

## AN ABSTRACT OF THE THESIS OF

Christopher J. Dunn for the degree of Master of Science in Forest Resources presented on October 12, 2010.

Title: Coarse Woody Detritus Dynamics, Variable Decay Rates and their Contribution to Wildland Fuel Succession Following High-Severity Fire Disturbance in Dry-Mixed Conifer Forests of Oregon's Eastern Cascades.

Abstract approved:

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John Duff Bailey

Reducing future fire severity is a proposed ecological benefit of salvage logging following wildfire disturbance. Considerable debate continues over the ability of such management practices to achieve this objective given limited understanding of coarse woody detritus (CWD) dynamics, fuel bed alterations, and post-fire vegetative growth. The objective of this study was to estimate the dynamics of snags and logs in conjunction with surface fuel accumulation following high-severity fire disturbance in dry-mixed conifer forests of Oregon's eastern Cascades.

Snag dynamics (fall and breakage rates) were estimated for *Abies sp.*, *Pinus ponderosa* and *Pinus contorta* in three DBH classes of <23 cm (small), 23-41 cm (medium) and >41 cm (large). A total of 5,103 snags in thirty 0.25-ha plots were sampled at seven different fire sites, covering a 24 year chronosequence following high-severity fire disturbance. *Pinus ponderosa* and *Pinus contorta* snags had the quickest fall rates with estimated half-lives of 7-8 and 12-13 years for small and medium sized snags, respectively. Large *Pinus ponderosa* snags had an estimated half-life of 17-18 years. *Abies sp.* snags fall rates were slower, with half-life estimates of 8-9, 14-15 and 20-21 years for small, medium and large snags respectively. Breakage rates were variable but correlated with wood strength, crown and stem weight and crown position (exposure to wind).

Decomposition loss rate-constants were obtained from the same fire sites, up to seven years post-fire, by removing three cross-sections from each of sixty fire-killed *Abies* sp. snags, sixty *Pinus ponderosa* snags, and forty *Pinus ponderosa* logs. *Abies* sp. snags exhibited significant decay with an estimated decomposition loss rate-constant of  $k = 0.0149 \text{ yr}^{-1}$ . *Pinus ponderosa* snags did not exhibit significant decay, but logs did. Sapwood and heartwood decomposition loss rate-constants equaled  $k = 0.0362 \text{ yr}^{-1}$  and  $k = 0.0164 \text{ yr}^{-1}$ , respectively. These values confirm hypothesized differences in decay rates among species and between snags and logs in dry forest environments.

An empirical model was developed to link snag fall and breakage with snag and log decomposition during succession in order to estimate the contribution of fire killed biological legacies to fine and coarse woody detritus accumulation. Legacy CWD is responsible for the largest total accumulation of surface fuel as snags break and fall, but primarily in 100- and 1000-hr fuel classes. Decomposition rates increase as CWD moves from standing to downed material, reducing total CWD biomass by 30-50% in 24 years. Fine fuels are primarily derived from post-fire vegetation and steadily increase over the 24-year period. Herbaceous fuel loads peak within 2-4 years but decrease rapidly as *Ceanothus velutinus* and *Arctostaphylos patula* shrubs establish quickly and steadily increase in total biomass over 24 years. Spread rates and flame lengths in post-fire environments are primarily driven by fuels generated from new growth.

The dynamic process of snag fall and breakage, and decomposition of snags and logs, limits CWD's effect on fire spread and intensity if reburning occurs, although soil heating and total heat release can be exacerbated by the combustion of decayed logs. Salvage logging significantly reduces CWD fuels but has limited impacts on other fuel bed components. Results of this study suggest post-fire management decisions consider vegetation dynamics as well as dead wood dynamics if reducing fire hazard is a primary objective.

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Coarse Woody Detritus Dynamics, Variable Decay Rates and their  
Contribution to Wildland Fuel Succession Following High-Severity Fire  
Disturbance in Dry-Mixed Conifer Forests of Oregon's Eastern Cascades

by  
Christopher J. Dunn

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I understand that my thesis will become part of the permanent collection.....

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Christopher J. Dunn, Author

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## TABLE OF CONTENTS

	<u>Page</u>
CHAPTER 1: INTRODUCTION .....	1
Background .....	1
Dead Wood and Carbon Dynamics .....	5
Post-fire Fuel Accumulation .....	9
Research Questions and Hypotheses .....	10
CHAPTER 2: COARSE WOODY DETRITUS DYNAMICS, VARIABLE DECOMPOSITION RATES AND THEIR EFFECTS ON CARBON EMISSIONS FOLLOWING HIGH-SEVERITY FIRE DISTURBANCE .....	12
Introduction .....	12
Methods .....	14
Site Selection .....	14
Snag Dynamics Sampling .....	18
Snag and Log Decomposition .....	18
Data Analysis .....	21
Snag Fall and Breakage .....	21
Decomposition .....	24
Results .....	25
Snag Fall Rates .....	25
Snag Top Breakage .....	28
Decomposition .....	31
Long-term Dynamics .....	33

## TABLE OF CONTENTS (Continued)

	<u>Page</u>
Discussion .....	36
Snag Fall.....	36
Snag Breakage.....	43
Snag and Log Decomposition .....	46
Integrated Snag Dynamics .....	49
Conclusions .....	54
CHAPTER 3: WILDLAND FIRE FUEL SUCCESSION FOLLOWING HIGH-SEVERITY FIRE IN DRY-MIXED CONIFER FORESTS OF OREGON'S EASTERN CASCADES .....	57
Methods.....	60
Site Selection.....	60
Snag Sampling.....	61
Fine and Coarse Woody Fuel Sampling.....	64
Live Herbaceous and Woody Fuel Loading.....	64
Litter and Duff.....	65
Data Analysis .....	66
Results .....	66
Discussion .....	73
Litter and Duff Fuel Succession.....	74
Live Shrub and Herbaceous Fuel Succession .....	75
1-Hr Fuel Succession .....	77
10-Hr Fuel Succession .....	79



## TABLE OF CONTENTS (Continued)

	<u>Page</u>
100-Hr Fuel Succession .....	81
1000-Hr Fuel Succession .....	82
Conclusions .....	84
CHAPTER 4: ESTIMATING THE IMPACTS OF SALVAGE LOGGING ON FUEL SUCCESSION USING A COARSE WOODY DETRITUS DYNAMICS MODEL .....	86
Introduction .....	86
Methods .....	89
Necromass Calculation .....	89
Snag Fall Rates .....	93
Snag Breakage .....	95
Multiple Snag Breakage Events .....	96
Transfer of Necromass from Breakage .....	96
Bark Transfer .....	98
Fine Woody Fuel Transfer .....	98
Decomposition .....	99
Combustion of CWD by Decay State .....	103
Salvage Logging .....	103
Results .....	104
Heartwood Biomass and Effective Decay Rate .....	104
Woody Fuel Succession .....	106
Discussion .....	112

## TABLE OF CONTENTS (Continued)

	<u>Page</u>
Fine Woody Detritus Succession .....	112
Coarse Woody Detritus Succession .....	116
Modeled Fuel Succession Following Salvage Logging .....	119
Role of Fine Woody Detritus .....	120
Role of Coarse Woody Detritus .....	122
Conclusions .....	123
CHAPTER 5: CONCLUSIONS .....	126
Snag Dynamics.....	126
Fuel Succession.....	128
Future Considerations and Opportunities.....	129
REFERENCES.....	132

## LIST OF FIGURES

<u>Figure</u>	<u>Page</u>
1.1 Conceptual model of the microclimatic conditions leading to variable moisture and temperature regimes in snags and logs .....	8
2.1 Map of fire site locations sampled in this study.....	15
2.2 Proportion of fallen snags by year since fire.....	27
2.3 Annual increase in the proportion of standing snags with top breakage .....	29
2.4 Density loss from decomposition over 150 years .....	32
2.5 Stand characteristics of example stands used to depict long-term dynamics .....	34
2.6 Snag deposition and log accumulation for three stands representing a biomass gradient.....	25
2.7 The process of snag fragmentation following high-severity fire .....	37
2.8 Decay trajectories for <i>Abies sp.</i> necromass using surface and standing decomposition rates. ....	49
2.9 Mean carbon flux from CWD following high-severity fire .....	53
2.10 Mean carbon flux from legacy CWD and FWD following high-severity fire.....	55
3.1 Map of fire site locations sampled in this study.....	62
3.2 Snag dynamics and fuel succession plot layout .....	65
3.3 Twenty-four years of fine fuel and shrub accumulation following high-severity fire .....	67
3.4 1000-hr fuel accumulation following high-severity fire during 24 years of succession. ....	70

## LIST OF FIGURES (Continued)

<u>Figure</u>	<u>Page</u>
3.5 Litter and duff accumulation during 24 years of development .....	72
3.6 Live shrub and herbaceous biomass accumulation .....	73
3.7 1-hr fuel accumulation during 24 years of succession following high-severity fire .....	78
3.8 10-hr fuel accumulation during 24 years of succession following high-severity fire. ....	80
3.9 100-hr fuel accumulation during 24 years of succession following high-severity fire .....	82
3.10 1000-hr fuel accumulation during 24 years of succession following high-severity fire .....	83
4.1 “NecroDynamics” individual snag fragmentation and decomposition process .....	90
4.2 Example of the effective decay rate for snags of varying sizes that did not exhibit, or are assumed to not have, statistically significant decomposition throughout the stem .....	102
4.3 Percentage of total <i>P. ponderosa</i> biomass made up of heartwood. ....	105
4.4 Effective decomposition loss rate-constant for <i>P. ponderosa</i> logs .....	106
4.5 1-hr fuel accumulation during 24 years of succession following high-severity fire .....	108
4.6 10-hr fuel accumulation during 24 years of succession following high-severity fire. ....	109
4.7 100-hr fuel accumulation during 24 years of succession following high-severity fire .....	110
4.8 CWD fuel accumulation during 24 years of succession following high-severity fire .....	111

## LIST OF FIGURES (Continued)

<u>Figure</u>	<u>Page</u>
4.9     Average 100-hr biomass estimates from regression analysis and “NecroDynamics” without decomposition .....	113
4.10   Modeled FWD fuel succession for 3 different stands .....	115
4.11   CWD succession and fuel availability. The y-axis scale varies by stand condition .....	117
4.12   FWD succession for a salvaged and unsalvaged stand following high-severity fire .....	121
4.13   Post-fire necromass dynamics after salvage logging in high- severity stands .....	124

## LIST OF TABLES

<u>Table</u>	<u>Page</u>
2.1 Sampled fire sites within the dry-mixed conifer forests of Oregon's eastern Cascades.....	17
2.2 The proportion of snags which had fallen at each fire site sampled .....	22
2.3 Type 3 tests of fixed effects and parameter estimates for predicting the mean proportion of fallen snags .....	26
2.4 Half-life estimates for snags by species and size class. ....	28
2.5 Regression coefficients for estimating snag breakage height .....	30
2.6 Parameter estimates from decomposition analysis .....	31
2.7 Summary of snag fall rates from relevant literature.....	39
3.1 Sampled fire sites within the dry-mixed conifer forests of Oregon's eastern Cascades.....	63
3.2 Regression coefficients for estimating woody detritus biomass .....	68
3.3 Regression coefficients for estimating litter, duff and shrub biomass.....	71
4.1 Allometric equations used to estimate live biomass .....	92
4.2 Back-transformed logistic regression equations for estimating the proportion of snags fallen by species, diameter class and years since fire .....	94
4.3 A summary of all k-constants used in the "NecroDynamics" model.....	101
4.4 Linear regression coefficients for estimating <i>P. ponderosa</i> 's heartwood diameter from diameter inside bark (Adjusted R-squared = 0.9298).....	104
4.5 Comparison of predicted values from "NecroDynamics" and independently sampled fuels transect data .....	107

## LIST OF TABLES (Continued)

<u>Table</u>	<u>Page</u>
4.6 Peak surface CWD and available fuel biomass for three stand conditions .....	118

# **Coarse Woody Detritus Dynamics, Variable Decay Rates and their Contribution to Wildland Fuel Succession Following High-Severity Fire Disturbance in Dry-Mixed Conifer Forests of Oregon's Eastern Cascades**

## **CHAPTER 1: INTRODUCTION**

### **Background**

The ecological importance of trees in forested ecosystems begins at an early age, continuing throughout their long life and well beyond. The importance of live trees is evident in their many ecological functions as primary producers, vegetative structures used by numerous wildlife species, nutrient cyclers, climate amelioration at micro and macro scales, and their influence on the long-term development of soils. Trees typically regulate structure, composition and function of forests. The ability to live from several hundred to greater than 2000 years highlights their dominance of many ecological processes over long temporal scales.

The ecological function of trees does not end with death, but rather continues in the form of standing dead trees (snags) and downed coarse woody detritus (logs). In this stage of a tree's function, they continue to act as a primary energy source for saprophagous organisms, as wildlife habitat, long-term nutrient and carbon stores, and contributors to soil development. A tree may continue to provide these ecological functions for 100-200 years after death, depending on the dynamic nature of their post-mortality life.

Natural tree mortality in forested landscapes can be caused by multiple mechanical and biological agents. Common biological disturbances include insects and pathogens (e.g., bark beetles and root rot) that generally cause mortality at the individual tree or small patch scale, although broader scale mortality does occur when environmental conditions are favorable (Franklin et al. 1987). Windthrow and fire are the most common forms of natural tree mortality in Oregon, but the prevalence of these disturbances varies by geographic region. Wetter western forests experience more windthrow than eastern Cascade forests while fire is more common in the east.



In all geographic regions multiple mortality agents exist, causing structural changes at small and large spatial scales at varying temporal intervals. These disturbances alter forest structure by creating dead biological legacies important to future stand development and ecosystem function (Franklin et al. 2002).

Prior to European settlement, the dominant disturbance within the dry-mixed conifer forests of Oregon's eastern Cascades was fire. The fire regime in this forest type is classified as mixed-severity with fire free intervals ranging from 9-62 years (Agee 1993). Across landscapes, variation in time since fire across, topography, fire weather conditions and broad climate patterns creates a mosaic of burn severities within an individual fire boundary during any given year. Individual stands will experience either high-severity (>75% tree mortality), moderate severity (25% - 50% tree mortality) or low severity (< 25% tree mortality) fire within a single fire event (Agee 1993). The landscape severity pattern will be different during subsequent fires, although there is evidence that high-severity fire may be spatially correlated during subsequent fire events (Thompson et al. 2007).

Dry-mixed conifer forests in Oregon's eastern Cascade's are dominated by *Pinus ponderosa* (ponderosa pine), *Abies grandis* (grand fir), *Abies concolor* (white fir), *Pseudotsuga menziesii* (Douglas-fir), *Pinus contorta* (lodgepole pine) and *Calocedrus decurrens* (incense-cedar) (Franklin and Dyrness 1988). The density and composition of a stand within these forests at a particular time is dependent on site conditions (elevation, slope, aspect, soil type and climate) and disturbance history. The cumulative effects of disturbance at long temporal scales creates a complex mosaic of stand conditions with varying composition and abundance of live trees, snags and logs (Hessburg et al. 2005).

European settlement altered natural disturbance regimes in these forests beginning in the mid-19<sup>th</sup> century through a reduction in human ignited fire, livestock grazing, and more recently, mechanized fire suppression (Agee 1993). The lack of fire has resulted in an increase in the total number of stems present in dry-mixed conifer forests, with particular increase in the density of *Abies* sp. stems <41cm DBH.

Evidence of structural change was noticed as early as the 1940's by Harold Weaver (1943). At the same time that natural disturbance decreased, human disturbances increased across Oregon. Commercial logging on private and public lands began early in the 20<sup>th</sup> century and increased until the early 1990's when logging on public lands precipitously dropped following the Northwest Forest Plan. Harvesting practices during this period included salvage logging fire killed trees in an attempt to recover the economic value of the standing timber before it diminished due to decomposition of the woody material (Sessions et al. 2004).

The effects of altered disturbance regimes on stand structure are obvious when considering live trees. The lack of fire in natural stands has increased the number of small diameter trees present on site, resulting in increased susceptibility to stand replacement fire across broad landscapes (Spies et al. 2006) and increased competitive stress on large legacy trees (Fellows and Goulden 2008). Past logging has typically modified stand structure by creating even-aged, single species stands that lack the vertical and horizontal diversity historically developed on dry mixed-conifer sites. Today, all species in dry-mixed conifer forests of Oregon have higher densities of stems <41cm DBH and lower densities of larger diameter trees (Merschel 2010). The ecological effects of these changes are becoming apparent as our forests are valued for ecosystem services beyond just timber products.

Less obvious, perhaps, are the effects of these changes on CWD. Harvest regimes facilitated live tree growth and removed the standing timber before competitive stresses, biotic agents and/or irregular mortality (e.g., lightning) created CWD. In the event of moderate to large disturbances, salvage logging would remove much of the newly created CWD, further reducing the presence of this material on the landscape. The loss of these biological legacies and the reduction of large trees that develop into large CWD have reduced the presence of this material on landscapes.

Increasingly, public land managers are tasked with providing multiple resources from the land they manage. In western forests, contemporary management strategies tend towards ecological restoration and/or ecologically sustainable

management practices that include management of dead wood resources as important structural components of forest ecosystems. The retention of snags and logs are increasingly blended with fuels reduction treatments in dry forests in order to meet ecological and management needs (Brown et al. 2003).

Inputs of dead material into the forest system occur as individual tree or small-patch mortality events and large pulse events associated with landscape-scale disturbances often caused by insect outbreaks or fire. During the past decade, dry-mixed conifer forests of Oregon's eastern Cascades have experienced large scale pulse events caused by an increase in fire activity relative to prior decades (Spies et al. 2006). Contemporary fires span low to high-severity fire behavior given complex interactions of topography, fire weather, fuel conditions and suppression activities, although contemporary landscapes disproportionately burn at high-severity fire relative to historical conditions (Spies et al. 2006).

Considerable debate continues about appropriate management strategies following fire disturbance, particularly in high-severity areas (Donato et al. 2006, Noss et al. 2006, Sessions et al. 2004). Salvage logging has been proposed for economic and ecological reasons. Rapid salvage logging operations attempt to capture the economic value of fire killed trees and improve local economies (Sessions et al. 2004). These operations have been conducted without a comprehensive understanding of their ecological impact even though they are often ecologically justified based on the premise that reforestation is required to reestablish trees and reduce future fire hazard by removing available fuel. Recent research has shown reforestation is not required to reestablish trees in dry or wet forest environments (Larson and Franklin 2005, Donato et al. 2006, Shatford et al. 2007). The ability of salvage logging to reduce future fire hazard is still debatable; some research suggests salvage logging exacerbates the fuel hazard (Donato et al. 2006), improves the conditions (Monsanto and Agee 2008), or remains relatively neutral in its effect (McGinnis et al. 2010).

## Dead Wood and Carbon Dynamics

Total ecosystem carbon storage in forests includes carbon present in above- and below-ground pools. Above-ground carbon pools include live trees, shrubs and herbaceous plants as well as dead pools; including fine woody detritus, logs, snags, and plant litter. Below-ground pools include live and dead fine and coarse roots, soil organic matter and soil mineral carbon. Fluctuations in the abundance of carbon present in each pool over long temporal lengths depend on complex processes of primary productivity, decomposition and disturbance history (Meigs et al. 2009).

Net ecosystem productivity (NEP) is the balance between net primary productivity (NPP) of a vegetative community and heterotrophic respiration ( $R_h$ ). In forest ecosystems, NPP includes the above- and below-ground growth of trees, shrubs and herbaceous communities.  $R_h$  is quantified as the decomposition of plant litter, fine woody detritus, snags, logs, root decomposition and soil respiration. NEP is positive when carbon assimilation into vegetative growth (i.e. NPP) is greater than respiration by saprophagous organisms (i.e.  $R_h$ ), quantified by the carbon flux into or out of the system (Chapin et al. 2002).

Substrate quality is the dominant control of decomposition rates among plant components, leading to increased decay rates of plant litter relative woody material. Carbon and nitrogen ratios and lignin concentrations affect the potential rate of decay. Woody material has lower concentrations of nitrogen, higher concentrations of lignin and increased concentrations of decay resistant hydrocarbons, making leaf litter substrate quality higher than woody plant material as an energy source for decay organisms. Decomposition loss rate-constants of plant litter in dry-mixed conifer forests have been found to be as high as  $k = 0.28 \text{ yr}^{-1}$ , while woody decomposition loss rate-constant have been reported as low as  $k = 0.011 \text{ yr}^{-1}$  (Monleon and Cromack 1996, Harmon et al. 2007).

At broad spatial scales, climatic conditions control the rate at which decomposition of plant material occurs. In general, the rate of decomposition

decreases with decreasing mean annual temperature (i.e. higher decay rates in tropical versus boreal forests). Within a forest system, temperature and moisture regimes control the rate of decomposition. When a system is too arid, decomposition becomes limited because dry environmental conditions are not conducive to survival and growth of decay organisms. For wetter, more coastal temperate systems decomposition is limited when decomposing substrate becomes saturated, developing an anaerobic environment that limits heterotrophic respiration and consumption of dead organic matter. A similar trend exists for temperature; low temperatures limit biotic activity and where as very high temperatures can cause oxidation and mortality of saprophagous organisms, also leading to an optimal temperature range for decomposition (Harmon et al. 1986).

Tradeoffs between optimal moisture and temperature regimes develop unique patterns in decay rates of woody material. At local scales, environmental conditions lead to varying rates of decomposition of dead wood among species and their physical position within a forest. Temperate dry-mixed conifer forests experience dry, hot summers and cold, moderately wet winters with snowfall being the dominant form of precipitation. In these systems, snags are hypothesized to have lower decay rates than logs with the opposite trend occurring in wetter forest types (Harmon et al. 1986).

The vertical orientation of snags causes them to receive increased levels of solar radiation (relative to logs) because of a lack of shading and exposure to reflected solar radiation from surface vegetation and material. Increased exposure and absorption of solar radiation reduces moisture by accelerating evaporation and increasing snag temperature, potentially limiting decomposition by pushing the decomposition environment of this material beyond optimal moisture regimes. Additionally, logs receive less solar radiation than snags because of shading by snags and new vegetation occupying the site. Shading from vegetation increases over succession as a closed canopy forest system develops.

Wind exposure reduces moisture content in snags relative to logs by disrupting the boundary layer forming around this material. A physical boundary layer develops

along the forest floor as shrubs and tree regeneration occupy the site, increasing the physical obstruction of wind. A lack of wind limits the disruption of the boundary layer immediately surrounding the woody material, maintaining higher relative humidity within the boundary layer as compared to the surrounding air, lowering the rates of evaporation from logs. Snag stems and crowns remain fully exposed to wind, effectively increasing evaporation rates of this material.

Lastly, snags absorb less moisture than logs. The vertical orientation of snags limits moisture absorption as precipitation is sloughed off (snow) or runs off (rain) more rapidly than absorption occurs except along the lower portion of the stem. Higher moisture absorption occurs in snags along the lower stem from wicking and snow burial, as well as percolation into the broken tops of snags. Pooling of precipitation on logs, direct contact with the forest floor and incorporation of logs into the snowpack layer increases the length of time this material is directly exposed to moisture, allowing for increased absorption. Additionally, some logs remain elevated above the soil for extended periods of time after falling, limiting surface contact and increasing air drying. Figure 1.1 summarizes the local site conditions that we believe create variable standing and surface decay rates.

The dry climatic condition of dry-mixed conifer forests in Oregon suggests that snags should decay at a slower rate than logs. No published study has explicitly evaluated this difference in these forests even though they may lead to dramatic shifts in the timing of carbon and mass loss due to decomposition. Immediately following high-severity fire, CWD represents the largest carbon store and potentially the largest carbon flux into or out of the system. Assuming there are differences in decomposition rates between snags and logs, the rate at which carbon is lost from large dead wood is dependent on the decomposition loss rate-constant for snags and logs and the deposition of snags to the soil surface.

