. 2017).

Provincial governmental guidance documents may define criteria for replacing OGMAs with the understanding that replacement OGMAs outside protected areas should be of equal or greater value (Integrated Land Management Bureau 2007, Nicholls et al. 2018). However, with increased wildfire activity and a fragmented landscape of residual old-growth patches there may be a point where a management area is so highly disturbed that there are no undisturbed old-growth areas left that meet the patch, connectivity and representation standards of usual OGMAs. In those cases, decision makers may need to analyze remnant forests to target disturbed areas for protection as they meet at least some old-growth value, while other burned old forest may be targeted for salvage logging or ecological restoration if they are so highly disturbed they functionally no longer provide old-growth value.

In this analysis, data were collected from representative samples of unburned old-growth forest plots over landscapes. These plots varied somewhat in elevation, aspect and slope so as to capture the natural range of variation of old-growth for the values of interest within a biogeoclimatic zone. The data were then compared to plots within moderate and high severity burned areas of the same biogeoclimatic variant to understand where a particular burned plot stood in relation to unburned plots. A broader sampling of unburned CWHms1 locations would likely have revealed a broader NRV for purposes of burned plot comparisons.

Comparing the total AGC to reference sites across the whole landscape (Figure 14) shows that 24 of the burn plots have similar levels of carbon as unburned plots and can be eliminated for consideration for restoration or replacement (Table 10). These sites may contain few or any live trees, but just in terms of carbon pools, they are storing similar amounts of carbon to unburned reference sites and are fulfilling potential old growth values for carbon storage. Of the 36 sites below the NRV threshold, more were found within one fire (Nahatlatch, n=11) than any other fire (Appendix A), indicating that region may have experienced the greatest loss of AGC and would potentially be the highest priority area for restoration (e.g., replanting) of the five sites.

Table 10 – Summary table of AGC and index threshold values

Variable	Minimum reference value	Number of burned plots (n=60) below threshold	% of burned plots below threshold
AGC (Mg ha^{-1})	174	36	60%
% Change* in relative AGC	-57%	32	53%
Structural Index	0.33	50	83%
% Change* in relative structural index	-56%	44	73%
Jaccard Similarity Index Value	0.242	59	98%
% Change* in relative Jaccard value	-45%	48	80%

** % Change in relative value refers to dotted line values fromFfigures 15, 19, 24*

An alternative threshold, or as an additional way to winnow the candidate sites down further, a decision maker could consider relative change in carbon between a plot and its specific reference sites as opposed to the whole landscape. Consider Figure 15 where one plot had approximately 57% less AGC than its local corresponding reference site (lowest green point at the dashed line). This plot still contained a total 190 Mg C ha^{-1}, which is more carbon than the lowest unburned site (174 Mg C ha^{-1}) over the entire landscape. Another plot had a total of 172 Mg C ha^{-1}, which is below the NRV for the entire landscape, but compared to its local reference sites was a difference of -60% in AGC. In this case, the first plot (-57%) would be a new minimum threshold, which eliminates the second plot (-60%) as a restoration candidate site. There were 32 burned plots that were below the threshold of -57%, and these plots would be the least likely to still maintain old forest carbon pools and therefore better candidates for restoration. Of these, most again came from a single fire (Nahatlatch, n=11).

Following the same method, the structural index found 10 plots within the unburned NRV when considering the landscape-level assessment of residual structure (Figure 18) while the relative assessment (Figure 19) found that 16 burned plots contained up to 56% less structure than their local reference plots but were still within the unburned NRV of the landscape. These plots could be ranked highest for conservation. When considering understory vegetation, plots can be found which have less commonality (Figure 23 and Figure 24) with unburned reference sites. Plots with high diversity likely have a high amount of residual resources to support the community, and plots with high commonality indicate good recovery or colonization by old-growth species.

The results of the above analysis could be put into a spreadsheet to see which plots were below the threshold in all categories (Table 11). This final list of plots represents the worst of the worst for burn severity, the plots that have the lowest old growth values. Future work would compare other historical burn severity plots to RBR maps to understand if similar predictions can be made.

Table 11–- Truncated table showing example of plot selection

Plots below structure threshold	Plots below carbon threshold	Plots below both thresholds
Boulder-High-1	Boulder-High-2	Boulder-High-6
Boulder-High-2	Boulder-High-3	
Boulder-High-3	Boulder-High-4	
Boulder-High-4	Boulder-High-5	Grizzly-High-3
Boulder-High-5	Boulder-High-6	Grizzly-High-4
Boulder-High-6	Boulder-Moderate-1	Grizzly-Moderate-3
Boulder-Moderate-2	Elaho-High-2	Meager-High-4
...	...	...

Kneeshaw and Burton (1998) developed a rating guide or score card to rank sub-boreal forest stands according to old-growth indicators that their research suggested. As in their paper, a similar score card could be created to rank all sites based on the combined criteria of AGC, structure and Jaccard indexes (Table 12). In these guides for the CWHms1 BEC zone, the threshold values are the minimum values for the unburned NRV from this analysis across all plots. The weighting factors are hypothetical values that sum to one, in the scenario that one variable is considered more important than another and can be weighted accordingly, perhaps reflecting stakeholder input.

Table 12–- A rating guide for burned plots within the CWHms1. Weighting factors sum to 1 and would be adjusted for the needs of the particular old-growth value

Rating Guide for CWHms1 post-fire stands					
	Observed Value	Threshold Values	Observed / Threshold	Weighting Factor	Product
Carbon	______	109	______		______
Structure	______	0.33	______		______
Commonality	______	0.24	______		______
				Sum	
Example of a lower value plot					
B-H-6	Observed Value	Threshold Values	Observed / Threshold	Weighting Factor	Product
Carbon	50	109	0.46	0.50	0.22
Structure	0.09	0.33	0.27	0.30	0.08
Commonality	0.00	0.24	0.00	0.05	0.00
				Sum	**0.30**
Example of a higher value plot					
E-M-3	Observed Value	Threshold Values	Observed / Threshold	Weighting Factor	Product
Carbon	247	109	2.3	0.50	1.13
Structure	0.38	0.33	1.16	0.30	0.35
Commonality	0.19	0.24	0.77	0.05	0.04
				Sum	**1.52**

In conclusion, this rating guide method is meant to help land managers and stakeholders weight different old-growth values to reflect the local, regional or provincial priorities for post-wildfire landscape management. By using a limited amount of field sampling, or through judicious use of remotely sensed data, weighting and ranking plots decisions can be made. Decisions may be about whether to retain burned areas in their naturally disturbed state in order to conserve the carbon or structure, versus targeting areas for replanting because they are so severely degraded that rehabilitation through rapid planting should be a priority. Decision making can be complicated by economic or political incentive for salvage logging to recover some forestry costs, however this further impacts already disturbed areas and there are a wide array of ecological and landscape ramifications from this (Lindenmayer et al. 2008) which needs to be considered first.

References Cited

Arsenault, A., and G. E. Bradfield. 1995. Structural – compositional variation in three age-classes of temperate rainforests in southern coastal British Columbia. Canadian Journal of Botany 73:54–64.

Bartels, S. F., H. Y. H. Chen, M. A. Wulder, and J. C. White. 2016. Trends in post-disturbance recovery rates of Canada's forests following wildfire and harvest. Forest Ecology and Management 361:194–207.

Bauhus, J., K. Puettmann, and C. Messier. 2009. Silviculture for old-growth attributes. Forest Ecology and Management 258:525–537.

B.C. BigTree Website and University of British Columbia. 2022. Conifers | BC BigTree. https://bigtrees.forestry.ubc.ca/bc-bigtree-registry/conifers/.

B.C. Ministry of Forests and Range, and B.C. Ministry of Environment, editors. 2010. Field manual for describing terrestrial ecosystems. 2nd ed. B.C. Ministry of Forests and Range, Victoria, B.C.

B.C. Ministry of Forests, and B.C. Ministry of Environment. 1995. Biodiversity Guidebook. Government of British Columbia.

B.C. Ministry of Forests, Lands and Natural Resource Operations, and Province of British Columbia, Victoria, B.C. 2016. Great Bear Rainforest Land Use Objectives Order.

B.C. Wildfire Service. 2020. Wildfire Averages. Province of British Columbia. https://www2.gov.bc.ca/gov/content/safety/wildfire-status/about-bcws/wildfire-statistics/wildfire-averages.

