Impacts of wildfire on forest surface fuel carbon in Pacific Northwest forests

by

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The following individuals certify that they have read, and recommend to the Faculty of Graduate and Postdoctoral Studies for acceptance, a thesis entitled:

Impacts of wildfire on forest surface fuel carbon in Pacific Northwest Forests

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Abstract

Forest fires are a common disturbance agent throughout the Pacific Northwest (PNW) and affect stand structure, age, species composition, and carbon storage of the productive PNW forests. Fire regimes in the PNW are predicted to shift towards more frequent and severe fires with climate change, which has important implications for carbon storage in the region. This study examines how fire severity (defined by remote sensing) impacts forest surface carbon pools (duff, litter, and downed woody materials). These carbon pools store a high proportion of stand carbon, and they have short- and long-term impacts on ecosystem function and fire behaviour.

I examined one atypically large and severe fire in coastal British Columbia to obtain baseline measurements of post-fire forest floor fuel carbon (duff, litter, woody materials of all sizes) in the region. I found that there were no differences in total surface carbon between burned and unburned plots, but there were less duff and litter fuels in burned plots. This study provides baseline data for studies of post-fire forest floor carbon dynamics in the Boulder Creek region.

Data from the United States Forest Inventory and Analysis program were utilized to estimate regional wildfire consumption factors for forest surface carbon pools (duff, litter, fine woody materials) in Oregon and Washington that are representative of the current fire regime. While forest surface pools were consumed in the fire, there were no significant differences in consumption between fire severity classes. 30 – 40% of carbon in each pool were left behind, even after high-severity fire.

This research provides both a case study and a regional study on the effects of wildfire on carbon in forest surface pools. Both types of studies provide information that is beneficial for the study of post-fire carbon, giving insights into landscape level impacts or single extreme events.

Lay Summary

Productive forests of the Pacific Northwest are an important part of the global carbon cycle. Forest carbon sinks are vulnerable to consumption during wildfire, especially as forest fires are projected to become more frequent and severe with climate change. This research focuses on the effects of wildfire on forest surface carbon pools like duff, litter, and downed woody materials throughout British Columbia, Canada, and Oregon and Washington, United States. These carbon pools store a high proportion of stand carbon, and they have short- and long-term impacts on ecosystem function and fire behaviour. The results of this research suggest that even high-severity fire can leave large amounts of surface fuel carbon in stands, which has important implications for future fire risk, and fire management decision making. The findings from this study provide valuable information about post-fire fuels in the Pacific Northwest, which inform our understanding of how wildfires affect forest carbon.

Preface

The research project in the Boulder Creek fire was initiated by my supervisor Dr. Bianca Eskelson (funded by NSERC Discovery Grant) and supported by Dr. Lori Daniels (NSERC Engage Grant in collaboration with Ecofish Research Ltd.). I designed the research methods and sampling design, with input from Drs. Bianca Eskelson, Vicente Monleon, and Lori Daniels. I was responsible for leading all data collection activities, with one field assistant, Nicole Prehn. All laboratory analysis and data entry was performed by me or Brett Tripp, under my supervision. I performed the data organization, exploration, and analysis with the aid of Drs. Bianca Eskelson and Vicente Monleon. The research for this chapter was conducted using resources, measurement equipment, and laboratory equipment provided by Dr. Bianca Eskelson (Forest Biometrics Laboratory), Dr. Lori Daniels (Tree Ring Laboratory) and Dr. Cindy Prescott (Belowground Ecosystem Group Laboratory).

In Chapter 3, the data were provided by Dr. Vicente Monleon of the Pacific Northwest Research Station from the US Forest Service, Department of Agriculture as part of a larger study on wildfire consumption factors initiated by Drs. Vicente Monleon and Bianca Eskelson. I was responsible for analyzing the data, with input from Drs. Bianca Eskelson and Vicente Monleon. This work was partially funded by the USDA Forest Service, Pacific Northwest Research Station, through agreement 14-JV-11261959-069.

This thesis is the original unpublished work of the author, Kate Frances Peterson. All figures and tables, and writing are my own work, with the exception of Figure 3-2, created by Dr. Bianca Eskelson. Manuscript and figure/table reviews were kindly provided by Drs. Bianca Eskelson, Vicente Monleon, and Lori Daniels. Chapter 2 and 3 will be submitted for publication upon acceptance of the thesis. Chapter 2 will have Drs. Bianca Eskelson, Vicente Monleon, and

Lori Daniels as co-authors. Chapter 3 will be part of a larger work with Drs. Vicente Monleon and Bianca Eskelson as co-authors.

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List of Abbreviations

BC – British Columbia

CI – confidence interval

CWHms1 – coastal western hemlock moist submaritime subzone 1

CWM – coarse woody materials

DBH – diameter at breast height

dNBR – delta normalized burn ratio

FIA – Forest Inventory and Analysis program (USDA Forest Service)

FWM – fine woody materials

Mg ha^{-1} – megagrams per hectare

MTBS – Monitoring Trends in Burn Severity program

PNW – Pacific Northwest

SD – standard deviation

SWM – small woody materials

YSF – years since fire

Acknowledgements

I would like to thank my supervisor, Dr. Bianca Eskelson, for her patience, encouragement, and incredible support throughout this process. I have learned so much from you, and I am extremely grateful for everything you have done for me. Thank you to my supervisory committee, Drs. Vicente Monleon and Lori Daniels, who provided invaluable guidance, expertise, and feedback in every step of this project. I would also like to thank Dr. Valerie LeMay for providing me with so many opportunities to grow academically and professionally.

To Veronica Woodruff, Nicole Prehn, and Brett Trip, thank you so much for providing the support that allowed me to complete my field and laboratory work. To everyone in the Biometrics Lab, thank you for always being available to help.

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I would like to thank my incredible friends and family for countless hours of support, encouragement, and company. Special thanks to Grace, Jill, and Marie-Eve, for always being there to listen and share your wisdom. Lastly, to Lauren, Mom, Dad, Adam, Amie, and Alyssa, I cannot thank you enough for everything that you have done for me; I would not have been able to do this without you.

In memory of Charles McNaughton.

Chapter 1: Introduction

In the Pacific Northwest (PNW), wildfire plays an important role in most forest ecosystems, affecting ecological processes (Wimberly and Liu 2014), stand structure and age (Wimberly and Liu 2014), and carbon cycling (Raymond and McKenzie 2012). Forests in the PNW are categorized into two types, moist and dry forests (Franklin and Johnson 2012; Wimberly and Liu 2014), with transitional ecosystems between the two. Historically, the drier regions of the PNW were adapted to frequent mixed severity fires (Hessburg et al. 2007; Perry et al. 2011; Heyerdahl et al. 2012; Law and Waring 2014; Marcoux et al. 2015), with fire return intervals ranging from 5-35 years (Franklin and Johnson 2012). The wetter regions of the PNW are also subject to fire, but typically experience much longer return intervals (Franklin and Johnson 2012), ranging from 250 to upwards of 1,000 years (Daniels et al. 1995; Lertzman et al. 2002; Gavin et al. 2003a, 2003b; Hallet et al. 2003; Daniels and Gray 2006). However, current fire regimes are shifting due to climate change, and are predicted to vary more throughout the next century (Littell et al. 2016). Generally, fire severity, area burned, and length of the fire season are projected to increase with climate change (Wotton and Flannigan 1993; Cansler and McKenzie 2014; Westerling et al. 2006; Wotton et al. 2017). It is uncertain how future wildfires will continue to impact forest ecosystems in the PNW, especially in the wetter forests not adapted to frequent fire (Wimberly and Liu 2014). Understanding both the immediate and long term effects of fire on forest carbon pools is important to understand how these temperate forests will continue to store carbon (Raymond and McKenzie 2012). As fire regimes alter with climate change, the relationships between fire and forest carbon pools could shift (Meigs et al. 2009). Therefore, we must continuously update our knowledge on wildfire effects on forest carbon.

PNW forests store very large amounts of carbon per unit area (Raymond and McKenzie 2012), up to a mean of 1,200 Megagrams per hectare (Mg ha^{-1}) in the wetter areas, and a mean of 150 Mg ha^{-1} in the drier regions (Smithwick et al. 2002), and they play an important role in the global carbon cycle (Meigs et al. 2009). Forest fires have both immediate and long-term effects on forest carbon (Meigs et al. 2009). In the short term, fires consume biomass, cause tree mortality, and remove understorey plants (Agee 1993). In the long term, fires can make forests more vulnerable to future reburns (Thompson et al. 2007), and it can take decades for forests to regenerate (Agee and Huff 1987) and for carbon pools to accumulate to pre-fire levels (Ryan et al. 2010; Eskelson et al. 2016).

Regional estimates of forest fuel consumption rates are integral in post-fire carbon accounting across larger regions, like the PNW. As fire activity is projected to increase with climate change (Wotton et al. 2017), it is necessary to monitor the impacts of fire on forest carbon (Meigs et al. 2009) on a regional basis. That being said, it is also important to study individual fires as they happen, especially when they are unusually large or severe compared to historic fires in the region. While regional estimates are invaluable for understanding fire effects on a broad scale, they will not capture atypical or anomalous fires in the same way a case study will. Case studies provide more detailed insights into the impacts of anomalous fires, which can be used in conjunction with regional estimates to monitor how fire and carbon are interacting.

1.1 Research Goals

The research goal of this thesis is to contribute to the understanding of fire effects on forest carbon in PNW forests. I focused on aboveground carbon pools, specifically forest floor and surface fuels, and the effects of fire severity on these carbon pools.

In Chapter 2, I performed a case study that aimed to establish baseline measurements of post-fire surface fuels—duff, litter, and fine, small, and coarse woody materials—on year after an unusually severe wildfire in a transitional coastal-interior ecosystem in British Columbia, Canada. The main questions addressed in this chapter were: a) Are there differences in post-fire surface fuel carbon pools among remotely-sensed fire severity classes?, and b) Are there differences in any surface fuel pools between burned and unburned plots?

In Chapter 3, I provide consumption factors for forest floor carbon pools that can be applied to wildfires across the US states of Oregon and Washington. The specific questions asked were: a) What proportions of each forest floor carbon pool, on average, were consumed during wildfire?, and b) Are differences in carbon consumption associated with fire severity classes, as defined by remote sensing?

Chapter 2: Surface fuel loads following a coastal-transitional fire of unprecedented severity: Boulder Creek fire case study

2.1 Introduction

Wildfire is a common disturbance in many forest ecosystems, with historical fire regimes ranging from frequent surface fires that cause minimal overstorey tree mortality to infrequent but intense stand-replacing crown fires (Schoennagel et al. 2004). In western North America, mixed-severity fire regimes are most common. They are well-documented in the interior conifer forests growing in dry climates of the Pacific Northwest region where fire return intervals range from years to decades (Hessburg et al. 2007; Perry et al. 2011; Heyerdahl et al. 2012; Law and Waring 2014; Marcoux et al. 2015). Historically, mixed-severity fires likely burned in the coastal temperate rainforests of the region as well, albeit with long fire return intervals of centuries to millennia at landscape scales (Daniels et al. 1995; Lertzman et al. 2002; Gavin et al. 2003a, 2003b; Hallet et al. 2003; Daniels and Gray 2006). The transitional forests between the coastal and interior ecosystems are likely exposed to a combination of both of these regimes. Forest fire characteristics are shifting with climate change, where increasing temperatures may cause future increases in the total area burned, fire behaviour, and the severity of the impacts (Flannigan et al. 2005; Westerling et al. 2006; Wotton et al. 2017). In the PNW region, wildfire activity is associated with short- and long-term variations in climate (Gedalof et al. 2005; Heyerdahl et al. 2008). Therefore, projected changes in climatic conditions toward longer, warmer and drier growing seasons have important implications for future fire regimes in this region (Daniels et al. 2017).

In southwestern British Columbia (BC), coastal maritime rainforests transition to interior continental forests over a linear distance of 200 kilometres (km). These forests lie on the boundary between the Pacific Maritime and Montane Cordillera ecozones of Canada (Government of Canada 2017). Temperate forests across this gradient are important in the global carbon balance, as they sequester and store large amounts of carbon (Smithwick et al. 2002; Nave et al. 2011). Decaying surface fuels consist of several distinct types: forest floor (duff, litter, fine woody material (FWM)), small woody material (SWM), and coarse woody material (CWM) (McRae et al. 1979). Combined, these fuels can account for a quarter of total ecosystem carbon in some Pacific Northwest forests (Smithwick et al. 2002). These fuels are also important in fire risk and behaviour, affecting surface fire intensity and spread (Agee and Huff 1987). Fire-caused tree mortality in western hemlock/Douglas-fir forests can be mostly due to overheating roots and scorched crowns, both of which can be influenced by surface fire intensity (Agee and Huff 1987). Post-fire fuel loads can be especially important, as increases in surface fuels after a large disturbance can lead to greater risk of reburns, delaying post-fire recovery (Agee and Huff 1987; Parks et al. 2014).

In BC, three of the past four fire seasons included uncharacteristically large and severe wildfires that burned during droughts and extreme fire weather. In the summer of 2015, an area of 24,789 hectares (ha) of coastal and transitional forest burned in the Coastal Fire Zone (BC Wildfire Service 2017). Fires in these coastal-transitional ecosystems can occur on long fire return intervals, upwards of 300 years in some cases (Daniels et al. 2017), suggesting that large stand-replacing fires are relatively rare (Appendix Figure B-2). Due to these long fire return intervals in coastal ecosystems, little information is available about post-fire conditions and dynamics of surface fuels. Yet, this kind of information is going to become more important as climatic changes are purported to alter existing fire regimes (Daniels et al. 2017), and we need to understand how these altered fire regimes can affect forest stand conditions. While coastal fires can be small relative to fires in drier interior ecosystems, they can negatively affect the long-living and productive coastal-transitional forests for centuries (Agee and Huff 1987). Years with anomalously large and severe fires emphasize the importance of understanding the effect of fires on forest carbon storage and fuel loads in ecosystems that historically had long fire return intervals but may be exposed to more frequent fires due to changing climate (Flannigan et al. 2005; Westerling et al. 2006; Wotton et al. 2017).

Understanding the impact of contemporary wildfires on forest composition, structure and post-fire recovery is essential to project future carbon storage dynamics in the region (Dymond et al. 2016). Post-fire fuel loads play and important role in the risk and severity of reburns, which can substantially impair post-fire recovery (Agee and Huff 1987; Prichard et al. 2017b). Quantifying the amount of carbon remaining after fires of varying severity provides information to aid in fire management decisions (Dunn and Bailey 2015), allowing managers to prescribe adequate fuel treatment plans.