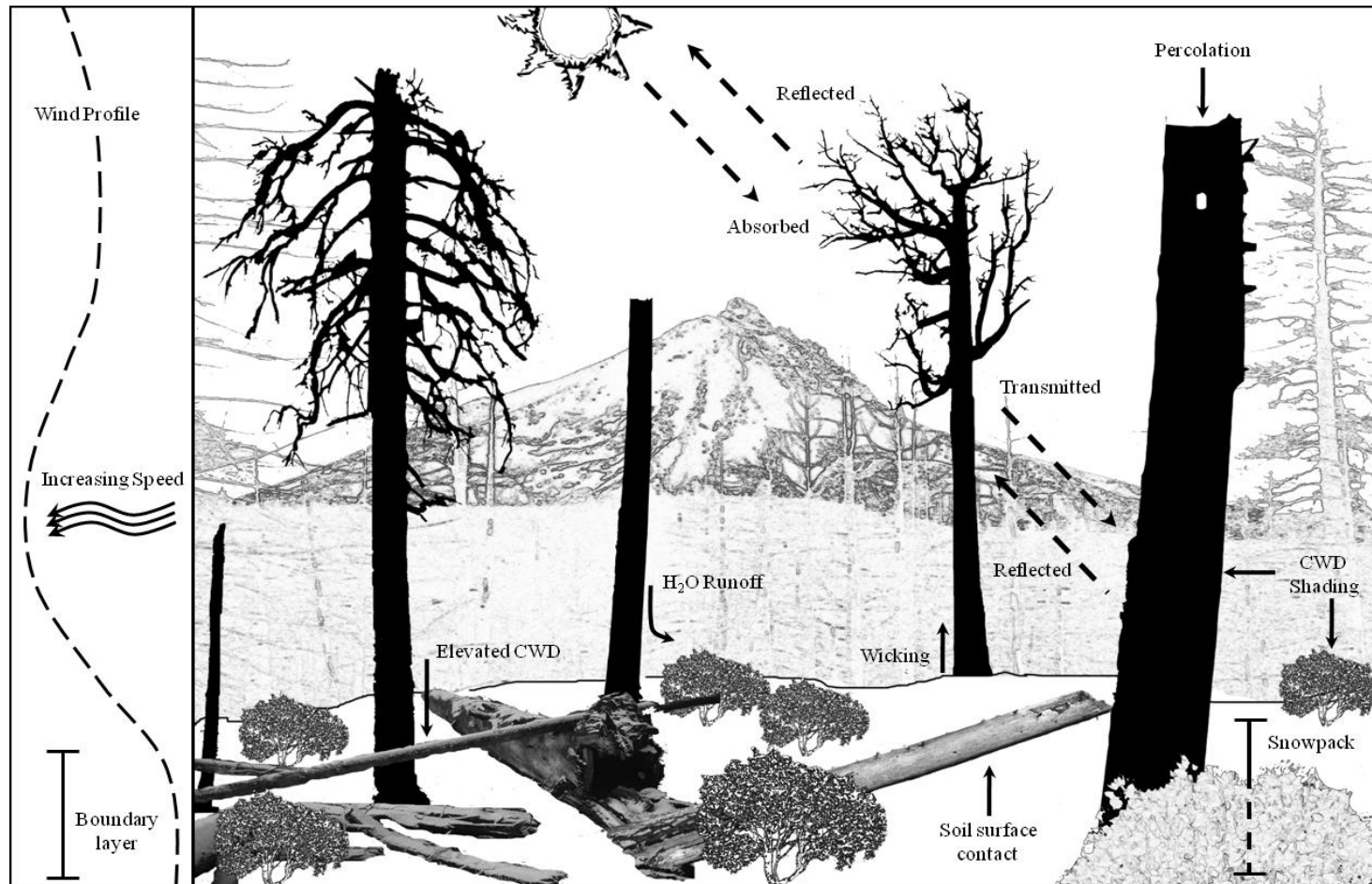


Figure1.1: Conceptual model of the microclimatic conditions leading to variable moisture and temperature regimes in snags and logs. In dry forest environments, more favorable conditions for decomposition occur closer to the soil surface leading to higher decay rates in logs, relative to snags.

## Post-fire Fuel Accumulation

Salvage logging in high-severity fire areas has been proposed as an ecologically necessary management action in order to reduce future fire severity associated with combustion of CWD (Sessions et al. 2004). Monsanto and Agee (2008) further illicit this potential by quantifying total soil surface area covered by logs following contemporary high-severity fires, relative to historical conditions in eastern Washington. Combustion of CWD in this environment can result in tree mortality (*via* root damage) on 25% of an area, an increase of 15% from historical conditions. The amount of CWD present as surface fuel in post-fire environments is dependent on the deposition of snags and decomposition and fragmentation of snags and logs. Differences in decay rates by species and position lead to a complex pattern of fuel contribution by dead biological legacies created in a fire event.

The decay state of CWD is also important in determining the amount available for combustion. In the Yellowstone Fires of 1988, combustion of beetle killed logs amounted to only 8% of fuel consumed seven years following mortality by bark beetles (Tinker and Knight 2000). Other studies have found that combustion efficiency is high for logs in decay states 4 and 5, but low for all other decay states (Josh Hyde, personal communication). Current combustion models categorize logs as sound or decayed material, identifying decayed logs by its propensity to break apart when kicked following the fuel sampling procedures of Brown (1974). Class IV log characteristics are consistent with this decayed state metric, with most decay class IV logs converging around a density of approximately  $0.20 \text{ g cm}^{-3}$  (Harmon et al. 2008). Variations in deposition rates and decay rates among species and their position control the amount of CWD available to combust at the time of a subsequent fire.

Pyrolysis of dried woody fuels occurs at temperatures as low as 200-280°C in the formation of char, but flaming combustion generally doesn't occur until woody material reaches 280-340°C (Johnson and Miyanishi 2001). For CWD to combust, radiant and convective heat transfer from fine fuel particles (e.g., surface litter and



duff layers) must be large enough and sustained for a long enough period to bring CWD to this critical temperature. Additional heat is required to evaporate moisture present in the wood before combustion occurs.

In a post-fire environment, sufficient time for accumulation of fine fuels is typically necessary to develop enough material to combust at temperatures high enough to ignite CWD. The required fuel development includes enough horizontal continuity in the fuel layer to sustain a flaming front as it progresses across the forest floor. Fine woody detritus particles generally are not spatially connected in early post-fire environments until a litter and duff layer develops, increasing the time required for sustained combustion to occur across a stand previously burned in a high-severity fire.

### **Research Questions and Hypotheses**

High-severity, high-mortality fire causes large shifts in biomass stores by transferring large quantities of live biomass to snag biomass which later become logs. If decomposition rates are less for snags relative to logs in dry forest environments, total carbon loss will vary based on the fragmentation rate of snags and different decay rates of snags and logs.

Coarse woody detritus' availability for combustion, in the event of a future fire, is also dependent on the rate at which this material decays and will vary based on the position and, thus, deposition of this material. Linking snag deposition rates and variable CWD decomposition rates is necessary to more accurately understand the effects of snags and logs on ecosystem processes and potential impacts on future fire severity. For this reason, this research focuses on the following questions:

1. What are the fall and breakage rates of *Pinus ponderosa*, *Abies sp.* and *Pinus contorta* snags following high mortality fire in dry-mixed conifer forests of Oregon's eastern Cascade Mountains?

*Hypothesis 1:* Fall and breakage rates will decrease with increasing diameter at breast height for all species and will be higher for *Abies sp.* relative to *Pinus sp.*

2. Are decomposition loss rate-constants of *Pinus ponderosa* and *Abies sp.* CWD different by species and position?

*Hypothesis 2:* Lower wood density and higher substrate quality leads to higher decomposition loss rate-constants for *Abies sp.* relative to *Pinus ponderosa*, and variation in moisture and temperature regimes caused by microclimatic differences increase loss rate-constants of logs relative to snags in dry-mixed conifer forests of Oregon.

3. What is the rate of accumulation of fine woody detritus, litter, duff, herbaceous and shrub fuel layers during the first 24 years following high mortality fire in dry-mixed conifer forests along the east slope of the Oregon Cascade Mountains?

*Hypothesis 3:* Fine woody detritus, litter, duff, herbaceous and shrub fuel layers will have positive accumulation rates following high-severity fire during the first 24 years of succession.

4. Does salvage logging reduce future fire hazard by removing combustible biomass following high-severity fire?

*Hypothesis 4:* Salvage logging directly reduces CWD and its availability for combustion in future fires but does not reduce FWD and its contribution to fire hazard.

## **CHAPTER 2: COARSE WOODY DETRITUS DYNAMICS, VARIABLE DECOMPOSITION RATES AND THEIR EFFECTS ON CARBON EMISSIONS FOLLOWING HIGH-SEVERITY FIRE DISTURBANCE**

### **Introduction**

Coarse woody detritus (CWD) management continues to gain importance as our knowledge improves about the ecological benefits of this material in forested landscapes (Harmon 2002, Brown et al. 2003). The ecological role of a tree is only partially fulfilled as a living organism (Harmon et al. 1986, Franklin et al. 1987, Spies et al. 1988). Snags and logs provide structural habitat valuable to multiple vertebrate and invertebrate species in dry-mixed conifer forests of Oregon (Bull et al. 1997, Thomas et al. 1979). CWD is also important for saprophyte communities, ecosystem nutrient cycling, and the carbon balance of a forest (Harmon et al. 1986, Triska and Cromack 1979, Harmon 2009). The availability of snags and logs to meet ecological needs is dependent on their creation, fragmentation and decomposition (Harmon 2002).

Inputs of CWD into ecosystems occur through multiple pathways at varying spatial and temporal scales. In Western U.S. forests, isolated incidents of annual tree mortality occur mainly by insects, disease, wind, and lightning. Stand- and landscape-level tree mortality occur at broader spatial and temporal scales as discrete pulse events mainly by wind, insect outbreaks, and/or fire disturbance (Franklin et al. 1987, Harmon et al. 1986, Spies et al. 1988).

Dry mixed-conifer forests in Oregon are fire adapted ecosystems with a mixed-severity fire regime, experiencing fire disturbance on intervals from 9 - 62 years, although high-severity events at a given site may have fire free intervals >200 years (Agee 1993). One of the most evident effects of fire disturbance is a pulse of tree mortality, dramatically increasing the amount CWD within the system. The magnitude of this transfer is most evident in high-severity fire areas where >75% of tree mortality occurs (Agee 1993). The extent of fire disturbance has increased across

the Western United States during the past 30 years, and future climate scenarios project a continuation of this trend (Brown 2004, Westerling et al. 2006). If fire extents do become greater, the amount of CWD in fire adapted systems will increase beyond contemporary levels.

Climate change poses potentially severe threats to global forested ecosystems, requiring effective mitigation strategies to prevent dramatic alterations in their structure, composition and function. Forests are, conversely, considered important for their potential to help mitigate climate change by sequestering carbon for long periods of time and reducing atmospheric carbon concentrations (IPCC 2007). Carbon sequestration in forests is dependent largely upon the magnitude of net primary productivity and heterotrophic respiration (Chapin et al. 2002).

The effects of forest disturbance on ecosystem productivity is gaining relevance as stand, landscape and regional carbon budgets are being quantified (Law et al. 2001, Meigs et al. 2009). High-severity fire will alter carbon budgets in forest systems by causing dramatic shifts in carbon pools, primarily by moving biomass from live to dead pools. This will affect net ecosystem productivity by decreasing primary productivity and increasing heterotrophic respiration.

Ecosystem-level heterotrophic respiration from woody biological legacies is dependent on species specific decomposition loss rate-constants and the total amount of necromass on site. In dry-climate forests, snags are hypothesized to decay at a slower rate than logs because of differences in moisture content and temperatures (Harmon et al. 1986). Quantification of long-term decay dynamics in these systems requires estimates of decomposition loss rate-constants for woody material oriented as aerial or surface necromass, as well as the rate of fragmentation from aerial to surface material. Little information is available integrating CWD dynamics in estimates of carbon emissions and net ecosystem productivity in post-fire environments, and no studies have focused on dry-mixed conifer forests along the eastern slope of the Oregon Cascade Mountains.

In this study, we estimated the decomposition rate of *P. ponderosa* and *Abies* *sp.* snag and *P. ponderosa* logs in dry-mixed conifer forests of Oregon's eastern Cascades. We also obtained estimates of snag fall rates for *P. ponderosa*, *Abies* *sp.*, and *P. contorta* snags across a 24 year chronosequence in high-severity fire sites. This information will provide long term perspectives about the dynamics of coarse woody detritus, aiding manager's decisions regarding the fate of this material.

## Methods

CWD dynamics occur over long temporal scales requiring long term research projects to obtain estimates from controlled studies, which are often longer than financially supported. For this reason, we used a chronosequence approach in order to make observations and obtain results that would otherwise take >20 years to receive. This approach substitutes space for time but requires the assumption that site conditions and history are the same for each site used in the analysis (Harmon et al. 1986). This assumption may not be true for all environmental variables, but we are confident broader trends in CWD dynamics are valid.

Management practices, including fire suppression and salvage logging, limited the number of fire sites available for sampling. A total of 7 fire sites were suitable for this study. Each fire site was included in the chronosequence beginning 1 year post-fire and ending 24 years post-fire (Figure 2.1).

## Site Selection

We focused on snag and log dynamics in dry-mixed conifer forests of Oregon's eastern Cascades. Sample plots were targeted at mature to old-growth dry-mixed conifer forests that have experienced high-severity fire (> 75% overstory mortality). These forest conditions provide the greatest diameter distribution and largest population of snags to sample, making them the most useful sites for obtaining estimates for numerous species and DBH distributions while limiting confounding impacts of other management practices.

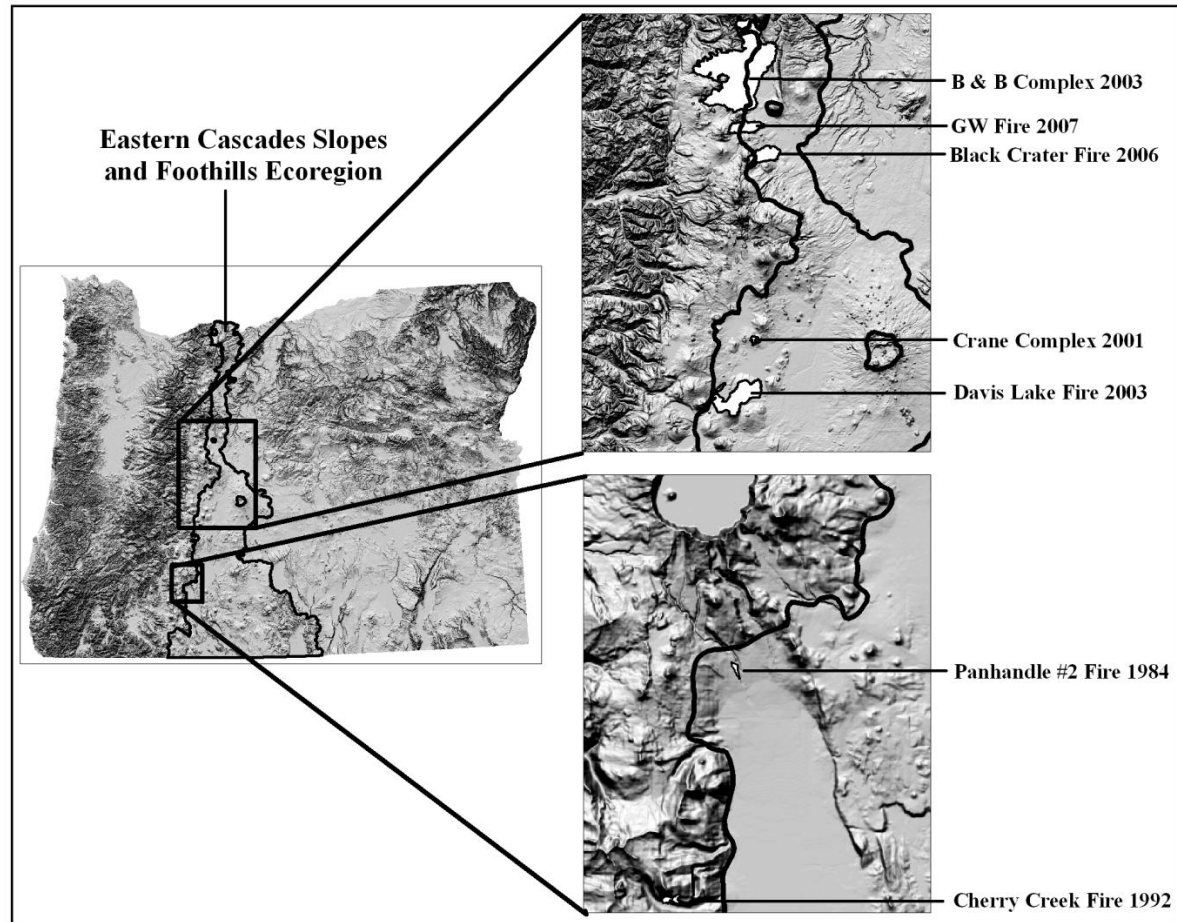


Figure 2.1: Map of fire site locations sampled in this study. The “Eastern Cascades Slopes and Foothills Ecoregion” is consistent with ecoregions developed by the Western Ecology Division of the EPA (Omernick 1987).

All fires occurring during the past 100 years in Oregon's eastern Cascades were considered for use in this study. Deschutes and Winema National Forests, BLM, Oregon Department of Forestry, and Crater Lake National Park GIS Fire Databases were queried for potential fires. Fire boundaries were reduced by dry-mixed conifer plant association groups consistent with the *Abies grandis* and *Abies concolor* Zones of the Eastern Cascades (Franklin and Dyrness 1988). Any portion of fire with evidence of more than a single fire event, or salvage logging/harvesting during this period was excluded as a potential site. Table 2.1 summarizes the average environmental conditions of fires sampled in this study.

When available, burn severity maps were used to extract high-severity fire areas within a fire boundary (Miller and Thode 2006). When burn severity maps were unavailable, the boundaries of high-severity fire areas were delineated using aerial photography. Silvicultural records, stand age data and aerial photo interpretation removed areas with a history of harvesting pre- or post-fire. High-severity fire sites proposed for salvage logging within the Black Crater and GW fires were excluded to maintain plot integrity after sampling.

GIS polygons were created around all potential sampling sites. The polygons were further reduced by a 50 meter buffer from roads, perennial streams, and green forest edges to ensure sampling occurred within high-severity areas only. All plots were spatially separated by a minimum of 200 meters to reduce spatial autocorrelation and randomly located using ARC GIS Hawth's Tools random point generator (Beyer 2004, ESRI 2009).

Table 2.1: Sampled fire sites within the dry-mixed conifer forests of Oregon's eastern Cascades. All sampling occurred during the summers of 2007 and 2008. Average elevation is the average of all plots from that fire. Climate data obtained from PRISM Climate Group website (PRISM Climate Group 2010).

Fire Site	Fire Year	Location	Year Sampled	Total Hectares	Average Elevation (m)	Average PPT (mm)	Average Tmax (°C)	Average Tmin (°C)
GW Fire	2007	Deschutes N.F.	2008	2977	1301	1052.07	12.72	0.43
Black Crater Fire	2006	Deschutes N.F.	2008	3807	1445	909.54	11.82	-0.60
B & B Complex	2003	Deschutes N.F.	2008	36733	1159	1147.56	13.18	0.72
Davis Lake Fire	2003	Deschutes N.F.	2008	8572	1551	843.53	12.12	-0.84
Crane Complex	2001	Deschutes N.F.	2008	289	1478	804.73	12.82	-0.54
Cherry Creek Fire	1994	Winema N.F.	2007	134	1722	1418.16	11.20	-0.14
Panhandle 2	1984	Crater Lake N.P.	2008	49	1419	889.89	14.19	-1.13



### ***Snag Dynamics Sampling***

A total of thirty 0.25 hectare plots were sampled across the seven fire sites. Plot centers were located using a Trimble GEO XT GPS unit. 50m x 50m square plots were used since shrub conditions prevented consistent lines to the plot centers. All plots were slope corrected and oriented on true north.

All snags were marked and measured within each 50 x 50 m plot using a systematic grid across slope. Snag characteristics included species, diameter at breast height (DBH), total height, decay class, condition, time of death and whether or not insects had colonized the stem. A snag's condition was noted as standing whole, standing broken, or fallen. Broken snags had a visible break point along the main stem greater than 2 meters in height. Snags broken below 2 meters in height were considered fallen. Height was not collected for fallen snags because of the difficulty of locating all portions broken during the snags life. Species was determined using characteristics described by Parks et al. (1997).

Pre-fire killed snags and live trees were also sampled during the inventory. Snags were determined to be dead at the time of the fire when > 5% of the sapwood was consumed or converted to char during the fire. Within the Biscuit Complex Fire of 2002, it was found that < 2% of live tree stem wood was combusted during high-severity fire (Campbell et al. 2007). Relatively high amounts of pre-fire killed snags occurred on the B & B Complex due to spruce budworm mortality in the preceding decade. The same characteristics were collected for live trees and pre-fire killed snags.

### ***Snag and Log Decomposition***

To more accurately quantify post-fire carbon emissions and mass loss, decomposition loss rate-constants need to be obtained for snags and logs. Decomposition loss rate-constants for *Abies sp.* and *P. ponderosa* snags and *P. ponderosa* logs were obtained in this study using a chronosequence approach up to 7

years post-fire. Older fires were not used since they were located in areas where such extensive sampling for decomposition was not feasible. Additionally, snag decomposition loss rate-constants could be biased by favoring slower decaying or initially higher density woody material that has the potential to remain standing longer relative to other snags of similar size.

Twelve *Abies sp.* and *P. ponderosa* snags were randomly sampled from each of the recent fire sites (1, 2, 5, 5, and 7 years post-fire) for a total of 60 snags of each species. Randomly located 10 x 50-meter belt transects were used to select snag samples located within the same polygons as snag inventory plots but outside of the plot boundaries. Target DBH of snags was 41cm, although samples ranging from 35-51cm were considered acceptable. The first encountered snag from the random starting point, meeting the size and species requirements, was selected for felling.

Prior to felling, species, DBH, height, and decay class was recorded for each snag. After felling, total snag height was divided into 4 equal sections, and 3 cross-sections were systematically removed at heights of 25%, 50%, and 75% of total snag height using a chainsaw. The height of each cross-section was recorded at the time of removal. Each cross-section was labeled, wrapped in plastic wrap and transported back to the Wildland Fire Lab at OSU for cold storage until processing.

We opportunistically sampled *P. ponderosa* trees felled as hazard trees during fire suppression operations to obtain surface decomposition loss rate-constants for logs. Limiting our sampling to trees felled during suppression operations allowed us to confidently date their year of fall. Log samples were primarily located along travel routes and were felled to prevent injury to firefighters and/or road blockage. All logs sampled had evidence of active felling and mop-up from fire suppression activities. All the same information was collected for logs as snags and cross-sections were removed from these samples using the same process as snags.

Wood density was calculated as oven-dry weight per unit pre-dried volume ( $\text{g cm}^{-3}$ ). Total volume of all cross-section samples was calculated as the volume of a cylinder using the cross-sections diameter after removal of outer and inner bark.

Longitudinal thickness (height of the cylinder) was determined from the average of 6 longitudinal measurements taken systematically around the outer edge of the cross-section. Total weight of each cross-section was obtained using an Ohaus Ranger 12kg scale (to the nearest gram).