B.C. Wildfire Service. 2022. Wildfire Season Summary. Province of British Columbia. https://www2.gov.bc.ca/gov/content/safety/wildfire-status/about-bcws/wildfire-history/wildfire-season-summary.

Bezzola, A., and D. Coxson. 2020. Lifeboat or sinking ship: will the size and shape of Old-Growth Management Areas provide viable future habitat for temperate rainforest lichens? Canadian Journal of Forest Research 50:774–787.

Black, T. A., R. S. Jassal, A. L. Fredeen, Pacific Institute for Climate Solutions, B.C. Ministry of Environment, University of British Columbia, University of Northern British Columbia, University of Victoria, and Simon Fraser University. 2008. Carbon sequestration in British Columbia's forests and management system. Pacific Institute for Climate Solutions, Victoria, B.C.

Burton, P. J., M.-A. Parisien, J. A. Hicke, R. J. Hall, and J. T. Freeburn. 2008. Large fires as agents of ecological diversity in the North American boreal forest. International Journal of Wildland Fire 17:754–767.

Chang, C. C., and B. L. Turner. 2019. Ecological succession in a changing world. Journal of Ecology 107:503–509.

Clason, A. J., P. M. F. Lindgren, and T. P. Sullivan. 2008. Comparison of potential non-timber forest products in intensively managed young stands and mature/old-growth forests in south-central British Columbia. Forest Ecology and Management 256:1897–1909.

Committee on the Status of Endangered Wildlife in Canada. 2012. COSEWIC assessment and status report on the marbled murrelet, Brachyramphus marmoratus, in Canada.

Daniels, L. D., and R. W. Gray. 2006. Disturbance regimes in coastal British Columbia. BC Journal of Ecosystems and Management 7:44–56.

Daniels, L. D., R. W. Gray, B. Brett, P. D. Pickell, M. F. J. Pisaric, R. D. Chavardès, G. A. Greene, H. M. Marcoux, and V. Comeau. 2017. Disturbance Regimes in the Maritime to Submaritime Forests of the South Coast of British Columbia: Status of Knowledge and Understanding. Page 98. Report to the British Columbia Ministry of Forests, Lands and Natural Resource Operations, Victoria, BC.

DeLong, S. C., and W. B. Kessler. 2000. Ecological characteristics of mature forest remnants left by wildfire. Forest Ecology and Management 131:93–106.

Donato, D. C., J. B. Fontaine, and J. L. Campbell. 2016. Burning the legacy? Influence of wildfire reburn on dead wood dynamics in a temperate conifer forest. Ecosphere 7:13.

Eskelson, B. N. I., V. J. Monleon, and J. S. Fried. 2016. A 6 year longitudinal study of post-fire woody carbon dynamics in California's forests. Canadian Journal of Forest Research 46:610–620.

European Space Agency. 2020, September. ESA - Sentinel Online. https://sentinel.esa.int/web/sentinel/home.

Finegan, B. 1984. Forest succession. Nature 312:109–114.

Food and Agriculture Organization of the United Nations. 2020. Global Forest Resources Assessment 2020. FAO, Rome.

Franklin, J. F. 1990. Biological Legacies: A Critical Management Concept from Mount St. Helens. Pages 216–219 *in* R. E. McCabe, editor. Transactions of the Fifty-fifth North American Wildlife and Natural Resources Conference. Washington, DC, United States.

Franklin, J. F., W. Laudenslayer, F. Hall, C. Maser, and T. Spies. 1986. Interim definitions for old-growth Douglas-fir and mixed-conifer forests in the Pacific Northwest and

California. U.S. Dept. of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Or.

Franklin, J. F., D. Lindenmayer, J. A. MacMahon, A. McKee, J. Magnuson, D. A. Perry, R. Waide, and D. Foster. 2000. Threads of continuity: Ecosystem disturbance, recovery, and the theory of biological legacies. Conservation in Practice 1:8–17.

Franklin, J. F., T. A. Spies, R. V. Pelt, A. B. Carey, D. A. Thornburgh, D. R. Berg, D. B. Lindenmayer, M. E. Harmon, W. S. Keeton, D. C. Shaw, K. Bible, and J. Chen. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management 155:399–423.

Fraver, S., A. Ringvall, and B. G. Jonsson. 2007. Refining volume estimates of down woody debris. Canadian Journal of Forest Research 37:627–633.

Frazer, G. W., J. A. Trofymow, and K. P. Lertzman. 2000. Canopy openness and leaf area in chronosequences of coastal temperate rainforests. Canadian Journal of Forest Research 30:239–256.

Gauslaa, Y., P. Bartemucci, and K. A. Solhaug. 2019. Forest edge-induced damage of cephalo- and cyanolichens in northern temperate rainforests of British Columbia. Canadian Journal of Forest Research 49:434–439.

Gavin, D. G., L. B. Brubaker, and K. P. Lertzman. 2003. Holocene fire history of a coastal temperate rain forest based on soil charcoal radiocarbon dates. Ecology 84:186–201.

Gerzon, M., B. Seely, and A. MacKinnon. 2011. The temporal development of old-growth structural attributes in second-growth stands: a chronosequence study in the Coastal

Western Hemlock zone in British Columbia. Canadian Journal of Forest Research 41:1534–1546.

Gilhen-Baker, M., V. Roviello, D. Beresford-Kroeger, and G. N. Roviello. 2022. Old growth forests and large old trees as critical organisms connecting ecosystems and human health. A review. Environmental Chemistry Letters 20:1529–1538.

Gill, N. S., D. Jarvis, T. T. Veblen, S. T. A. Pickett, and D. Kulakowski. 2017. Is initial post-disturbance regeneration indicative of longer-term trajectories? Ecosphere 8:e01924.

Gonzalez, J. S. 1990. Wood density of Canadian tree species. Northern Forestry Centre, Forestry Canada, Edmonton, Alberta.

Gorley, A. 2012. Conserving Old Growth Forests in BC. Implementation of old-growth retention objectives under FRPA. Special Investigation, Forest Practices Board BC, Victoria, B.C.

Gorley, A., and G. Merkel. 2020. A New Future for Old Forests: A Strategic Review of How British Columbia Manages for Old Forests Within Its Ancient Ecosystems. Independent Strategic Review, Victoria, B.C.

Gray, A. N., V. J. Monleon, and T. A. Spies. 2009. Characteristics of remnant old-growth forests in the northern Coast Range of Oregon and comparison to surrounding landscapes. General Technical Report PNW-GTR-790, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

Gray, A. N., T. R. Whittier, and M. E. Harmon. 2016. Carbon stocks and accumulation rates in Pacific Northwest forests: role of stand age, plant community, and productivity. Ecosphere 7:e01224.

Green, R. N., and K. Klinka. 1994. A Field Guide to Site Identification and Interpretation for the Vancouver Forest Region. Land Management Handbook No. 28. B.C. Ministry of Forests, Research Program, Victoria, B.C.

Guindon, L., S. Gauthier, F. Manka, M.-A. Parisien, E. Whitman, P. Bernier, A. Beaudoin, P. Villemaire, and R. Skakun. 2021. Trends in wildfire burn severity across Canada, 1985 to 2015. Canadian Journal of Forest Research 51:1230–1244.

Halofsky, J. E., D. L. Peterson, and B. J. Harvey. 2020. Changing wildfire, changing forests: the effects of climate change on fire regimes and vegetation in the Pacific Northwest, USA. Fire Ecology 16:4.

Hamilton, E., and A. Nicholson. 1991. Defining British Columbia's old-growth forests: Discussion paper. B.C. Ministry of Forests, Research Branch, Victoria, BC.

Harper, K. A., S. E. Macdonald, P. J. Burton, J. Chen, K. D. Brosofske, S. C. Saunders, E. S. Euskirchen, D. Roberts, M. S. Jaiteh, and P.-A. Esseen. 2005. Edge Influence on Forest Structure and Composition in Fragmented Landscapes. Conservation Biology 19:768–782.

Haughian, S., P. Burton, S. W. Taylor, and C. Curry. 2012. Expected effects of climate change on forest disturbance regimes in British Columbia. Journal of Ecosystems and Management 13:1–24.

Hessburg, P. F., C. L. Miller, S. A. Parks, N. A. Povak, A. H. Taylor, P. E. Higuera, S. J. Prichard, M. P. North, B. M. Collins, M. D. Hurteau, A. J. Larson, C. D. Allen, S. L. Stephens, H. Rivera-Huerta, C. S. Stevens-Rumann, L. D. Daniels, Z. Gedalof, R. W. Gray, V. R. Kane, D. J. Churchill, R. K. Hagmann, T. A. Spies, C. A. Cansler, R. T. Belote, T. T. Veblen, M. A. Battaglia, C. Hoffman, C. N. Skinner, H. D. Safford, and R.