The purpose of this case study was to examine surface fuel loads after the Boulder Creek fire, a relatively large, high-severity wildfire that burned coastal-interior transitional forests in the Coastal Fire Zone of BC in 2015 (BC Wildfire Service 2018a). To understand how wildfire severity affects surface fuel loads in these forests, I quantified differences in surface fuels remaining in plots that burned at different severities and tested for differences in fuel loads between burned and unburned plots. This case study provides baseline fuel loads one year after the Boulder Creek fire. The study established permanent sample plots to be revisited in the future to document post-fire surface fuel dynamics. Our findings improve understanding on how fire severity impacts surface fuel loads in BC's coastal-interior transitional forests that have not experienced frequent large fires over the last 60 years.

2.2 Materials and methods

2.2.1 Study area

The focus of this study is the 2015 Boulder Creek fire (50.626 ˚N 123.401 ˚W) located along the upper Lillooet River valley, 60 km northwest of the village of Pemberton in southwestern BC (Figure 2-1; Appendix Figure B-1). The Boulder Creek fire, ignited by lightning on June 14, 2015, was one of seven fires from that year that were notable due to their size, severity, and the risk they posed to communities. It burned 6,735 ha of forest, largely at high-severity (BC Wildfire Service 2017). This study area is in the southern variant of the Coastal Western Hemlock moist submaritime subzone (CWHms1; elevation ≤ 1,050 m) and leeward variant of the Mountain Hemlock moist maritime subzone (elevation > 1,050 m) biogeoclimatic zones (Fairbairns 2011). Mean annual temperature is 5.8˚C and total precipitation is 875 mm; values for the fire season from May through September are 13.4˚C and 191 mm (1981–2010 climate normals derived using the program ClimateWNA v.5.50; Wang et al. 2012). The CWHms1 subzone is the third driest of 10 subzones along the coastal-interior transitional zone, with a continentality index value of 23 (Coastal western hemlock range = 3 to 24; Pojar et al. 1991). The river valley ranges in elevation from 400 to 2,000 m above sea level and is characterized by steep slopes and fast-moving creeks in deep ravines (Fairbairns 2011). The region consists of a combination of moist submaritime ecosystems and drier interior ecoregions. This transition is strongly influenced by complex physiography and steep climatic gradients between the windward and lee sides of the Coast Mountain range (Daniels et al. 2017). Forest composition, structure, and productivity also vary, with transitional ecosystems encompassing a wide range of forest types, such as mesic submaritime western hemlock and dry interior Douglas-fir (Green and Klinka 1994).

Within the study area, tree species also vary with elevation and topographic position (Fairbairns 2011). In the lower elevation CWHms1 variant, the dominant tree species include western redcedar (*Thuja plicata* Donn *ex* D.Don), western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) with minor components of amabilis fir (*Abies amabilis* Douglas ex. J. Forbes) (Hamann et al. 2005). On valley-bottom floodplains, red alder (*Alnus rubra* Bong.), black cottonwood (*Populus trichocarpa* Torr. & A. Gray ex. Hook.), and bigleaf maple (*Acer macrophyllum* Pursh 1813) dominate. Red alder is also common on steep slopes (Hamann et al. 2005).

The region surrounding the study area is prone to coarse-scale disturbances including seismic activity, landslides, avalanches, and floods (Green et al. 1988). However, wildfires are relatively rare, depending on the dominant vegetation of the area. Forests in the CWHms1 variant are classified as Natural Disturbance Type 2, with infrequent stand-replacing fires at mean intervals of 200 years (BCMOF and BCMOE 1995). At high elevations in the region, the forests are classified as Natural Disturbance Type 1, with mean fire return intervals of 350 years (BCMOF and BCMOE 1995). Consistent with these representations of the stand-replacing fire regime, clearcut harvesting and even-aged silivicultural systems have been applied throughout much of the valley since 1977. Following harvesting, tree planting supplemented natural regeneration to ensure adequate stocking of the economically desirable species Douglas-fir and western redcedar.

2.2.2 Sample plots

I used Landsat derived fire severity maps (BC MFLNRO 2016) and biogeoclimatic zones (Fairbairns 2011) in a geographic information system to stratify the study area by fire severity (unburned, low, medium, and high) within biogeoclimatic variants. I generated a grid of 200x200 m squares across the burned area and randomly selected plots from the centre point of the grid cells. During field reconnaissance, I determined that burned areas on steep slopes at high elevation lacked roads and were not safely accessible. Similarly, a large landslide in 2010 made the area west of the Lillooet River inaccessible. Therefore, I conducted field sampling east of the Lillooet River (Figure 2-1), where most of the area burned was in the CWHms1 variant. Much of this area had been harvested prior to the year 2000 and was covered by second-growth forests < 50 years old; therefore, I only sampled in plots that were not old-growth. I sampled 37 plots in the second-growth forests across the four severity classes (Figure 2-1): 10 unburned plots, and 10, 8, and 9 plots that burned at low-, moderate- and high-severity, respectively.

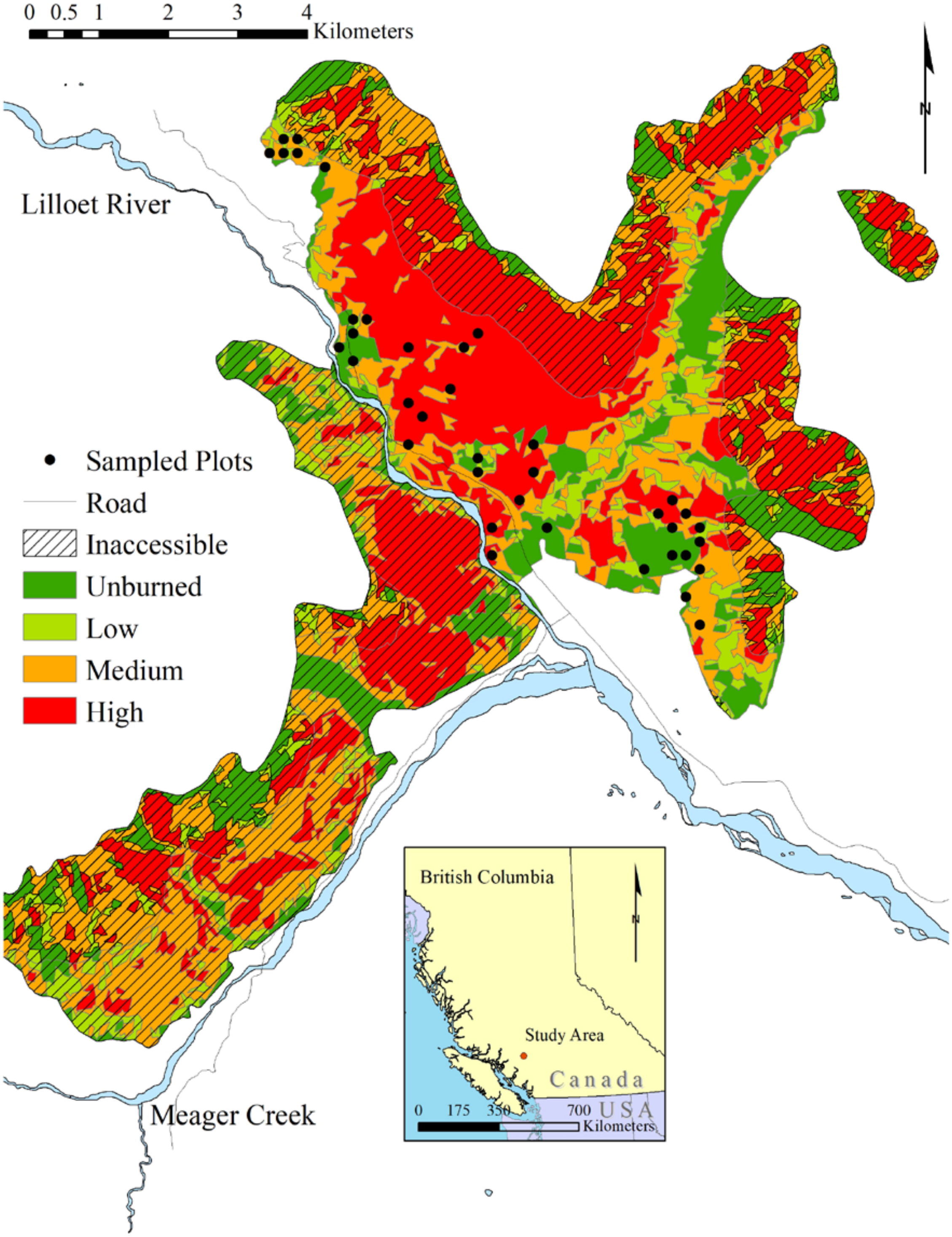

Figure 2-1. Fire severity map of the 2015 Boulder Creek fire. Political boundary data provided by U.S. Geological Survey.

2.2.3 Field sampling

At each plot, we measured elevation (m), slope angle (%), and aspect (degrees) (BC MFLNRO 2010) and visually assessed the stage of stand development to differentiate mature and second-growth forests based on tree sizes. To verify the remotely-assessed fire severity class, we visually estimated fire severity. Following Keeley (2009), we assessed the level of scorch on fuels, the amount of fine fuels consumed, as well as the amount of foliage killed or consumed in the stand. We visually estimated the amount of forest floor fuels (duff, litter, FWM) remaining, the height of scorch marks on standing trees, the size of the finest twigs and branches remaining, as well as crown mortality and consumption.

Eight to 10 plots (n = 37) were sampled in each of the unburned, low-, medium- and high-severity classes. Following the protocol for the Canadian National Forest Inventory (CFIC 2008), plot centers were permanently staked and random azimuths were chosen to establish a 30 m fuels transect with the plot center bisecting the transect (Figure 2-2). A second fuels transect was established at a 90 degree angle from the first transect. Using the line intercept method (Thompson 2012), large CWM (> 30 cm in diameter) was measured along the entire transect, medium CWM (7.5-30 cm in diameter) was measured on a total distance of 20 m, from 0-10 m and 20-30 m. For both large and medium CWM fuels, we recorded diameter at the point of intersection (cm), the angle at which the piece was tilted (degrees), and the distances along the transect (m). For each piece of wood, we recorded the species, if discernable, and assigned a decay class (1 – 5) (Maser et al. 1979). For all analyses, we combined the large and medium CWM data, referred to as CWM. SWM was tallied into one of three intersection diameter size classes (1.1 – 3.0 cm, 3.1 – 5.0 cm, 5.1 – 7.5 cm) along 10 m of the transect—from 0-5 m and 25-30 m.

In each plot, we collected FWM fuels (< 1.0 cm in diameter) in 30 x 30 cm quadrats, established at the beginning and end of both transects. Using a sampling frame, all FWM pieces were collected and clippers were used to cut any pieces that straddled the frame border. Roots and stems still attached to the ground were not collected. The samples were then dried at 70˚C until they reached constant mass, and dry weight (biomass) in grams was recorded.

For the purposes of this study, we defined litter as non-woody plant material such as foliage and cones (Keane 2015) that are not decayed, or not decayed past the point of recognition. While litter is often expanded to include the smallest woody materials (Chojnacky et al. 2009, Jain and Fried 2010), we chose to exclude those from litter measurements because we collected and measured them separately. Duff was considered to be decayed and unrecognizable plant materials (Keane 2015). The depths of the duff and litter layers above the mineral soil were measured to the nearest millimeter at 10 and 20 m along both transects in each plot and in each FWM quadrat (Brown 1974; CFIC 2008). If there was no apparent duff or litter or if the measurement point intersected rock or fallen logs, the depths were recorded as zero. To estimate duff and litter bulk density, a 10 x 10 cm sample of the duff and litter was taken from each plot, where possible, and the depth of the sample was measured. To record dry weight biomass (g), each duff/litter sample was dried at 70˚C until it reached constant mass.

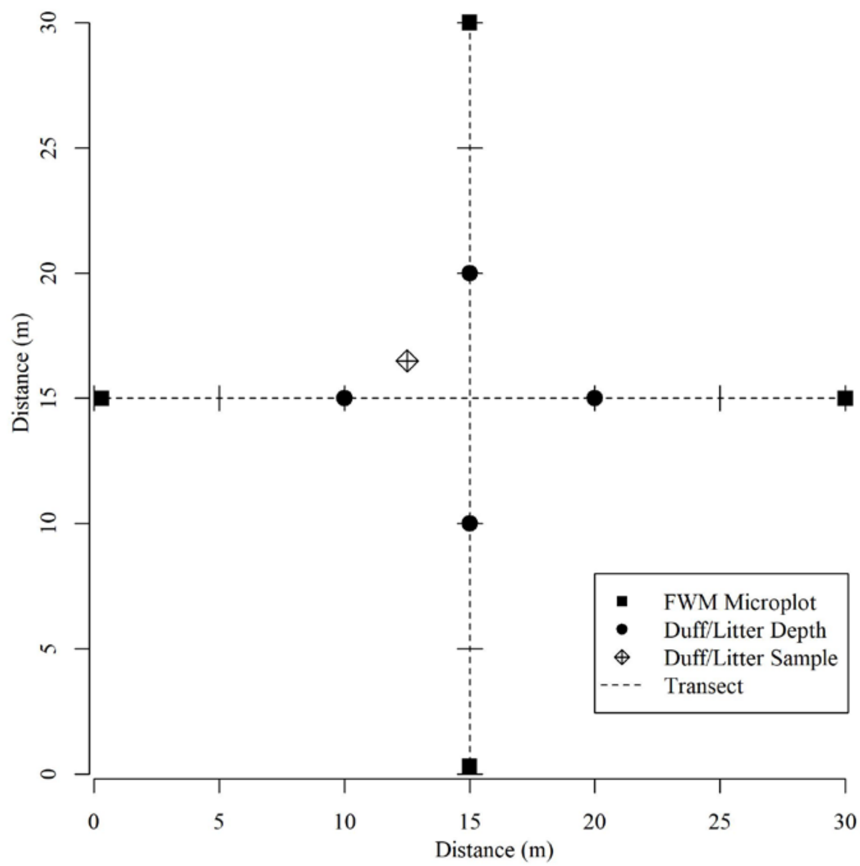

Figure 2-2. Ground plot layout, adapted from protocols used in the Canadian National Forest Inventory program (CFIC 2008).

2.2.4 Biomass and Carbon calculations

For large and medium CWM, fuel biomass was calculated using equations from the United States Forest Inventory and Analysis program (Woodall and Monleon 2008, p. 19, Eq. 4) with species- and region-specific bulk density and decay reduction factors (BC Ministry of Forests, Lands, Natural Resource Operations and Rural Development, personal communication, 2016). For SWM, I used the midpoints of each SWM diameter class in the analysis. The midpoint was 2.05 cm in class 1 (range = 1.1 – 3.0 cm), 4.05 cm in class 2 (range = 3.1 – 5.0 cm), and 6.3 cm in class 3 (range = 5.1 – 7.5 cm). The volume of SWM in each class was calculated from the midpoint diameter and number of pieces of wood using the volume formula by Woodall and Monleon (2008). I converted volume to biomass using bulk density averages reported by Fasth et al. (2010).