Drying cross-section samples in their entirety required more time and space than was reasonably available. To obtain oven-dried weights, subsamples representing the variation in moisture content of the entire sample were removed from each cross-section. Each subsample weighed between 100-150 grams; weighed to the nearest 0.01grams on an Ohaus Scout Pro. The samples were oven dried at 55 °C until reaching low, stable moisture condition (approximately 5 days). The weight of each cross-section sample was reduced by the proportion of dry weight to initial weight of its subsample.

Volume was calculated for sapwood and heartwood separately for *P. ponderosa* cross-sections. After determining the volume of the entire cross-section, sapwood and heartwood were separated, using a hammer and chisel, along the outer ring of heartwood. Diameter and longitudinal thickness of the heartwood and radial thickness of the sapwood was recorded. Total volume of the heartwood was subtracted from the volume of the entire cross-section to obtain the volume of the sapwood only. Subsamples were removed from heartwood and sapwood separately and processed for moisture content as described above.

Each cross-section represents a different proportion of the total tree stem since proportionately more stem biomass is located in the lower reaches of a tree. In order to account for this difference, total snag density is estimated as the sum of the weighted average densities of the cross-sections. Density of each cross-section is weighted based on its area relative to the average area of the three cross-sections.

## Data Analysis

All plot level snag data was pooled to the fire site level to meet the assumption of independent samples and avoid autocorrelation found within a single fire event. The proportion of snags fallen at each site was separated into species and DBH size classes. Size classes were delineated as small (<23 cm DBH), medium (23 – 41 cm DBH), and large (> 41 cm DBH) snags following McIver and Ottmar (2006).

### *Snag Fall and Breakage*

A total of 5103 snags were measured in the field to estimate fall and breakage rates for *P. ponderosa*, *P. contorta*, and *Abies sp.* Table 2.2 provides the proportion of snags that had fallen at each fire site by the time they were sampled. Sample sizes were not large enough to estimate fall or breakage rates for *P. menziesii* or *C. decurrens*.

A generalized linear model (PROC GLM, SAS Institute Inc. 2008) was used to estimate the proportion of fallen and broken snags as predicted by multiple independent variables. Logistic regression analysis with a logit link function was required because of the binomial distributions indicative of count data (Ramsey and Shafer 2002). Model assumptions were evaluated and met before continuing with interpretation of the parameter estimates and model results.

Break height was recorded for all snags in field measurements. Equations to estimate break height were obtained from linear regression (PROC Reg, SAS Institute Inc. 2008). Break heights were log transformed to meet assumptions of normality (Ramsey and Shafer 2002).

Table 2.2: The proportion of snags which had fallen at each fire site sampled. Values in parenthesis represent number of snags measured for each species and size class at each fire site.

Diameter Species Class		Fire Site							Total N (Species/Size)
		GW 2007	Black Crater 2006	B & B Complex 2003	Davis Lake 2003	Crane Complex 2001	Cherry Creek 1992	Panhandle 2 1984	
Proportion Fallen (N)									
Abies sp.	<23cm	0.000 (188)	0.051 (642)	0.197 (152)	0.293 (624)	0.337 (1497)	0.868 (91)	0.993 (141)	3335
	23-41cm	0.000 (64)	0.019 (107)	0.055 (55)	0.109 (92)	0.025 (119)	0.663 (80)	0.881 (59)	576
	>41cm	0.000 (47)	0.000 (45)	0.000 (19)	0.050 (20)	0.200 (5)	0.381 (21)	0.556 (9)	166
	Total N	(299)	(794)	(226)	(736)	(1621)	(192)	(209)	4077
P. ponderosa	<23cm	0.057 (229)	0.211 (19)	0.391 (23)	0.400 (10)	0.536 (56)	1.000 (9)	1.000 (4)	350
	23-41cm	0.000 (16)	0.000 (9)	0.000 (23)	0.222 (9)	0.263 (19)	0.641 (39)	1.000 (4)	119
	>41cm	0.024 (42)	0.000 (39)	0.000 (21)	0.058 (52)	0.219 (64)	0.346 (26)	0.680 (25)	269
	Total N	(287)	(67)	(67)	(71)	(139)	(74)	(33)	738
P. contorta	<23cm		0.048 (63)		0.217 (23)	0.521 (96)		1.000 (72)	254
	23-41cm				0.200 (10)	0.200 (5)		0.938 (16)	32
	Total N		(63)		(33)	(101)		1 (90)	288
Total N (Site)		(586)	(924)	(293)	(840)	(1861)	(266)	(332)	5103

The full model tested for predicting both the proportion of fallen and broken snags was:

$$Y_{ijk} = \mu + \beta_i + \gamma_j + \lambda_k + \beta\lambda_{ik} + \beta\gamma\lambda_{ik} + \varepsilon_{ijk}$$

where,

$Y_{ijk}$  is the proportion of fallen snags in the  $i^{\text{th}}$  year for the  $j^{\text{th}}$  species in the  $k^{\text{th}}$  diameter class

$\mu$  is the overall mean proportion of fallen snags

$\beta_i$  is the effect of the  $i^{\text{th}}$  year since the fire ( $i = 1, 2, 5, 7, 15, 24$ )

$\gamma_j$  is the effect of the  $j^{\text{th}}$  species ( $j = P. ponderosa, P. contorta, Abies. sp.$ )

$\lambda_k$  is the effect of the  $k^{\text{th}}$  diameter class ( $k = <23 \text{ cm}, 23-41\text{cm}, >41\text{cm}$ )

$\beta\lambda_{ik}$  is the interaction effect of the  $i^{\text{th}}$  year since fire and the  $k^{\text{th}}$  diameter class

$\beta\gamma\lambda_{ik}$  is the interaction effect of the  $i^{\text{th}}$  year since fire for the  $j^{\text{th}}$  species of the  $k^{\text{th}}$  diameter class

$\varepsilon_{ijk}$  is the random error term representing the variability among years within the species and diameter classes

Snag height was not included in this model since it covaries with DBH and the initial heights of broken snags cannot be determined as a consequence of the chronosequence methodology. Drop-in-deviance tests were used to select the most significant regression model (Ramsey and Shafer 2002, Littell et al. 2006). A deviance scale parameter was calculated by dividing the models Pearson's chi-squared statistic by the degrees of freedom determined with the Satterthwaite method (SAS Institute Inc. 2008).

### ***Decomposition***

A decrease in density of wood samples over time is indicative of a loss of wood mass due to decay. Simple linear regression analysis using PROC REG in SAS 9.2 (SAS Institute Inc. 2008) was conducted on *Abies sp.*, *P. ponderosa* snag and log sapwood, and *P. ponderosa* snag and log heartwood in separate analyses. The natural logarithm of wood density values were used to obtain the loss rate-constant (k-constant) for a negative exponential model, the most commonly used and accepted decay model (Means et al. 1980, Harmon et al. 1986). The slope of the regression line represents the k-constant for the woody material.

Model assumptions were evaluated and met before continuing with interpretation of the parameter estimates and model results. Density values were log transformed to meet assumptions of normality and fit a negative exponential decay model. Analysis for outliers and leverage were included in the statistical procedure.

Allometric equations for estimating heartwood biomass have not been developed for *P. ponderosa* but are necessary to accurately estimate the decomposition rate of this species with varying DBH. The internal heartwood structure is modeled as a cone. Diameter inside bark at the base of the snag is estimated using *P. ponderosa* taper equations (Garber and Maguire 2003). Heartwood base diameter is estimated from diameter inside bark. Statistical analysis consisted of simple linear regression predicting heartwood diameter from diameter inside bark using PROC REG in SAS 9.2 (SAS Institute Inc. 2008). Model assumptions were evaluated and met before continuing with interpretation of the parameter estimates and model results.

## Results

### *Snag Fall Rates*

A statistically significant model was found for predicting the proportion of fallen snags following a fire. The best fit model was:

$$Y_{ij} = \mu + \beta_i + \gamma_j + \lambda_k + \beta\lambda_{ik} + \epsilon_{ijk}$$

where,

$Y_{ij}$  is the proportion of fallen snags in the  $i^{\text{th}}$  year for the  $j^{\text{th}}$  species in the  $k^{\text{th}}$  diameter class

$\mu$  is the overall mean proportion of fallen snags

$\beta_i$  is the effect of the  $i^{\text{th}}$  year since the fire ( $i = 1, 2, 5, 7, 15, 24$ )

$\gamma_j$  is the effect of the  $j^{\text{th}}$  species ( $j = P. ponderosa, P. contorta, Abies. sp.$ )

$\lambda_k$  is the effect of the  $k^{\text{th}}$  diameter class ( $k = <23 \text{ cm}, 23-41\text{cm}, >41\text{cm}$ )

$\beta\lambda_{ij}$  is the interaction effect of the  $i^{\text{th}}$  year since fire and the  $k^{\text{th}}$  diameter class

$\epsilon_{ijk}$  is the random error term representing the variability among years within the species and diameter classes

On average, the ratio between the Pearson Chi-Square statistic and its degrees of freedom should equal one in a generalized linear model (SAS Institute Inc. 2008). The ratio for this data set was 2.75, suggesting the data exhibited more dispersion than expected under a binomial model with a logit link function. A deviance scale parameter was included in the statistical analysis to account for the overdispersion. Table 2.3 provides coefficients for the statistically significant predictor variables used to estimate mean proportion of fallen snags on the logit scale. Figure 2.2 provides estimates of the mean proportion of snags fallen during 50 years of succession following high-severity fire disturbance.



Table 2.3: Type 3 tests of fixed effects and parameter estimates for predicting the mean proportion of fallen snags. \* Indicates an interaction between the parameters. The reference group is *P. ponderosa* snags >41cm DBH.

Type 3 Tests of Fixed Effects				
Fixed Effect	Num DF	Den DF	F-Value	Pr > F
Years Since Fire	1	43	211.9	<0.0001
Species	2	43	4.34	0.0191
Diameter Class	2	43	2.41	0.1019
Years Since Fire * Diameter Class	2	43	8.16	0.001
Parameter Estimates				
Fixed Effect	Estimate	Std Error	DF	p-value
Intercept	-3.5046	0.5031	43	< 0.0001
Years Since Fire	0.197	0.03629	43	< 0.0001
<i>Abies sp.</i>	-0.5624	0.2268	43	0.0171
<i>P. contorta</i>	-0.06696	0.3473	43	0.848
< 23cm DBH	0.7122	0.553	43	0.2047
23- 41cm DBH	-0.2529	0.6844	43	0.7136
Years Since Fire * <23cm	0.1976	0.04893	43	0.0002
Years Since Fire * 23-41cm	0.1024	0.05135	43	0.0526

Snag half-lives provide the year when 50% of the total snag population is estimated to have fallen. Half-life estimates provide an opportunity to compare fall rates among multiple studies, particularly those repeatedly sampling a given population of snags through the early portion of the populations total attrition time. Doubling the half-life does not estimate the potential longevity of snags in a given population since logistic regression lines are non-linear (Figure 2.2). Table 2.4 provides half-life estimates for each species and DBH class sampled.

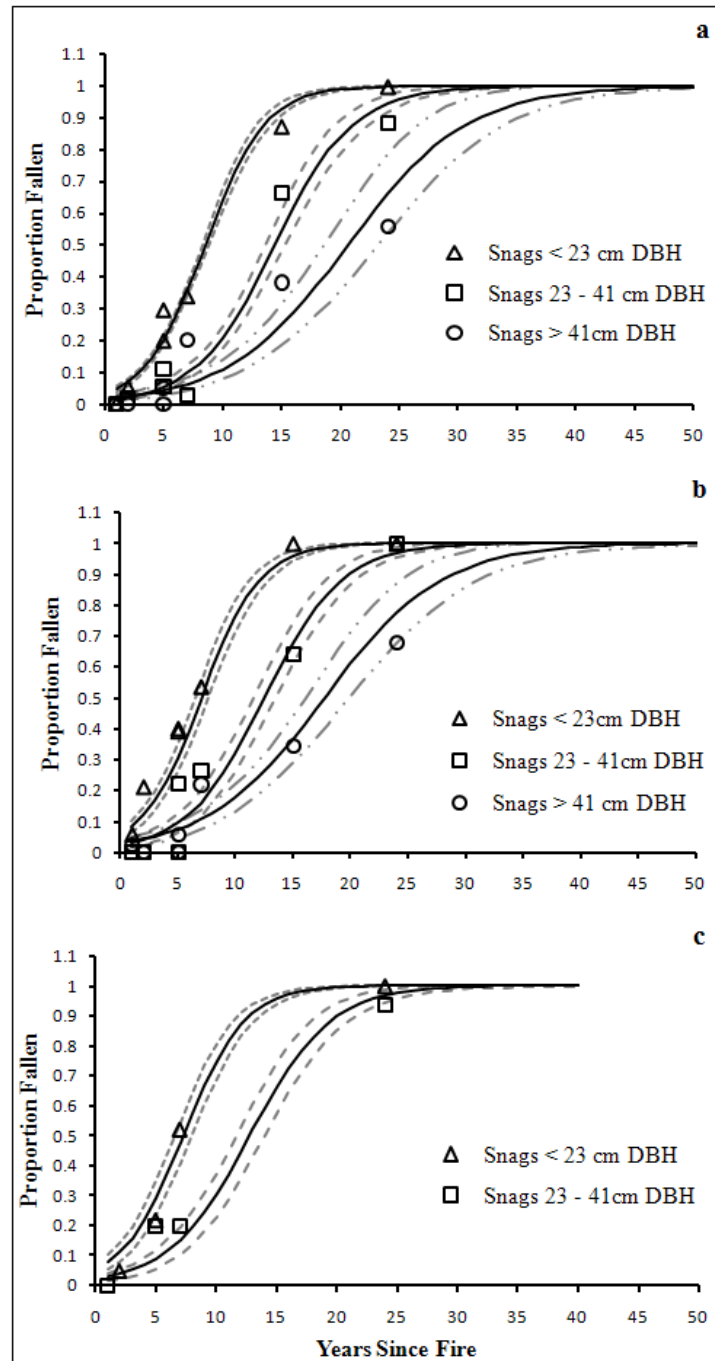


Figure 2.2: Proportion of fallen snags by year since fire. a = *Abies sp.* b = *P. ponderosa*. c = *P. contorta*. Species, size class and years since fire are statistically significant ( $p < 0.05$ ) independent variables for predicting the proportion of snags fallen. Solid lines are mean proportions and dashed lines are estimates of standard errors. Open symbols are sampled proportions from fire sites.

Table 2.4: Half-life estimates for snags by species and size class.

<b>Half-Life Estimates</b>			
Species	DBH Class	Estimated Half-Life	95% Confidence Range
<i>Abies sp.</i>	< 23 cm	8 - 9 years	8 - 9 years
	23- 41 cm	14 - 15 years	13 - 16 years
	> 41 cm	20 - 21 years	17 - 27 years
<i>P. ponderosa</i>	< 23 cm	7 - 8 years	7 - 8 years
	23- 41 cm	12 -13 years	11 - 14 years
	> 41 cm	17 - 18 years	15 - 23 years
<i>P. contorta</i>	<23 cm	7 - 8 years	6 - 8 years
	23-41 cm	12 -13 years	11 - 15 years

*P. ponderosa* and *P. contorta* did not exhibit significant differences in their fall rates, resulting in these species having the same half-lives for small and medium snags. Only one *P. contorta* snag >41 cm DBH was sampled so an estimate wasn't made for this size class. The lack of large *P. contorta* snags also suggests few trees reach this size class in Oregon's dry-mixed conifer forests.

*Abies sp.* had the slowest fall rates and, thus, the highest half-life for all size classes. For snags <23 cm DBH, half-lives for *Abies sp.* were estimated to be about 1 year greater than *Pinus sp.* This difference in half-life increases to 2 years for snags 23 – 41 cm DBH and 3 years for snags > 41cm DBH. For all species, DBH class had a greater effect on fall rates than species. Half-lives increased by 5 years for *Pinus sp.* and by 6 years for *Abies sp.* as one moved to the next larger diameter class.

### ***Snag Top Breakage***

No statistically significant linear trends were found for the proportion of snags broken by species, DBH class, or time since fire (snag age). Although a significant linear trend was not found, it was evident that top breakage begins as soon as the first year following a fire and the total proportion of snags with broken tops increases with time for almost all species and diameter classes (*P. contorta* exhibits little breakage).

A general annual increase in the proportion of snags with broken tops was estimated by dividing the total number of snags broken at a fire site by the number of years since the fire, and then averaging this value across all fire sites. Multiplying the average breakage rate by the number of years since fire estimated the total proportion of standing snags broken by that year. The average annual increase in proportion of snags broken by species and size class are shown in Figure 2.3. This method assumed breakage rates were linear over time and represents the best estimate that can be obtained from this study.

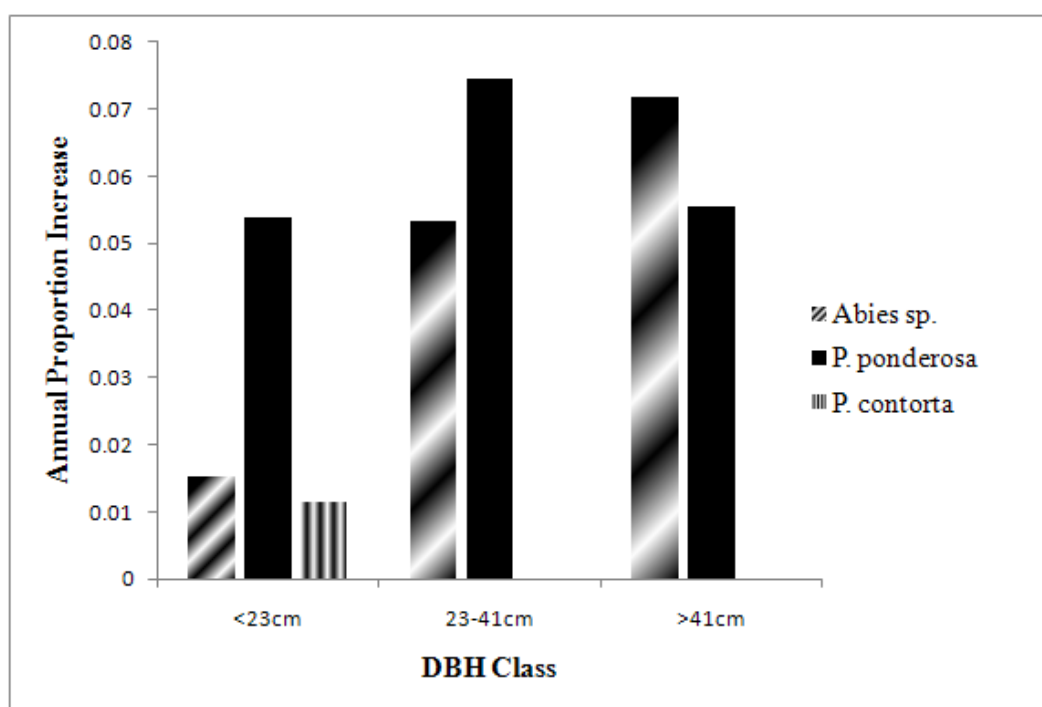


Figure 2.3: Annual increase in the proportion of standing snags with top breakage. Multiplying the number of years since fire by this proportion estimates the total number of snags still standing that have a broken top.

For *Abies sp.*, the rate of top breakage increased with increasing diameter size class. Each year, 1.5% more snags will break in the smallest size class. Over time, little necromass is transferred by this process since most snags of this type fell prior to breaking due to their high fall rates and low breakage rates. The opposite trend occurs

for large *Abies sp.* snags; an additional 7.2% of snags break each year but these snags have relatively low fall rates.

*P. ponderosa* snags don't appear to have a distinct trend in breakage rate associated with size class. Each year, 7.4% of standing snags were estimated to break in the medium size class compared to 5.4% and 5.5% in the small or large size classes, respectively. *P. contorta* showed no evidence of breaking in the medium size classes and only an additional 1.2% break in the smallest size class each year.

Break height increased with increasing DBH and decreases with years since fire. Species did not exhibit a statistically significant effect. Table 2.5 summarizes the linear regression coefficients in log scale.

Table 2.5: Regression coefficients for estimating snag breakage height. <sup>1</sup> Lower and upper 95% confidence limits. Estimates are on log scale. DBH = diameter at breast height. Time since fire = number of years since fire.

Parameter	Estimate (SE)	p-value	LCL <sup>1</sup>	UCL <sup>1</sup>
Intercept	1.39962 (0.06920)	< 0.0001	1.2636	1.5357
DBH	0.02182 (0.00158)	< 0.0001	0.0187	0.02493
Time Since Fire	-0.05682 (0.00591)	< 0.0001	-0.06844	-0.0452

The full statistical model included species as a categorical variable but was not significant at an  $\alpha = 0.05$  level (p-value = .1121, F-value = 2.20 and df = 384). DBH was positively correlated with broken height. An increase of 1cm DBH increased the median break height by 2.2% (95% confidence interval of 1.8% to 2.5%, p-value < 0.0001, F = 176.82 and df = 384). Age of snag was negatively correlated with height of break. A one year increase in time since fire decreased the median break height by 5.5% (95% confidence interval of 4.4% to 6.6%, p-value < 0.0001, F-value = 93.16 and df = 384). The adjusted R-square for this linear regression model is  $r^2 = 0.3865$ , so considerable variability remains unexplained in this model.

### Decomposition

Decomposition loss rate-constants estimated for *Abies sp.* and *P. ponderosa* snags, and *P. ponderosa* logs are provided in Table 2.6. *Abies sp.* snags were found to exhibit statistically significant decay throughout the stem, with an estimated decomposition loss rate-constant of  $k = 0.0149 \text{ yr}^{-1}$ . *P. ponderosa* snag sapwood and heartwood decomposition was not statistically significant at an  $\alpha = 0.05$  level.

Table 2.6: Parameter estimates from decomposition analysis. Slope = - (k-constant). All values are in log scale. Bold values indicate statistically significant decay rates. <sup>1</sup> 95% confidence limits

Substrate	Estimate	Std Error	p-value	LCL <sup>1</sup>	UCL <sup>1</sup>
<i>P. ponderosa</i> snag sapwood					
slope	-0.00379	0.00549	0.493	-0.01477	0.0072
<i>P. ponderosa</i> snag heartwood					
slope	-0.01013	0.00622	0.1087	-0.02257	0.00231
<i>P. ponderosa</i> log sapwood					
slope	<b>-0.0362</b>	<b>0.0093</b>	<b>0.0004</b>	<b>-0.05506</b>	<b>-0.0173</b>
<i>P. ponderosa</i> log heartwood					
slope	<b>-0.01642</b>	<b>0.00635</b>	<b>0.0139</b>	<b>-0.0293</b>	<b>-0.0035</b>
<i>P. ponderosa</i> combined					
slope	<b>-0.024</b>	<b>0.0073</b>	<b>0.0023</b>	<b>-0.388</b>	<b>-0.0092</b>
<i>A. grandis</i> snag					
slope	<b>-0.01488</b>	<b>0.00476</b>	<b>0.0028</b>	<b>-0.02442</b>	<b>-0.0053</b>

*P. ponderosa* log sapwood and heartwood were found to have statistically significant decomposition loss rate-constant of  $k = 0.0362 \text{ yr}^{-1}$  and  $k = 0.01642 \text{ yr}^{-1}$ , respectively. When heartwood and sapwood are combined as whole samples, we estimated a statistically significant decomposition loss rate-constant of  $k = 0.024 \text{ yr}^{-1}$ . Figure 2.4 depicts density loss due to decomposition for these decomposition loss rate-constants and the uncertainty surrounding the estimate.