B. Salter. 2019. Climate, Environment, and Disturbance History Govern Resilience of Western North American Forests. Frontiers in Ecology and Evolution 7:239.

Hilbert, J., and A. Wiensczyk. 2007. Old-growth definitions and management: A literature review. BC Journal of Ecosystems and Management 8:15–32.

Hoffman, K. M., A. J. Trant, W. Nijland, and B. M. Starzomski. 2018. Ecological legacies of fire detected using plot-level measurements and LiDAR in an old growth coastal temperate rainforest. Forest Ecology and Management 424:11–20.

Holt, R., K. Price, L. Kremsater, A. MacKinnon, and K. Lertzman. 2008. Defining old growth and recovering old growth on the coast: discussion of options. Ecosystems Based Management Working Group. B.C. Ministry of Forests, Research Program, Victoria, B.C.

Integrated Land Management Bureau, editor. 2007, August. OGMA Guidance Thompson Okanagan Service Centre Southern Interior Region. B.C. Ministry of Forests.

Ishii, H. T., S. Tanabe, and T. Hiura. 2004. Exploring the Relationships Among Canopy Structure, Stand Productivity, and Biodiversity of Temperate Forest Ecosystems. Forest Science 50:342–355.

Keeley, J. E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire 18:116–126.

Key, C. H., and N. C. Benson. 2006. Landscape assessment: Sampling and analysis methods: Firemon: Fire effects monitoring and inventory system.". General Technical Report GTR-164-CD, US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Kleinn, C., B. Traub, and C. Hoffmann. 2002. A note on the slope correction and the estimation of the length of line features. Canadian Journal of Forest Research 32:751–756.

Kneeshaw, D. D., and P. J. Burton. 1998. Assessment of Functional Old-Growth Status: A Case Study in the Sub-Boreal Spruce Zone of British Columbia, Canada. Natural areas journal : a quarterly publication of the Natural Areas Association. 18:293–308.

Kolden, C. A., J. A. Lutz, C. H. Key, J. T. Kane, and J. W. van Wagtendonk. 2012. Mapped versus actual burned area within wildfire perimeters: Characterizing the unburned. Forest Ecology and Management 286:38–47.

Kopra, K., and M. C. Feller. 2007. Forest Fires and Old-Growth Forest Abundance in Wet, Cold, Engelmann Spruce – Subalpine Fir Forests of British Columbia, Canada. Natural Areas Journal 27:345–353.

Landres, P. B., P. Morgan, and F. J. Swanson. 1999. Overview of the Use of Natural Variability Concepts in Managing Ecological Systems. Ecological Applications 9:1179–1188.

Larson, A. J., and J. F. Franklin. 2005. Patterns of conifer tree regeneration following an autumn wildfire event in the western Oregon Cascade Range, USA. Forest Ecology and Management 218:25–36.

Latifi, H. 2012. Characterizing Forest Structure by Means of Remote Sensing: A Review. Pages 1–26 *in* B. Escalante, editor. Remote Sensing - Advanced Techniques and Platforms. InTech Open Access.

Lentile, L. B., P. Morgan, A. T. Hudak, M. J. Bobbitt, S. A. Lewis, A. M. S. Smith, and P. R. Robichaud. 2007. Post-Fire Burn Severity and Vegetation Response Following Eight Large Wildfires Across the Western United States. Fire Ecology 3:91–108.

LePage, P., and A. Banner. 2015. Long-term recovery of forest structure and composition after harvesting in the coastal temperate rainforests of northern British Columbia. Forest Ecology and Management 318:250–260.

Lertzman, K., D. Gavin, D. Hallett, L. Brubaker, D. Lepofsky, and R. Mathewes. 2002. Long-Term Fire Regime Estimated from Soil Charcoal in Coastal Temperate Rainforests. Conservation Ecology 6:5–5.

Lertzman, K. P., G. D. Sutherland, A. Inselberg, and S. C. Saunders. 1996. Canopy Gaps and the Landscape Mosaic in a Coastal Temperate Rain Forest. Ecology 77:1254–1270.

Lindenmayer, D. B., P. J. Burton, and J. F. Franklin. 2008. Salvage logging and its ecological consequences. Island Press, Washington, DC, United States.

Lindenmayer, D. B., and W. F. Laurance. 2017. The ecology, distribution, conservation and management of large old trees: Ecology and management of large old trees. Biological Reviews 92:1434–1458.

Lindenmayer, D. B., W. F. Laurance, and J. F. Franklin. 2012. Global Decline in Large Old Trees. Science 338:1305–1306.

Little, S. N., and J. L. Ohmann. 1988. Estimating nitrogen lost from forest floor during prescribed fires in Douglas-fir/western hemlock clearcuts. Forest Science 34:152–164.

Lutz, J. A., T. J. Furniss, D. J. Johnson, S. J. Davies, D. Allen, A. Alonso, K. J. Anderson-Teixeira, A. Andrade, J. Baltzer, K. M. L. Becker, E. M. Blomdahl, N. A. Bourg, S. Bunyavejchewin, D. F. R. P. Burslem, C. A. Cansler, K. Cao, M. Cao, D. Cárdenas, L.-W. Chang, K.-J. Chao, W.-C. Chao, J.-M. Chiang, C. Chu, G. B. Chuyong, K. Clay, R. Condit, S. Cordell, H. S. Dattaraja, A. Duque, C. E. N. Ewango, G. A. Fischer, C. Fletcher, J. A. Freund, C. Giardina, S. J. Germain, G. S. Gilbert, Z. Hao, T. Hart, B. C. H.

Hau, F. He, A. Hector, R. W. Howe, C.-F. Hsieh, Y.-H. Hu, S. P. Hubbell, F. M. Inman-Narahari, A. Itoh, D. Janík, A. R. Kassim, D. Kenfack, L. Korte, K. Král, A. J. Larson, Y. Li, Y. Lin, S. Liu, S. Lum, K. Ma, J.-R. Makana, Y. Malhi, S. M. McMahon, W. J. McShea, H. R. Memiaghe, X. Mi, M. Morecroft, P. M. Musili, J. A. Myers, V. Novotny, A. de Oliveira, P. Ong, D. A. Orwig, R. Ostertag, G. G. Parker, R. Patankar, R. P. Phillips, G. Reynolds, L. Sack, G.-Z. M. Song, S.-H. Su, R. Sukumar, I.-F. Sun, H. S. Suresh, M. E. Swanson, S. Tan, D. W. Thomas, J. Thompson, M. Uriarte, R. Valencia, A. Vicentini, T. Vrška, X. Wang, G. D. Weiblen, A. Wolf, S.-H. Wu, H. Xu, T. Yamakura, S. Yap, and J. K. Zimmerman. 2018. Global importance of large-diameter trees. Global Ecology and Biogeography 27:849–864.

Lutz, J. A., S. Struckman, T. J. Furniss, C. A. Cansler, S. J. Germain, L. L. Yocom, D. J. McAvoy, C. A. Kolden, A. M. S. Smith, M. E. Swanson, and A. J. Larson. 2020. Large-diameter trees dominate snag and surface biomass following reintroduced fire. Ecological Processes 9:41.

Luyssaert, S., E.-D. Schulze, A. Börner, A. Knohl, D. Hessenmöller, B. E. Law, P. Ciais, and J. Grace. 2008. Old-growth forests as global carbon sinks. Nature 455:213–215.

Mackey, B., E. Skinner, and P. Norman. 2021. A Review of Definitions, Data, and Methods for Country-level Assessment and Reporting of Primary Forest : A Discussion Paper for the Food and Agriculture Organisation of the United Nations. Griffith Climate Action Beacon, Griffith University, Queensland, Australia.

MacKinnon, A. 1998. Biodiversity and old-growth forests. In Conservation biology principles for forested landscapes. Page (J. Voller and S. Harrison, Eds.). UBC Press, Vancouver, BC.

MacKinnon, A. 2003. West coast, temperate, old-growth forests. The Forestry Chronicle 79:475–484.

Maestrini, B., E. C. Alvey, M. D. Hurteau, H. Safford, and J. R. Miesel. 2017. Fire severity alters the distribution of pyrogenic carbon stocks across ecosystem pools in a Californian mixed-conifer forest: Pyrogenic Carbon Stock and Fire Severity. Journal of Geophysical Research: Biogeosciences 122:2338–2355.