Duff and litter fuel carbon mass were calculated from the measured depth and estimated mean density values. To estimate the mean density value, the volume of each sample collected in the microplot was calculated and divided by the dry weight. The volume of the duff and litter layer was calculated for an area of one ha with the average duff and litter depths used as height. To obtain biomass per ha, this volume was multiplied by the density.

To obtain carbon mass for woody fuels, the biomass values in megagrams per hectare (Mg ha^{-1}) were multiplied by 0.5, a standard conversion factor for woody fuels (Campbell et al. 2007). The same conversion constant was used for litter, as the proportion of carbon in fresh Douglas-fir and cedar litter has been found to approximate 50% (Moore et al. 2006). Decayed fuels such as duff typically contain a smaller proportion of carbon—approximately 39% for Douglas-fir and 45% for cedar forests (Moore et al. 2006). As Douglas-fir was one of the most common species in the sample plots, I applied a biomass to carbon conversion constant of 40% for duff.

2.2.5 Statistical analysis

Poisson pseudo maximum likelihood models (Santos Silva and Tenreyro 2006) were fit with the PROC GLIMMIX procedure in SAS 9.4 to test for differences in mean carbon mass between remotely-sensed fire severity classes. This modelling approach was chosen because our data were strictly positive and had many zero values. I applied these methods for carbon mass in the following fuels: woody material (SWM, CWM), forest floor (duff, litter, FWM), and total surface carbon (all of the examined fuel types combined). The one fixed factor used in our analysis was severity (four levels – unburned, low, moderate, and high). If severity levels were not significantly different from each other, a burn indicator variable (two levels – unburned, burned) was used in place of severity. The response variable was carbon mass (Mg ha^{-1}) by fuel type.

2.3 Results

The dominant overstorey species in the 37 sample plots was Douglas-fir, followed by western hemlock and western redcedar, with scattered stands of black cottonwood and amabilis fir in the unburned areas (Table 2-1). Plot elevation ranged from 432 – 1,065 m, with a minimum slope of 0% and a maximum of 80%. (Table 2-1). With a mean elevation of 857 m, plots that burned at low-severity had the highest elevation (Table 2-1). Moderate-severity plots had the steepest slopes overall, with mean of 48% (Table 2-1). Low- and moderate-severity plots were typically found on southwest slopes, whereas most unburned plots and high-severity plots were south-facing. Thirty-nine and 32% of the fire area were classified as high- and moderate-severity, respectively. Low-severity and unburned classes covered 10% and 19% of the fire, respectively (BC MFLNRO 2016). Five low-severity plots and two moderate-severity plots had been logged in the 5 years prior to the fire.

Wildfires above 3,000 ha are rare in the coastal fire zone of BC (Appendix Figure B-2). When large fires (>200 ha, Stocks et al. 2002) occur in the coastal region, they are commonly

300 to 800 ha in size (BC Wildfire Service 2018a). The Boulder Creek fire is only the third wildfire that has burned more than 5,000 ha of coastal forest between 1950 and 2015.

The Boulder Creek fire burned large areas of managed second-growth forests that have experienced several logging operations as well as a large-scale hydroelectric dam project (Woodruff, personal communication, 2016), with few scattered old-growth management areas. One moderate-severity plot was found to be old-growth, with several large Douglas-fir trees that were approximately 1m DBH in size. Because the rest of the plots were second-growth stands, I excluded the old-growth plot from further analyses as it was not representative of second-growth forests, our population of interest. During field sampling I found that the remote sensing information misclassified one plot as unburned when there was clear evidence of a low-severity fire. This plot was treated as low-severity in the analyses.

Table 2-1. Plot characteristics by fire severity.

		Elevation (m)			Slope (%)			Slope direction (degrees)	Dominant Species*		Secondary Species*
Severity	n	Mean	SD	Range	Mean	SD	Range	Mean	Species	Count	Species
Unburned	10	713	194.8	493-1065	25.0	22.7	0 - 60	199	Ba	4	Fd, Act
Low	10	857	246.6	475-1058	32.9	20.6	0 - 68	233	Cw, Fd	4	Hw
Moderate	8	811	173.4	486-965	48.1	22.2	21-80	218	Cw	4	Fd
High	9	722	211.7	432-1059	36.0	27.4	0-65	199	Fd	6	Hw

**Dominant species is defined as the most frequently occurring overstorey species in each plot, count denotes the number of plots with the same dominant species. The secondary species refers to the next most frequently occurring overstorey species. Only trees that were part of the general crown were included in the count. Species codes: Ba – amabilis fir, Cw – western redcedar, Fd – Douglas-fir, Act – black cottonwood, Hw – western hemlock.*

Table 2-2. Carbon mass in Mg ha^{-1} per fuel type by fire severity, with standard deviations in parentheses

Severity	n	Duff	Litter	FWM	SWM	CWM	Total Fine Fuels	Total Surface Fuels
Unburned	10	0.42 (0.39)	0.16 (0.09)	0.63 (0.40)	5.77 (6.11)	17.5 (25.02)	6.99 (6.53)	24.50 (24.63)
Low	10	0.12 (0.15)	0.08 (0.13)	0.59 (0.49)	3.85 (3.26)	18.31 (10.58)	4.65 (3.40)	22.96 (10.10)
Moderate	8	0.03 (0.04)	0.05 (0.07)	0.55 (0.37)	4.27 (4.33)	8.78 (7.67)	4.89 (4.26)	13.68 (9.53)
High	9	0.01 (0.01)	0.005 (0.008)	0.37 (0.33)	3.76 (3.21)	21.22 (24.89)	4.15 (3.28)	25.37 (26.17)

The total surface column represents the sum of all surface fuel types examined (duff, litter, FWM, SWM, and CWM). Total fine fuels are the sum of duff, litter, FWM, and SWM.

2.3.1 Total surface carbon

Overall, there was neither a significant statistical difference in total surface carbon between burned and unburned plots ($p = 0.6537$) (Table 2-2) nor between fire severity classes ($p \geq 0.1302$). CWM was separated from the analysis to ensure that any differences in the total fine fuels (duff, litter, FWM, SWM) were not obscured by the relatively large amount of carbon mass found in CWM. No significant differences were found between burned and unburned plots in total fine fuel carbon ($p = 0.1831$) (Table 2-2). While the differences were not significant, there were decreases in carbon mass in several of the examined fuel types leading to an apparent decrease in total fine fuel carbon mass between burned and unburned plots, with minimal differences between low-, moderate-, and high-severity (Table 2-2).

2.3.2 Woody fuels

While not statistically significant, CWM carbon mass was higher in low- and high-severity plots compared to unburned plots (Table 2-2). Moderate-severity plots had significantly less CWM carbon mass than low-severity plots ($p = 0.0365$). Both SWM and FWM did not differ significantly among any severity classes ($p = 0.77$ and 0.21, respectively). There was also no significant difference between burned and unburned plots in these fuel types (SWM $p = 0.296$, FWM $p = 0.3674$); however, FWM carbon mass did decrease as fire severity increased. Like CWM, SWM fuel carbon mass at moderate-severity differed from low-, and high-severity, however in the opposite way, with more carbon mass at moderate-severity compared to low- and high-severity plots. These differences were not statistically significant.

2.3.3 Non-woody fuels

For both duff and litter, there were significant differences in carbon between burned and unburned plots ($p < 0.0196$) as well as among fire severity classes ($p < 0.0257$) (Table 2-2). Litter decreased as fire severity increased, with significantly less litter carbon mass on moderate- and high-severity plots compared to unburned plots ($p < 0.0196$). High-severity plots also had less litter carbon than low-severity plots ($p = 0.0002$), however there were no significant differences in litter carbon mass between unburned and low-severity ($p = 0.2025$) and between low- and moderate-severity ($p = 0.469$) plots. Similar to litter, duff carbon decreased with increasing fire severity. Unburned plots had significantly more duff carbon than plots that burned at any severity ($p < 0.011$). Duff carbon mass in low-severity plots was significantly higher than in both moderate- ($p = 0.0257$) and high-severity plots ($p < 0.0001$). However the difference between duff carbon mass at moderate- and high-severity was suggestive but inconclusive ($p = 0.06$).

2.4 Discussion

2.4.1 Regional significance of the Boulder Creek fire

The 2015 Boulder Creek fire burned in a season of aggressive fire activity, with extremely hot and dry weather leading to a larger-than-average total area burned and number of fires in the coastal region of British Columbia (BC Wildfire Service 2017). Only four fires in the coastal region have been larger than 5,000 ha since the 1950s, with three of those occurring in the past three years (BC Wildfire Service 2018a) (Appendix Figure B-2). The lack of fires is likely due to a combination of environmental controls and successful fire suppression. Historically, mixed- or low-severity fires on long return intervals were most likely for the region

(Daniels and Gray 2006); in contrast, the Boulder Creek fire burned mostly at high- and moderate-severity, with scattered low-severity patches. It is important to understand how these extreme events might affect forest stand conditions. In southwestern Oregon, one large fire in 1987 appeared to play a role in the severity of the large 2002 Biscuit fire (Thompson et al. 2007). This suggests that the Boulder Creek region could be vulnerable to reburns, especially considering the large patches that burned at high-severity, as initial burn severity can indicate the subsequent reburn severity (Thompson et al. 2007). This has important implications for post-fire fuel management in the region, because understanding of reburn risk can show where fire suppression, fuels treatments, and/or prescribed burns might be required (Prichard et al. 2017b).

2.4.2 Pre-fire differences in stand characteristics

All measurements for this study were taken post-fire and pre-burn data were unavailable. Due to this, it is impossible to know that control plots and plots across the three fire severities were comparable with regards to pre-fire fuel carbon. It is also possible that our plots burned at different severities because of pre-fire differences in species composition or stand structure. Our unburned plots had more deciduous trees than plots that burned at any severity, suggesting that there are differences in species composition that could have led to differences in fire behaviour, as different forest stands have different degrees of flammability (Alexander et al. 2012). Mixed-conifer forests often burn at high-severity due to their tendency of growing densely with several canopy layers (Prichard and Kennedy 2014). In contrast, deciduous trees can often be less flammable than conifers and usually have fewer ladder fuels making it more difficult for fires to climb into the crown (Fitzgerald and Bennett 2017). It is possible that our plots burned at different severities, or did not burn at all, due to pre-fire differences in stand characteristics.

Topography is a possible driver of fire severity, as steeper slopes could be more likely to burn at low- or moderate-severity (Bigler et al. 2005), partly consistent with our findings that moderate-severity plots were the steepest on average. Topography also plays an important role in stand structure and species composition (Harris and Taylor 2015), suggesting that our moderate-severity plots may have differed from plots in the other remotely-sensed burn severity classes or the control group, even before the fire occurred. However, without pre-burn data this is impossible to determine. Overall, the possible impacts of topography and forest type on fire severity demonstrate the need for pre-fire data. The permanent sample plots established in this study will provide pre-fire data for any future reburns that may occur within the Boulder Creek fire boundary.

2.4.3 Post-fire surface fuel carbon storage

While there were no statistically significant differences in total surface fuel carbon, there was less fine fuel carbon mass in burned plots compared to unburned plots. Observed decreases in duff and litter carbon with increasing fire severity are consistent with other studies on forest floor consumption in the region (Campbell et al. 2007). Duff carbon was significantly lower in low-severity compared to unburned plots, but the same was not the case for litter carbon. This was unexpected as litter is typically consumed at a higher rate than duff (Campbell et al. 2007). The similarities in litter fuel carbon mass between unburned and low-severity plots could be explained by post-fire accumulation. Litter carbon in low-severity plots would have accumulated in the year after the Boulder Creek fire prior to measurement. Litter from scorched and dead trees as well as herbaceous understorey growth can accumulate quickly post-fire (Agee and Huff 1987; Dunn and Bailey 2015; Stalling et al. 2017), but duff accumulation may not begin to occur for five to 10 years after a fire (Dunn and Bailey 2015; Eskelson and Monleon 2018). A low/moderate-severity fire may have consumed all of the duff and litter but did not burn, or scorched the tree crowns (Campbell et al. 2007). Therefore, the fire may have killed the trees but not consumed the foliage, which would remain to become litter input in the year post-fire.

While not statistically significant, FWM carbon mass decreased as fire severity increased, with similar amounts from unburned to moderate-severity and a sharp drop between moderate- and high-severity. This is consistent with results from other studies that suggest that up to 80% of small and medium downed woody fuels are consumed at high-severity (Campbell et al. 2007). The sharp decrease in FWM carbon mass between moderate- and high-severity could also be attributed to post-fire accumulation in low- and moderate-severity, as FWM carbon can increase quickly post-disturbance (Dunn and Bailey 2015). Unlike FWM carbon, there was no apparent association between severity increases and differences in SWM carbon mass. Moderate-severity plots had more SWM carbon than both low- and high-severity plots, which could be explained by post-fire accumulation from rapidly decomposing snags, as outlined in Dunn and Bailey (2015). A moderate-severity fire most likely caused greater overstorey mortality than a low-severity fire, but may not have consumed as many finer branches and twigs as a high-severity fire, leaving remnant branches to become downed woody fuel input in the year post-fire.

CWM can be quite abundant in hemlock/Douglas-fir stands after large-scale disturbances (Agee and Huff 1987). In the Boulder Creek fire, CWM carbon masses did not significantly differ between unburned plots and burned plots at any severity, which confirms similar findings of Maestrini et al. (2017). This may be because pre-existing CWM in the burned plots were not fully consumed by the fire. It also aligns with results from Eskelson and Monleon (2016), where pre- and post-fire CWM carbon mass did not differ significantly. Mitchell et al. (2009) also found that pre- and post-fire carbon stored in larger downed woody fuels do not differ substantially, even after high-severity fires, possibly due to input during fire. However, I did find that there was less CWM carbon in moderate-severity than in high-severity plots. One possible reason for this could be that pre-existing CWM was consumed at both severities; however, the moderate-severity fire may not have weakened the standing trees enough to become immediate input into the CWM pool, whereas high-severity likely did. This is further indicated by the fact that high-severity plots appear to have less standing tree carbon (Appendix Figure A-1; Appendix Table A-3) than any other severity class, suggesting that most standing trees transitioned into CWM either during or after the fire.

For SWM and CWM, carbon mass in moderate-severity plots did not follow the expected decrease in carbon mass as fire severity increased. One reason for this could be that moderate-severity plots had much steeper slopes than any other severity classes. As previously observed by Bassett et al. (2015), sloped areas that burn at higher severities can have less CWM and more SWM volume when compared to low-lying, less sloped areas, which aligns with the lower amounts of CWM carbon and higher amounts of SWM carbon in the moderate-severity plots of this study.