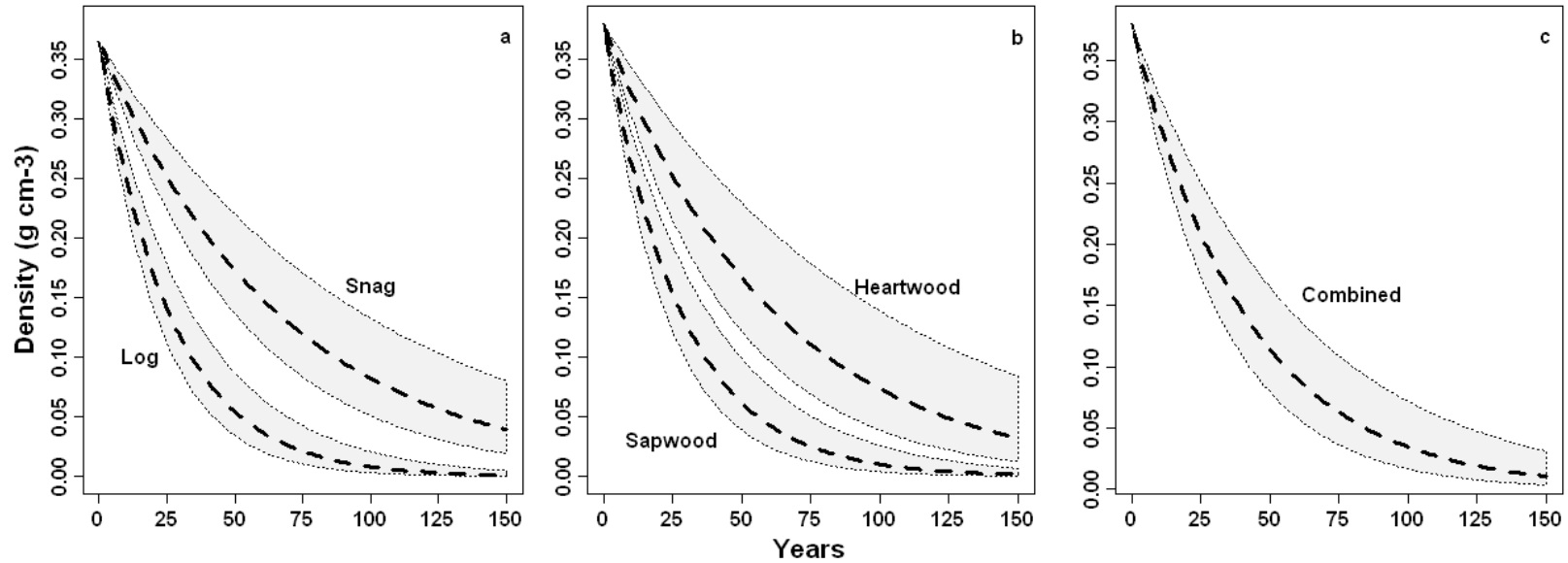


Figure 2.4: Density loss from decomposition over 150 years. Dashed lines represent mean decay rates and shaded areas represent variability as  $\pm 1$  standard error. a = *Abies sp.* snag and log decay. *Abies sp.* log decay rate estimate is from Harmon et al. (2007) and is included for comparison. The estimated variability for *Abies sp.* log is from *P. ponderosa* sapwood standard errors from this study. b = *P. ponderosa* sapwood and heartwood density loss from decay. c = *P. ponderosa* combined sapwood and heartwood decay loss. Using the combined decay rate assumes all trees exhibit the same proportion of heartwood and sapwood as the samples used to obtain this estimate.

### *Long-term Dynamics*

The cumulative effect of species and size interactions developed a non-linear pattern of biomass fragmentation following high-severity fire. Figure 2.5 provides stand characteristics for three stands used to depict long-term dynamics. Figure 2.6 depicts the fall and fragmentation of snags and accumulation of logs with and without snag breakage for three 1-hectare stands spanning the biomass gradient sampled. Decomposition is occurring for standing and surface material at rates dependent on species and their position, as described previously.

Snag necromass exhibited an initial lag in its transfer rate for about 5 years, followed by increased rates as more snags break and fall. The amount of necromass transferred slows after approximately 20 years, primarily due to a lack of necromass remaining standing. The total amount standing varies by initial biomass, but half-lives for necromass are surprisingly similar. In the highest biomass stand, 50% of total necromass will have fallen by 13 – 14 years post-fire. Without breakage, the half-life estimate was 16 – 17 years with a maximum difference occurring 15 – 16 years post-fire and amounting to  $40.8 \text{ Mg ha}^{-1}$  (15.8% of initial necromass).

The medium biomass stand has the same half-life as the high biomass stand (13 – 14 years) with breakage and 17 – 18 years without. The maximum difference occurs 16-17 years post-fire and amounts to  $26.74 \text{ Mg ha}^{-1}$  (16.5% of initial necromass). The low biomass stand had a slightly earlier half-life of 12-13 years with breakage and 16-17 years without. The maximum difference occurs 14 – 15 years post-fire and equals  $24.73 \text{ Mg ha}^{-1}$  (21.5% of initial necromass).

The non-linear trend in biomass deposition accompanied by variable decomposition rates leads to peak CWD accumulations at 26, 25 and 23 years post-fire for the high, medium and low biomass stands. Maximum CWD accumulation amounts to 68%, 59%, and 52% of initial biomass estimates for high, medium and low biomass stands. After peak accumulation, decomposition loss is greater than input rates and results in a negative exponential loss of CWD necromass.



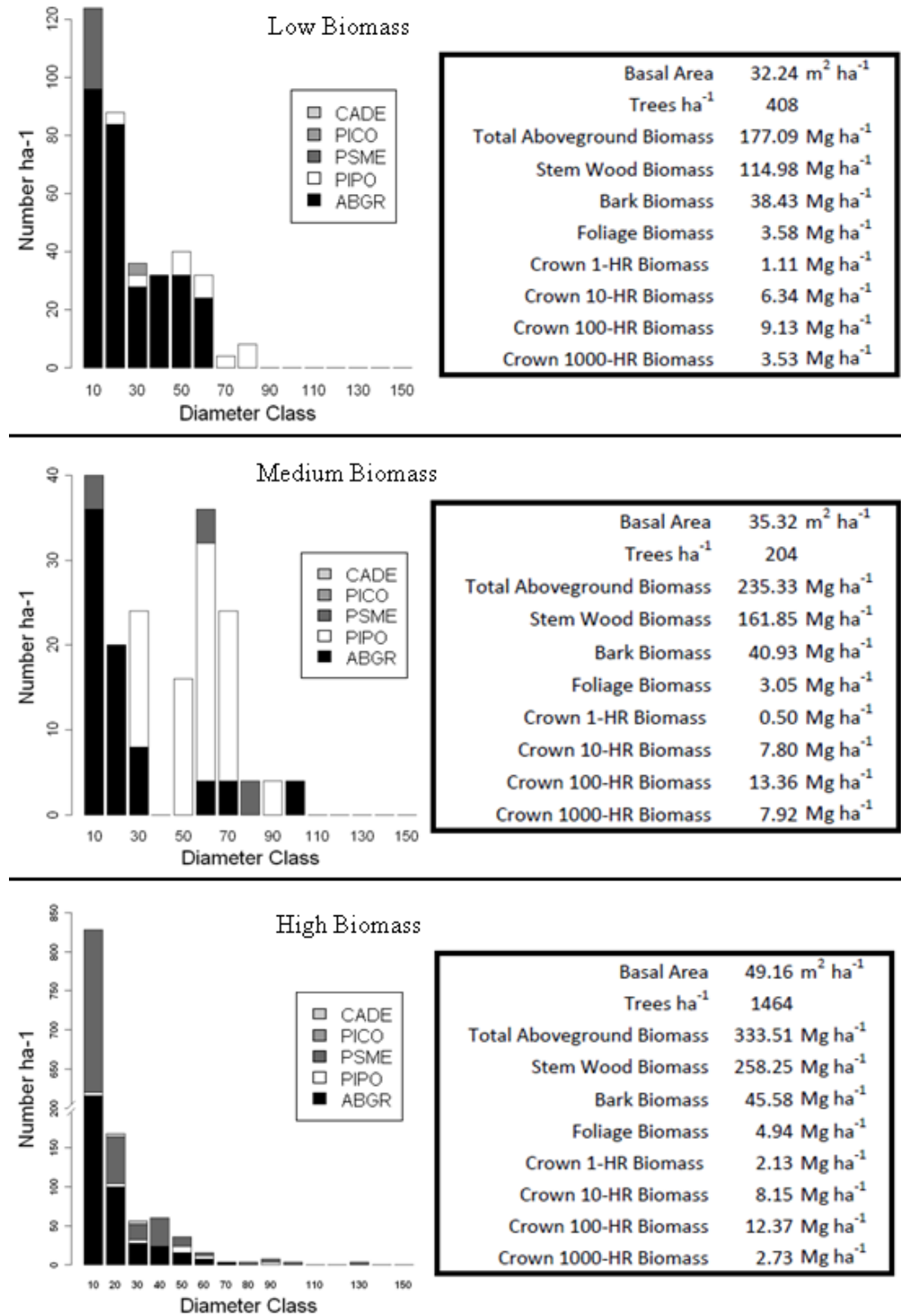


Figure 2.5: Stand characteristics of example stands used to depict long-term dynamics. These stands span the low to high biomass gradient for stands sampled in this study.

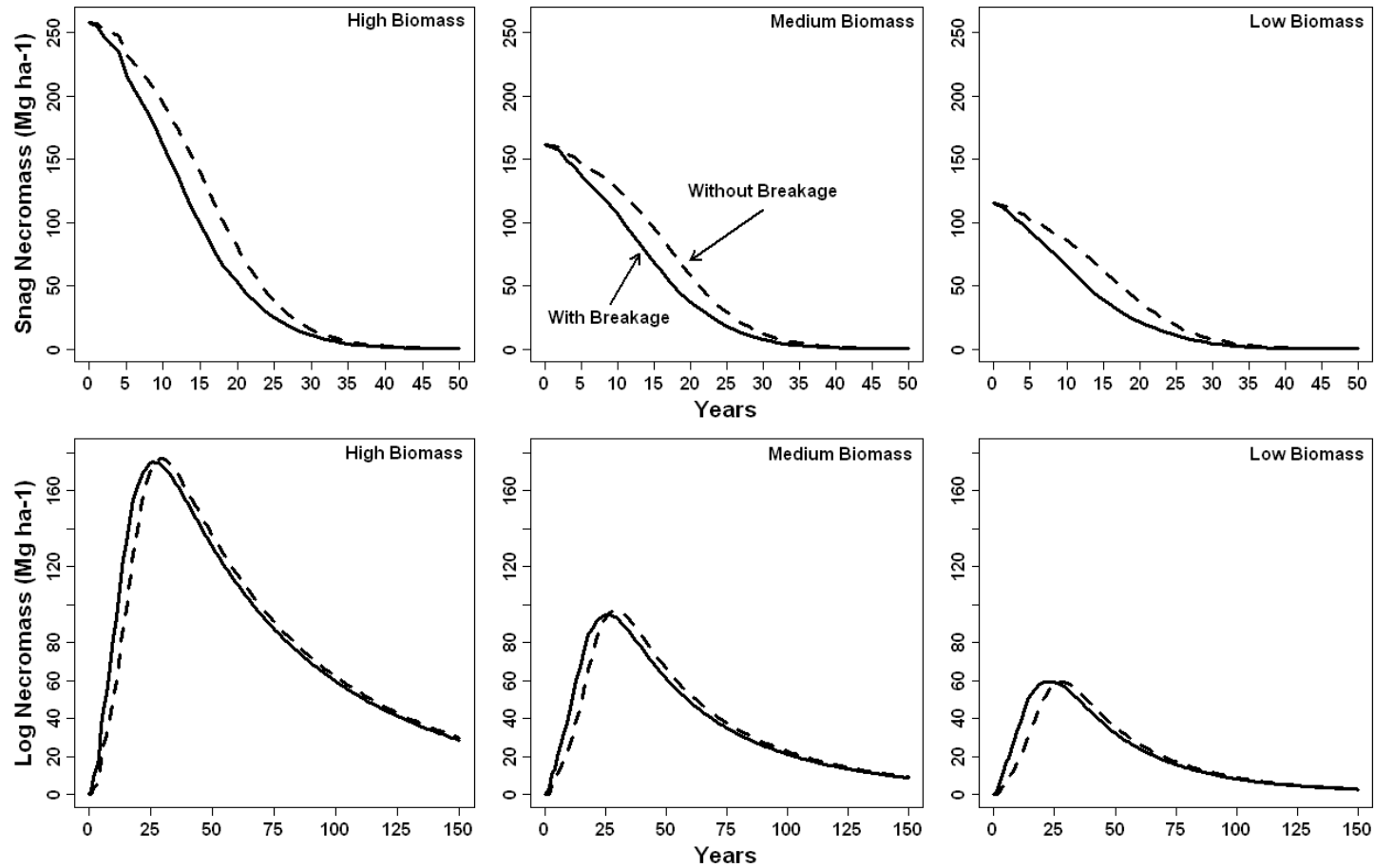


Figure 2.6: Snag deposition and log accumulation for three stands representing a biomass gradient. Scales are different by rows to allow for time related details to be examined.

## Discussion

Snag fall and fragmentation is a dynamic process directly affecting forest structural habitat, ecosystem carbon emissions and fuel succession (Bull et al. 1997, Harmon et al. 1986, McIver and Ottmar 2006). Fire is a pulse disturbance that creates landscape-level changes in snag abundance, removing existing snags at the same time as creating others. High-severity fire environments are generally described as having >75% overstory tree mortality and represent areas within a fire boundary having the largest proportion of pre-fire trees converted to snags (Agee 1993, Meigs et al. 2009). The large snag populations develop with a range of characteristics dependent on pre-fire stand conditions (largely described by species composition and density) and offer opportunities to explore the dynamics of snags created by the same mortality agent and experiencing similar local environmental conditions.

Following mortality, individual snags may follow multiple pathways before they have completely fallen. Figure 2.7 depicts potential trajectories a snag may take prior to falling, dependent on an individual snags characteristics and resultant dynamics. At broader scales, snag populations exhibit varying rates of attrition dependent on broad geographic and local site environmental conditions.

### *Snag Fall*

The initiation of snag fall varies by species and size class. In the eastern Cascades, there is a lag period before *Abies sp.* snags begin to fall. Small and medium snags begin falling after 1 year and large snags begin falling after 4 years. Medium and large *P. ponderosa* snags had a lag period of 2 years, but small snags did not exhibit a lag. *P. contorta* medium snags may exhibit a 1 or 2 year lag but our small sample size cannot confirm this. These values are consistent with other studies where lags for *P. ponderosa*, *P. contorta* and *P. menziesii* were between 2 and 3 years with evidence of little snag fall up to 5 years, particularly for larger size classes (Bull 1983, Everett et al. 1999, Keen 1955, Mitchell and Preisler 1998, Schmid et al. 1985).

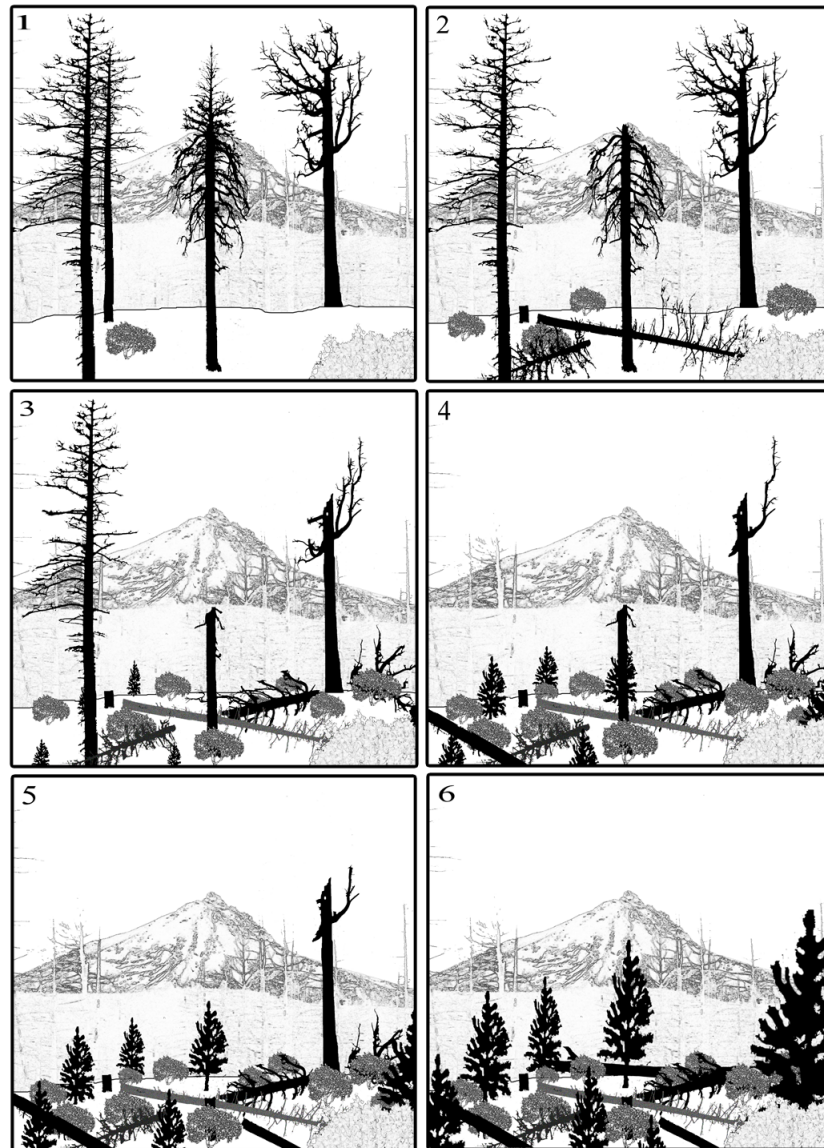


Figure 2.7: The process of snag fragmentation following high-severity fire. 1) Snags begin standing whole with slow decomposition of standing necromass , 2) Snags fall whole and/or break, depositing CWD to the forest floor and accelerating decomposition of that material, 3) Some snags remain standing whole while others break for the first or second time, depositing more CWD to the forest floor, 4) Most snags are broken or fallen and most decomposition occurs on the soil surface while live tree biomass begins to recover, 5) Broken snags fall at slower rates and decomposition continues, 6) All snags have fallen, necromass transfer is complete and accelerated surface decomposition occurs.

Diameter-at-breast-height appears to be the dominant factor affecting the rate of snag fall for conifer snags. For *P. ponderosa* and *P. contorta*, we found an increase in diameter class results in a 5 years increase in their estimated half-life. *Abies sp.* experience a 6 year increase in estimated half-life as one increases in diameter class.

Decreasing fall rates with increasing DBH are consistent across geographic regions and species. Along Oregon's eastern Cascades, Dahms (1949) reported that 81% of *P. ponderosa* snags 23-41 cm DBH fell by year 10 but only 40% of snags >41cm DBH fell by the same year. In Arizona, Chambers and Mast (2005) reported 50% of *P. ponderosa* snags <25 cm DBH fell by year 7, but only 40% of snags 25-50.9 cm DBH and 38% of snags 51-71 cm DBH fell by the same year. Morrison and Raphael (1993) reported that 85 – 93% of *Abies sp.* snags < 23cm fell by year 10, but only 62 – 81% of snags 24-53 cm DBH and 56% of snags > 53 cm DBH had fallen. Everett et al. (1999) reported 50% of *P. contorta* snags < 23 cm DBH fell by year 10-11 but it took 15 years for 50% of snags 23 -41 cm DBH to fall. Regardless of species, fall rates will decrease as DBH increases. Table 2.7 summarizes ours and published half-life estimates for *P. ponderosa*, *Abies sp.* and *P. contorta* snags within the DBH range sampled in this study.

The effect of DBH on fall rates is likely continuous, having an effect at smaller ranges than the diameter classes used in this or other studies. For example, 7 years following mortality Keen (1929) reported 42% of *P. ponderosa* snags 51-96 cm DBH fell but only 22% of snags 96-122cm fell. Chambers and Mast (2005) reported 38% of snags 51-76 cm DBH and 0% of snags 76 – 107 cm DBH had fallen by year 7 and Dahms (1949) reports 34% and 25% of snags fallen for the same DBH classes. Lack of time and ability to follow a large population of snags, distributed sufficiently across the range of DBH of interest, limits most studies ability to evaluate the effect of smaller increases in DBH on fall rates. We incorporated all snags >41 cm DBH into a single group which limits predictive accuracy of fall year for an individual large snag, especially those >100cm DBH.

Table 2.7: Summary of snag fall rates from relevant literature.

Species	DBH Distribution	Half-life (years)	Region	Length of Study	Source
<i>P. ponderosa</i>	<23 cm	7-8	OR (eastern Cascades)	24 years	Current Study
	<25 cm	7-8	northeastern OR	8 years	Bull (1983)
	<25 cm	7	AZ	7 years	Chambers and Mast (2005)
	<23 cm	7	WA (eastern Cascades)	81 years	Everett et al. (1999)
	<26 cm	5-6	southwestern CO	10 years	Harrington (1996)
	13 - 36 cm	6	northern CA, southern OR	9 years	Landram et al. (2002)
	not reported	7-8	AZ	up to 27 years	Passovoy and Fule (2006)
	15 - 50 cm	5-10	Front Range of CO	10 years	Schmid et al. (1985)
<i>P. ponderosa</i>	23 - 41 cm	12-13	OR (eastern Cascades)	24 years	Current study
	25 - 49 cm	45% fallen by year 8	northeastern OR	8 years	Bull (1983)
	25 - 50.9 cm	40% fallen by year 7	AZ	7 years	Chambers and Mast (2005)
	23 - 41 cm	81% fallen by year 10, 89% by year 22	OR (eastern Cascades)	22 years	Dahms (1949)
	23 - 38 cm	6-7	southwestern CO	10 years	Harrington (1996)
	25 - 46 cm	58% by year 7	northern CA	7 years	Keen (1929)
	13 - 36 cm	6	northern CA, southern OR	9 years	Landram et al. (2002)
	39.47 cm	9-10	AZ	8 -9 years	Russell et al. (2006)
	15 - 50 cm	5-10	Front Range of CO	10 years	Schmid et al. (1985)
<i>P. ponderosa</i>	> 41 cm	17-18	OR (eastern Cascades)	24 years	Current study
	> 50 cm	~ 33% fallen by year 8	northeastern OR	8 years	Bull (1983)
	51 - 71 cm	38% by year 7	AZ	7 years	Chambers and Mast (2005)
	>41 cm	40% fallen by year 10, 78% by year 22	OR (eastern Cascades)	22 years	Dahms (1949)
	> 41 cm	only 21% fallen after 60 years	WA (eastern Cascades)	81 years	Everett et al. (1999)
	51 - 96 cm	42 - 43% by year 7	northern CA	7 years	Keen (1929)
	96 - 122 cm	22 % by year 7	northern CA	7 years	Keen (1929)
	38-74	8 (5 to > 9)	northern CA, southern OR	9 years	Landram et al. (2002)

Table 2.7 continued: Summary of snag fall rates from relevant literature.