Maser, C., R. G. Anderson, K. Cromack Jr, J. T. Williams, and R. E. Martin. 1979. Dead and Down Woody Material. Pages 78–95 Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. U.S. Department of Agriculture, Forest Service.

Matsuzaki, E., P. Sanborn, A. L. Fredeen, C. H. Shaw, and C. Hawkins. 2013. Carbon stocks in managed and unmanaged old-growth western redcedar and western hemlock stands of Canada's inland temperate rainforests. Forest Ecology and Management 297:108–119.

McElhinny, C., P. Gibbons, C. Brack, and J. Bauhus. 2005. Forest and woodland stand structural complexity: Its definition and measurement. Forest Ecology and Management 218:1–24.

McKinley, D. C., M. G. Ryan, R. A. Birdsey, C. P. Giardina, M. E. Harmon, L. S. Heath, R. A. Houghton, R. B. Jackson, J. F. Morrison, B. C. Murray, D. E. Pataki, and K. E. Skog. 2011. A synthesis of current knowledge on forests and carbon storage in the United States. Ecological Applications 21:1902–1924.

Meddens, A. J. H., C. A. Kolden, and J. A. Lutz. 2016. Detecting unburned areas within wildfire perimeters using Landsat and ancillary data across the northwestern United States. Remote Sensing of Environment 186:275–285.

Meigs, G., and M. Krawchuk. 2018. Composition and Structure of Forest Fire Refugia: What Are the Ecosystem Legacies across Burned Landscapes? Forests 9:243.

Mildrexler, D. J., L. T. Berner, B. E. Law, R. A. Birdsey, and W. R. Moomaw. 2020. Large Trees Dominate Carbon Storage in Forests East of the Cascade Crest in the United States Pacific Northwest. Frontiers in Forests and Global Change 3:594274.

Miller, J. D., and A. E. Thode. 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). Remote Sensing of Environment 109:66–80.

Ministry of Forests, R. B. 2022. Biogeoclimatic Ecosystem Classification Program. Government. https://www2.gov.bc.ca/gov/content/safety/wildfire-status/about-bcws/wildfire-history/wildfire-season-summary#previous.

Mitchell, A. L., A. Rosenqvist, and B. Mora. 2017. Current remote sensing approaches to monitoring forest degradation in support of countries measurement, reporting and verification (MRV) systems for REDD+. Carbon Balance and Management 12:9.

Moyer, J. M., P. N. Duinker, and F. G. Cohen. 2010. Old-growth forest values: A narrative study of six Canadian forest leaders. The Forestry Chronicle 86:256–262.

Mulverhill, C., N. C. Coops, J. C. White, P. Tompalski, and P. L. Marshall. 2019. Structural development following stand-replacing disturbance in a boreal mixedwood forest. Forest Ecology and Management 453:117586.

National Aeronautical and Space Agency (NASA). 2020, September. Landsat Science. https://landsat.gsfc.nasa.gov/.

Nicholls, D., T. Ethier, M. Eng, and P. Dykstra. 2018. Post-Natural Disturbance Forest Retention Guidance : 2017 Wildfires. Ministry of Forests, Lands, Natural Resource Operations and Rural Development, Victoria, B.C.

Oksanen, J., F. G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, E. Szoecs, and H. Wagner. 2020. vegan: Community Ecology Package Version 2.5.7. Downloaded from https://CRAN.R-project.org/package=vegan.

Oliver, C. D., and B. A. Larson. 1996. Forest Stand Dynamics. Update Edition. John Wiley & Sons, New York.

Parish, R., and J. A. Antos. 2004. Structure and dynamics of an ancient montane forest in coastal British Columbia. Oecologia 141:562–576.

Parks, S., G. Dillon, and C. Miller. 2014. A New Metric for Quantifying Burn Severity: The Relativized Burn Ratio. Remote Sensing 6:1827–1844.

Parks, S., L. Holsinger, M. Voss, R. Loehman, and N. Robinson. 2018. Mean Composite Fire Severity Metrics Computed with Google Earth Engine Offer Improved Accuracy and Expanded Mapping Potential. Remote Sensing 10:879.

Parminter, J., British Columbia, and L. and N. R. O. Ministry of Forests. 2014. Natural disturbance bibliography for British Columbia. Ministry of Forests, Lands and Natural Resource Operations, Victoria, B.C.

Peterson, K. F., B. N. I. Eskelson, V. J. Monleon, and L. D. Daniels. 2019. Surface fuel loads following a coastal–transitional fire of unprecedented severity: Boulder Creek fire case study. Canadian Journal of Forest Research 49:925–932.

Ping, X., Y. Chang, M. Liu, Y. Hu, W. Huang, S. Shi, Y. Jia, and D. Li. 2022. Carbon Emission and Redistribution among Forest Carbon Pools, and Change in Soil Nutrient Content after Different Severities of Forest Fires in Northeast China. Forests 13:110.

Pollock, M. M., and T. J. Beechie. 2014. Does Riparian Forest Restoration Thinning Enhance Biodiversity? The Ecological Importance of Large Wood. JAWRA Journal of the American Water Resources Association 50:543–559.

Price, K., R. Holt, and D. Daust. 2020. BC's Old Growth Forest: A Last Stand for Biodiversity. Page 25.

Price, K., R. F. Holt, and D. Daust. 2021. Conflicting portrayals of remaining old growth: the British Columbia case. Canadian Journal of Forest Research 51:742–752.

Pulsford, S. A., D. B. Lindenmayer, and D. A. Driscoll. 2016. A succession of theories: purging redundancy from disturbance theory: Purging redundancy from disturbance theory. Biological Reviews 91:148–167.

R Core Team. 2021. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Radies, D. N., and D. S. Coxson. 2004. Macrolichen colonization on 120–140 year old Tsuga heterophylla in wet temperate rainforests of central-interior British Columbia: a comparison of lichen response to even-aged versus old-growth stand structures. The Lichenologist 36:235–247.

Roach, W. J., S. W. Simard, C. E. Defrenne, B. J. Pickles, L. M. Lavkulich, and T. L. Ryan. 2021. Tree diversity, site index, and carbon storage decrease with aridity in Douglas-fir forests in western Canada. Frontiers in Forests and Global Change 4:682076.

Schaedel, M. S., A. J. Larson, C. J. Weisbrod, and R. E. Keane. 2017. Density-dependent woody detritus accumulation in an even-aged, single-species forest. Canadian Journal of Forest Research 47:1215–1221.

Seidl, R., W. Rammer, and T. A. Spies. 2014. Disturbance legacies increase the resilience of forest ecosystem structure, composition, and functioning. Ecological Applications 24:2063–2077.

Simard, S. W., D. A. Perry, J. E. Smith, and R. Molina. 1997. Effects of soil trenching on occurrence of ectomycorrhizas on *Pseudotsuga menziesii* seedlings grown in mature forests of *Betula papyrifera* and *Pseudotsuga menziesii*. New Phytologist 136:327–340.

Spies, T. A. 2004. Ecological concepts and diversity of old-growth forests. Journal of Forestry 102:14–20.

Spies, T. A., and J. F. Franklin. 1991. The Structure of Natural Young, Mature, and Old-Growth Douglas-Fir Forests in Oregon and Washington. Page 21. General Technical Report, U.S. Department of Agriculture, Forest Service. Pacific Northwest Research Station, Portland, OR.

Spies, T. A., J. F. Franklin, and T. B. Thomas. 1988. Coarse Woody Debris in Douglas-Fir Forests of Western Oregon and Washington. Ecology 69:1689–1702.

Spies, T. A., P. A. Stine, and R. Gravenmier. 2018. Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area. Page 200. General Technical Report PNW-GTR-970, U.S. Department of Agriculture, Forest Service. Pacific Northwest Research Station, Portland, OR.

Spiess, A.-N., and N. Neumeyer. 2010. An evaluation of R2 as an inadequate measure for nonlinear models in pharmacological and biochemical research: a Monte Carlo approach. BMC Pharmacology 10:1–11.

Stephenson, N. L., A. J. Das, R. Condit, S. E. Russo, P. J. Baker, N. G. Beckman, D. A. Coomes, E. R. Lines, W. K. Morris, N. Rüger, E. Álvarez, C. Blundo, S. Bunyavejchewin, G.