Yocom-Kent et al. (2015) found that the differences in post-fire carbon between severity classes widen over time. Thus, I hypothesize that post-fire carbon will decrease the most in stands that burned at high-severity as we monitor change in surface fuel carbon at the Boulder Creek fire. Our data show that much of the remaining carbon in high-severity stands is found in snags (Appendix Table A-3) and CWM. CWM will likely increase over the next several years (Dunn and Bailey 2015, Eskelson and Monleon 2018), as long as the rate of snag input into CWM pools is greater than the rate of CWM decay (Dunn and Bailey 2015). A recent study from

California, USA, suggests that CWM increases throughout the first nine years after fire (Eskelson and Monleon 2018), which is consistent with the idea that most snags convert to CWM within eight years of a fire (Ritchie et al. 2013). While our study was performed in a different region and different forest types, the distribution of stand carbon in plots that had not been logged prior to the Boulder Creek fire suggests it is likely that the same trend would occur here. Once most of the snags have fallen, it stands to reason that the high-severity plots will lose carbon due to decomposition until regeneration has firmly established (Agee and Huff 1987; Ryan et al. 2010). The large patches of high-severity in the Boulder Creek fire have important implications for post-fire fuel carbon dynamics, where it could take decades for carbon mass to return to pre-fire levels (Ryan et al. 2010).

Seven out of 37 plots had been logged within 7 years prior to the Boulder Creek fire, and all burned at low- (n=5) or moderate-severity (n=2). In this study, I initially performed the analyses with and without the logged plots (Appendix Table A-1, A-2) and compared the results. Because only minimal differences between the two analyses were found, the logged plots were included in all presented analyses. Logging can alter the amount of surface fuels (Tinker and Knight 2000), increasing the amount of fine and coarse woody fuels, leading to changes in the fire risk and flammability of the stand (Donato et al. 2006; Lindenmayer et al. 2009). Changes in surface fuels can also increase the short-term risk of burning in the adjacent, less flammable stands (Lindenmayer et al. 2009). For future research, it would be beneficial to incorporate forest management practices into fire area stratification for plot selection, in addition to severity and forest type, for a better view of the interacting disturbances across the landscape.

2.4.4 Implications of fire severity classifications

I used Landsat delta normalized burn ratio (dNBR) derived fire severity classes for our study, which relies on changes in forest canopy to determine severity levels (Eidenshink et al. 2007). Our fire severity maps were not validated in the field prior to sampling. Stand-replacing disturbances, such as high-severity fires, are easy to discern using Landsat imagery, but disturbances that do not result in stand replacement can be more difficult to distinguish from normal variations in spectral indices (Cohen et al. 2018). Due to this uncertainty, along with the lack of ground-truthing, it is possible that some of our plots were assigned to an incorrect fire severity class, especially in the areas where mixed-severity fire caused uneven tree mortality and carbon consumption. In particular, moderate-severity areas can be a major source of errors as they often occur around the edges of high-severity patches, which are difficult to discern on a larger scale (Miller et al. 2009). Because Landsat-derived fire severity classes are based on vegetation changes, changes in the forest floor and surface fuels may not be fully captured (Alonzo et al. 2017) potentially leading to a lack of significant differences in fuel carbon masses across fire severity classes. During field sampling I found that one plot was misclassified as unburned when there was clear evidence of scorched litter and duff after a low-severity surface fire. If any plots were misclassified to a less obvious degree, low-severity plots may have been treated as moderate-severity and vice versa. Clearly, there is a need for ground-based fire severity classifications that allow for greater detection of changes in surface fuels (Jain et al. 2012), in order to discern how ground fire severity affects surface fuels.

2.5 Conclusions and future work

The Boulder Creek fire was highly important in the coastal region, as it was atypically large for the region. I found that the total amount of surface fuel carbon did not differ between burned and unburned plots one year post-fire. However, there was significantly less carbon mass in the finest fuels, duff and litter in burned plots. Impacts of the Boulder Creek fire on the woody fuels may have been obscured by post-fire accumulation, and by issues associated with using crown-based fire severity classifications to assess surface fire severity. To improve future studies of this and other fires in the region, it would be prudent to take into account land management practices, forest type, and stand age when stratifying the burn area and selecting plots. Measures of ground-based fire severity would allow for a more correct picture of post-fire differences among severity classes. Understanding the impacts of forest fires on surface fuels will allow us to develop post-fire forest management plans. This case study provides baseline post-fire surface fuel data for the Boulder Creek fire, which can be used as a starting point for longitudinal studies of post-fire fuel and carbon dynamics. The established permanent plots provide a valuable opportunity for analyzing post-fire forest carbon dynamics in coastal-interior transition forests of BC. Information on disturbances in these transitional zones is currently lacking due, in part, to the historically long fire return intervals and relatively few fires in the documentary records. Fire behaviour, severity, size, and frequency are expected to shift with climate change, which highlights the importance of understanding long-term fire effects on these forests.

Chapter 3: Consumption factors for forest floor carbon pools after wildfire in Washington and Oregon, USA

3.1 Introduction

Wildfire is one of the impactful disturbance agents in the Pacific Northwest (PNW) region. Fire regimes are anticipated to be altered by climate (Littell et al. 2010; Littell et al. 2016) resulting in longer fire seasons (Wotton and Flannigan 1993; Flannigan et al. 2005; Westerling et al. 2006; Wotton et al. 2017) with larger and more severe fires (Cansler and McKenzie 2014). Area burned in the PNW has already increased since the 1980s, which has been attributed to both fire suppression as well as higher temperatures and less precipitation (Cansler and McKenzie 2014). The amount of forested area burned in the PNW has been projected to double or triple (Littell et al. 2010), or exhibit a 3.8-fold increase by the 2080s (Wimberly and Liu 2014).

Carbon is sequestered in many different forest carbon pools, including soils, woody and non-woody live biomass, CWM, FWM, duff, and litter (Smithwick et al. 2002). Forest floor carbon pools have direct impacts on ecosystem function, carbon storage, and fire behavior, and they are sensitive to anthropogenic and natural disturbances (Cameron et al. 2015). These carbon pools are substantial carbon sinks (Cameron et al. 2015), storing proportionately large amounts of carbon (Chojnacky et al. 2009). FWM, litter, and duff are believed to be fully consumed in fires that burn at high-severity, but consumption rates are more variable in lower severity fires (Campbell et al. 2007). Of all forest carbon pools, estimating forest floor carbon consumption can carry the most uncertainty, as there are many different distinct pools with differences in moisture and flammability (de Groot et al. 2009). Understanding the impact of fire on forest carbon is necessary to manage forests for their carbon sequestration abilities (Kurz et al. 2013).

While there is information on the consumption factors of different carbon pools in the PNW region, they are often based on case studies of individual fires (Campbell et al. 2007), which do not allow regional inference to other fires in the PNW. Other estimates of consumption factors are estimated from prescribed burns (van Leeuwen et al. 2014; Prichard et al. 2017a). The effects of prescribed burns on forest carbon stocks have been widely studied (Little et al. 1982). However, prescribed burns often do not behave in the same way as wildfire, leading to substantial differences in fire pattern and severity, which drive how fires affect carbon pools (Ryan et al. 2013).

In this study, I quantify forest floor fuel consumption in Oregon and Washington, USA, from a sample of burned and unburned forest inventory plots. To create comparable populations, a spatial matching approach was conducted to select unburned plots that had similar covariate distributions as the burned plots (Stuart 2010). Non-experimental studies, such as this one, rely on the assumption that there are no differences between the treatment and control groups that were caused by factors other than the treatment (Stuart and Green 2008; Stuart 2010). Matching allows us to balance the covariates of the control and treatment groups to better satisfy this assumption (Stuart 2010).

The first objective of this study was to estimate regional consumption factors for each forest floor carbon pool (duff, litter, FWM). The second objective was to test for differences in consumption factors among fire severity classes (determined by remote sensing), to determine if fire severity affected the amount of each carbon pool consumed. It is crucial to continuously update our understanding of how forest carbon pools interact with fire, as our predictions for future fire regimes rely on fire/climate/ecosystem relationships remaining constant, when this may not always be the case (Cansler and McKenzie 2014). The US national inventory data provide a probability sample of the region, and thus a sample of burned plots that represent the probability of wildfire conditions as they occurred on the landscape. Therefore, the results of this study can be used to make inferences about all large wildfires in Oregon and Washington. The consumption factors presented in this study can be applied to other wildfires in the region, because they are not derived from a single fire but a representative sample of all large wildfires of the region.

3.2 Methods

3.2.1 Fire severity classification

Burned Forest Inventory and Analysis (FIA) plots were identified by overlaying the locations of all FIA plots within Oregon and Washington with fire severity data from the Monitoring Trends in Burn Severity (MTBS) program (Eidenshink et al. 2007). Burn severity classes were applied at the whole-plot level (0 = unburned control, 1= unburned/low-severity, 2=low, 3=moderate, 4= high severity burns), using the most common fire severity in the nine 30x30m pixels overlaying each plot. Plots designated as severity class 1 (unburned/very low-severity plots within fire boundaries) were removed from the analysis because many of them were unburned plots within wildfire boundaries and the remaining plots burned at very low severity. Along with the MTBS data, field-gathered disturbance codes as well as disturbance years for the plots were examined in order to identify which plots showed fire-damage in the field. Plots that had burned more than 10 years prior to measurement were considered to be unburned and were used as potential control plots in the matching process. 134 FIA plots that were measured within 2 years of the fire were used in the analysis (Table 3-1; Figure 3-1). For the duff and litter analysis, six plots measured in 2001/2002 had to be removed from the analysis because the duff and litter measurement protocol was slightly different when the annual inventory was first implemented. This resulted in 128 burned FIA plots used in the duff and litter analysis (Table 3-1; Figure 3-1).

3.2.2 Data collection

The data for this project have been collected by the US Forest Service Forest Inventory and Analysis (FIA) program (PNW Region) from permanent forest inventory plots established throughout the states of Oregon and Washington—approximately one FIA plot per 2,500ha (6,000 acres)—which are re-measured every 10 years on a rotating basis (Bechtold and Patterson 2005). At each permanent FIA plot, stand characteristics are measured using a tri-areal plot design and transects. The measurements taken include FWM and duff and litter characteristics. The measurement procedures are outlined by the USDA Forest Service (2015).

FWM is collected as count data along transects in the plot. FWM is divided into three size classes: small (0.03 – 0.61 cm) and medium (0.62 – 2.4 cm) FWM pieces are counted on a 1.83m long transect, while large (2.5 – 7.4 cm) FWM pieces are counted along a 3 m transect. FWM volume and biomass were calculated as in Woodall and Monleon (2008). Carbon is proportional to biomass, as typically carbon amounts are considered to be 50% of the mass for woody materials (Woodall et al. 2012).

The FIA program defines litter as freshly fallen non-woody plant material such as leaves and needles that is not decayed, or not decayed past the point of recognition (Keane 2015). Duff was considered to be decayed and unrecognizable plant materials (Keane 2015). Duff and litter depths (in cm) were measured to the nearest quarter of a centimetre at the 7.3m mark of each of eight transects. These measurements were used to estimate litter and duff loads following Woodall and Monleon (2008).

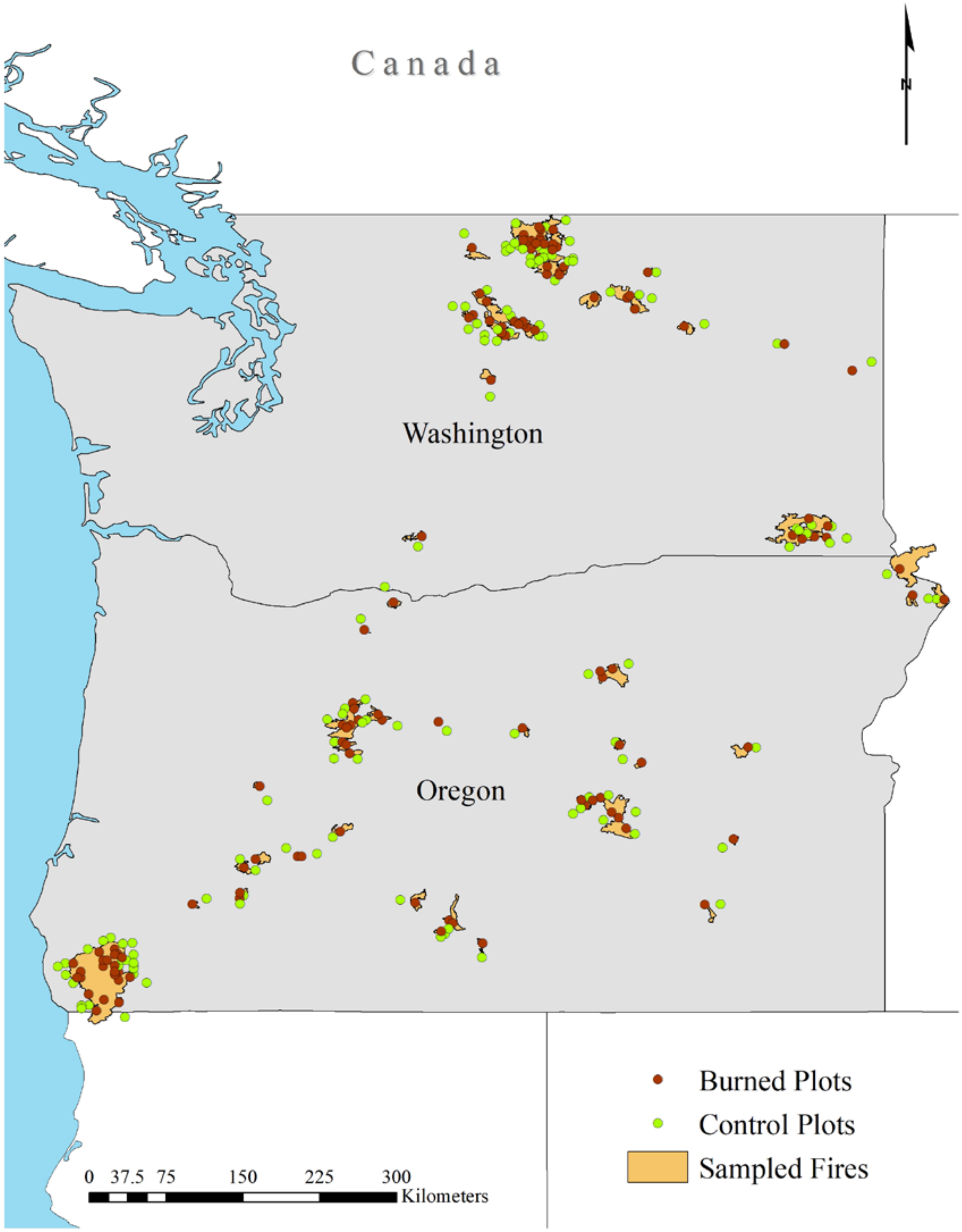

Figure 3-1. Locations of all sample plots and fires studied in Oregon and Washington. Fire boundary data created by mtbs.gov. Political boundary data created by U.S. Geological Survey.