Species	DBH Distribution	Half-life (years)	Region	Length of Study	Source
<i>Abies sp.</i> (dry forests)	<23 cm	8 - 9	OR (eastern Cascades)	24 years	Current Study
	13 - 36 cm	7 (range 5 - 10)	northern CA, southern OR	9 years	Landram et al. (2002)
	<23 cm	85 - 93% by year 10	CA (Sierra Nevada Mts.)	10 years	Morrison and Raphael (1993)
<i>Abies sp.</i>	23 - 41 cm	14 - 15 years	OR (eastern Cascades)	24 years	Current Study
	24 - 53 cm	62 - 81% by year 10	CA (Sierra Nevada Mts.)	10 years	Morrison and Raphael (1993)
<i>Abies sp.</i>	> 41 cm	20 - 21 years	OR (eastern Cascades)	24 years	Current Study
	38-74 cm	44% fallen by year 9	northern CA, southern OR	9 years	Landram et al. (2002)
	>53 cm	56 % fallen by year 10	CA (Sierra Nevada Mts.)	10 years	Morrison and Raphael (1993)
<i>P. contorta</i>	< 23 cm	7-8	OR (eastern Cascades)	24 years	Current Study
	< 25 cm	7	northeastern OR	8 years	Bull (1983)
	3 - 23 cm	10 - 11	WA (eastern Cascades)	81 years	Everett et al. (1999)
	7.62 - 30.5	9	MT	15 years	Lyon (1977)
	mean = 19.7	7 - 8	OR (eastern Cascades)	15 years	Mitchell and Preisler (1998)
<i>P. contorta</i>	23 - 41 cm	12-13	OR (eastern Cascades)	24 years	Current study
	> 25 cm	8 - 9	northeastern OR	8 years	Bull (1983)
	23 - 41 cm	15	WA (eastern Cascades)	81 years	Everett et al. (1999)
	not reported	37% fallen by year 9	northern CA, southern OR	9 years	Landram et al. (2002)
	mean = 29 cm	9 - 10	OR (eastern Cascades)	15 years	Mitchell and Preisler (1998)

Although DBH has a larger effect, species does affect half-life estimates and fall rates of snags. No statistical difference was found between *P. ponderosa* and *P. contorta* fall rates in our study. Bull (1983) reported that half-lives for *P. ponderosa* and *P. contorta* were within 1 year of each other in northeastern Oregon, supporting the similarity found in our estimates. At a stand level, we would expect total snag populations to fall more rapidly if *P. contorta* snags are the dominant species because these snags have a smaller average DBH than *P. ponderosa* in multi-age stands.

Somewhat unexpectedly, *Abies sp.* snags have slower fall rates than *Pinus sp.* Half-life estimates are 1, 2 and 3 years later for *Abies sp.* than *Pinus sp.* small, medium and large snags in Oregon's eastern Cascade forests. Landram et al. (2002) report half-lives for *Abies sp.* to be about 2 years longer than *P. ponderosa* snags of the same DBH. Morrison and Raphael (1993) also report that *Abies sp.* snags persisted longer than *Pinus sp.* snags, although the average DBH of *Abies sp.* snags was greater so direct species comparison is confounded by this discrepancy. These estimates support our finding that *Abies sp.* snags persist slightly longer than *P. ponderosa* snags, likely a result of different decay rates, wood strength and crown/stem weight.

Half-lives for small snags of all species correlate well with findings from other published studies, but our half-life estimates occur later for *P. ponderosa* and *Abies sp.* medium and large snags. Our *P. contorta* estimates fall within the range of other published values (see Table 2.7 for comparisons). Everett et al. (1999) reported half-lives for *P. contorta* and *P. menziesii* snags 23-41 cm DBH of 15 years, very similar to the results for *Abies sp.* and *Pinus sp.* in this study. They also found that only 21% of *P. ponderosa* snags >41 cm DBH had fallen by year 60 and reported half-lives for *P. menziesii* >41 cm DBH to be ~60 years post-fire, which is much longer than our expectation but supports the possibility that half-lives can be >20 years.

We believe much of the observed differences among studies are attributed to variation in environmental conditions. Environmental conditions (i.e. snow loads, winds, season lengths) experienced by a particular snag population will contribute to



the rate of fall and fragmentation of that population. At broad scales, variation occurs across geographic regions due to climate and weather variability. For example, along the Front Range of Colorado Chinook winds can exceed 75 mph, increasing fall rates. Schmid (1985) found half-lives to be as low as 5 years and as high as 10 years, depending on the snag population evaluated. These wind events do not always lead to increased snag fall, but as snags become increasingly decayed at the base or have snow and ice loads, the annual fall rate can appear high because of a single wind event. The effect of these events is likely more pronounced in medium and large snags because of their increased exposure to wind and snow/ice loads. Additionally, Chambers and Mast (2005) found that the basal area of live trees adjacent to snags reduced snag fall rates. Our study focused on high-severity fire areas, limiting the influence of adjacent live trees but suggests variability between studies can be expected; especially since live tree basal area is not always reported.

Small snags (i.e. <23 cm DBH) exhibit surprisingly similar fall rates among species and across broad geographic regions. The majority of published half-life estimates for small snags are ~7 years, which is equivalent to our estimate for *P. ponderosa* and *P. contorta* but 1 year less than *Abies sp.* in dry forests. Published half-lives suggest 50% of snags <23 cm DBH may fall as soon as 5 years and as long as 10 years. The upper range may be caused by snags > 23 cm DBH being incorporated into the estimate, suggesting 10 years may be a longer half-life than actually observed by small snags.

Differences in study design (i.e. length of study, time series vs. chronosequence approach) may explain some of the discrepancies in half-life estimates between this and other studies. Tracking individual snag populations over time allows for the investigation of many processes lost in chronosequence studies, particularly at the individual snag or local environment scale. At the same time, the inability to observe snag populations over long periods (i.e. >20 years) limits the reliability of these fall rate estimates across the entire life of a snag. Most studies observed snag populations for no more than 10 years, fewer years than some of our

estimated half-lives. In some cases, snag populations did not reach their half-life, limiting estimation of a snag population's true half-life (Bull 1983, Chambers and Mast 2005) and evaluation of the reliability of early years to predict long term attrition rates.

For example, Dahms (1949) found only 81% of snags 23-41cm DBH had fallen after 10 years, a fall rate much faster than our estimate. After re-sampling the snags at 22 years post-mortality, he found 89% of these snags had fallen. Our model estimates are similar to Dahms' 22 year estimates, predicting 89% of 23 – 41 cm snags falling by years 19 – 20 (95% confidence range of 17 to 23 years) but not his half-life estimates. The discrepancy between early estimates and correlation with older estimates suggest a random event (i.e. heavy snow loads, high winds) occurred early in the existence of Dahms' snag population, accelerating fall rates early but subsequently slowing under average environmental conditions. In this case, one of his point estimates is within the range of our estimates while the other is not. We see similar results for large snags, although even more variability is present.

Comparisons of half-lives are limited by the environmental conditions experienced by the sampled population(s) before reaching their half-life. Our chronosequence approach links multiple snag populations together across a broad geographic region and relatively long period. The estimated mean proportion of snags standing over time and variability around that mean incorporate the effects of stochastic environmental events that occur during each of these snag population's existence.

### ***Snag Breakage***

In general, over time there is an increasing proportion of snags with broken tops. There doesn't appear to be any general trends across diameter classes for all snag species but instead breakage trends vary by species (Figure 2.3). This variation likely arises because of the interacting effects of decomposition and tree physiological characteristics.

*P. contorta* snags due not readily break before they fall, and only snags <23 cm DBH showed any signs of top breakage. Our estimate of the number of *P. contorta* snags broken is supported by other research. Bull (1983) found that <10% of *P. contorta* snags broke during the first 8 years following mortality but by year 8 these snags had fallen, leaving no broken *P. contorta* snags standing. After 8 years, our estimates suggest no more than 8-9% of standing snags would have a broken top but more than 50% of the snag population will have fallen by this time. Everett et al. (1999) also found that *P. contorta* fell intact rather than breaking, leaving the dominant control of necromass transfer for *P. contorta* as whole snag fall.

*P. ponderosa* snags show significant breakage for all DBH classes, with no clear trend found across the diameter classes. We found that an additional 5% of small and large snags will have broken tops and an additional 7.4% of medium snags will have broken tops each year. Published studies report that *P. ponderosa* snags break at higher rates than *P. contorta* snags, although there may be an increasing breakage rate with increasing DBH (Bull 1983, Everett et al. 1999). Bull (1983) found that *P. ponderosa* snags >50 cm DBH had an average annual breakage rate of ~ 9% and a 3.5 year lag before initiation of breakage, resulting in >70% of standing snags having broken tops by 8 years after death. Snags <50 cm DBH had an average annual breakage rate of 5.63% and the same lag as larger snags.

Chambers and Mast (2005) also reported an increase in the proportion of *P. ponderosa* snags broken over time following fire. In the initial measurements of their two sites, 1% of the snags were broken 1 year following fire and 28% were broken 4 years following fire. No lag was present before top breakage occurred and the average annual rate of breakage for all snags was 1% and 7% for the same two fire sites. Three years later, 3% and 53% were broken at the two fire sites, indicating an average breakage rate of 1% and 7.6% for these two sites. The rates found in these studies are surprisingly similar to those found in our study, although the trends do not match exactly. Both studies indicate a non-linear trend in breakage rate similar to fall rates. In some cases, a lag may occur before the initiation of top breakage followed by a

rapid increase in top breakage. We assume that breakage rates are linear through time when modeling the fragmentation of snags, which may overestimate necromass transfer in the first 5 years.

*Abies sp.* snags exhibit a trend of increasing top breakage rate as DBH class increases, similar to *P. ponderosa* rates reported by Bull (1983). For large snags, the rate of breakage is higher for *Abies sp.* than *P. ponderosa* and is approximately the same rate as medium *P. ponderosa* snags. When averaging across all snag size classes, Landram et al. (2002) found that 14% of *Abies sp.* snags had broken tops after 8 years but only 11% of *P. jeffreyi* were broken. Although the diameter distributions are not known, this supports our conclusion that *Abies sp.* snags break at a higher rate than *P. ponderosa* snags.

Snag break-height increases with DBH and decreases with time since fire although no statistical significant difference was found between species. Everett et al. (1999) reported that *P. ponderosa* and *P. menziesii* heights decreased incrementally with top breakage. In that study, average snag height of *P. menziesii* 15 years following fire was 50% of the average height of a site only 1 year post-fire. According to our model, the average break height for a single 70 cm DBH snag 15 years following fire would be 45% of the original height. Considering snags may be standing intact or have broken once or more, the average reported by Everett et al. (1999) is close to our expected height average as modeled by “NecroDynamics”.

Studies on snag dynamics should consider snag top breakage because of its effects on wildlife habitat suitability and necromass transfer (Bull 1983, Chambers and Mast 2005). For example, Bull (1983) reported that 13 of 15 of the snags used for nesting had broken tops and that the cavities were excavated between 3 and 8 years following mortality. Chambers and Mast (2005) found that larger DBH snags with broken tops were more likely to be used as cavity nesting sites relative to unbroken, smaller snags. Additionally, snags that remain standing for > 25 years have a larger DBH and broken tops, suggesting snags will remain standing longer if their tops are broken (Morrison and Raphael 1993). The ecological value of broken snags and the

increased deposition of necromass (via this process) warrant incorporation of this information into the comprehensive understanding of snag dynamics.

### ***Snag and Log Decomposition***

We obtained decay estimates for snags and logs up to seven years post-mortality. We were limited to this time frame by the availability of fire sites, the ability to accurately date the time of death for snags, and our ability to know the year when snags were felled to the ground. At older fires, the year of death becomes difficult to determine since some trees survive several years before dying after a fire (Thies et al. 2006). Additionally, physiological (i.e. initial wood density) or microsite characteristics (i.e. soil depth, physiographic depressions) of snags might promote their retention and bias the sample towards snags with these qualities.

Different decomposition rates have been hypothesized for snags and CWD, caused primarily by variations in moisture infiltration and retention (Harmon et al. 1986, Harmon 2009). In dry forest systems, snags are hypothesized to have lower decay rates than CWD with the opposite trend occurs in wetter forest types (Harmon et al. 1986, Harmon 2009). Tradeoffs between optimal moisture and temperature regimes develop unique patterns in decay rates of woody material. At local scales, these conditions create different rates of decomposition between dead wood among species and their position within a forest. Refer to Chapter 1 for an explanation of the decomposition environment and Figure 1.1 for a graphical display.

Comparison of our *P. ponderosa* snag and CWD decay rates, obtained from the same fire sites, confirms that snags have a lower decomposition loss rate-constant than CWD in dry forest environments. Our estimates show no statistically significant decay for *P. ponderosa* snag sapwood or heartwood but statistically significant decay for both stem components in CWD.

Snags have reduced decay rates relative to CWD, but the difference varies by species. *Abies* sp. snags exhibit significant decay, with an estimated decomposition loss rate-constant of  $k = 0.01488 \text{ yr}^{-1}$ , which is a higher decay rate than *P. ponderosa*

snag sapwood and heartwood ( $k \approx 0.0 \text{ yr}^{-1}$  for both substrates). Although *P. contorta* was not included in this decomposition analysis, Busse (1994) reported that *P. contorta* snags do not decay, as indicated by the lack of a difference in density between live *P. contorta* trees and fallen snags.

*P. ponderosa* sapwood is not as decay resistant as its heartwood, but it has been suggested that *P. ponderosa* sapwood is slightly more decay resistant than *Abies sp.* wood (Bull et al. 1997). Additionally, Bull et al. (1997) suggest *Abies sp.* snags retain bark longer than *P. ponderosa* snags, increasing moisture retention and improving the decomposition environment of this species. We are unable to discern what causes different decay rates, but our estimates confirm the effect of species variation under the environmental conditions snags are exposed to in dry environments.

The effect of substrate quality is evident in CWD, but does not appear to be as different in less decay resistant substrates (i.e. *P. ponderosa* sapwood, *Abies sp.*). Our *P. ponderosa* sapwood decomposition loss rate-constant is similar to other non-decay resistant woody necromass in this geographic region. *Abies grandis* decomposition loss rate-constants obtained in central Oregon are reported as  $k = 0.038 \text{ yr}^{-1}$ , compared to our estimate of  $k = 0.0362 \text{ yr}^{-1}$  for *P. ponderosa* sapwood (Harmon et al. 2007). Additionally, in a study conducted on the Winema National Forest of southern Oregon, *Abies concolor* was found to have a decomposition loss rate-constant of  $k = 0.035 \text{ yr}^{-1}$  (Harmon et al. 1996, unpublished data). The similarity in these decay rates suggests non-decay resistant woody material will decay at a similar rate, regardless of species, after the substrate has been transferred to the soil surface (Harmon 2009).

Although variation in decay rates is less noticeable in non-decay resistant material, *P. ponderosa* heartwood does decay slower than its sapwood. Heartwood of this species decays at 45% of the rate of sapwood, primarily as a result of heartwood's decay resistant tannins, resins, waxes, polyphenols, etc. (Harmon et al. 1986). When combined, sapwood and heartwood decomposition loss rate-constant is  $k = 0.0240 \text{ yr}^{-1}$ . An estimate from Pringle Falls Experimental Forest found the *P. ponderosa* decay

rate constant to be  $k = 0.011 \text{ yr}^{-1}$ , although the uncertainty is considered high (Harmon et al. 2007). The latter estimate is within the 95% confidence bounds of ours, although close to the lower bound. Differences in environmental conditions between these sites, related primarily to microsite conditions, likely contribute to this disparity. Additionally, difference between these rates is partially a function of the proportion of heartwood within the sample, related to DBH and age of the sampled logs.

The ratio of heartwood to sapwood is important for understanding how long a species with decay resistant heartwood will remain on the landscape (Bull et al. 1997). The development of heartwood in ponderosa pine ecosystems is variable and primarily dependent on the amount of leaf area being supported (Waring et al. 1982). Allometric equations for estimating sapwood/heartwood ratios are not available, but some research provides insight into differences across broad diameter classes. For example, Bull et al. (1997) reported that the proportion of heartwood increases with tree age and may constitute from 25% to 75% of tree stem volume. Small diameter trees will have little to no heartwood so the decomposition loss rate-constant for the total CWD will be at or close to the rate of sapwood decay found in this study. The opposite is true for large trees, where the majority of the stem may be heartwood. In this case, the total log decay will be closer to the heartwood rate and persist on the landscape longer.

The effect of reduced decay rates in standing versus surface woody material creates different decomposition trajectories. For example, *Abies sp.* snag decay rate is <50% of its log decay rate resulting in different decay trajectories for each piece of a snag that transfers to the soil surface, dependent on break and/or fall year. Figure 2.8 depicts the decay trajectory of individual pieces of necromass that transfer at year 10 and 20 for an *Abies sp.* snag. Each species within a given stand will have its own decomposition trajectory depending on breakage events, year of fall and the difference in surface versus standing decay rates.

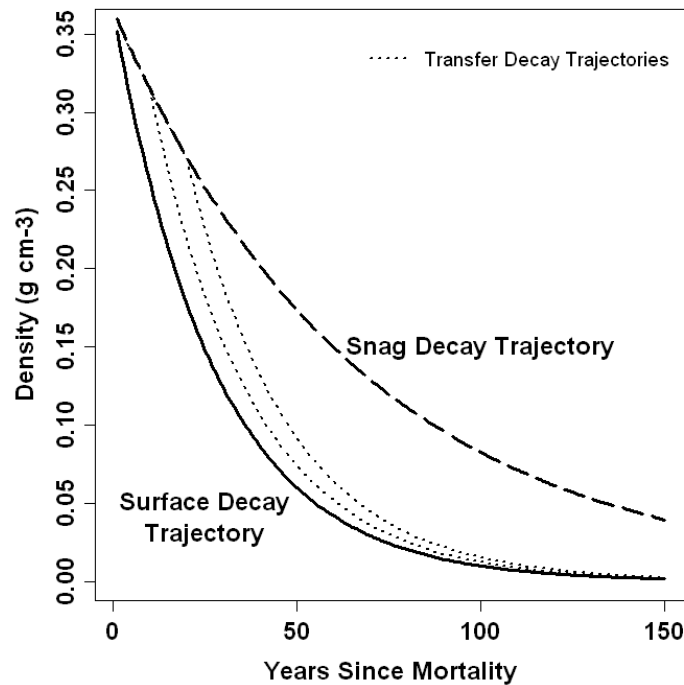


Figure 2.8: Decay trajectories for *Abies sp.* necromass using surface and standing decomposition rates. Dotted lines depict necromass density trajectories when transferred at years 10 and 20. Each broken or fallen piece of necromass will follow its own density loss trajectory depending on its transfer year and variable decay rates.

### ***Integrated Snag Dynamics***

Snag fall, breakage and decomposition interact to develop unique patterns of snag biomass deposition for each species within the dry-mixed conifer forests of the eastern Cascades. Horizontal and vertical physical forces act upon snags causing snag fall and breakage. Horizontal force develops as wind penetrates into snag crowns, encountering resistance that applies physical force to the snag stem. Vertical force is applied as gravitational forces pull on crown and snag biomass. The total amount of applied force experienced by an individual snag is dependent on wind speed, crown position, architecture and biomass. As these forces are applied, breakage and fall is resisted the by tensile and compressive strength of a tree stem and anchoring effect of roots (Quine and Gardiner 2007).



High breakage rates for *P. ponderosa* (across all diameter classes), increasing breakage rates with increasing diameter class for *Abies sp.* snags, and low breakage rates for *P. contorta* snags suggest crown and upper stem biomass has a significant effect on breakage rates. Allometric equations have been developed for predicting crown biomass from DBH, increasing exponentially with increasing DBH. *P. contorta* has the lowest crown biomass of these species, followed by *Abies sp.* and *P. ponderosa*. Our breakage rates suggest the ratio of crown biomass to stem diameter effects breakage potential. More biomass present in upper portions of snags and the “sail area” of the crown increases the forces exerted on tensile and compression strength of stem wood as it sways in windy conditions. Breakage occurs when a critical threshold of applied force is reached, dependent on the total applied force and resistive forces of the wood (Quine and Gardiner 2007).

Wood strength provides tensile and compressive forces that resist breakage, but ultimately fails with increasing force and/or decreasing wood strength. Wood strength is directly related to wood density (Bergman et al. 2010). For the sampled species, *Abies sp.* has the lowest wood density followed by *P. ponderosa* and *P. contorta* (0.365, 0.38 and 0.41 g cm<sup>-3</sup>, respectively). Additionally, *Abies sp.* are the only snags that exhibit significant decomposition throughout the stem. Low strength (low density) and persistent decomposition increase the breakage potential of this species. The effect of wood strength is most apparent when comparing large *Abies sp.* and *P. ponderosa* snags, where *Abies sp.* break at a higher rate than *P. ponderosa* snags. *P. contorta* breakage rates further support the cumulative effect of crown biomass and stem strength on top breakage. This species has low crown biomass, relatively high wood density and lower breakage rates than the other species in this study. Whether crown biomass or wood strength is the controlling factor evidently depends on species and snag DBH, but the critical thresholds for these variables are beyond the scope of this study.

Snag fall is affected by the same factors that affect breakage, but additional interactions occur with top breakage. Coarse root biomass can be directly estimated

from total aboveground biomass (Grier and Logan 1977), suggesting that the total anchoring biomass of roots is higher for larger snags. Fall rates decrease with increasing DBH, supporting this concept. Additionally, the amount of biomass in upper portions of a snag affects the rate of fall. *Abies sp.* fall at a slower rate than *P. ponderosa* snags partially because of their lower initial biomass, high breakage rates, and standing decomposition. Combined, these factors reduce the amount of biomass remaining standing in *Abies sp.* relative to *P. ponderosa* over time. A tipping point is reached more rapidly for *P. ponderosa* snags since the amount of biomass maintained within the standing stem and crown over time are greater. The combined effect is slight, causing an increase of 1, 2 and 3 years (small, medium and large snag DBH classes, respectively) in half-lives for *Abies sp.* relative to *P. ponderosa* snags.

When snag dynamics are modeled together, the non-linear trend of snag fall is preserved in snag biomass fragmentation but there is a reduction in the early lag period, regardless of pre-fire stand conditions (Figure 2.6). This is a result of snag decomposition (slight effect) and biomass transfer through top breakage. High biomass sites maintain more biomass longer than other sites, but this is a result of the total number of snags present on site and the increased number of large snags. The proportion of initial biomass remaining on site over time is similar across the biomass gradient.

Although the proportion of snag biomass remaining standing over time is similar across each example stand, there is a difference in CWD accumulation primarily due to variation in decay rates of the species present in that stand. Higher biomass stands tend to support large *P. menziesii* and *P. ponderosa* snags. Both species exhibit moderate to high decay resistance that increases with snag age and size, effectively decreasing the total stand decay rate. This effect is most evident when evaluating the accumulation of CWD (refer to Figure 2.6).

Prior to peak CWD accumulation, necromass input from snag fragmentation is greater than decay loss. In the example stands provided, peak accumulation of CWD totals ~68%, 59%, and 52% of initial biomass for the high, medium and low biomass

sites, respectively. Necromass loss represents the sum total of standing decomposition and surface decomposition from transferred material. Since fragmentation rates are similar for each site, the variation in total necromass loss prior to the peak is a result of snag species and size. The lower biomass site has smaller snags and higher proportions of *Abies sp.*, resulting in increased standing and surface decay rates. This difference primarily affects the amount of total CWD, but can shift the timing of peak accumulation as evident in peak accumulation occurring 3 years (23 years post-fire) earlier for the low biomass site relative to the other sites.

Since surface decay rates are greater than standing decay rates, carbon emissions follow the same trend as biomass accumulation. The apparent discrepancy in the actual peak year ( $\sim 1$  year) is a result of the models accounting (i.e. emissions are estimated prior to transfer of and summation of total necromass). Figure 2.9 depicts the carbon flux from aboveground CWD for the three example sites. Peak emissions occur at the same time as peak CWD accumulation for all sites. Included in the figure is an estimation of carbon flux from CWD when negative exponential decay functions using surface decay rates are applied to all CWD immediately post-fire.

Simulation models assuming surface decay rates immediately following mortality over predict the flux of carbon from CWD necromass during the first 20 years following fire and under predict thereafter. The magnitude of the over prediction ranges from 1.2 to 2 Mg ha<sup>-1</sup>, depending on the amount of biomass present on site. This difference decreases with time until the estimates are equal at approximately year 20. After this point, assuming surface decay rates immediately following mortality under predicts the carbon flux. Figure 2.10 represents the same effect but includes FWD decomposition.