Chuyong, S. J. Davies, Á. Duque, C. N. Ewango, O. Flores, J. F. Franklin, H. R. Grau, Z. Hao, M. E. Harmon, S. P. Hubbell, D. Kenfack, Y. Lin, J.-R. Makana, A. Malizia, L. R. Malizia, R. J. Pabst, N. Pongpattananurak, S.-H. Su, I.-F. Sun, S. Tan, D. Thomas, P. J. van Mantgem, X. Wang, S. K. Wiser, and M. A. Zavala. 2014. Rate of tree carbon accumulation increases continuously with tree size. Nature 507:90–93.

Storch, F., C. F. Dormann, and J. Bauhus. 2018. Quantifying forest structural diversity based on large-scale inventory data: a new approach to support biodiversity monitoring. Forest Ecosystems 5:34.

Sutherland, I. J., E. M. Bennett, and S. E. Gergel. 2016. Recovery trends for multiple ecosystem services reveal non-linear responses and long-term tradeoffs from temperate forest harvesting. Forest Ecology and Management 374:61–70.

Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. B. Lindenmayer, and F. J. Swanson. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9:117–125.

Thompson, I. D., editor. 2009. Forest resilience, biodiversity, and climate change: a synthesis of the biodiversity / resiliende / stability relationship in forest ecosystems. Secretariat of the Convention on Biological Diversity, Montreal.

Trofymow, J. A., J. Addison, B. A. Blackwell, F. He, C. A. Preston, and V. G. Marshall. 2003. Attributes and indicators of old-growth and successional Douglas-fir forests on Vancouver Island. Environmental Reviews 11:S187–S204.

Trofymow, J. A., G. Stinson, and W. A. Kurz. 2008. Derivation of a spatially explicit 86-year retrospective carbon budget for a landscape undergoing conversion from old-growth to managed forests on Vancouver Island, BC. 6th North American Forest Ecology Workshop: From science to sustainability 256:1677–1691.

Ung, C.-H., P. Bernier, and X.-J. Guo. 2008. Canadian national biomass equations: new parameter estimates that include British Columbia data. Canadian Journal of Forest Research 38:1123–1132.

Wang, T., A. Hamann, D. Spittlehouse, and C. Carroll. 2016. Locally downscaled and spatially customizable climate data for historical and future periods for North America. PLOS ONE 11:e0156720.

Wells, R. W., K. P. Lertzman, and S. C. Saunders. 1998. Old-growth definitions for the forests of British Columbia, Canada. Natural Areas Journal 18:279–292.

Whitman, E., M.-A. Parisien, L. M. Holsinger, J. Park, and S. A. Parks. 2020. A method for creating a burn severity atlas: an example from Alberta, Canada. International Journal of Wildland Fire 29:995.

Whitman, E., M.-A. Parisien, D. K. Thompson, R. J. Hall, R. S. Skakun, and M. D. Flannigan. 2018. Variability and drivers of burn severity in the northwestern Canadian boreal forest. Ecosphere 9:26.

Wong, I. 2004. Range of natural variability 4:15.

Woodall, C. W., and V. J. Monleon. 2008. Sampling protocol, estimation, and analysis procedures for the down woody materials indicator of the FIA program. Page 22. USAD Forest Service, Northern Research Station, Newtown Square, PA.

Yin, X. 1999. The decay of forest woody debris: numerical modeling and implications based on some 300 data cases from North America. Oecologia 121:81–98.

Appendices

Appendix A – Final Values for Selected Plot-Level Attributes

Plot ID	Site Name	Burn Severity	Relativized Burn Ratio	CBI Score	Total Aboveground Carbon (Mg ha^{-1})	Structural Index	Understory Plant Species Richness	Mean Jaccard Similiarity Index	Plot ID	Site Name	Burn Severity	Relativized Burn Ratio	CBI Score	Total Aboveground Carbon (Mg ha^{-1})	Structural Index	Understory Plant Species Richness	Mean Jaccard Similiarity Index
B-C-1	Boulder	None	0	0.0	469	0.60	9	0.70	G-H-7	Grizzly	High	548	2.3	220	0.15	7	0.25
B-C-2	Boulder	None	0	0.0	292	0.33	8	0.70	G-M-1	Grizzly	Moderate	446	2.1	209	0.12	13	0.05
B-H-1	Boulder	High	590	3.0	155	0.18	9	0.06	G-M-2	Grizzly	Moderate	410	2.0	149	0.27	9	0.17
B-H-2	Boulder	High	609	3.0	113	0.18	10	0.06	G-M-3	Grizzly	Moderate	458	2.2	103	0.12	5	0.08
B-H-3	Boulder	High	634	2.8	206	0.17	9	0.06	G-M-4	Grizzly	Moderate	431	2.4	101	0.09	9	0.13
B-H-4	Boulder	High	633	3.0	89	0.16	9	0.00	G-M-5	Grizzly	Moderate	426	2.4	86	0.12	7	0.07
B-H-5	Boulder	High	629	2.9	108	0.17	8	0.00	G-M-6	Grizzly	Moderate	248	2.4	89	0.29	10	0.13
B-H-6	Boulder	High	639	3.0	97	0.09	6	0.00	M-C-1	Meager	None	-36	0.0	386	0.83	10	0.31
B-M-1	Boulder	Moderate	289	2.4	128	0.47	12	0.05	M-C-2	Meager	None	-20	0.0	238	0.71	11	0.31
B-M-2	Boulder	Moderate	254	2.5	241	0.25	14	0.05	M-H-1	Meager	High	431	2.4	25	0.08	7	0.13
B-M-3	Boulder	Moderate	429	2.4	178	0.21	10	0.09	M-H-2	Meager	High	504	2.8	35	0.11	8	0.12
B-M-4	Boulder	Moderate	399	2.5	199	0.55	7	0.07	M-H-3	Meager	High	469	2.4	16	0.07	7	0.13
B-M-5	Boulder	Moderate	460	2.6	157	0.21	10	0.06	M-H-4	Meager	High	331	3.0	14	0.07	5	0.15
B-M-6	Boulder	Moderate	535	2.6	204	0.20	8	0.06	M-H-5	Meager	High	539	2.5	68	0.30	8	0.12
E-C-1	Elaho	None	59	0.0	464	0.89	11	0.64	M-H-6	Meager	High	520	2.8	110	0.10	9	0.11
E-C-2	Elaho	None	48	0.0	276	0.76	12	0.64	M-M-1	Meager	Moderate	314	2.1	178	0.16	11	0.20
E-H-1	Elaho	High	493	2.8	254	0.23	11	0.10	M-M-2	Meager	Moderate	309	1.9	222	0.48	15	0.19
E-H-2	Elaho	High	502	2.7	96	0.36	7	0.06	M-M-3	Meager	Moderate	144	2.4	195	0.34	14	0.20
E-H-3	Elaho	High	472	2.5	258	0.54	15	0.04	M-M-4	Meager	Moderate	250	2.0	177	0.26	11	0.13
E-H-4	Elaho	High	442	2.8	374	0.23	5	0.06	M-M-5	Meager	Moderate	212	2.2	115	0.34	12	0.15
E-H-5	Elaho	High	513	2.6	319	0.14	9	0.11	M-M-6	Meager	Moderate	198	2.0	196	0.44	14	0.23
E-H-6	Elaho	High	485	2.5	244	0.19	13	0.09	N-C-1	Nahatlatch	None	-19	0.0	312	0.52	22	0.24
E-M-1	Elaho	Moderate	211	2.2	376	0.23	21	0.16	N-C-2	Nahatlatch	None	-18	0.0	565	0.77	19	0.24
E-M-2	Elaho	Moderate	223	2.0	430	0.41	18	0.16	N-H-1	Nahatlatch	High	369	2.8	30	0.11	11	0.05
E-M-3	Elaho	Moderate	262	1.9	433	0.38	11	0.19	N-H-2	Nahatlatch	High	364	2.9	53	0.12	11	0.12
E-M-4	Elaho	Moderate	326	2.0	478	0.27	9	0.21	N-H-3	Nahatlatch	High	463	2.9	28	0.12	11	0.12
E-M-5	Elaho	Moderate	310	2.0	325	0.28	9	0.21	N-H-4	Nahatlatch	High	454	2.8	49	0.14	16	0.16
E-M-6	Elaho	Moderate	394	1.9	268	0.17	12	0.12	N-H-5	Nahatlatch	High	456	2.9	30	0.05	10	0.07
G-C-1	Grizzly	None	-15	0.0	306	0.78	7	0.46	N-H-6	Nahatlatch	High	495	2.9	45	0.10	12	0.20
G-C-2	Grizzly	None	-16	0.0	174	0.43	9	0.46	N-M-1	Nahatlatch	Moderate	483	2.7	173	0.22	11	0.17
G-H-1	Grizzly	High	626	3.0	142	0.23	4	0.09	N-M-2	Nahatlatch	Moderate	448	2.6	145	0.27	14	0.19
G-H-2	Grizzly	High	619	2.5	122	0.21	10	0.13	N-M-3	Nahatlatch	Moderate	330	2.5	61	0.14	15	0.11
G-H-3	Grizzly	High	396	2.9	24	0.25	6	0.08	N-M-4	Nahatlatch	Moderate	350	2.5	57	0.18	13	0.16
G-H-4	Grizzly	High	446	2.6	44	0.30	4	0.20	N-M-5	Nahatlatch	Moderate	403	2.3	65	0.12	20	0.19
G-H-6	Grizzly	High	575	2.4	125	0.15	9	0.13	N-M-6	Nahatlatch	Moderate	397	2.6	190	0.20	18	0.22