3.2.3 Plot matching

In order to minimize the amount of post-fire forest floor fuel accumulation on the burned FIA plots, only plots that had burned within two years prior to measurement were selected as burned plots for the analysis. Burned plots were spatially matched with unburned plots with similar characteristics (i.e., ownership and forested proportion of the plot) to create comparable populations of burned and unburned plots. Specifically, for each burned plot, the spatially closest unburned plot that belonged to the same ownership group (public vs. private) and had similar forested proportion was picked as match. The spatial matching process should result in similar covariate distributions of burned and unburned plots (Stuart 2010). To assess the quality of the matching process, elevation, latitude as well as PRISM climate variables (Daly et al. 1997—mean annual temperature, maximum August temperature, minimum December temperature, annual precipitation) were used as covariates. A plot of standardized differences of means of these six covariates before and after matching was used to assess the covariance balance (Stuart 2010).

3.2.4 Statistical analysis

The SAS 9.4 PROC GLIMMIX procedure (SAS Institute Inc. 2011) was used to fit Poisson pseudo-maximum likelihood models (Santos Silva and Tenreyro 2006) for each carbon pool—duff, litter, and small, medium and large FWM. The response variable was carbon mass per pool in Mg ha^{-1}. Consumption factors (the proportion of carbon mass consumed in the fire) are multiplicative, allowing us to use the estimates from the log-linked models (Appendix D). The models were fit with fire severity, a factor with four levels, as the explanatory variable. If there were no significant differences among the remotely-sensed severity classes, the models were re-run with all severity classes combined into a burn indicator variable. To account for the plot matching, a random effect for the matched pairs was included. Fine forest floor fuels can accumulate quickly post-fire, making it necessary to examine the effects of using different years since fire (YSF) limits for selecting the data used in the model development. Models were run again after removing plots where YSF was equal to two, thus only including measurements that were taken within one year post-fire.

3.3 Results

Non-woody forest floor pools—duff and litter—had more pre- and post-fire carbon than the fine woody pools (FWM of all sizes) (Table 3-1; Figure 3-4). The mean and median values show that duff and large FWM carbon were substantially lower for burned than unburned plots, regardless of severity (Table 3-1). This was true in every other pool as well, but to a lesser degree (Table 3-1). For every pool excluding large FWM, the median carbon mass decreased with increasing fire severity. The data have a right skew in all carbon pools, suggesting many plots with low carbon mass with a few plots with high carbon mass values, which is demonstrated by the number of plots with 0 Mg ha^{-1} of carbon (Table 3-1).

Table 3-1. Summary statistics for carbon mass (Mg ha^{-1}) for each carbon pool, SD - standard deviation. The number of 0s column shows a count of the number of plots with no carbon mass.

	Severity	n	Median	Mean (SD)	# of 0s
Duff	Control	128	6.17	7.94 (7.27)	7
	Low	28	0.97	2.37 (3.92)	7
	Moderate	44	0.69	2.16 (3.33)	12
	High	56	0.51	2.95 (5.95)	20
Litter	Control	128	3.77	4.42 (3.51)	0
	Low	28	1.55	2.12 (1.97)	0
	Moderate	44	1.09	1.84 (1.95)	3
	High	56	0.69	1.52 (1.89)	7
Small FWM	Control	134	0.08	0.13 (0.15)	2
	Low	28	0.03	0.05 (0.05)	0
	Moderate	47	0.03	0.05 (0.07)	7
	High	59	0.02	0.04 (0.03)	4
Med FWM	Control	134	0.64	0.75 (0.53)	5
	Low	28	0.35	0.44 (0.45)	2
	Moderate	47	0.28	0.39 (0.41)	5
	High	59	0.15	0.26 (0.26)	10
Large FWM	Control	134	2.06	2.77 (2.68)	12
	Low	28	0.79	1.06 (1.28)	8
	Moderate	47	0.45	0.8 (0.98)	18
	High	59	0.48	0.9 (1.18)	18

3.3.1 Matching efficiency

Based on the standardized differences of covariate means before and after matching (Figure 3-2), covariance balance between burned and unburned plots was achieved and thus the matching process was considered to be successful. All standardized means after the matching were below 0.25, a cut-off that should be achieved as suggested by Stuart and Green (2008). The largest change in the standardized difference of the mean was observed for latitude (Figure 3-2). This was one of the major goals for choosing a spatial matching approach. The standardized differences in means were also clearly reduced in the remaining five covariates with large changes in standardized differences for the three temperature variables and smaller changes for elevation and annual precipitation (Figure 3-2).

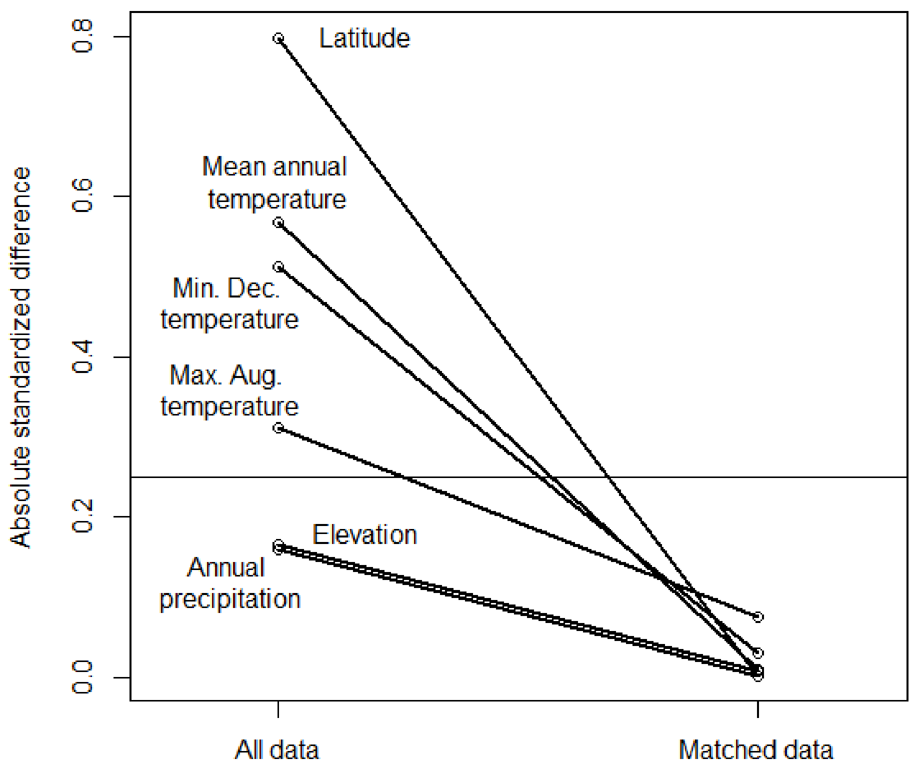

Figure 3-2. Plot of standardized difference of means of six covariates before and after matching

3.3.2 Consumption gradients across fire severity classes

For all carbon pools, there were no significant differences among consumption factors in different remotely-sensed fire severity classes at the 0.05 level (Table 3-2). Although not statistically significant, general trends suggest that carbon consumption increases as fire severity increases, with the exception of large FWM, where consumption was highest in moderate-severity and lowest in low-severity plots (Table 3-2). In all carbon pools excluding large FWM, the largest differences between consumption factors were between moderate- and high-severity; consumption factors at moderate- and low-severity were typically more similar to each other (Table 3-2). Duff consumption varied the least between fire severity classes, with a difference in consumption factors of 0.03 between low- and high-severity. Medium FWM is the pool with the largest differences between consumption in low- and high-severity, ranging from 0.41 at low-severity to 0.65 at high-severity (Table 3-2).

Table 3-2. Consumption factors (CF; proportion of carbon consumed) and their 95% confidence intervals (95% CI) by carbon pool and severity class.

	Severity	CF	95% CI
Duff	Low	0.66	0.40 - 0.75
	Moderate	0.68	0.45 - 0.73
	High	0.69	0.64 - 0.81
Litter	Low	0.53	0.31 - 0.67
	Moderate	0.54	0.36 - 0.67
	High	0.68	0.53 - 0.78
Small FWM	Low	0.6	0.40 - 0.74
	Moderate	0.62	0.45 - 0.73
	High	0.74	0.64 - 0.81
Med FWM	Low	0.4	0.13 - 0.60
	Moderate	0.47	0.27 - 0.62
	High	0.65	0.54 - 0.74
Large FWM	Low	0.61	0.38 - 0.75
	Moderate	0.73	0.59 - 0.82
	High	0.66	0.49 - 0.77

3.3.3 Consumption factors for burned plots

There were no significant differences in consumption factors among fire severity classes for any pool, therefore I reported average consumption factors across all severity classes (Table 3-3). Duff and large FWM had the highest consumption factors, where 68% of the carbon in each pool was consumed in the fires. Medium FWM was the least consumed – only 54% was consumed (Table 3-3). The estimated consumption factor for litter (0.60) was smaller than for duff and small and large FWM (Table 3-3).

Table 3-3. Consumption factors (CF - proportion of pre-fire carbon mass consumed) for carbon pools comparing burned with unburned.

	n	CF	95% CI
Duff	256	0.68	0.54 - 0.78
Litter	256	0.6	0.49 - 0.69
Small FWM	268	0.67	0.59 - 0.73
Medium FWM	268	0.54	0.43 - 0.63
Large FWM	268	0.68	0.57 - 0.75

3.3.4 Comparison of time since fire values for estimating consumption factors

Overall, there is a general trend that is similar in each carbon pool over time (Figures 3-3 & 3-4). In every pool, there is a decrease in carbon mass from unburned plots to plots measured immediately post-fire, which continues to decrease within the next year post-fire (Figures 3-3 & 3-4) and then increases slightly past one year post-fire. However, when the severity classes are examined separately there are apparent differences in post-fire carbon dynamics (Figures 3-5 & 3-6). The impacts of low-severity fires on post-fire carbon accumulation varied between carbon pools, which may be attributed to the small amount of sample size in low-severity fires. Carbon mass of duff decreased during the period of interest (≤2 YSF) after a low-severity fire while carbon mass of FWM increased during that time frame (Figures 3-5 & 3-6). Post-fire carbon dynamics after moderate- and high-severity fire were more similar among pools. Both duff and litter increased in the first two years after moderate-severity fire. However, in high-severity fire, carbon mass in both pools stayed mostly similar with a slight decrease in the first year after fire (Figure 3-5). Small and large FWM appeared to experience minimal changes in carbon mass after moderate-severity fire (Figure 3-6). Post-fire carbon dynamics in medium FWM were different from the rest of the carbon pools (Figure 3-6), where carbon mass increased after fire in low- and high-severity, but decreased with time after moderate-severity fire.

I compared results between models that were developed using plots that burned ≤2 years prior to measurement (Tables 3-2 & 3-3) and models that were based on data that only included plots that burned within one year prior to measurement (Appendix C). In all carbon pools except for duff, consumption factors increased (by 0.02 for small FWM, 0.07 for litter, and 0.09 for large FWM) when only plots that burned within one calendar year prior to measurement were included in the models (Appendix Table C-1). For duff the estimate of the consumption factor did not change for the reduced data set, however variability of the estimate increased (Table 3-1; Appendix Table C-2). The estimated litter consumption factor was higher (0.67, 95% CI: 0.52 – 0.78; Appendix Table C-2) when only plots that burned within one year prior to measurement were included in the analysis. For medium FWM, the consumption factors for low- and high-severity were significantly different from each other (at the 0.05 level) when examining plots that burned within one year of measurement (Appendix Table C-1). This difference was no longer present when plots that burned between one and two years prior to measurement were added (Table 3-1).

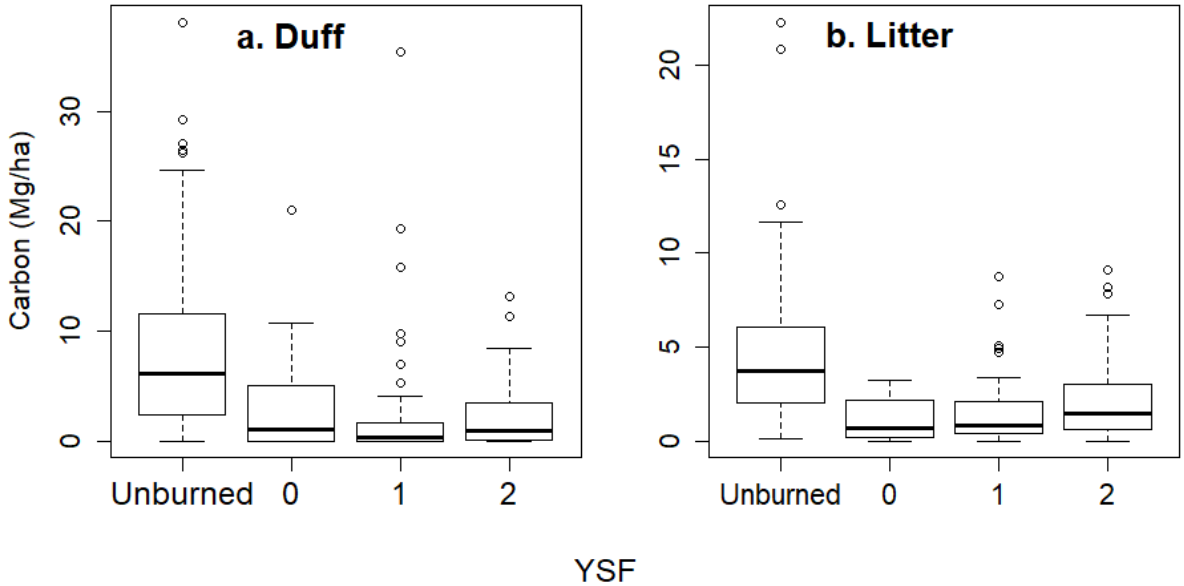

Figure 3-3. Duff and litter carbon mass (Mg ha^{-1}) by years since fire (YSF) across all severity classes.