The apparent lag in decay flux due to reduced decay rates for snags conserves carbon in forests sites relative to an assumption of equal decay rates by necromass position.

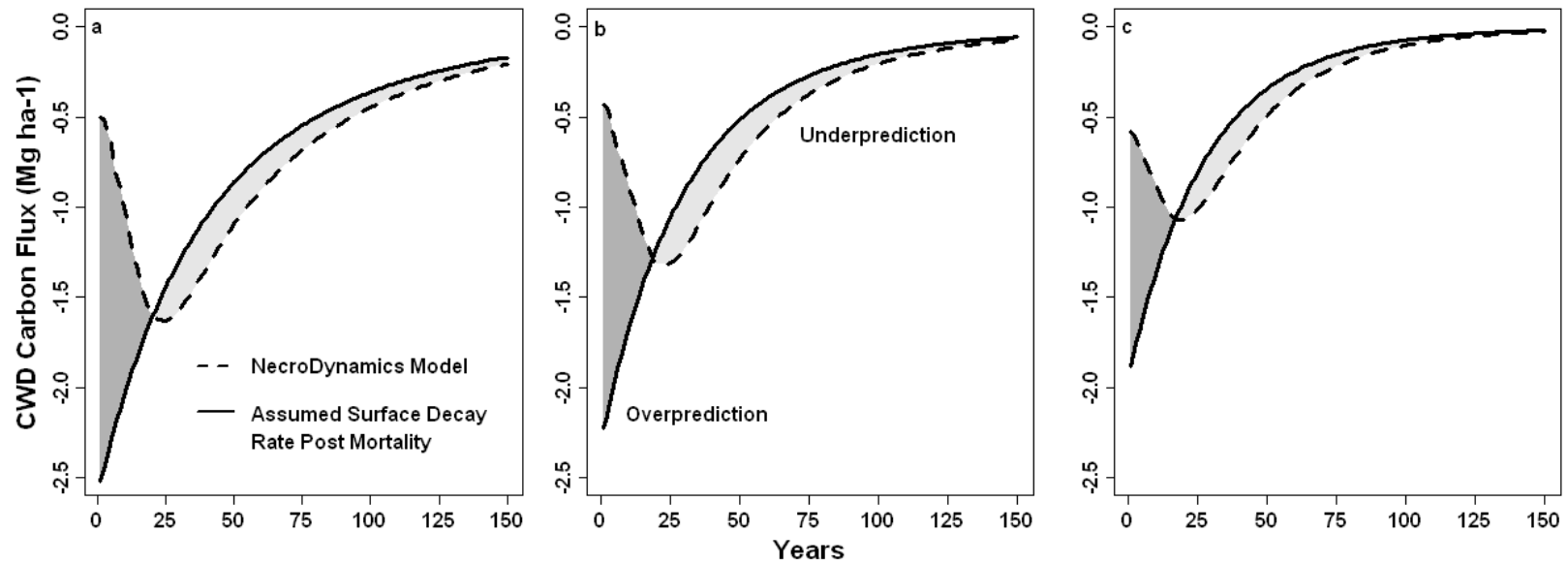


Figure 2.9: Mean carbon flux from CWD following high-severity fire. a = high biomass stand, b = medium biomass stand, and c = low biomass stand. Peak flux is delayed due to the difference in decomposition loss rate-constants for snags and logs and the process of snag fragmentation.

This effect reduces the expected impact of fire on net ecosystem productivity (NEP) relative to the assumed surface decay rate for all material, although NEP will remain negative due to the reduction in NPP from mortality and persistent soil carbon and detritus emissions (Meigs et al. 2009). New growth offsets much of the carbon flux, reducing the slope of NEP but positive recovery occurs later and at a lower rate relative to the negative exponential model.

Ultimately, recovery of pre-fire carbon stocks will take longer than recovery to positive NEP, likely taking longer than 50 years (Harmon 2009). In the event of a subsequent fire and the additional loss of CWD, recovery of carbon stocks could take several hundred years depending on consumption of CWD and additional mortality from the fire event.

## **Conclusions**

Snags and logs provide structural habitat valuable to multiple vertebrate and invertebrate species in dry-mixed conifer forest of Oregon (Thomas et al. 1979, Bull et al. 1997) and are important for saprophyte communities, ecosystem nutrient cycling, and the carbon balance of a forest (Franklin et al. 1987, Harmon et al. 1986, Harmon 2009, Triska and Cromack 1979). The availability of snags and logs to meet ecological needs is dependent on their creation, fragmentation and decomposition (Harmon 2002).

One of the most evident effects of fire disturbance is a pulse of tree mortality, dramatically increasing the abundance of snags within an ecosystem. Once created, snags begin a dynamic process of breaking, decaying and falling. Once transferred to the forest floor, snags become logs and continue to provide ecological functions. The rate and amount of snag transfer is dependent on decomposition, top breakage and fall. Within a particular geographic region, each of these aspects of a snags dynamic life varies dependent primarily on DBH and species.

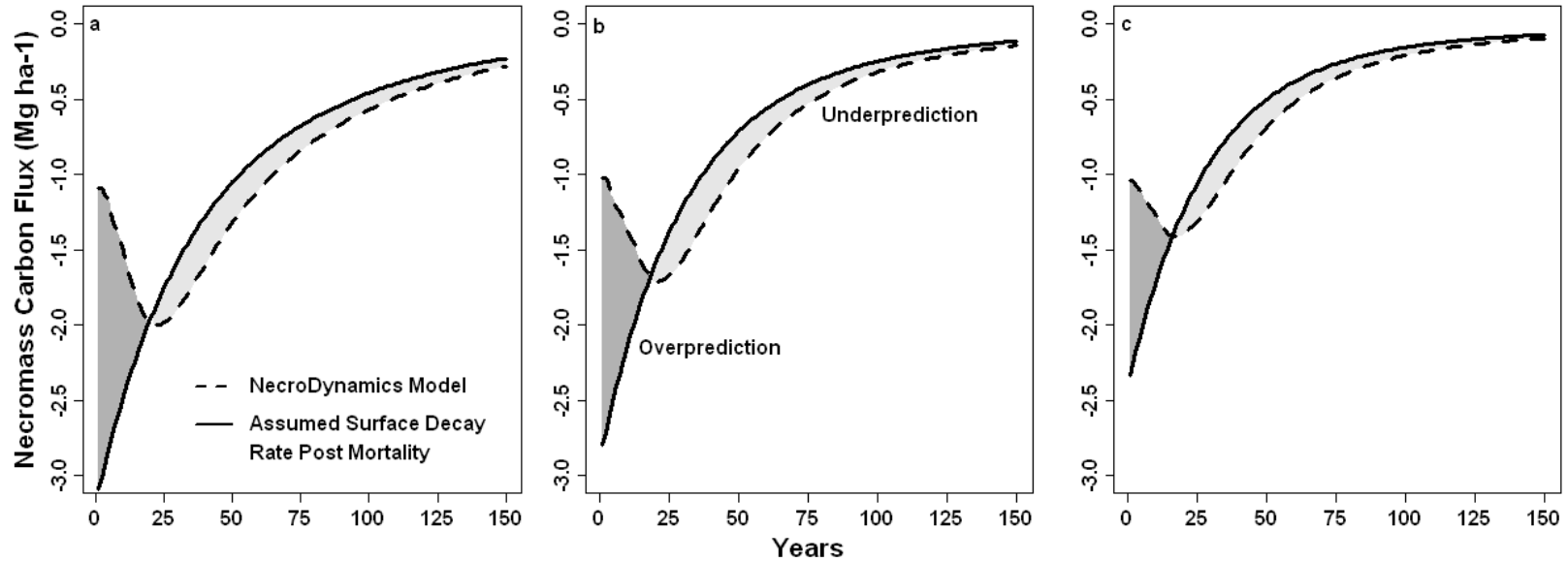


Figure 2.10: Mean carbon flux from legacy CWD and FWD following high-severity fire. a = high biomass stand, b = medium biomass stand, and c = low biomass stand. Peak flux is delayed due to the difference in decomposition loss rate-constants for snags and logs and the process of snag fragmentation.

Within the dry-mixed conifer forests of Oregon's eastern Cascades, *P. contorta* and *P. ponderosa* of the same DBH fall at rates that are essentially the same, but *P. ponderosa* breaks much more readily and *P. contorta* averages much lower DBH in natural stands. *Abies sp.* snags tend to persist longer than *Pinus sp.* snags, but large *Abies sp.* snags break more readily. Combined, the physical transfer of snag necromass in this region follows a predictable trend that includes a lag period, followed by accelerated deposition, ultimately slowing after about 20 years.

In dry forest environments, decomposition occurs in snags and logs but snags decay at a reduced rate relative to logs. The magnitude of this difference varies by species. *P. ponderosa* does not appear to have significant stem decomposition while standing while *Abies sp.* does. As these snags transfer to the forest floor, their decay rates accelerate but still vary by species. The abundance of CWD available for ecological needs is dependent on the species composition, DBH distribution and snag necromass transfer rate.

Coarse woody detritus dynamic and variable decay rates result in a complex pattern of carbon emissions from this material. Maximum carbon emissions occurs greater than 20 years post-fire, suggesting carbon is conserved on the site longer than assumed by using log decay rates for all CWD immediately following mortality. If maximizing carbon storage with active management becomes the primary objective of a dry-mixed conifer forest, this study provides a comprehensive basis for comparing the success of accomplishing this objective.

### **CHAPTER 3: WILDLAND FIRE FUEL SUCCESSION FOLLOWING HIGH-SEVERITY FIRE IN DRY-MIXED CONIFER FORESTS OF OREGON'S EASTERN CASCADES**

Fire is a natural ecosystem process that exerts significant force on the spatial distribution of forest vegetation and stand structure, with cumulative effects occurring over long temporal periods (Franklin et al. 2002). Dry-mixed conifer forests in Oregon are fire adapted ecosystems with a mixed-severity fire regime, experiencing fire disturbance at intervals ranging from 9 - 62 years, although high-severity fire may have fire free intervals >200 years (Agee 1993). One of the most evident effects of fire disturbance is a pulse of tree mortality, dramatically increasing CWD within the system. The magnitude of this transfer is most evident in high-severity fire areas where >75% of tree mortality occurs (Agee 1993). The extent of fire disturbance has increased across the Western United States during the past 30 years and future climate scenarios predict a continuation of this trend (Brown et al. 2004, Westerling et al. 2006). As fire extents become greater, the amount of CWD in fire adapted systems will increase beyond contemporary levels.

Mixed-severity fires alter stand structure and understory composition by creating a spatial mosaic of stand structures within a fire boundary (Hessburg et al. 2005). Post-fire stands vary in number of live trees, snags, logs and surviving understory vegetation (Meigs et al. 2009). These biological legacies are key components of forest structure and wildlife habitat altered by fire and management practices. Unique bird communities have been found across burn-severity gradients, relying on biological legacies for nesting, roosting, and foraging (Fontaine et al. 2009, Saab 2007). Increased variation in stand structure provides opportunities for movements of avian species across the landscape as habitats increase or decrease in suitability. Stands develop on successional trajectories depending largely on the distribution of these biological legacies following a fire (Franklin et al. 2002).

European settlement altered natural disturbance regimes in these forests beginning in the mid-19<sup>th</sup> century through a reduction in human ignited fire, livestock



grazing, timber harvesting and more recently, mechanized fire suppression (Agee 1993). The effects of altered disturbance regimes on stand structure are obvious when considering live trees. The lack of fire in natural stands has increased the number of small diameter trees present on site, resulting in increased susceptibility to stand replacement fire across broad landscapes (Spies et al. 2006) and increased competitive stress on large legacy trees (Fellows and Goulden 2008). Today, all species in dry-mixed conifer forests of Oregon have higher densities of stems <41cm DBH and lower densities of larger diameter trees (Merschel 2010). The ecological effects of these changes are becoming apparent as our forests are valued for ecosystem services beyond just timber products.

Less obvious, perhaps, are the effects of these changes on CWD. Harvest regimes facilitated live tree growth and removed the standing timber before competitive stresses, biotic agents and/or irregular mortality (e.g., lightning) created CWD. In the event of moderate to large disturbances, salvage logging would remove much of the newly created CWD, further reducing the presence of this material on the landscape. The loss of these biological legacies and the reduction of large trees that develop into large CWD have reduced the presence of this material on landscapes.

Increasingly, public land managers are tasked with providing multiple resources from the land they manage. In western forests, contemporary management strategies tend towards ecological restoration and/or ecologically sustainable management practices that include management of dead wood resources as important structural components of forest ecosystems. The retention of snags and logs are increasingly blended with fuels reduction treatments in dry forests in order to meet ecological and management needs (Brown et al. 2003). As more post-fire environments remain unsalvaged, there are increasing concerns regarding the impacts of leaving high amounts of woody fuel in these environments, potentially increasing future fire severity at the time of the next fire disturbance (Sessions et al. 2004).

Fire severity is often measured by the amount of overstory vegetation killed during a fire (Miller and Thode 2006, Thompson et al. 2007). It is important to

consider the impacts of management strategies on all fuel types since vegetative mortality may occur through three distinct mechanisms that are a result of different fire behavior: 1) consumption and/or scorch of photosynthetic structures, 2) cambial necrosis, and 3) root necrosis (Agee 1993). Fuel particles <0.62 cm and cured herbaceous fuels are the primary driver of rate-of-spread because of their rapid combustion due to high surface-area-to-volume-ratio (Rothermel 1972). Byram (1959) showed that rate of spread is positively related to fireline intensity and Van Wagner (1973) provided equations to estimate scorch height from fireline intensity, which could lead to the initiation of a passive or active crown fire (Scott and Reinhardt 2001). Additionally, shrubs can contribute to significant rates-of-spread and heat release as noted by chaparral fuel models exhibiting the greatest fireline intensity (Scott and Burgan 2005).

Cambial necrosis of the aboveground tree stem is a function of heat release and residence time of flames (Dickinson and Johnson 2001), while root necrosis is a result of flaming and glowing combustion from CWD as well as the smoldering surface organic matter (Dickinson and Johnson 2001, Monsanto and Agee 2008). The potential for mortality by multiple mechanisms suggests the alteration of one fuel bed component may mitigate vegetative mortality through a single pathway without effectively reducing the potential of tree mortality by all pathways.

The objective of this study was to quantify succession of wildland fire fuels following high-severity fire. To date, only 1 empirical study has quantified fuel accumulation following high-severity fire (McGinnis et al. 2010). To meet these objectives, we explored the following questions:

1. What are the successional trajectories of fire generated CWD following high-severity fire and how do they contribute to fuel succession?
2. What is the rate of accumulation of fine woody detritus, litter, duff, herbaceous and shrub fuel layers during the first 24 years following high-severity fire in dry-mixed conifer forests of Oregon's eastern Cascade Mountains?

## Methods

Successional fuel dynamics occur over relatively long temporal scales making controlled studies difficult to conduct. For this reason, we used a chronosequence approach to make observations and obtain results that would otherwise take >20 years to gather. This approach substitutes space for time, but requires making the assumption that site conditions and history are the same for each site used in the analysis (Harmon et al. 1986). This assumption may not be true for all environmental variables, but we are confident broader trends in fuel dynamics are valid.

Management practices, including fire suppression and salvage logging, limited the number of fire sites available for sampling. A total of 7 fire sites were suitable for this study. Each fire site was included in the chronosequence beginning 1 year post-fire and ending 24 years post-fire (Figure 3.1).

### *Site Selection*

We focused on fuel successional dynamics in dry-mixed conifer forests of Oregon's eastern Cascades. Sample plots were targeted at mature to old-growth dry-mixed conifer forests that have experienced high-severity fire (> 75% overstory mortality). We focused on these forest types because they represent the most common post-fire environment remaining unsalvaged because of their ecological benefits, but also contain large amounts of fire generated CWD that may contribute significantly to future fire severity.

All fires occurring during the past 100 years in the eastern Cascades were considered for use in this study. Deschutes and Winema National Forests, BLM, Oregon Department of Forestry, and Crater Lake National Park GIS Fire Databases were queried for potential fires. Fire boundaries were reduced by dry-mixed conifer plant association groups consistent with the *Abies grandis* and *Abies concolor* Zones of the Eastern Cascades (Franklin and Dyrness 1988). Any portion of fire with evidence of more than a single fire event, or salvage logging/harvesting during this

period was excluded as a potential site. Table 3.1 summarizes the average environmental conditions of the fires sampled in this study.

When available, burn severity maps were used to extract high-severity fire areas within a fire boundary (Miller and Thode 2006). When burn severity maps were unavailable, the boundaries of high-severity fire areas were delineated using aerial photography. Silvicultural records, stand age data and aerial photo interpretation removed areas with a history of harvesting pre- or post-fire. High-severity fire sites proposed for salvage logging within the Black Crater and GW fires were excluded to maintain plot integrity after sampling.

GIS polygons were created around all potential sampling sites. The polygons were further reduced by a 50 meter buffer from roads, perennial streams, and green forest edges to ensure sampling occurred within high-severity areas only. All plots were spatially separated by a minimum of 200 meters to reduce spatial autocorrelation and randomly located using ARC GIS Hawth's Tools random point generator (Beyer 2004, ESRI 2009).

### ***Snag Sampling***

All snags were marked and measured within each 50 x 50 m plot by systematically gridding the entire plot. Snag characteristics included species, diameter at breast height (DBH), total height, decay class, condition, time of death and whether or not insects had colonized the stem. A snag's condition was noted as standing whole, standing broken, or fallen. Broken snags had a visible break point along the main stem greater than 2 meters in height. Snags broken below 2 meters in height were considered fallen. Height was not collected for fallen snags because of the difficulty of locating all portions broken during the snags life. Species was determined using characteristics described by Parks et al. (1997).

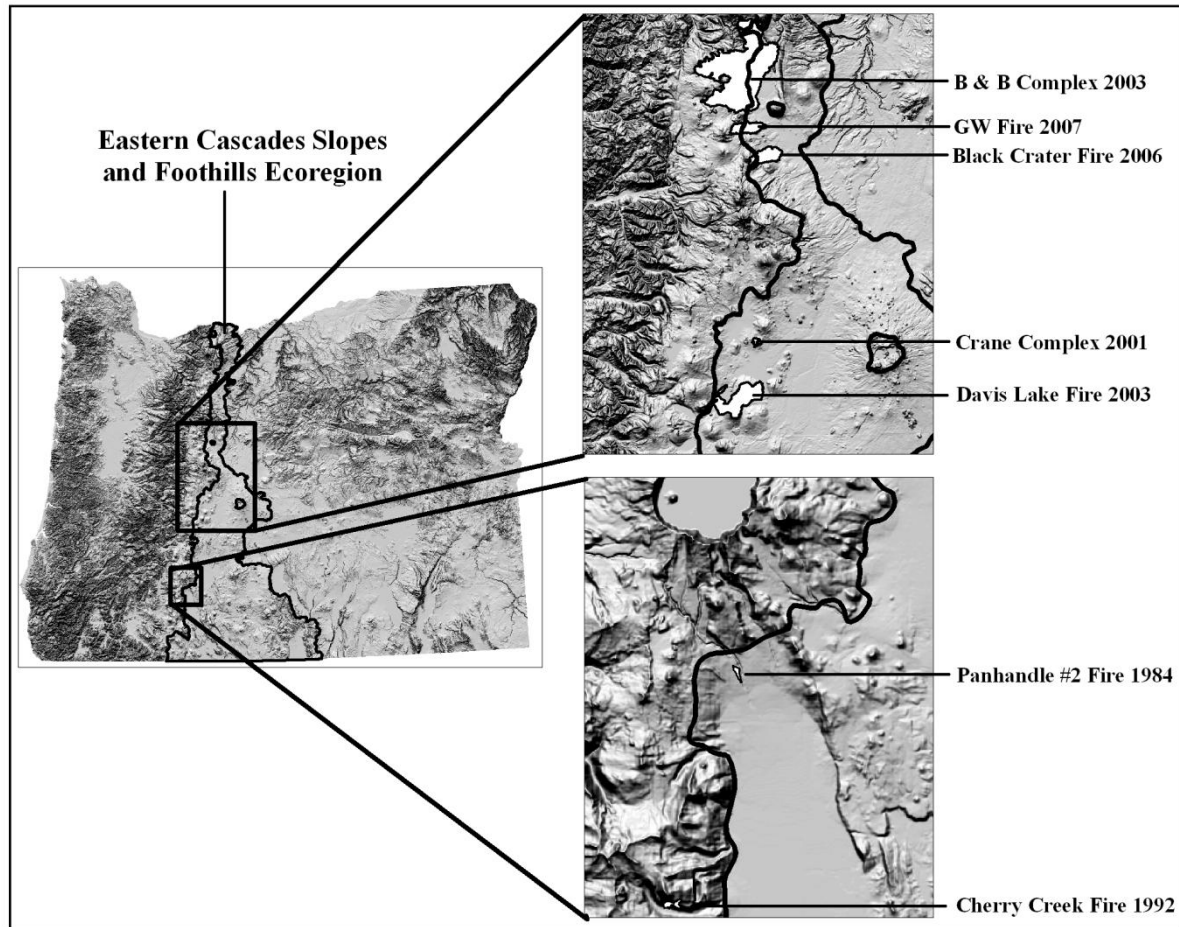


Figure 3.1: Map of fire site locations sampled in this study. The “Eastern Cascades Slopes and Foothills Ecoregion” is consistent with ecoregions developed by the Western Ecology Division of the EPA (Omernick 1987).

Table 3.1: Sampled fire sites within the dry-mixed conifer forests of Oregon's eastern Cascades. All sampling occurred during the summers of 2007 and 2008. Average elevation is the average of all plots from that fire. Climate data obtained from PRISM Climate Group website (PRISM Climate Group 2010).

Fire Site	Fire Year	Location	Year Sampled	Total Hectares	Average Elevation (m)	Average PPT (mm)	Average Tmax (°C)	Average Tmin (°C)
GW Fire	2007	Deschutes N.F.	2008	2977	1301	1052.07	12.72	0.43
Black Crater Fire	2006	Deschutes N.F.	2008	3807	1445	909.54	11.82	-0.60
B & B Complex	2003	Deschutes N.F.	2008	36733	1159	1147.56	13.18	0.72
Davis Lake Fire	2003	Deschutes N.F.	2008	8572	1551	843.53	12.12	-0.84
Crane Complex	2001	Deschutes N.F.	2008	289	1478	804.73	12.82	-0.54
Cherry Creek Fire	1994	Winema N.F.	2007	134	1722	1418.16	11.20	-0.14
Panhandle 2	1984	Crater Lake N.P.	2008	49	1419	889.89	14.19	-1.13

Pre-fire killed snags and live trees were also sampled during the inventory. Snags were determined to be dead at the time of the fire when > 5% of the sapwood was consumed or converted to char during the fire. Within the Biscuit Complex Fire of 2002, it was found that < 2% of live tree stem wood was combusted during high-severity fire (Campbell et al. 2007). Relatively high amounts of pre-fire killed snags occurred on the B & B Complex due to spruce budworm mortality in the preceding decade. The same characteristics were collected for live trees and pre-fire killed snags. Additional details on snag dynamics and the “NecroDynamics” model development can be found in previous chapters of this document.

### ***Fine and Coarse Woody Fuel Sampling***

The line intercept method was used to sample fine and coarse woody detritus by 1hr (0-.62cm), 10hr (.63-2.54cm), 100 hr (2.55-7.62cm), and 1000hr (>7.62cm) size classes (Brown 1974). All fine woody transects (FWD, <7.62 cm diameter) began at 15 meters from plot center and proceeded towards plot corners (NW, NE, SW, SE) and surface CWD transects were 30 meters diagonal to corners and 20 meters in the cardinal directions (Figure 3.2). Surface CWD biomass was calculated using volume equations and mean decay class densities reported by Harmon and Sexton (1996) for each surface CWD encountered.