Appendix B – Structural Index Base Values and Scores for Each Plot

PlotID	Site Name	Remotely Sensed Burn Severity	Relativized Burn Ratio	Quadratic Mean DBH of Live Trees	SD of Live Tree DBH	SD of Tree Height	Mean DBH of CWD	Mean DBH of Standing Deadwood	Number of Decay Classes	Volume of Large Live Trees ≥ 40cm DBH	Species Richness of Live Trees ≥ 7cm DBH	Score of Quadratic Mean DBH of Live Trees	Score of SD of Live Tree DBH	Score of SD of Tree Height	Score of Mean DBH of CWD	Score of Mean DBH of Standing Deadwood	Score of Number of Decay Classes	Score of Volume of Large Live Trees (≥ 40cm DBH)	Score of Species Richness of Live Trees ≥ 7cm DBH	Final Structural Index Score
B-C-1	Boulder	None	0	0.49	0.08	5.52	0.18	0.46	4.00	22.74	2.00	0.43	0.07	0.13	0.37	0.44	0.60	0.94	0.40	0.60
B-C-2	Boulder	None	0	0.58	0.08	2.99	0.20	0.00	1.00	16.99	1.00	0.55	0.08	0.03	0.46	0.00	0.00	0.68	0.20	0.33
B-H-1	Boulder	High	590	0.36	0.12	9.19	0.22	0.34	2.00	0.00	0.00	0.27	0.11	0.27	0.53	0.28	0.20	0.00	0.00	0.18
B-H-2	Boulder	High	609	0.38	0.23	10.96	0.16	0.31	2.00	0.00	0.00	0.30	0.22	0.33	0.32	0.23	0.20	0.00	0.00	0.18
B-H-3	Boulder	High	634	0.34	0.12	9.04	0.20	0.32	2.00	0.00	0.00	0.25	0.11	0.26	0.47	0.25	0.20	0.00	0.00	0.17
B-H-4	Boulder	High	633	0.41	0.17	8.29	0.14	0.37	2.00	0.00	0.00	0.33	0.16	0.23	0.24	0.32	0.20	0.00	0.00	0.16
B-H-5	Boulder	High	629	0.36	0.12	10.54	0.12	0.34	3.00	0.00	0.00	0.28	0.12	0.32	0.15	0.28	0.40	0.00	0.00	0.17
B-H-6	Boulder	High	639	0.28	0.13	5.70	0.08	0.26	2.00	0.00	0.00	0.18	0.12	0.13	0.00	0.17	0.20	0.00	0.00	0.09
B-M-1	Boulder	Moderate	289	0.66	0.32	6.31	0.22	0.72	3.00	4.87	1.00	0.65	0.30	0.16	0.55	0.79	0.40	0.15	0.20	0.47
B-M-2	Boulder	Moderate	254	0.47	0.21	10.95	0.26	0.43	2.00	0.00	0.00	0.42	0.20	0.33	0.70	0.40	0.20	0.00	0.00	0.25
B-M-3	Boulder	Moderate	429	0.43	0.20	12.62	0.17	0.39	2.00	0.00	0.00	0.37	0.19	0.40	0.35	0.35	0.20	0.00	0.00	0.21
B-M-4	Boulder	Moderate	399	0.43	0.15	11.39	0.19	0.35	4.00	9.86	2.00	0.36	0.14	0.35	0.43	0.29	0.60	0.37	0.40	0.55
B-M-5	Boulder	Moderate	460	0.33	0.24	12.88	0.21	0.23	3.00	0.00	0.00	0.23	0.22	0.41	0.51	0.13	0.40	0.00	0.00	0.21
B-M-6	Boulder	Moderate	535	0.36	0.21	12.57	0.21	0.30	2.00	0.00	0.00	0.28	0.20	0.40	0.51	0.22	0.20	0.00	0.00	0.20
E-C-1	Elaho	None	59	0.33	0.12	7.74	0.21	0.31	3.00	10.00	5.00	0.23	0.12	0.21	0.48	0.23	0.40	0.37	1.00	0.89
E-C-2	Elaho	None	48	0.36	0.20	10.03	0.24	0.00	2.00	12.30	4.00	0.28	0.19	0.30	0.63	0.00	0.20	0.47	0.80	0.76
E-H-1	Elaho	High	493	0.49	0.37	5.99	0.23	0.36	2.00	0.00	0.00	0.44	0.35	0.14	0.59	0.30	0.20	0.00	0.00	0.23
E-H-2	Elaho	High	502	0.43	0.18	8.36	0.20	0.46	3.00	0.00	1.00	0.36	0.17	0.23	0.44	0.44	0.40	0.00	0.20	0.36
E-H-3	Elaho	High	472	0.93	1.05	28.35	0.24	0.87	2.00	0.00	0.00	1.00	1.00	1.00	0.62	1.00	0.20	0.00	0.00	0.54
E-H-4	Elaho	High	442	0.27	0.14	13.05	0.34	0.23	2.00	0.00	0.00	0.15	0.13	0.41	1.00	0.13	0.20	0.00	0.00	0.23
E-H-5	Elaho	High	513	0.24	0.10	3.61	0.32	0.22	1.00	0.00	0.00	0.12	0.10	0.05	0.93	0.11	0.00	0.00	0.00	0.14
E-H-6	Elaho	High	485	0.27	0.13	6.93	0.31	0.24	2.00	0.00	0.00	0.16	0.12	0.18	0.87	0.14	0.20	0.00	0.00	0.19
E-M-1	Elaho	Moderate	211	0.42	0.25	10.10	0.15	0.35	4.00	0.00	0.00	0.35	0.23	0.30	0.26	0.29	0.60	0.00	0.00	0.23
E-M-2	Elaho	Moderate	223	0.54	0.22	14.86	0.10	0.48	4.00	5.52	1.00	0.50	0.21	0.48	0.09	0.47	0.60	0.18	0.20	0.41
E-M-3	Elaho	Moderate	262	0.34	0.19	12.70	0.23	0.31	4.00	0.00	1.00	0.25	0.18	0.40	0.58	0.23	0.60	0.00	0.20	0.38
E-M-4	Elaho	Moderate	326	0.62	0.39	11.23	0.15	0.49	3.00	0.00	0.00	0.60	0.37	0.34	0.25	0.48	0.40	0.00	0.00	0.27
E-M-5	Elaho	Moderate	310	0.59	0.23	14.45	0.17	0.54	3.00	0.00	0.00	0.56	0.22	0.47	0.34	0.55	0.40	0.00	0.00	0.28
E-M-6	Elaho	Moderate	394	0.27	0.11	7.01	0.17	0.24	4.00	0.00	0.00	0.15	0.10	0.18	0.36	0.14	0.60	0.00	0.00	0.17
G-C-1	Grizzly	None	-15	0.33	0.17	6.84	0.14	0.19	5.00	13.69	4.00	0.24	0.16	0.18	0.23	0.06	0.80	0.54	0.80	0.78
G-C-2	Grizzly	None	-16	0.29	0.10	4.30	0.16	0.21	4.00	4.44	2.00	0.19	0.09	0.08	0.30	0.10	0.60	0.13	0.40	0.43
G-H-1	Grizzly	High	626	0.31	0.16	7.23	0.22	0.27	5.00	0.00	0.00	0.21	0.16	0.19	0.54	0.18	0.80	0.00	0.00	0.23
G-H-2	Grizzly	High	619	0.30	0.12	6.88	0.24	0.27	4.00	0.00	0.00	0.19	0.12	0.18	0.60	0.18	0.60	0.00	0.00	0.21