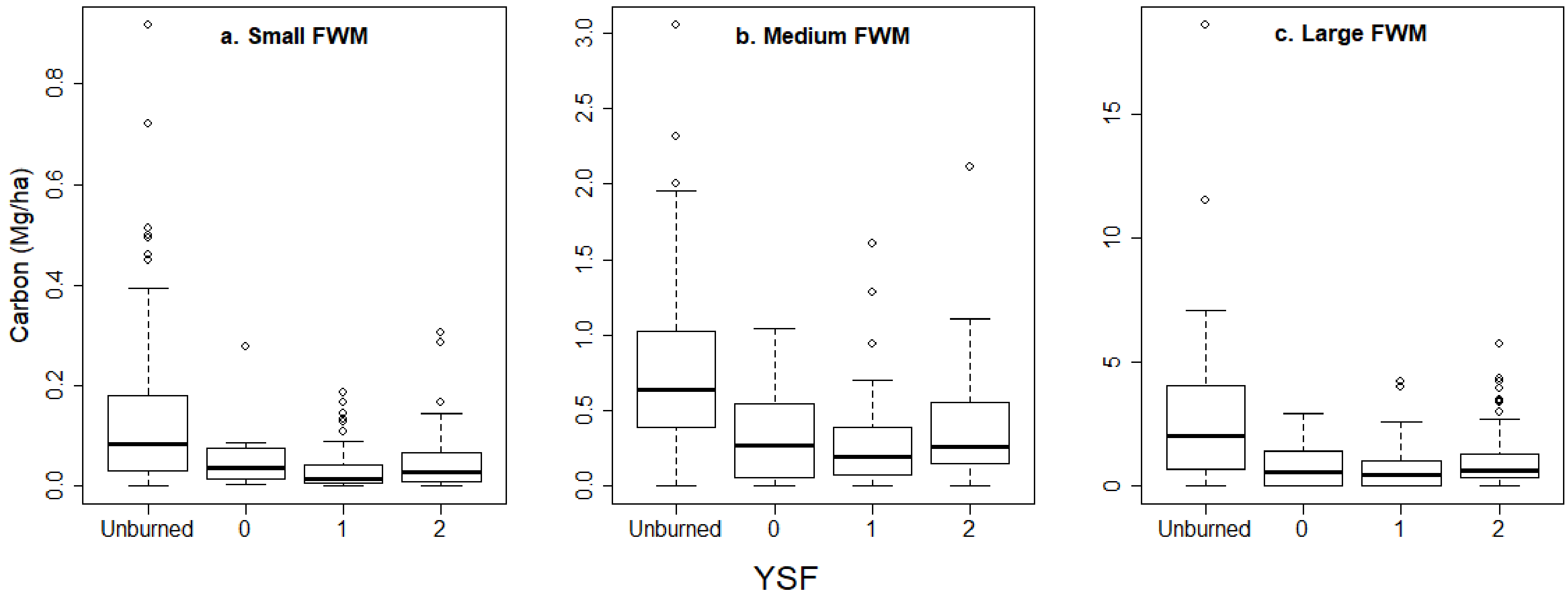

Figure 3-4. FWM carbon mass (Mg ha^{-1}) by years since fire (YSF) across all severity levels.

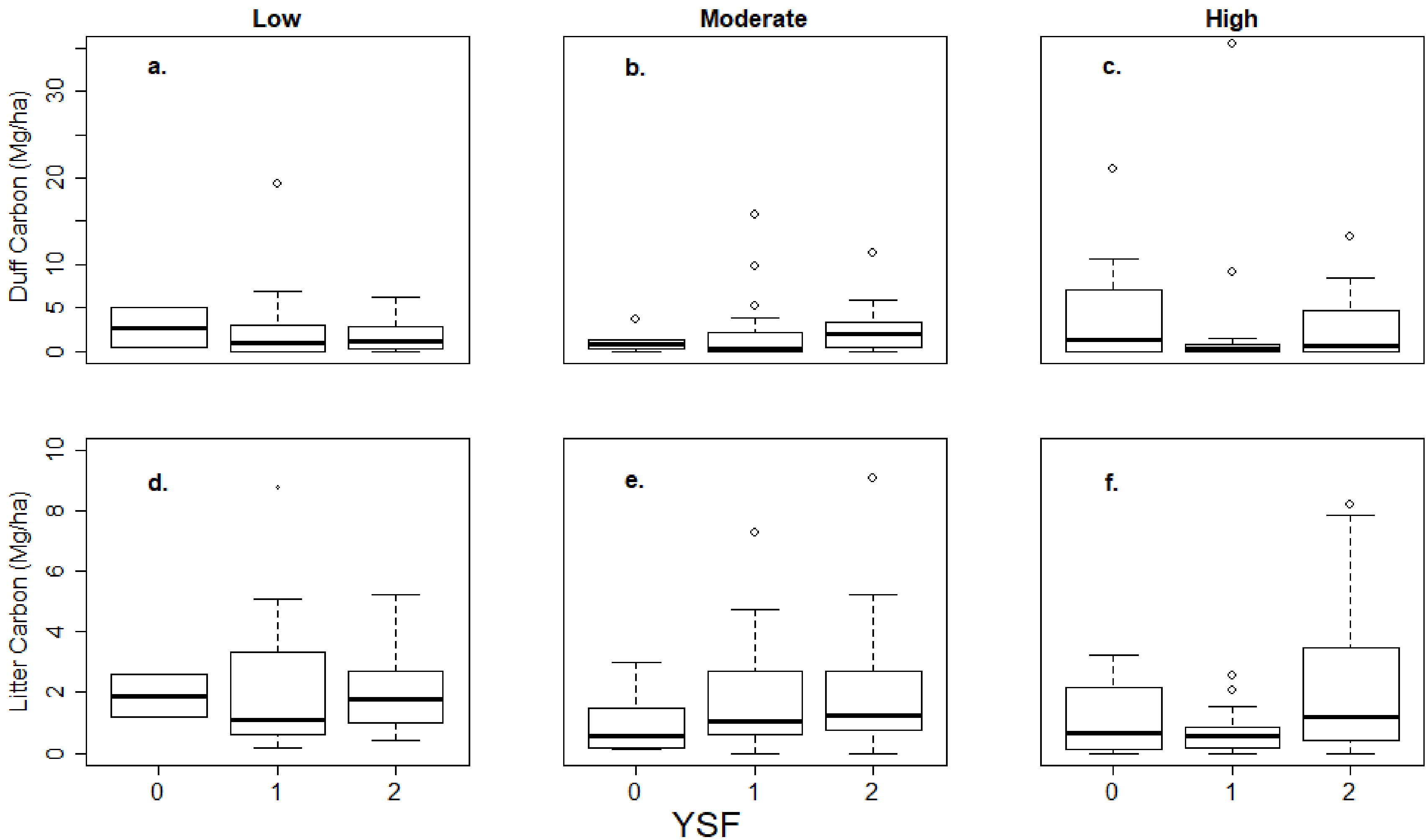

Figure 3-5. Duff and litter carbon mass (Mg ha^{-1}) by YSF and fire severity class.

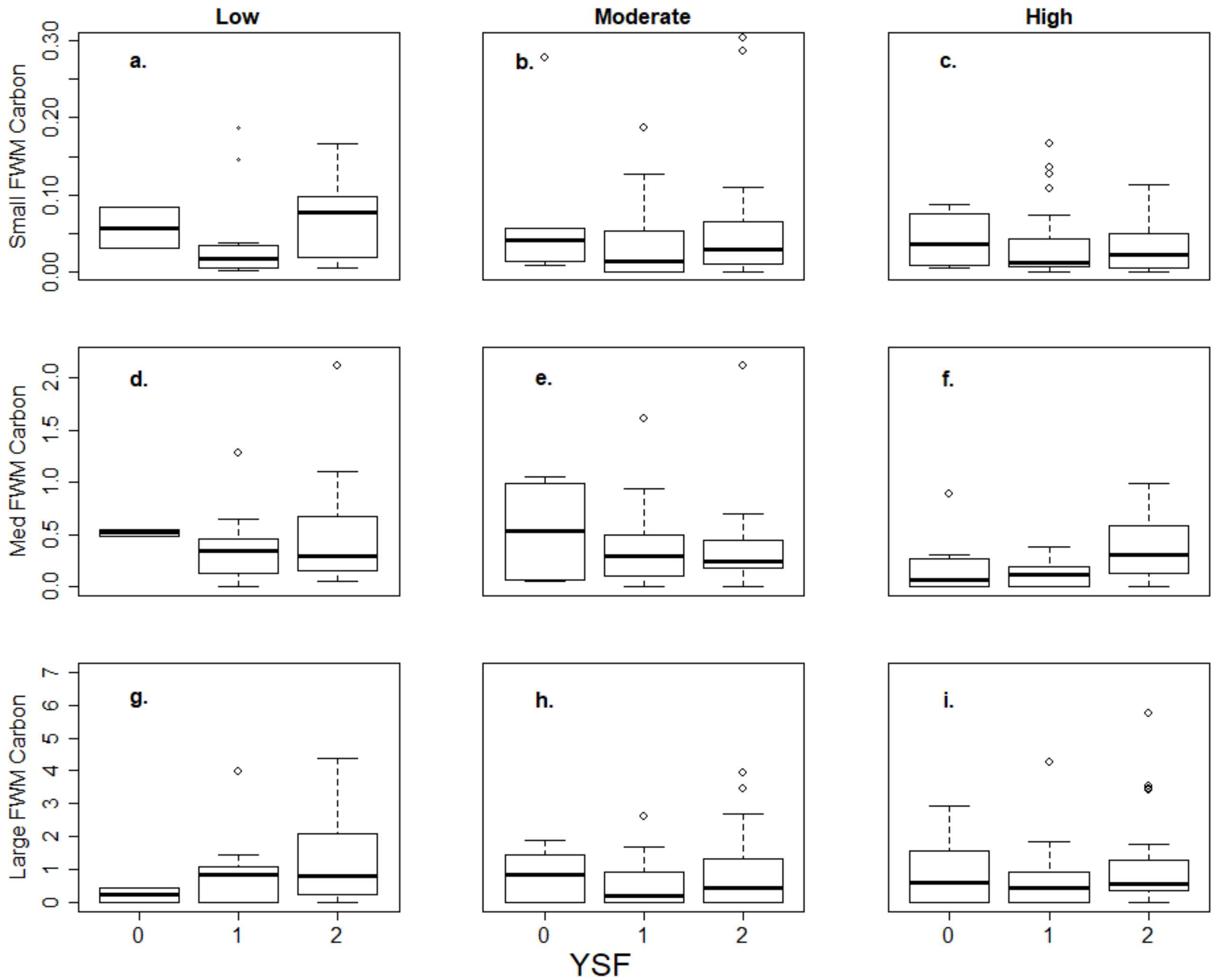

Figure 3-6. FWM carbon mass (Mg ha^{-1}) by YSF and severity class.

3.4 Discussion

3.4.1 Comparison of consumption factors

The regional litter consumption factors generated in this study are similar to results from prescribed burns in the region. Prichard et al. (2017a) found that mean litter consumption in dry pine sites in the western PNW was approximately 60%, equivalent to the estimate in this study. A study using averages from three prescribed burns in the southwestern US found that litter consumption factors were much higher (81%, standard deviation (SD): 8.9) (van Leeuwen et al. 2014), however differences in ecosystems, such as overstorey species or climate, could explain the gap between the estimates by van Leeuwen et al. (2014) and this study. In contrast, the estimated duff consumption factor of 68% reported in this study is higher than consumption factors previously reported for prescribed burns that ranged from 37% (SD: 23.6) (Prichard et al. 2017a) to 60% (van Leeuwen et al. 2014). Depending on the objectives of the prescribed burn, the managers may aim for minimal duff consumption to protect the forest soils and tree roots that can be damaged in fires (Tiedemmann et al. 2000), which would explain lower duff consumption factors for prescribed burns. Ocular estimates of duff consumption from one large wildfire in the region (Campbell et al. 2007; 54% and 51% for low- and moderate-severity) are also lower than our estimated duff consumption factors of 66 and 68%, however they are within the 95% confidence intervals of this study. The litter consumption factors generated in this study were significantly lower than the numbers presented by Campbell et al. (2007), for all severity classes. Campbell et al. (2007) used post-fire ocular estimates of pre-fire duff and litter to generate consumption factors, which could explain why the consumption factors in this study, which were based on unburned control plots varied substantially from their estimates. Burned plots may have had minimal duff or litter carbon pre-fire, which would be difficult to discern from plots in which duff and litter carbon was present pre-fire but fully consumed. I found that seven unburned plots used in this study had no duff carbon.

Prescribed burns in the PNW region typically consumed more small and medium FWM carbon than the wildfires examined in this study. In drier pine sites, a mean of 83% of the small and medium FWM was consumed (Prichard et al. 2017a). van Leeuwen et al. (2014) generated small and medium FWM consumption factors using data from prescribed fires in California. They found that of 87 and 79% of small and medium FWM respectively was consumed, in contrast with our consumption factors of 67% and 54%, respectively. Our results for large FWM (CF: 68%) align more closely with estimates from prescribed fires, compared to 69% from Prichard et al. (2017a) and 73% from van Leeuwen et al. (2014). Consumption factors previously presented for wildfires were more similar to the results of this study for small and large FWM. Small FWM consumption factors tend to increase as fire severity increases, ranging from 61% at low-severity to 78% at high-severity (Campbell et al. 2007), which was corroborated by the results of this study (small FWM consumption 60 – 74%). For large FWM, consumption factors at low- (61%) and moderate- (73%) severity were similar to Campbell et al. (2007) (67 and 73% respectively). However, our study estimated a consumption factor of 66% (CI: 57 – 75%) for high-severity, which is lower than the estimate presented by Campbell et al. (2007) of 79%. Campbell et al. (2007) used consumption factors from one severe fire, which could be more extreme than the average of high-severity fires across the landscape. Very high severity fires often consume all fine fuels, which may have led to values much higher than the regional consumption factors.

3.4.2 Differences in consumption among fire severity classes

In this study, there were no significant differences among severity classes of consumption factor estimates in any carbon pool, suggesting that there is a high amount of variability in consumption factors within fire severity classes. The similarities in consumption factors between severity classes could be connected to the use of remotely-sensed fire severity classifications, especially given that this study focuses solely on forest floor carbon pools. Forest floor consumption is highly variable, and wildfires often do not consume the forest floor in a uniform manner (Jain et al. 2012). Landsat-derived fire severity classifications use differences in vegetation characteristics to compute fire severity (Eidenshink et al. 2007), but they do not fully capture the changes in the forest floor and organic soil layers of the forest (Alonzo et al. 2017). The lack of significant differences between consumption factors at different fire severity classes could be attributed to a disconnect between crown fire severity and the physical impact of fire on forest floor fuels. Using Landsat imagery for fire severity classification without any additional information could lead to underestimating carbon stock changes after fire (Alonzo et al. 2017), which suggests a need for a ground- or soil- based fire severity classification system. I found that consumption factors for low- and moderate-severity fires were more similar to each other than to high-severity consumption factors in almost every pool. These similarities could be due to the greater uncertainty associated with remotely-sensed moderate-severity fire (Miller et al. 2009), leading to overlap between low- and moderate-severity plots.