### ***Live Herbaceous and Woody Fuel Loading***

Four 2-meter diameter circular subplots were placed 15 meters from plot center in the cardinal directions (Figure 3.2). All live herbaceous vegetation was collected in within theses subplots and brought back to the Wildland Fire Lab at Oregon State University for processing. This material was oven dried at 55 °C until its weight reached low, stable moisture condition ( $\pm 0.3$  g, ~72 hours). Stem basal diameters and species were collected for all woody stems originating within the subplot. Average height for woody species was also recorded. Species specific allometric equations

were used to estimate the total above ground biomass of woody species at these sites (Means et al. 1994).

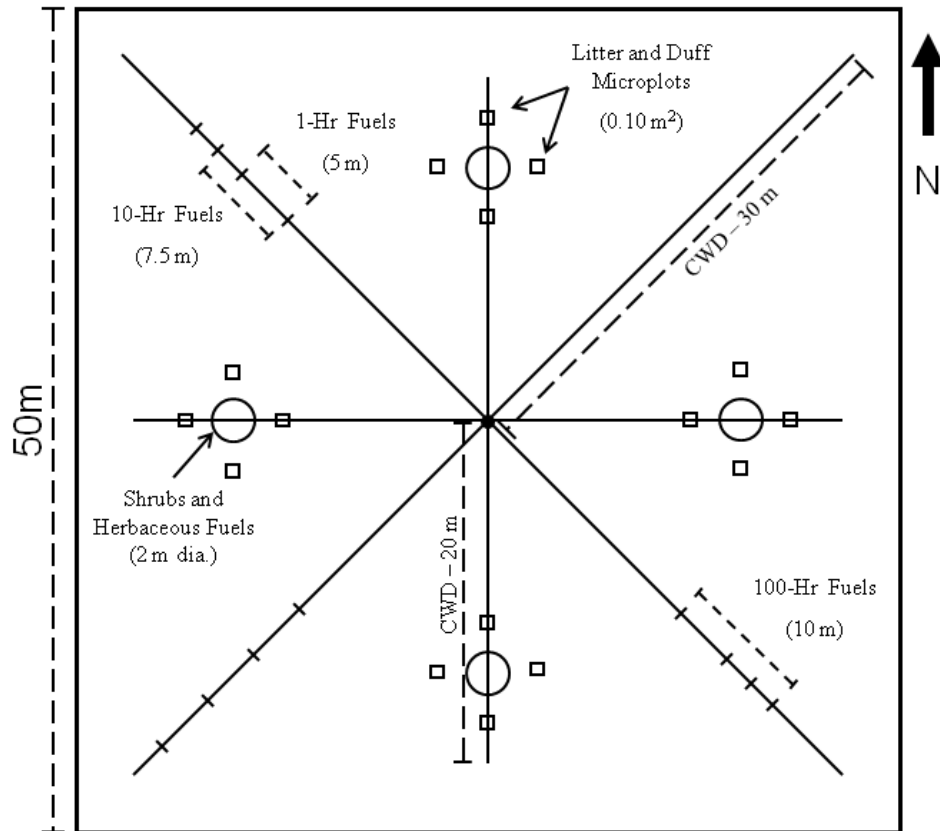


Figure 3.2: Snag dynamics and fuel succession plot layout. Total transect length varied by size class (1-hr = 20 m, 10-hr = 30 m, 100-hr = 40 m, CWD = 200 m).

### ***Litter and Duff***

Litter and duff were sampled at 16 - 0.10 m<sup>2</sup> microplots located 2 meters to the N, S, E, and W of the subplot centers (Figure 3.2). Litter and duff layers were separated by their compactness, decayed state, and presence of fungal hyphae. Average depth of each layer was measured to the nearest 0.5 cm, collected and weighed. One litter and duff sample was randomly selected from each “subplot group” (i.e. 4 samples per plot) and taken back to the lab for processing. This material



was oven dried at 55 °C until weight equilibrium was reached ( $\pm 0.5$  g, ~96 hours). The moisture content found in the dried samples was applied to the litter and duff sample field weights from their respective “subplot groups”. All 16 samples were averaged to obtain the mean plot litter and duff weights.

### **Data Analysis**

Stepwise model selection was used to determine the best fit model for predicting fuel biomass during 24 years of succession. Estimates for surface CWD, FWD, litter and duff accumulations were determined using linear regression analysis for each fuel bed separately using PROC REG in SAS 9.2 (SAS Institute Inc. 2008). All model assumptions were evaluated and met before proceeding with the analysis.

### **Results**

Figure 3.3 presents the fuels data for FWD, litter, duff and shrubs with loess smoothed lines. Time since fire was a significant predictor of 1-hr fuel biomass during 24 years of succession, accumulating within 1 year of the fire event. The positive intercept value was a result of rapid deposition of scorched canopy twigs, not incomplete combustion (Table 3.2). Each year there was an accumulation of 0.0376 Mg ha<sup>-1</sup> of 1-hr biomass, but the majority of accumulating 1-hr biomass was assimilated into the litter layer. Input rates gradually exceeded assimilation and decomposition rates as portions of individual shrub canopies died and senesced dead branches (estimate = 0.03967 Mg ha<sup>-1</sup> of 1-hr fuel biomass for each 1% increase in dead shrub canopy). Our observations suggest shrub canopy mortality occurred primarily after 15 years of post-fire development.

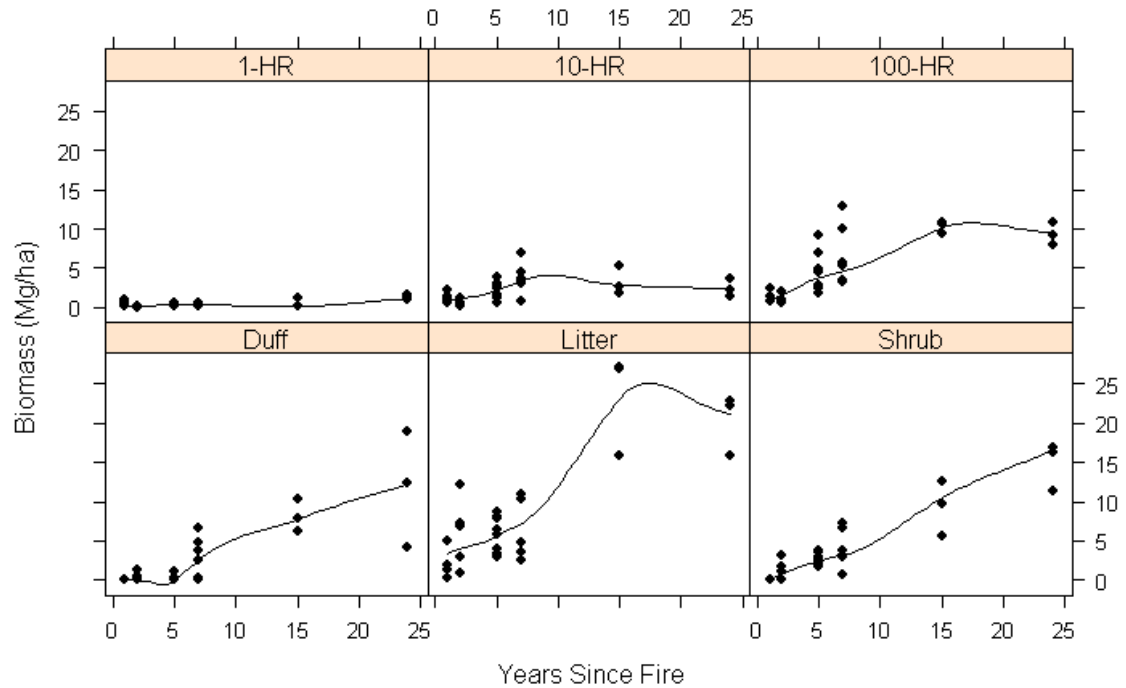


Figure 3.3: Twenty-four years of fine fuel and shrub accumulation following high-severity fire. The lines are loess smoothed lines, not predicted regression lines. In most cases, multiple predictor variables are significant for each fuel layer.

10-hr biomass exhibited the highest variability and least amount of variation explained by regression analysis. 10-hr biomass accumulated within 1 year following fire disturbance (back-transformed intercept =  $0.91 \text{ Mg ha}^{-1}$ , 95% CI = 0.67 to 1.25) primarily as a result of small branch fall from scorched snags within 1 year. Accumulation in successive years was best predicted by the number of snags fallen or broken (Table 3.2). For each snag that fell or broke, median 10-hr biomass increased by a multiplicative factor of 1.01 (95% CI= 1.006 to 1.013).

A  $1 \text{ Mg ha}^{-1}$  increase in litter biomass resulted in a decrease of  $0.0384 \text{ Mg ha}^{-1}$  of 1-hr biomass. On average, litter biomass accumulated at  $0.897 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ , resulting in relatively little net total accumulation of 1-hr fuel biomass each year (sample mean of  $1.23 \text{ Mg ha}^{-1}$  after 24 years of succession).

Table 3.2: Regression coefficients for estimating woody detritus biomass. <sup>1</sup> indicates log transformation of dependent variable. Biomass estimates were determined using the line intercept method (Brown 1974).

Dependent		Independent	95% Confidence Intervals				
Fuel Type	Variable	Variable	Estimate	Std Error	p-value	Lower	Upper
1-hr (0-0.62cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	0.29885	0.06438	< 0.0001	0.16626	0.43144
		time since fire (yrs)	0.03758	0.01177	0.0038	0.01334	0.06183
		avg. litter biomass (Mg ha <sup>-1</sup> )	-0.03838	0.00736	< 0.0001	-0.05353	-0.02322
		% shrub crown dead	0.03967	0.01	0.0005	0.01907	0.06026
Model Adj. R2 = 0.8011							
<sup>1</sup> 10-hr (0.63-2.54cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	-0.08537	0.1538	0.5834	-0.40095	0.23021
		# of snags fragmented	0.00915	0.00165	< 0.0001	0.00576	0.01255
Model Adj. R <sup>2</sup> = 0.5142							
100-hr (2.55 - 7.62cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	-0.40263	0.91906	0.6649	-2.29178	1.48651
		time since fire (yrs)	0.90487	0.2801	0.0033	0.32911	1.48063
		time since fire squared	-0.02351	0.01035	0.0317	-0.04478	-0.00223
		# of snags fragmented	0.01662	0.00864	0.0652	-0.00113	0.03438
Model Adj. R <sup>2</sup> = 0.6808							
Total FWD (< = 7.6cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	0.48819	1.03096	0.6399	-1.63511	2.6115
		time since fire (yrs)	1.11094	0.31491	0.0016	0.46237	1.7595
		time since fire squared	-0.0282	0.01161	0.0227	-0.05211	-0.00428
		# of snags fragmented	0.02194	0.01035	0.044	0.00063584	0.04325
Model Adj. R <sup>2</sup> = 0.7378							
1000-hr (> 7.62cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	-49.09889	19.21334	0.0166	-88.5214	-9.67638
		time since fire (yrs)	5.17671	0.87985	< 0.0001	3.37142	6.98201
		stem wood biomass (Mg ha <sup>-1</sup> )	0.35573	0.10428	0.002	0.14177	0.56968
Model Adj. R <sup>2</sup> = 0.6282							

An increase in 1 Mg ha<sup>-1</sup> of litter biomass also reduced 10-hr biomass by 0.13 Mg ha<sup>-1</sup> suggesting that, along with 1-hr biomass, 10-hr woody detritus becomes incorporated immediately into the litter layer. Since annual litter accumulation was 0.897 Mg ha<sup>-1</sup>, 0.12 Mg ha<sup>-1</sup> of 10-hr fuel was assimilated into the litter layer each year.

100-hr fuel biomass essentially started at 0.00 Mg ha<sup>-1</sup> and accumulated significantly with time. Biomass accumulation exhibited a non-linear trend accounted for by the quadratic term included in the model. Additional variation was explained by the total number of snags fallen or broken over time. Collectively, FWD (<7.62 cm diameter) followed the same trend as 100-hr fuel biomass primarily because a disproportionate amount of total FWD biomass is 100-hr fuel biomass.

Linear regression analysis suggested legacy 1000-hr fuel biomass (>7.62 cm diameter) was not present following a high-severity fire but increased linearly through 24 years of succession (Table 3.2). Our field observations suggest some 1000-hr fuel biomass remained following the fire event, dependent on stand history and log combustion. Linear regression analysis also suggested inputs of logs occurred rapidly and increased linearly through 24 years. Our observations suggest log accumulation rates were slow in the first few years, increased until approximately 20 years at which point the rate of accumulation decreased and total 1000-hr fuel biomass reached an asymptote (Figure 3.4). Additional variation in 1000-hr fuel biomass was explained by the inclusion of estimated total stem wood biomass using allometric equations (Table 3.2).

Following high-severity fire, the litter layer was essentially 0.00 Mg ha<sup>-1</sup> but accumulated rapidly at a rate of 0.90 Mg ha<sup>-1</sup> yr<sup>-1</sup> (95% CI, 0.65 to 1.15), reaching >20 Mg ha<sup>-1</sup> after 24 years. Each centimeter of litter depth accumulation averaged 505.8 g m<sup>-2</sup> (Table 3.3). In Figure 3.3, accumulation of litter appeared to be non-linear. A quadratic term for years since fire in the statistical analysis was not significant, leading to a linear model. Additional sampling is needed to determine whether the observed trend is better fit by a linear or non-linear model.

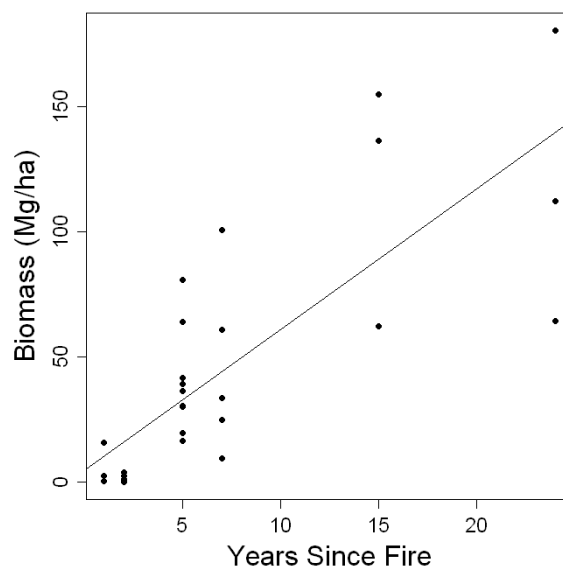


Figure 3.4: 1000-hr fuel accumulation following high-severity fire during 24 years of succession. Line represents linear regression line predicting 1000-hr biomass by age.

In contrast, duff was not completely consumed by high-severity fire and accumulated at an exponential rate. Each year, median duff biomass increased by a multiplicative factor of 1.13 (95% CI, 1.09 to 1.16), indicating a lag in development followed by an increasing accumulation rate over time (Figure 3.5). Duff had a higher bulk density than litter, resulting in each centimeter of duff accumulation averaging  $658.2 \text{ g m}^{-2}$  (Table 3.3).

Shrub growth was positively correlated with time since fire. There was no evidence of shrub survival following high-severity fire but shrubs re-established quickly and accumulated linearly over time (Figure 3.6). On average,  $0.64 \text{ Mg ha}^{-1}\text{yr}^{-1}$  of shrub biomass accumulated during the first 24 years of succession. No significant linear trends were found for estimating herbaceous biomass. There was no evidence of accumulation or difference across time with the exception of plots within the B&B Complex after 5 years of growth. Within this fire site, significant amounts of herbaceous biomass ( $>2.0 \text{ Mg ha}^{-1}$ ) were found but this amount of biomass is the exception and did not fit a linear trend with other sites (Figure 3.6).

Table 3.3: Regression coefficients for estimating litter, duff and shrub biomass.

Fuel Type	Dependent Variable	Independent Variable	Estimate	Std Error	p-value	95% Confidence Intervals	
						Lower	Upper
Litter	Biomass (Mg ha <sup>-1</sup> )	Intercept	2.24748	1.21638	0.0752	-0.24417	4.73912
		time since fire (yrs)	0.89701	0.12275	< 0.0001	0.64557	1.14844
		<b>Model Adj. R<sup>2</sup> = 0.6437</b>					
Litter	Biomass (g m <sup>-2</sup> )	Intercept	-80.27138	124.369	0.5239	-335.02968	174.48691
		depth (cm)	505.82022	54.42539	< 0.0001	394.33486	617.30557
		<b>Model Adj. R<sup>2</sup> = 0.7464</b>					
<sup>1</sup> Duff	Biomass (Mg ha <sup>-1</sup> )	Intercept	-0.09042	0.15317	0.5597	-0.40417	0.22332
		time since fire (yrs)	0.11815	0.01546	< 0.0001	0.08649	0.14981
		<b>Model Adj. R<sup>2</sup> = 0.6645</b>					
Duff	Biomass (g m <sup>-2</sup> )	Intercept	-15.44476	26.08589	0.5586	-68.87929	37.98976
		depth (cm)	658.21271	32.9499	< 0.0001	590.7179	725.70753
		<b>Model Adj. R<sup>2</sup> = 0.9321</b>					
Shrubs	Biomass (Mg ha <sup>-1</sup> )	Intercept	-0.44515	0.46892	0.3506	-1.40569	0.5154
		time since fire (yrs)	0.63839	0.04732	< 0.0001	0.54146	0.73532
		<b>Model Adj. R<sup>2</sup> = 0.8619</b>					

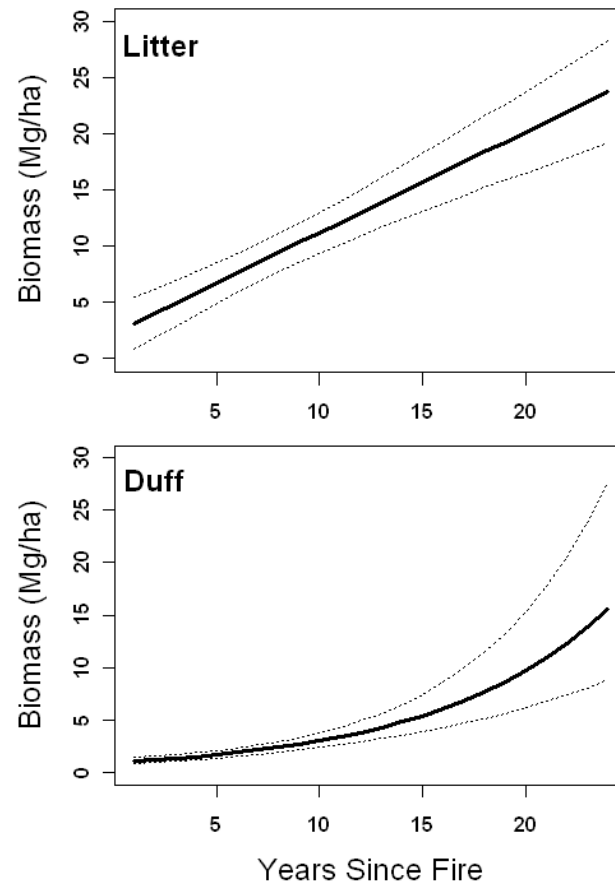


Figure 3.5: Litter and duff accumulation during 24 years of development.  $a$  = litter,  $b$  = duff. Dotted lines are 95% confidence intervals around the mean estimate. Duff biomass exhibits a lag in development relative to the litter layer.

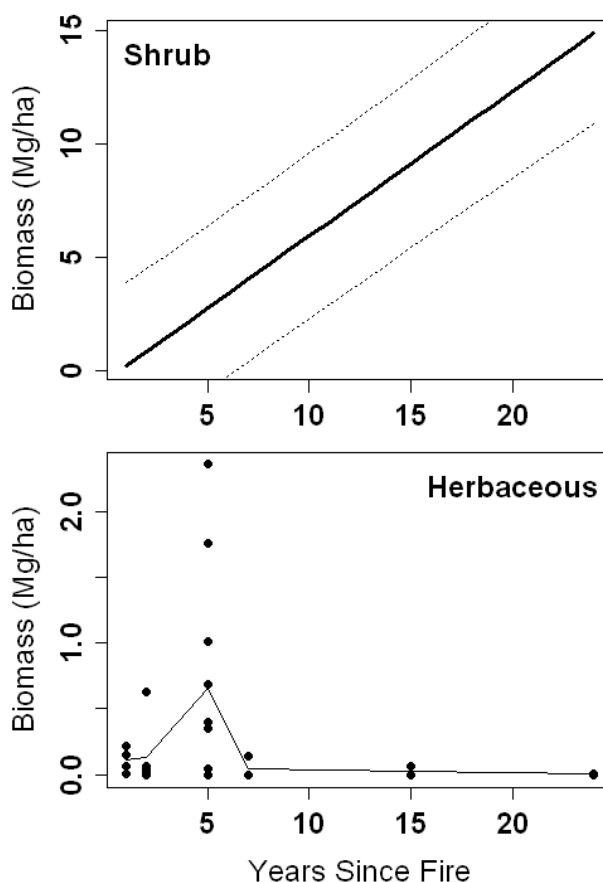


Figure 3.6: Live shrub and herbaceous biomass accumulation. The herbaceous layer is an insignificant component of post-fire fuel – note axes differences. Herbaceous trend line is a loess smoothed line and dotted lines are 95% confidence intervals for shrub estimates.

## Discussion

Fuel succession following high-severity fire is a complex interaction between pre-fire stand structure, fire intensity, fragmentation/decomposition of biological legacies and the reestablishment/dynamics of live vegetation. Each fuel bed layer interacts with these controlling factors to develop unique pathways of accumulation during the first decades following fire. The complexity of these interactions results in high variability in fuel conditions, at relatively small spatial scales, through time. However, there are strong patterns that make fuel succession predictable.



### ***Litter and Duff Fuel Succession***

Litter and duff fuel layers combust with high efficiency during wildfires. Campbell et al. (2007) reported that 99-100% of the forest floor combusted in high-severity portions of the Biscuit Fire in Southern Oregon. Analysis of litter and duff from our study extends these results to dry-mixed conifer forests of Oregon's eastern Cascades. The intercept estimate from regression analysis of litter biomass (Table 3.3) is not significantly different from zero, although there is suggestive evidence that some litter layer may persist. Knapp et al. (2005) reported approximately 97% consumption of litter and duff following late season prescribed fire (approximate residual forest floor =  $3.25 \text{ Mg ha}^{-1}$ ) and van Wagendonk and Sydoriak (1987) reported greater than 94% consumption of the forest floor (residual approximately equal to  $9.4 \text{ Mg ha}^{-1}$ ) following prescribed fire in dry-mixed conifer forests of the Sierra Nevada Mountains. Our estimates suggest  $2.24 \text{ Mg ha}^{-1}$  of litter layer and  $0.91 \text{ Mg ha}^{-1}$  of the duff layer remains post-fire as indicated by the positive intercept, which are within the bounds of the prescribed and wildfire results (Table 3.3). We believe that less litter remains after the fire than this suggests, primarily because a pulse of needle litter occurs 1-2 yrs post-fire as scorched needles are deposited to the forest floor.

Leaf litter decomposition following fire in Oregon's eastern Cascade forests is rapid relative to other plant material. Only a portion of total litter biomass is incorporated into the duff layer since plant litter is consumed rapidly by heterotrophs. Monleon and Cromack (1996) reported a leaf litter decomposition loss rate-constant, following fire in dry-mixed conifer forests, as  $k = 0.28 \text{ yr}^{-1}$ . Material not consumed by heterotrophs fragments further and becomes part of the duff and fermentation layers.

In contrast, bark decomposes at a slower rate than leaf litter, thus contributing significantly to the development of litter and duff layers in post-fire environments. *Abies sp.* bark was reported to have a decomposition loss rate-constant of  $k = 0.007 \text{ yr}^{-1}$

(Harmon 1996, unpublished data). The relatively recalcitrant nature of bark suggests it remains on the landscape for many decades.