G-H-3	Grizzly	High	396	0.19	0.05	3.29	0.10	0.19	5.00	0.00	1.00	0.06	0.04	0.04	0.07	0.07	0.80	0.00	0.20	0.25
G-H-4	Grizzly	High	446	0.20	0.08	3.77	0.20	0.18	5.00	0.00	1.00	0.07	0.07	0.06	0.45	0.06	0.80	0.00	0.20	0.30
G-H-6	Grizzly	High	575	0.22	0.11	6.61	0.21	0.19	3.00	0.00	0.00	0.10	0.10	0.17	0.49	0.07	0.40	0.00	0.00	0.15
G-H-7	Grizzly	High	548	0.28	0.11	6.30	0.24	0.26	2.00	0.00	0.00	0.17	0.10	0.16	0.59	0.16	0.20	0.00	0.00	0.15
G-M-1	Grizzly	Moderate	446	0.22	0.09	5.59	0.10	0.20	4.00	0.00	0.00	0.09	0.08	0.13	0.06	0.08	0.60	0.00	0.00	0.12
G-M-2	Grizzly	Moderate	410	0.26	0.13	6.19	0.10	0.21	4.00	2.13	1.00	0.14	0.13	0.15	0.07	0.10	0.60	0.03	0.20	0.27
G-M-3	Grizzly	Moderate	458	0.22	0.11	5.51	0.16	0.19	3.00	0.00	0.00	0.09	0.10	0.13	0.31	0.07	0.40	0.00	0.00	0.12
G-M-4	Grizzly	Moderate	431	0.18	0.07	5.75	0.16	0.17	2.00	0.00	0.00	0.04	0.06	0.13	0.30	0.04	0.20	0.00	0.00	0.09
G-M-5	Grizzly	Moderate	426	0.22	0.06	5.54	0.15	0.21	3.00	0.00	0.00	0.10	0.06	0.13	0.26	0.10	0.40	0.00	0.00	0.12
G-M-6	Grizzly	Moderate	248	0.30	0.16	7.25	0.12	0.22	4.00	2.33	1.00	0.19	0.15	0.19	0.14	0.11	0.60	0.04	0.20	0.29
M-C-1	Meager	None	-36	0.37	0.20	8.72	0.16	0.14	4.00	24.21	4.00	0.28	0.19	0.25	0.32	0.00	0.60	1.00	0.80	0.83
M-C-2	Meager	None	-20	0.41	0.15	12.06	0.19	0.43	4.00	13.77	3.00	0.34	0.14	0.38	0.43	0.40	0.60	0.54	0.60	0.71
M-H-1	Meager	High	431	0.25	0.08	2.86	0.13	0.24	2.00	0.00	0.00	0.13	0.07	0.02	0.18	0.13	0.20	0.00	0.00	0.08
M-H-2	Meager	High	504	0.30	0.09	6.20	0.11	0.29	2.00	0.00	0.00	0.20	0.08	0.15	0.11	0.21	0.20	0.00	0.00	0.11
M-H-3	Meager	High	469	0.19	0.04	2.72	0.15	0.19	2.00	0.00	0.00	0.06	0.04	0.02	0.25	0.07	0.20	0.00	0.00	0.07
M-H-4	Meager	High	331	0.21	0.06	2.52	0.13	0.20	2.00	0.00	0.00	0.09	0.06	0.01	0.17	0.09	0.20	0.00	0.00	0.07
M-H-5	Meager	High	539	0.35	0.19	5.02	0.16	0.29	3.00	0.00	1.00	0.26	0.18	0.11	0.30	0.21	0.40	0.00	0.20	0.30
M-H-6	Meager	High	520	0.24	0.09	4.94	0.15	0.22	2.00	0.00	0.00	0.12	0.08	0.10	0.27	0.11	0.20	0.00	0.00	0.10
M-M-1	Meager	Moderate	314	0.40	0.15	9.47	0.12	0.38	2.00	0.00	0.00	0.33	0.14	0.28	0.16	0.32	0.20	0.00	0.00	0.16
M-M-2	Meager	Moderate	309	0.36	0.17	8.81	0.15	0.38	4.00	0.00	2.00	0.28	0.16	0.25	0.27	0.33	0.60	0.00	0.40	0.48
M-M-3	Meager	Moderate	144	0.47	0.19	8.61	0.13	0.46	3.00	0.00	1.00	0.41	0.18	0.24	0.19	0.43	0.40	0.00	0.20	0.34
M-M-4	Meager	Moderate	250	0.30	0.14	7.68	0.14	0.27	2.00	0.00	1.00	0.20	0.13	0.21	0.24	0.18	0.20	0.00	0.20	0.26
M-M-5	Meager	Moderate	212	0.45	0.14	8.60	0.12	0.49	3.00	2.29	1.00	0.38	0.13	0.24	0.17	0.48	0.40	0.03	0.20	0.34
M-M-6	Meager	Moderate	198	0.41	0.20	8.24	0.16	0.36	2.00	0.00	2.00	0.33	0.19	0.23	0.30	0.30	0.20	0.00	0.40	0.44
N-C-1	Nahatlatch	None	-19	0.19	0.05	4.29	0.13	0.22	4.00	0.00	3.00	0.05	0.05	0.08	0.20	0.11	0.60	0.00	0.60	0.52
N-C-2	Nahatlatch	None	-18	0.35	0.18	6.64	0.17	0.22	3.00	17.84	4.00	0.27	0.17	0.17	0.33	0.11	0.40	0.72	0.80	0.77
N-H-1	Nahatlatch	High	369	0.21	0.05	4.64	0.11	0.21	4.00	0.00	0.00	0.09	0.05	0.09	0.11	0.09	0.60	0.00	0.00	0.11
N-H-2	Nahatlatch	High	364	0.25	0.10	4.87	0.13	0.24	3.00	0.00	0.00	0.14	0.09	0.10	0.19	0.13	0.40	0.00	0.00	0.12
N-H-3	Nahatlatch	High	463	0.24	0.10	4.53	0.10	0.22	4.00	0.00	0.00	0.12	0.10	0.09	0.06	0.11	0.60	0.00	0.00	0.12
N-H-4	Nahatlatch	High	454	0.30	0.12	6.81	0.13	0.27	3.00	0.00	0.00	0.19	0.11	0.18	0.17	0.18	0.40	0.00	0.00	0.14
N-H-5	Nahatlatch	High	456	0.14	0.00	2.23	0.14	0.14	2.00	0.00	0.00	0.00	0.00	0.00	0.22	0.01	0.20	0.00	0.00	0.05
N-H-6	Nahatlatch	High	495	0.26	0.13	7.00	0.12	0.23	2.00	0.00	0.00	0.15	0.12	0.18	0.15	0.12	0.20	0.00	0.00	0.10
N-M-1	Nahatlatch	Moderate	483	0.47	0.16	9.53	0.11	0.44	4.00	0.00	0.00	0.41	0.15	0.28	0.11	0.41	0.60	0.00	0.00	0.22
N-M-2	Nahatlatch	Moderate	448	0.30	0.13	8.45	0.12	0.26	3.00	1.53	1.00	0.20	0.12	0.24	0.15	0.17	0.40	0.00	0.20	0.27
N-M-3	Nahatlatch	Moderate	330	0.19	0.06	7.34	0.12	0.18	5.00	0.00	0.00	0.05	0.05	0.20	0.14	0.05	0.80	0.00	0.00	0.14
N-M-4	Nahatlatch	Moderate	350	0.21	0.09	7.49	0.12	0.20	6.00	0.00	0.00	0.08	0.09	0.20	0.15	0.08	1.00	0.00	0.00	0.18
N-M-5	Nahatlatch	Moderate	403	0.18	0.05	3.69	0.10	0.17	5.00	0.00	0.00	0.04	0.04	0.06	0.09	0.04	0.80	0.00	0.00	0.12
N-M-6	Nahatlatch	Moderate	397	0.36	0.22	11.21	0.13	0.29	4.00	0.00	0.00	0.27	0.21	0.34	0.20	0.20	0.60	0.00	0.00	0.20