3.4.3 Impacts of time since fire on estimated consumption

Finer forest fuels like litter and smaller FWM have been found to start accumulating within one year after fire (Dunn and Bailey 2015; Eskelson and Monleon 2018), which could imply that I am underestimating consumption factors by including plots that were measured up to two years after fire. In contrast with literature on forest floor consumption (Campbell et al. 2007; van Leeuwen et al. 2014; Prichard et al. 2017a), I found that the estimated consumption factor for litter was lower than the estimated value for duff. As the duff layer is found beneath the litter layer, it is unlikely that duff would be consumed while litter materials remain. The low consumption factor estimated for litter could be attributed to the inclusion of plots that burned between one and two calendar years prior to measurement, thus allowing time for understorey foliage and dead foliage from fire killed trees to become litter input. Litter accumulation has been found to peak in the first year after a fire (Stalling et al. 2017). The consumption factor for litter was more similar to post-wildfire estimates computed by Campbell et al. (2007) when plots that burned 1-2 years prior to measurement were removed from the analysis, which supports my hypothesis that litter had begun to accumulate post-fire. Unlike litter, the estimated consumption factor for duff did not change when the YSF criteria were restricted to only include plots that burned 0-1 years prior to measurement, which is consistent with findings from Eskelson and Monleon (2018), where duff carbon did not begin to increase for at least nine years post-fire. Dunn and Bailey (2015) also found that duff had not accumulated on several of their plots within five years post-fire. Medium FWM (small branches between 0.62 and 2.4cm in diameter) may accumulate rapidly post-fire (Dunn and Bailey 2015), which is consistent with the fact that the estimated consumption factor for medium FWM was lower than the consumption factors of any other pool in this study. The fires could have consumed the finer twigs both on the ground and on tree branches, leaving small branches to become medium FWM input post-fire. The low consumption factors for FWM could also be due to partial consumption of large FWM, leaving behind fragments that would be classified as medium FWM.

3.5 Conclusions

Significant proportions of each carbon pool were consumed by fire; however, these consumption factors did not differ significantly among fire severity classes defined using remote sensing. The results of this study show that the consumption of forest floor fuels is significant, even in low- and moderate-severity fires. Issues with capturing changes in surface fuels using remotely-sensed fire severity classifications could explain the lack of significant differences among severity classes, suggesting the need for a ground-based fire severity classification that would better explain the variability in forest floor consumption in mixed-severity fires. Time since fire also plays an important role in the consumption factors of surface fuels, due to rapid post-fire accumulation. The presented results provide regional estimates of consumption factors for forest floor carbon pools, which are applicable to all large wildfires across the states of Oregon and Washington.

Chapter 4: Conclusion

4.1 Overall conclusions

This thesis contributes new understanding of fire effects on forest surface carbon in PNW forests. In the Boulder Creek fire case study there were no significant differences in total post-fire surface fuel carbon mass between burned and unburned plots one year after the fire. While it is clear that the Boulder Creek fire impacted stand carbon, the lack of differences in carbon between burned and unburned plots is likely explained by snag conversion to downed fuels during or post-fire, resulting in abundant fuels on burned plots. Fine fuels were found to be consumed in the fire; however, they only make up a small proportion of total surface fuels and carbon values. In the PNW regional study, I found that there was significant consumption of forest floor pools during the fire; however, 30 to 40% of each pool remained unconsumed. Carbon consumption or post-fire carbon mass by pool rarely varied among fire severity classes, which could be partially attributed to the use of crown-based fire severity classes that do not properly capture changes in ground fuels. Overall, I found that using remotely-sensed fire severity classifications do not adequately explain fire severity on the forest floor, which suggests a need for ground-based measures of fire severity to assess fire effects on forest floor fuel and carbon pools.

4.2 Comparison of scopes of inference

Forestry research aims to expand knowledge and understanding of biological processes (von Gadow and Kleinn 2013). Forest management is influenced by two types of scientific research: narrowly focused studies and observations of large systems (Walters and Holling 1990). In the PNW regional study, I used national inventory data collected across Oregon and

Washington, USA, along with spatial plot matching to obtain estimates of carbon consumption factors that allow inference of the study results to the sampled population (Ramsey and Schafer 2013) of fires and forest types in the US PNW region (i.e., Oregon and Washington). The consumption factors presented in this study represent consumption factors for large fires across all severities in the forest types examined. Other studies examining carbon consumption in the region often use single fires (Campbell et al. 2007) or prescribed burns (van Leeuwen et al. 2014; Prichard et al. 2017a), which restricts the inferences that can be made using the study results to similar fires in similar forest types in the local area. The consumption factors computed in the PNW regional study can be used for regional post-fire carbon accounting studies and they can aid in predicting carbon emissions from forest wildfires in the region. The consumption factors are important for understanding how PNW landscapes respond to fire regimes in PNW forests that are projected to shift towards being more severe under climate change (Cansler and McKenzie 2014). This leads into a need to continue studying individual fires as they occur. Our results from the PNW regional study provide forest floor consumption estimates related based on the current state of forests and fires in the PNW, allowing us insights into the effects of current fire regimes of the region. However, as the current fire regimes change, we must continually study new fires, especially in areas that may not have burned frequently in the past. While we do not benefit from the same scope of inference we achieve using regional studies, case studies allow for closer examination of post-fire impacts, especially for fires that are atypical to the region, such as the Boulder Creek fire examined in Chapter 2. Post-fire dynamics in specific wildfires may not be representative of other fires in the region, therefore results from case studies of single fires should only be carefully compared to other fires and studies. However, case studies can establish baseline measurements for future studies, and they can give us valuable information on single extreme events. Information about extreme fires like the Boulder Creek fire cannot be found in current regional datasets, as more frequent, less severe fires are more common and thus have more representation in the samples. If these severe fires become more common, as predicted, it will become more important to understand how they affect forest carbon dynamics. Both case studies and regional studies provide important information that is crucial in accounting for fire impacts on carbon now and in the future.

4.3 Limitations and future work

Landsat imagery was used to stratify wildfires into severity classifications in both chapters of this thesis. This technology is extremely important in the study of forest fires (Cohen et al. 2017). Landsat-derived fire severity classifications, like NBR and dNBR, use changes in vegetation to calculate severity levels (Eidenshink et al. 2007), providing information that can be used to aid in estimating wildfire carbon consumption and emissions (Veraverbeke and Hook 2013). However, remotely-sensed fire severity classifications can be problematic for low magnitude disturbances (Cohen et al. 2017; Cohen et al. 2018) such as low-severity fire, where it is difficult to distinguish slight changes in vegetation cover from normal variations in spectral indices (Cohen et al. 2018). In Chapter 3, I found that carbon masses in plots that burned at low- and moderate-severity were more similar to each other than they were to plots that burned at high-severity. Difficulties in discerning moderate-severity using Landsat data (Miller et al. 2009) may have led to overlap or misclassifications between low- and moderate-severity, explaining the similarities between the two severity classes. Not all fires are crown fires, therefore Landsat-derived fire severity classifications, which rely on changes in vegetation cover (Eidenshink et al. 2007), may not capture all of the changes in understorey vegetation and surface fuels (Alonzo et al. 2017), which makes it difficult to discern the severity of the fire on the forest floor. In the Boulder Creek fire study, I found that one plot was classified as unburned when in reality it had been impacted by a low-severity surface fire that was not severe enough to cause substantial overstorey mortality. The uncertainties associated with using remotely-sensed fire severity classifications that were observed in both chapters demonstrate the need to develop ground-based fire severity classifications that can be used alongside or in the place of Landsat imagery, especially when examining forest floor carbon pools.

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Appendices

Appendix A Chapter 2 Additional Results

During the data collection portion of the Boulder Creek fire case study, I found that seven out of 37 plots had been clearcut logged within seven years prior to the fire. I compared the results of the models using a dataset that excluded plots that had been logged (Section A.1) and a dataset that included all measured plots regardless of logging status (Section A.2) in order to examine for possible differences in forest floor carbon mass due to logging.

The results differed slightly when plots that experienced logging were separated from the analysis. Overall, significant differences did not change drastically. I found that removing the plots that had experienced logging did not change the results for total surface carbon, where there were no significant differences between any remotely-sensed fire severity classes ($p > 0.07$). For CWM, only examining plots that had not been logged showed a suggestive, but inconclusive difference between moderate and high severity ($p = 0.037$) that was not seen when logged plots were included. For SWM, the difference between unburned and low-severity was significant when logged plots were separated from the analysis, but with a p-value of 0.04, this was only suggestive, but inconclusive. There were no changes in FWM results between both datasets, while duff and litter saw minor changes in significant differences between severity classes.

A.1 Results of analysis without plots that experienced logging.

Appendix Table A-1. Parameter estimates generated without plots that experienced logging.

a. Total Surface

Reference Severity	Comparison	Estimate	p
2	1	0.02306	0.951
2	3	0.7371	0.0729
2	4	-0.00773	0.9844
3	1	-0.7141	0.1207
3	4	-0.7449	0.1186
4	1	0.03079	0.945
1	234	-0.139	0.7054

b. CWM

Reference Severity	Comparison	Estimate	p
2	1	0.2294	0.6476
2	3	1.2623	0.0155
2	4	0.03678	0.9348
3	1	-1.0329	0.0966
3	4	-1.2255	0.0371
4	1	0.1926	0.7362

c. SWM

Reference Severity	Comparison	Estimate	p
2	1	-0.9732	0.0401
2	3	-0.8368	0.0926
2	4	-0.5459	0.2015
3	1	-0.1364	0.7779
3	4	0.2909	0.5208
4	1	-0.4273	0.3136

d. FWM

Reference Severity	Comparison	Estimate	p
2	1	0.1556	0.6945
2	3	0.1127	0.7777
2	4	0.6857	0.1318
3	1	0.0429	0.877
3	4	0.5731	0.1046
4	1	-0.5302	0.1278
1	234	-0.1392	0.5929

e. Litter

Reference Severity	Comparison	Estimate	p
2	1	-0.1088	0.8293
2	3	0.7121	0.2356
2	4	3.3932	<.0001
3	1	-0.8209	0.052
3	4	2.681	0.0002
4	1	-3.5019	<.0001

f. Duff

Reference Severity	Comparison	Estimate	p
2	1	-1.7069	0.0013
2	3	0.6499	0.2484
2	4	2.0109	0.0005
3	1	-2.3569	<.0001
3	4	1.361	0.0133
4	1	-3.7178	<.0001

A.2 Results of analysis with plots that experienced logging

Appendix Table A-2. Parameter estimates generated including plots that had been logged.

a. Total Surface

Reference Severity	Comparison	Estimate	p
2	1	-0.06785	0.8374
2	3	0.5186	0.0592
2	4	-0.09863	0.7799
3	1	-0.5865	0.1308
3	4	-0.6172	0.1302
4	1	0.03079	0.9449
1	234	-0.4283	0.1831

b. CWM

Reference Severity	Comparison	Estimate	p
2	1	0.0451	0.9229
2	3	0.735	0.0365
2	4	-0.1475	0.7197
3	1	-0.6899	0.1913
3	4	-0.8825	0.0685
4	1	0.1926	0.7356

c. SWM

Reference Severity	Comparison	Estimate	p
2	1	-0.4048	0.3272
2	3	-0.1032	0.8078
2	4	0.02248	0.9518
3	1	-0.3016	0.5188
3	4	0.1257	0.7716
4	1	-0.4273	0.3116
1	234	-0.3803	0.296

d. FWM

Reference Severity	Comparison	Estimate	p
2	1	-0.05974	0.8484
2	3	0.08489	0.7983
2	4	0.4704	0.2126
3	1	-0.1446	0.6236
3	4	0.3855	0.2854
4	1	-0.5302	0.1253
1	234	-0.2213	0.3674

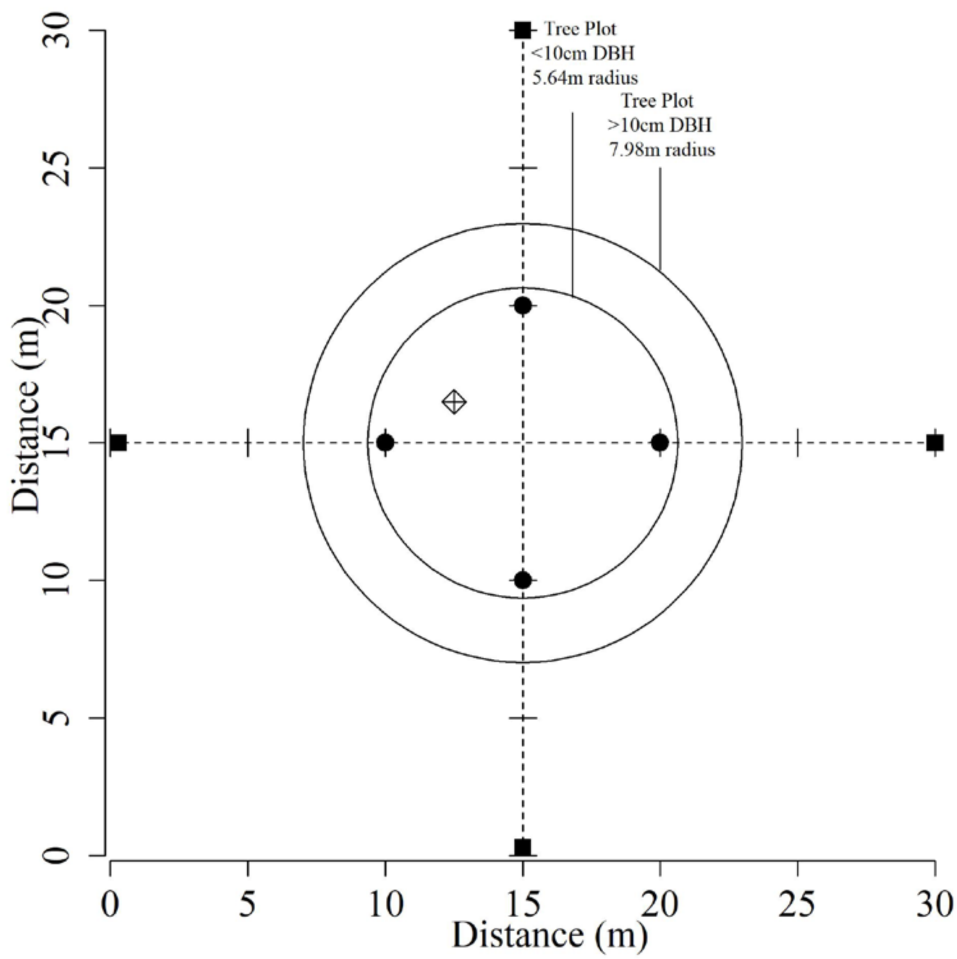
e. Litter

Reference Severity	Comparison	Estimate	p
2	1	-0.6524	0.2025
2	3	0.4562	0.469
2	4	2.8495	0.0002
3	1	-1.1086	0.0196
3	4	2.3933	0.0007
4	1	-3.5019	<.0001

f. Duff

Reference Severity	Comparison	Estimate	p
2	1	-1.2737	0.011
2	3	1.3708	0.0257
2	4	2.4441	<.0001
3	1	-2.6445	<.0001
3	4	1.0733	0.0602
4	1	-3.7178	<.0001

A.3 Standing tree carbon



Appendix Figure A-1. Location and size of tree plots in relation to transects.

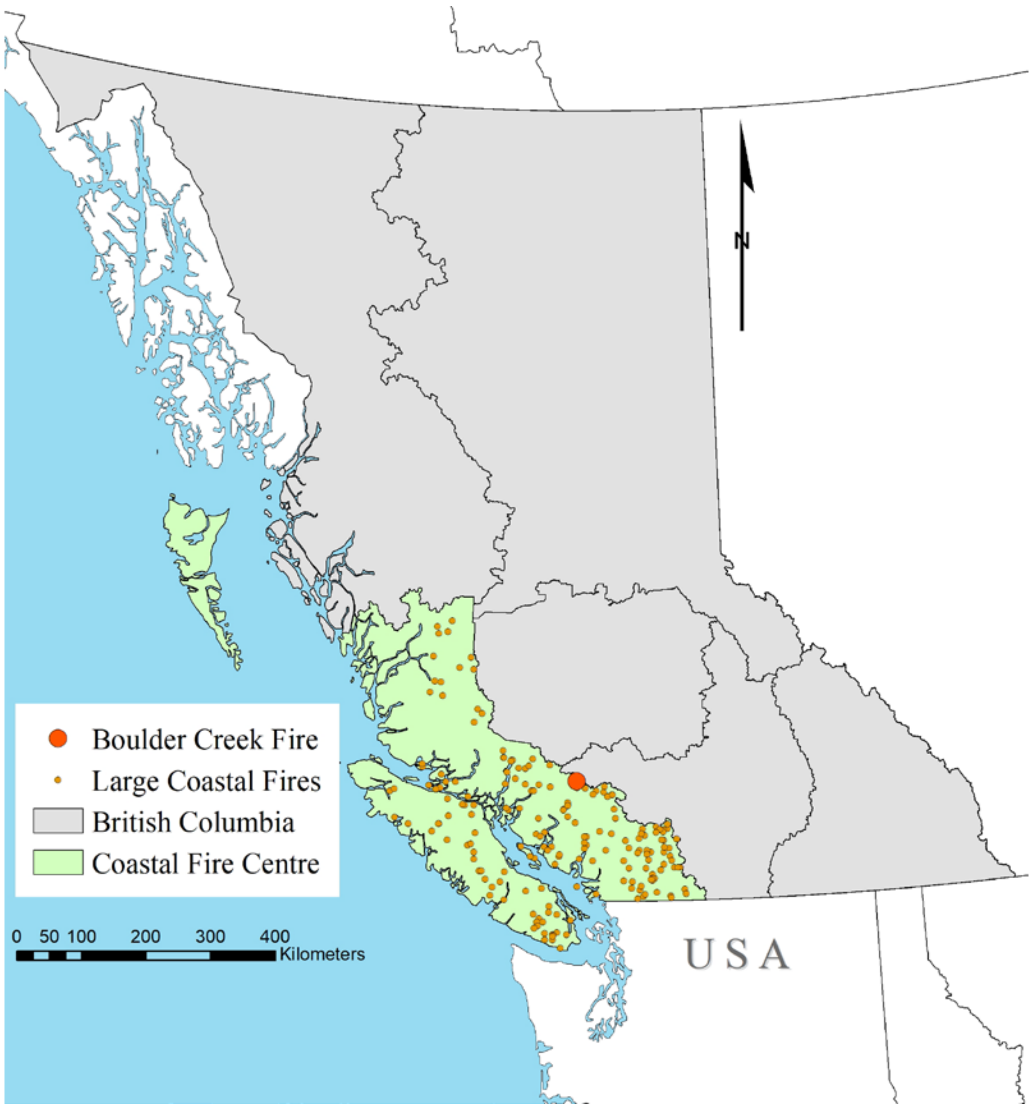
Trees with diameter at breast height (DBH) ≥ 10cm were measured in a 200m^2 circular plot (radius = 7.98m). We measured the height of the first tree per species in each of six DBH classes (in cm: 10.0 – 24.9, 25.0 – 39.9, 40.0 – 49.9, 50.0 – 64.9, 65.0 – 74.9, > 75). Trees with DBH < 10cm were measured in a 100m^2 circular plot (radius = 5.64m). Where the trees were burned too severely to determine species, the height of one tree in each DBH class was measured. Each tree was recorded as living if any green foliage was observed from the ground. For all trees and snags (standing dead trees), a decay class was assigned ranging from 1 (alive, healthy) to 7 (dead, bark and twigs absent, heartwood decay, stem broken) (Maser et al. 1979).

Appendix Table A-3. Mean and (standard deviation) standing tree carbon mass (Mg ha^{-1}) by remotely-sensed fire severity class.

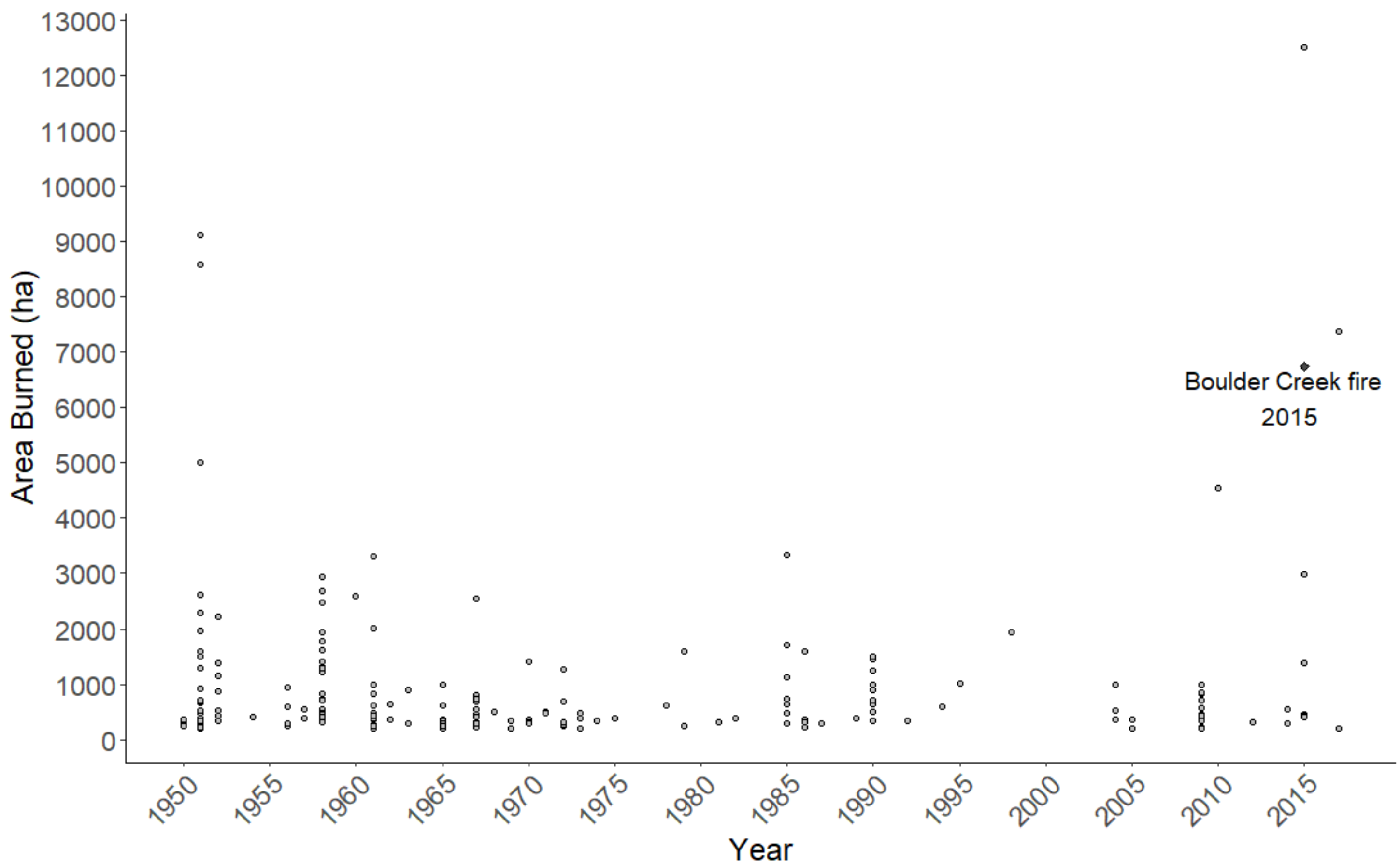
Severity	n	Live Trees	Snags
Unburned	10	31.24 (33.02)	1.52 (2.75)
Low	10	27.53 (55.30)	18.33 (33.49)
Moderate	8	23.11 (32.20)	23.11 (32.19)
High	9	0 (0)	33.97 (45.77)

Appendix B Area burned in coastal fire zone since 1950

The Boulder Creek fire was a relatively large fire in the coastal fire zone of British Columbia. The coastal fire zone covers 16.5 million hectares of forest. It encompasses all of Vancouver Island, and the coast of the mainland, extending approximately 200 km inland. The region stretches from the BC-Washington border up approximately 500 km, and it includes 75% of BCs population.



Appendix Figure B-1. Location of large coastal fires in British Columbia 1950 – 2017. Burn area and fire centre data provided by the BC Wildfire Service (2018a, ©2018b).



Appendix Figure B-2. Area burned (ha) in large fires per year in the coastal fire region of British Columbia, 1950 – 2017. Burn area data provided by the BC Wildfire Service (2018a).

Appendix C Comparison of YSF values

In the PNW regional study, plots were considered burned if they had been measured within two years post-fire, however the nature of the finer fuels studied suggests that they could start to accumulate in the first year post-fire. I performed the same consumption factor estimations using only plots that had burned within one calendar year prior to measurement, to compare results and examine for possible post-fire accumulation.

Appendix Table C-1. Estimated consumption factors (CF - proportion of carbon consumed) per carbon pool by severity using plots that burned within one year prior to measurement.

Pool	Severity	n	CF	95% CI
Duff	Low	16	0.61	(-0.07 - 0.85)
	Moderate	25	0.74*	(0.39 - 0.89)
	High	28	0.64+	(0.10 - 0.85)
Litter	Low	16	0.46+	(0.004 - 0.71)
	Moderate	25	0.59*	(0.34 - 0.74)
	High	28	0.83***	(0.71 - 0.89)
Small FWM	Low	16	0.70**	(0.42 - 0.85)
	Moderate	26	0.67***	(0.44 - 0.81)
	High	30	0.71***	(0.57 - 0.80)
Med FWM	Low	16	0.51*[a]	(0.25 - 0.67)
	Moderate	26	0.47*[a]	(0.21 - 0.65)
	High	30	0.81***[b]	(0.70 - 0.88)
Large FWM	Low	16	0.71***	(0.48 - 0.83)
	Moderate	26	0.81***	(0.63 - 0.89)
	High	30	0.76***	(0.57 - 0.86)

**** denotes significance at the <0.0001 level*

*** denotes significance at the 0.001 level*

** denotes significance at the 0.01 level*

+ denotes significance at the 0.05 level

Letters denote significance groups within carbon pools. For pools with no designations, no significant differences were found

Appendix Table C-2. Estimated consumption factors for each carbon pool average across fire severity classes, if there were no significant differences between fire severity classes.

Pool	n	CF	95% CI
Duff	69	0.66***	(0.41 - 0.81)
Litter	69	0.67***	(0.52 - 0.78)
Small FWM	72	0.69***	(0.59 - 0.77)
Medium FWM	72	N/A$^{\pm}$	N/A$^{\pm}$
Large FWM	72	0.77***	(0.66 - 0.84)

***** denotes significance at the <0.0001 level*

$^{\pm}$ *Due to significant differences between fire severity classes, the severity classes were not combined into one burn indicator variable.*

Consumption factors estimated using plots that burned less than one year prior to measurement (Table C-1, C-2) are almost all higher than consumption factors estimated using plots that burned up to two years prior to measurement (Table 3-2, 3-3), suggesting post-fire fuel accumulation may have occurred.

Appendix D Consumption factor calculations

For each carbon pool (duff, litter, FWM, CWM, total surface fuels), I fit a Poisson pseudo-maximum likelihood model (Santos Silva and Tenreyro 2006).

$$\log(\mu) = \beta_0 + \beta_1 burn$$

Where μ denotes mean carbon mass in Mg ha^{-1}, and burn is an indicator variable denoting the presence (burn = 1) or absence of wildfire activity (burn = 0). Our variable of interest is the proportion of pre-fire carbon remaining post-fire (consumption factor – CF), which is the ratio between log(μ|burn = 1) and log(μ|burn = 0).

For unburned plots:

$$\log(\mu|burn = 0) = \beta_0 \qquad \text{(Equation D.1)}$$

For burned plots:

$$\log(\mu|burn = 1) = \beta_0 + \beta_1 \qquad \text{(Equation D.2)}$$

Therefore, it follows that:

$$\log(\mu|burn = 1) - \log(\mu|burn = 0) \qquad \text{(Equation D.3)}$$

$$= (\beta_0 + \beta_1) - \beta_0 = \beta_1$$

Based on the quotient rule of logarithms, we know that:

$$\log(\mu|burn = 1) - \log(\mu|burn = 0) = \log\left[\frac{(\mu|burn = 1)}{(\mu|burn = 0)}\right]$$

Therefore, based on Equation D.3, the log of the ratio between mean carbon mass in burned and unburned plots is equal to the estimate, β_1:

$$\log\left[\frac{(\mu|burn = 1)}{(\mu|burn = 0)}\right] = \beta_1$$

$$\frac{(\mu|burn=1)}{(\mu|burn=0)} = \exp(\beta_1) \qquad \text{(Equation D.4)}$$

Because we are interested in the amount of carbon removed as opposed to the amount remaining:

$$CF = 1 - \exp(\beta_1) \qquad \text{(Equation D.5)}$$