Appendix C – CBI Final Values

plot ID	Site Name	Remotely Sensed Burn Severity	A Substrates	B Herbs & Low Shrubs	C Tall Shrubs and Small Trees	D Intermediate Trees	E Big Trees	Mean CBI Score
B-H-1	Boulder	High	2.8	3	3	3	3	3
B-H-2	Boulder	High	3	3	3	2.8	3	3
B-H-3	Boulder	High	2	3	3	3	3	2.8
B-H-4	Boulder	High	3	3	3	2.8	3	3
B-H-5	Boulder	High	2.8	3	2.5	3	3	2.9
B-H-6	Boulder	High	3	3	3	3	3	3
E-H-1	Elaho	High	2.2	3	3	3	3	2.8
E-H-2	Elaho	High	2.3	3	3	2.3	3	2.7
E-H-3	Elaho	High	1.8	3	2	2.8	3	2.5
E-H-4	Elaho	High	2.2	3	3	3	3	2.8
E-H-5	Elaho	High	1.3	3	3	2.8	3	2.6
E-H-6	Elaho	High	1.3	3	2.5	2.8	3	2.5
G-H-1	Grizzly	High	2.8	NA	3	3	3	3
G-H-2	Grizzly	High	1	NA	3	3	3	2.5
G-H-3	Grizzly	High	3	NA	3	2.8	3	2.9
G-H-4	Grizzly	High	1.8	NA	3	2.8	3	2.6
G-H-6	Grizzly	High	0.7	NA	3	2.8	3	2.4
G-H-7	Grizzly	High	0.5	NA	3	2.8	3	2.3
M-H-1	Meager	High	3	1	2	2.8	3	2.4
M-H-2	Meager	High	2.8	3	2.5	2.8	3	2.8
M-H-3	Meager	High	3	1	2.5	2.8	2.8	2.4
M-H-4	Meager	High	3	3	3	2.8	3	3
M-H-5	Meager	High	3	1	2.5	3	2.8	2.5
M-H-6	Meager	High	2.2	3	3	2.8	3	2.8
N-H-1	Nahatlatch	High	3	2.5	3	2.3	3	2.8
N-H-2	Nahatlatch	High	3	3	2.5	2.8	3	2.9
N-H-3	Nahatlatch	High	3	2.5	3	3	3	2.9
N-H-4	Nahatlatch	High	3	2.5	2.5	3	3	2.8
N-H-5	Nahatlatch	High	3	3	2.5	3	2.8	2.9
N-H-6	Nahatlatch	High	3	3	2.5	2.8	3	2.9
B-M-1	Boulder	Moderate	1.7	3	3	3	1.3	2.4
B-M-2	Boulder	Moderate	1.3	3	3	2.5	2.8	2.5
B-M-3	Boulder	Moderate	1.7	3	3	1.8	2.8	2.4
B-M-4	Boulder	Moderate	1.3	3	3	2.8	2.5	2.5
B-M-5	Boulder	Moderate	1.7	3	3	2.5	3	2.6
B-M-6	Boulder	Moderate	1.2	3	3	2.8	3	2.6
E-M-1	Elaho	Moderate	1	3	2.5	2.8	1.8	2.2
E-M-2	Elaho	Moderate	0.5	3	2.5	2	1.8	2
E-M-3	Elaho	Moderate	0.2	3	2	2.5	1.8	1.9
E-M-4	Elaho	Moderate	1	3	2.5	1.8	1.8	2
E-M-5	Elaho	Moderate	1.2	3	2.5	1.8	1.8	2
E-M-6	Elaho	Moderate	0.5	3	2.5	1.8	1.8	1.9
G-M-1	Grizzly	Moderate	0	NA	3	2.8	2.8	2.1
G-M-2	Grizzly	Moderate	0.8	NA	3	2.3	2	2
G-M-3	Grizzly	Moderate	0.8	NA	2.5	2.8	2.8	2.2
G-M-4	Grizzly	Moderate	1	NA	3	2.8	2.8	2.4
G-M-5	Grizzly	Moderate	2	NA	3	2.3	2.5	2.4
G-M-6	Grizzly	Moderate	2.3	NA	2.5	2	2.8	2.4
M-M-1	Meager	Moderate	1.7	3	2.5	1.8	1.8	2.1
M-M-2	Meager	Moderate	0.8	3	2.5	1.5	1.8	1.9
M-M-3	Meager	Moderate	1	3	2.5	2.5	3	2.4
M-M-4	Meager	Moderate	1	3	2.5	1.8	1.8	2
M-M-5	Meager	Moderate	1	3	2.5	1.3	3	2.2
M-M-6	Meager	Moderate	0.8	3	2.5	1.8	2	2
N-M-1	Nahatlatch	Moderate	2.8	2.5	2.5	2.8	3	2.7
N-M-2	Nahatlatch	Moderate	2.2	3	3	2.3	2.5	2.6
N-M-3	Nahatlatch	Moderate	2.2	2.5	2.5	2.3	3	2.5
N-M-4	Nahatlatch	Moderate	2.7	2	2	3	3	2.5
N-M-5	Nahatlatch	Moderate	1.3	2	2.5	2.8	2.8	2.3
N-M-6	Nahatlatch	Moderate	1.5	2.5	3	2.8	3	2.6

Note: Title letters (e.g., A, B,C…) refer to forest stratum. NA values are because no herb cover was recorded in Grizzly reference plots, therefore no difference could be recorded.

Appendix D – List of Plant Species

Scientific Name		
Abies Amabilis	*Linnaea borealis*	*Rubus spp.*
Acer glabum	*Listera caurina*	*Salix sitchensis*
Alectoria sarmentosa	*Liverwort*	*Salix spp.*
Alnus sitchensis	*Lupinus arcticus*	*Sedge 1*
Amelanchier alnifolia	*Lupinus Spp.*	*Smilacina racemosa*
Anaphalis margaritacea	*Lycopodia spp.*	*Sorbus sitchensis*
Apocynum androsaemifolium	*Lycopodium Spp.*	*Spiraea betulifolia ssp. Lucida*
Arctostaphylos uva-ursi	*Mahonia nervosa*	*Streptopus amplexifolius*
Asteraceae family	*Menziesia ferruginea*	*Streptopus spp.*
Balsamorhiza sagittata	*Mnium Spp.*	*Taxus brevifolia*
Berteroa incana	*Nephroma resupinatum*	*Thuja plicata*
Betula papyrifera	*Oplopanax horridus*	*Tiarella trifoliata*
Betula papyrifera	*Orthilia secunda*	*Trillium spp.*
Blechnum spicant	*Pachistima myrsinites*	*Tsuga heterophylla*
Boraginaceae family	*Pellia neesiana*	*Tsuga mertensiana*
Brachythecium frigidum	*Penstemon spp.*	*Tsuga hybrid*
Carex Spp. 1	*Phleum pratense*	*Unknown dicot 1*
Carex Spp. 2	*Picea hybrid*	*Unknown dicot 2*
Carex Spp. 3	*Pinus contorta*	*Unknown Moss 1*
Ceanothus spp.	*Pinus monticola*	*Unknown Moss 2*
Ceanothus velutinus	*Pleurozium schreberi*	*Vaccinium membranaceum*
Cedar Spp.	*Polygonum sp.*	*Vaccinium ovalifolium*
Ceratodon purpureus	*Polytrichum juniperinum*	*Vaccinium scoparium*
Chamaecyparis nootkatensis	*Prosartes hookeri*	*Vaccinium Spp.*
Chimaphila umbellata	*Pseudotsuga menziesii*	*Veratrum viride*
Cladina arbuscula	*Pseudotsuga menziesii glauca*	*Viola Spp.*
Cladonia	*Pseudotsuga menziesii var. menziesii*	
Cladonia bellidiflora	*Pteridium aquilinum*	
Clintonia uniflora	*Pyrolaceae spp.*	
Cornus canadensis	*Rhododendron albiflorum*	
Cornus stolonifera	*Rhytidiadelphus loreus*	
Dicranum scoparium	*Rhytidiadelphus sp.*	
Epilobium angustifolium	*Rhytidiadelphus triquetrus*	
Gaultheria shallon	*Rhytidiopsis robusta*	
Goodyera oblongifolia	*Ribes lacustre*	
Gymnocarpium dryopoteris	*Ribes spp.*	
Heuchera micanthra	*Rosa gymnocarpa*	
Hieracium albiflorum	*Rosa sp.*	
Hybrid Tsuga	*Rubus arcticus*	
Hylocomium splendens	*Rubus idaeus*	
Hypogymnia inactiva	*Rubus leucodermis*	
Kindbergia oregana	*Rubus parviflorus*	
Lichen 1	*Rubus pedatus*	
Lichen 2	*Rubus pubescens*	