Total forest floor biomass (litter and duff) accumulates at a higher rate than total shrub and herbaceous biomass, suggesting the development of these layers following high-severity fire is dominated by snag bark sloughing. Across our sites, allometric equations estimate average bark biomass to be  $38.73 \text{ Mg ha}^{-1}$  (~16.1% of total aboveground biomass). After 24 years of post-fire development, estimated litter and duff biomass averages  $39.39 \text{ Mg ha}^{-1}$ . Bark biomass accounts for a total of  $32.74 \text{ Mg ha}^{-1}$  and residual 1-hr biomass accounts for only  $0.386 \text{ Mg ha}^{-1}$  after 24 years. The net accumulation of litter and duff from *de novo* sources ( $6.26 \text{ Mg ha}^{-1}$ ) occurs from non-woody plants, shrubs and regenerating trees. Continued input from new vegetation will occur and accelerate as an overstory canopy develops and controls the rate of litter input (Hall et al. 2006).

Litter and duff are not considered important components of fire rate of spread since their relatively high bulk density inhibits oxygen availability and rapid combustion (Rothermel 1972). Although a well developed litter and duff layer do not contribute significantly to fire rates of spread, they do contribute to overall fire spread by increasing spatial continuity of surface fuels, which can lead to increased combustion of CWD (Reinhardt et al. 1991). Additionally, litter and duff are a significant source of total heat release and increased fire residence time. The main effect of litter and duff combustion is soil heating, potentially causing mortality through root necrosis and soil chemical and morphological changes (DeBano et al. 1998).

### ***Live Shrub and Herbaceous Fuel Succession***

The herbaceous plant community does not contribute significantly to total fuel accumulation in our sites. Herbaceous fuels loadings were insignificant (i.e.  $<0.5 \text{ Mg ha}^{-1}$ ) at all fire sites except the B&B Complex on the Sisters Ranger District of Deschutes National Forest. The highest plot level biomass sampled was  $2.3 \text{ Mg ha}^{-1}$ ,

but average herbaceous biomass 5 years post-fire was  $< 1.0 \text{ Mg ha}^{-1}$ . High levels of herbaceous biomass correlated with understory dominance of *Pteridium aquilinum* (bracken fern). The presence of bracken fern increases herbaceous loading, but this vegetation remains green until late summer or early fall reducing its availability to contribute to fire spread except during late season burns (Figure 3.6).

In contrast to herbaceous fuels, shrub colonization occurs rapidly and makes a major contribution to fuels within a few decades. Shrubs were the dominant vegetative cover on our sites by year 5, as species such as *Ceanothus velutinus* (snowbrush), a species adapted to post-fire environments by having seeds that benefit from fire scarification, rapidly colonized and occupied the sites (Brown and Smith 2000). *Arctostaphylos patula* (greenleaf manzanita), *Ribes sp.*, and *Castanopsis chrysophylla* (golden chinkapin) were other common shrub species encountered during sampling. After 24 years, shrub cover averaged 70% and was 2 meters in height.

Live shrubs not only contribute to post-fire fuels as carbon based biomass themselves, but they contribute to the development of other fuel beds. Higher litter biomass was evident under shrub drip lines as leaf cast contributed significant amounts of biomass to the litter layer annually. By year 15 on our sites, shrub heights and cover increased to a level where self pruning of lower crowns began. When maintenance respiration of individual plant parts supporting photosynthesis requires more energy than is gained, leaves and lower branches are senesced and input as surface fuel. We found that the percent of shrub crown that was dead significantly contributed to 1-hr fuel biomass (Table 3.2). Shrub biomass accumulation is expected to decrease over the next 10 years as shrubs approach their height limits and tree regeneration outcompetes the shrub layer for growing space. The gradual collapse of the shrub layer will result in a pulse of FWD  $> 20 \text{ Mg ha}^{-1}$  primarily in the 10-hr and 100-hr fuel size classes. The impact of this fuel input will last for several decades until decomposition or a subsequent fire event occurs.

### ***1-Hr Fuel Succession***

Rate of spread, fireline intensity and flame length are primarily driven by 1-hr fuel loadings (Rothermel 1972). High surface-area-to-volume-ratios of smaller material increase heat transfer between fuel particles of this size allowing them to combust rapidly and contribute significantly to the characteristics of the flaming front. Although available fuel of all size classes are important in fire management, high accumulations of 1-hr fuels can pose significant fire control problems, increased overstory scorch and fire severity.

Pre-fire surface 1-hr fuels combust with high efficiency in natural and prescribed fire. Campbell et al. (2007) reported 100% of 1-hr fuels combusted in high-severity fire areas within the Biscuit Fire in southwest Oregon and Van Wagtendonk and Sydoriak (1987) reported 95% of 1-hr fuels in dry-mixed conifer forests combusted during a prescribed fire. Residual 1-hr fuel biomass following high-severity fire is insignificant so sampling of 1-hr fuel biomass captures accumulation rates without the confounding effects of incomplete combustion. Accumulation of 1-hr fuels is relatively constant during the first 24 years of succession. Considerable variability occurs across the chronosequence due to 1-hr fuel inputs from shrub crowns and the assimilation of these fuels into the litter layer (Figure 3.7).

Fire killed trees are a major source of 1-hr fuels via rapid litter and twig fall. All or portions of a tree crown may be killed by a fire through direct combustion of foliage and small branches or heat scorch. Combustion directly consumes foliage and small branches while heat scorch may leave the entire canopy intact. Campbell et al. (2007) reported that 69% – 89% of foliage and 5% to 64% (large and small conifers, respectively) of branches were consumed in high-severity fire in southwest Oregon. Incomplete combustion of tree crowns may result in relatively large amounts of foliage and 1-hr fuels input to the forest floor by 1 year post-fire, even when >95% of the pre-fire surface amounts are combusted (Figure 3.7).

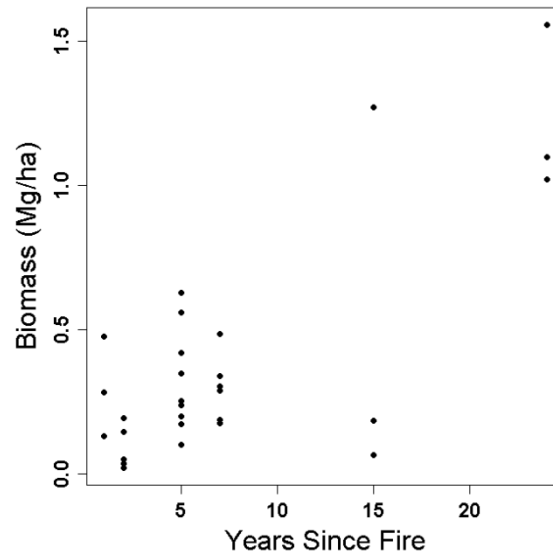


Figure 3.7: 1-hr fuel accumulation during 24 years of succession following high-severity fire. Points represent fuels transect estimates of 1-hr biomass

As canopy consumption increases, the amount of 1-hr fuel deposited within 1 year decreases. In year 1 and 2, we found low loadings of 1-hr fuel in plots where > 90% canopy consumption was apparent. At older sites it wasn't possible to estimate the amount of canopy consumption that occurred, so the legacy effect of this material on loadings is difficult to discern. In our case, nearly  $0.50 \text{ Mg ha}^{-1}$  of 1-hr fuel biomass was present 1 year post-fire, following complete overstory mortality without canopy consumption (mortality via root or cambial scorch). This amounts to only a small portion of total crown foliage and twig biomass so we expect continued deposition of this material for a couple years following fire. This material is quickly removed from the 1-hr surface fuel layer as it becomes incorporated into the litter and, ultimately, the duff layer. Our estimates suggest the legacy of this material may last for ~5 years before being lost to decomposition or being incorporated into the litter layer.

Regardless of whether legacy 1-hr fuels are deposited to the forest floor within the first couple years or delayed with snag fragmentation, 1-hr surface fuel

accumulation from biological legacies is reduced by decomposition and assimilation into litter layers. We found 1-hr biomass steadily increases through 24 years even though the amount of litter biomass is negatively correlated with 1-hr fuel biomass. Net biomass increase barely exceeds assimilation rates in the litter layer until substantial shrub canopy mortality occurs (Table 3.2).

As shrub canopy cover increases, foliage and lower branches are senesced as respiratory costs become greater than their ability to provide for or aid in photosynthesis (Chapin et al. 2002). These plant parts are deposited to the soil surface as litter, 1- and 10-hr fuels. Less than 2.5% shrub canopy mortality occurred on all plots prior to 15 years post-fire. After 15 years of shrub growth, we estimated 13-18% of the shrub canopy was dead, increasing up to 30% after 24 years. We anticipate the rate of shrub mortality to increase as the overstory canopy establishes and outcompetes the shrubs (Oliver and Larsen 1996). Our estimates suggest this will occur during the next 10 years, creating a pulse of 1-hr fuels in addition to persistent inputs from overstory trees which out compete the shrub layer.

### ***10-Hr Fuel Succession***

Similar to 1-hr fuels, 10-hr fuels combust with high efficiency during fire. 10-hr surface fuel consumption was reported to be between 86-92% following prescribed fire treatment (Parsons 1978, Van Wagtendonk and Sydoriak 1987, Knapp et al. 2005). Although some residual 10-hr surface biomass survives the fire, the majority of sampled biomass following high-severity fire is input from snag or overstory trees and *de novo* vegetation (Keyser et al. 2009).

Snag fall and breakage have the largest influence on 10-hr fuel biomass during several decades of fuel succession following high-severity fire. Statistically, the number of snags fragmented explained 51.4% of the variation in 10-hr biomass in our sites. Some of the unexplained variability can be attributed to incomplete combustion of surface 10-hr biomass and incomplete combustion of tree crowns, creating a significant source of fine fuel (Keyser et al. 2008). A portion of the scorched twigs

and branches <2.54 cm diameter transfers to the forest floor within 1 year following a fire, causing a pulse of 10-hr surface fuel consistent with 1-hr fuel deposition. In contrast, crown fires consume large portions of the tree canopy and result in a lower pulse following fire, as indicated by the range of 10-hr surface fuels biomass we found 1-year post-fire (i.e. 0.5 Mg ha<sup>-1</sup> to >2.0 Mg ha<sup>-1</sup>) and depicted in Figure 3.8.

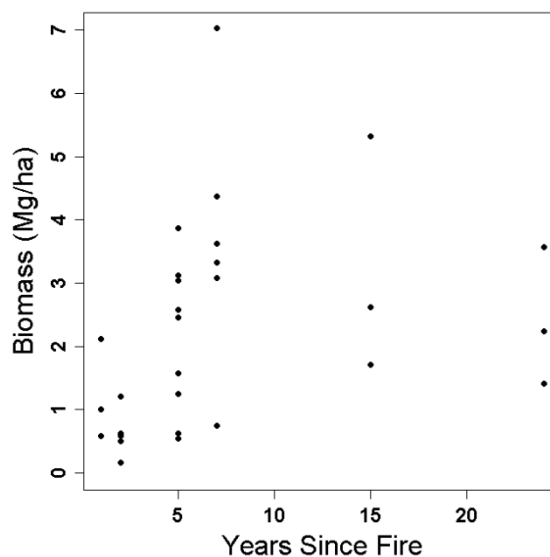


Figure 3.8: 10-hr fuel accumulation during 24 years of succession following high-severity fire. Points represent fuels transect estimates.

Accumulation of 10-hr biomass is steady for 12 – 15 years but begins to decrease during the next decade (Figure 3.8). The inflection point observed in the data represents the time when input from dead biological legacies was equal to decomposition losses. Additional variation evident in Figure 3.8 occurs because of assimilation rates of 10-hr biomass into the litter layer and, eventually, inputs from self-pruning or dying shrubs. McGinnis et al. (2010) report 10-hr fuel biomass equals 2.3 Mg ha<sup>-1</sup> and 3.0 Mg ha<sup>-1</sup>, following stand replacing (4-6 and 5-6 years respectively) fire at untreated, mixed conifer sites in California. Additionally, they report 1.3 Mg ha<sup>-1</sup> and 1.7 Mg ha<sup>-1</sup> of 10-hr biomass occurred 14-16 years and 19-21 years post high-severity fire. These values are similar to the range of values we found 5 and 7 years post-fire but are distributed at the lower end of our estimates at 15 and

24 years post-fire (Figure 3.8). The difference in later years is primarily a result of the low pre-fire basal area in untreated, older fire sites sampled by McGinnis et al. (2010).

During the first few decades of succession, shrubs are the dominant vegetative cover in dry-mixed conifer forests of Oregon's eastern Cascades. Overstory trees have successively established by 24 years post-fire and are overtopping the current shrub layer, as expected by successional theory and stand dynamics (Oliver and Larson 1996). Overtopping will result in shrub mortality and additional 10-hr fuel inputs, as described for 1-hr fuels. Within the next decade, we expect 10-hr fuel accumulation will be controlled by increased shrub mortality followed by branch deposition from overstory trees dependent on their canopy characteristics (Hall et al. 2006, Keyser et al. 2008).

### ***100-Hr Fuel Succession***

Accumulation of 100-hr fuel biomass has the strongest relationship with time since fire for all fuel size classes, but accumulation rates during the first several decades are non-linear (Table 3.2). 100-hr fuel biomass accumulation is linear for approximately 15 years, appearing to reach an asymptote as input rates decrease after total transfer from crowns ends (sample points, Figure 3.9). An asymptote suggests equilibrium has been reached between input rates and decomposition rates. We do not believe this is the case but rather this trend is a function of not applying decay classes to FWD so biomass can only be reduced by burial, combustion or transfer offsite.

We expect 100-hr fuel biomass to decrease after about 15 years of succession. This trend is partially supported by McGinnis et al. (2010), who reported 3.8 and 7.0 Mg ha<sup>-1</sup> of 100-hr biomass at two separate sites sampled 4-6 and 5-6 years following stand replacing fire. At two other sites 14-16 and 19-21 years post-fire, they reported 100-hr fuel biomass equal to 4.5 and 5.4 Mg ha<sup>-1</sup>, respectively. Their estimates at the two younger sites were within the bounds of 100-hr fuel biomass estimates from our sampled sites (Figure 3.9). In contrast, their older fire site estimates are lower than ours primarily because pre-fire basal area at these sites is much lower than the



basal area range we sampled (i.e. basal area averaged 1 and 12 m<sup>2</sup> ha<sup>-1</sup> for their sites 14-16 and 19-21 years post-fire, respectively, compared to our sampled range of 23.7 to 59.2 m<sup>2</sup> ha<sup>-1</sup>). Although we cannot directly compare the successional trajectory of their sites, their 100-hr fuel estimates are within the bounds of ours and, thus, support our expected successional trajectory.

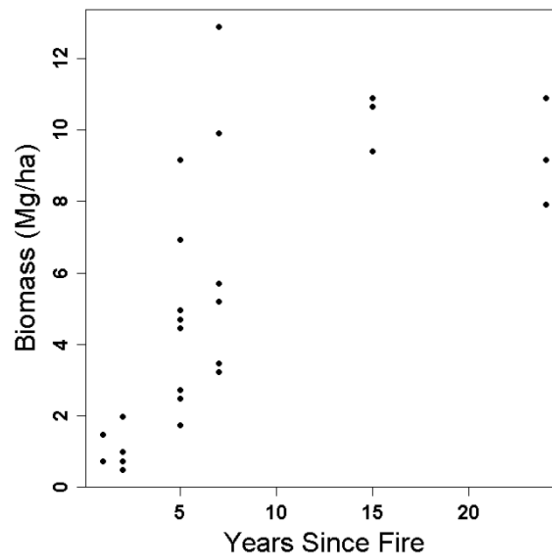


Figure 3.9: 100-hr fuel accumulation during 24 years of succession following high-severity fire. Linear regression analysis predicting biomass from time since fire explains 68% of the variability in the data.

### ***1000-Hr Fuel Succession***

Following high-severity fire, accumulation of logs occurs by snag fall and breakage until *de novo* trees grow large enough, die and fall to the forest floor (Harmon 2009). Snag stem wood is the primary source of logs although larger branches (>7.62 cm in diameter) contribute an average of 4% (range 1-7%) of surface CWD, as estimated from allometric equations (Brown 1978, Jenkins et al. 2004). In a separate modeling study, McIver and Ottmar (2007) found peak accumulations of logs occurred 25 years following high-severity fire, very close to our snag dynamics results (Chapter 2). Total transfer from snag fall and breakage is greater than decomposition losses through 24 years of post-fire succession at our sites, resulting in surface CWD

peak accumulation around 24 years (range 23 – 26 years) at which point decomposition losses become greater than deposition rates (Figure 3.10).

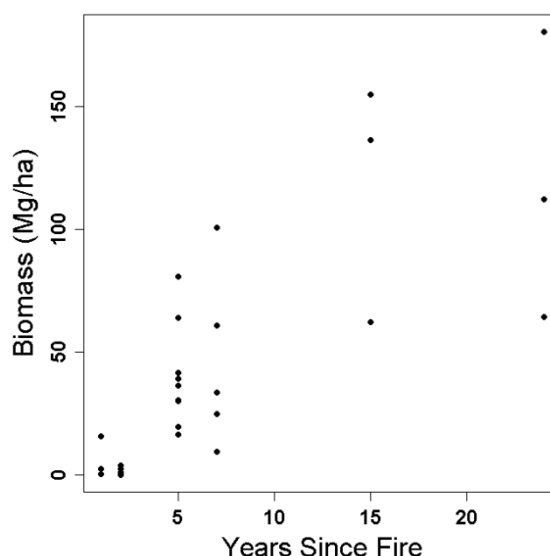


Figure 3.10: 1000-hr fuel accumulation during 24 years of succession following high-severity fire. Linear regression analysis predicting biomass from time since fire and initial stem wood biomass explains 63% of the variability in the data.

The amount of CWD available for combustion varies by decay state, position and interaction with other combustible plant material (Albini and Reinhardt 1997). To date, no published studies have quantified the proportion of log biomass combusted by decay class, but rotten logs have been reported to combust with greater efficiency and heat than sound logs (Albini and Reinhardt 1997, Monsanto and Agee 2008). Tinker and Knight (2000) reported that only 8% of log biomass was combusted in fires in Yellowstone National Park and another 8% was converted to charcoal. Although this is an average across an entire stand with varying abundances of log biomass by decay class, we used 8% as the combustion efficiency of sound wood, defined as woody detritus with a density  $>0.30 \text{ g cm}^{-3}$ . Brown (1974) assumed rotten woody detritus had a density of  $0.30 \text{ g cm}^{-3}$ , which does not correlate directly to studies of wood density by decay class but roughly equates to decay class 2-3 for species found in dry-mixed

conifer forest (Harmon et al. 2008). Decay class 2 logs are relatively sound and decay class 3 logs have friable sapwood and little bark remaining attached (Cline et al. 1988). By the time logs reach decay class 4, density values tend to converge around  $0.20 \text{ g cm}^{-3}$ , although variation does occur within and among species (Harmon et al. 2008). We believe decay class 4 logs represent rotten material and assume that wood densities between  $0.30$  and  $0.20 \text{ g cm}^{-3}$  represent partially rotten material.

## Conclusions

In dry-mixed conifer forests of Oregon's eastern Cascades, live fuels respond quickly to post-fire conditions and are dominated primarily by shrubs adapted to post-fire environments (Brown and Smith 2000). Herbaceous communities do not contribute significantly to fuel loadings, but shrub biomass increases linearly for the first several decades and have the potential to exacerbate fire behavior in the event of a subsequent fire event (Thompson et al. 2007). The risk posed by this fuel layer is only elevated during periods of high drought stress, transitioning this fuel layer from a heat sink to a heat source (Rothermel 1972). The main contribution of shrubs regarding post-fire fuels is increases in litter, duff and FWD biomass. After a few decades, this layer is beginning to show signs of competitive stress leading to increased mortality as overstory trees begin to occupy the site and outcompete this vegetative layer (Oliver and Larsen 1996). The impact of this transition will be a pulse of FWD from shrub biomass.

Total fine woody detritus biomass is controlled by snag breakage and fall during the first decade of succession, but each timelag fuel size class responds differently. 1-hr fuels remain in relatively low abundance following high-severity fire until shrub mortality occurs. 10-hr fuels are directly related to snag crown deposition, with inputs from shrubs occurring after 10-15 years of development. 100-hr fuels are directly related to snag breakage and fall with no new inputs during the first couple of decades of development.

Coarse woody detritus has the potential to input large amounts of dead fuels as snags break and fall. The amount of biomass present in future years is dependent on initial stand biomass, snag breakage and fall rates and surface and standing decay rates (Harmon 2009). Total surface CWD biomass peaks 23-26 years post-fire, but not all of this fuel is available for combustion. Since the combustion of surface CWD is correlated with the decay stage of this material, peak available surface CWD fuel is further delayed from the time since fire by at least another decade beyond the peak accumulation year. During this additional time period, the amount of total biomass present is further reduced since decomposition continues to reduce total surface CWD on site.

## CHAPTER 4: ESTIMATING THE IMPACTS OF SALVAGE LOGGING ON FUEL SUCCESSION USING A COARSE WOODY DETRITUS DYNAMICS MODEL

### Introduction

Prior to European settlement, the dominant disturbance within the dry-mixed conifer forests of Oregon's eastern Cascades was fire. The fire regime in this forest type is classified as mixed-severity with fire free intervals ranging from 9-62 years (Agee 1993). Variation in the time since fire across a landscape, topography, fire weather conditions and broad climate patterns creates a mosaic of burn severities within an individual fire boundary during a given year. Individual stands will experience either high-severity (>75% tree mortality), moderate severity (25% - 50% tree mortality) or low severity (< 25% tree mortality) fire within a single fire event. The landscape severity pattern will be different during subsequent fires, although there is evidence that high-severity fire may be spatially correlated during subsequent fire events (Thompson et al. 2007).

Dry-mixed conifer forests in Oregon's eastern Cascade's are dominated by *Pinus ponderosa* (ponderosa pine), *Abies grandis* (grand fir), *Abies concolor* (white fir), *Pseudotsuga menziesii* (Douglas-fir), *Pinus contorta* (lodgepole pine) and *Calocedrus decurrens* (incense-cedar) (Franklin and Dyrness 1988). The density and composition of a stand within these forests at a particular time is dependent on site conditions (elevation, slope, aspect, soil type and climate) and disturbance history. The cumulative effects of disturbance at long temporal scales creates a complex mosaic of stand conditions with varying composition and abundance of live trees, snags and logs (Hessburg et al. 2005).

European settlement altered natural disturbance regimes in these forests beginning in the mid-19<sup>th</sup> century through a reduction in human ignited fire, livestock grazing, and more recently, mechanized fire suppression (Agee 1993). The lack of fire has resulted in an increase in the total number of stems present in dry-mixed

conifer forests, with particular increase in the density of *Abies sp.* stems <41cm DBH. Evidence of structural change was noticed as early as the 1940's by Harold Weaver (1943). At the same time that natural disturbance decreased, human disturbances increased across Oregon. Commercial logging on private and public lands began early in the 20<sup>th</sup> century and increased until the early 1990's when logging on public lands precipitously dropped following the Northwest Forest Plan. Harvesting practices during this period included salvage logging fire killed trees in an attempt to recover the economic value of the standing timber before it diminished due to decomposition of the woody material (Sessions et al. 2004).

The effects of altered disturbance regimes on stand structure are obvious when considering live trees. The lack of fire in natural stands has increased the number of small diameter trees present on site, resulting in increased susceptibility to stand replacement fire across broad landscapes (Spies et al. 2006) and increased competitive stress on large legacy trees (Fellows and Goulden 2008). Past logging has typically modified stand structure by creating even-aged, single species stands that lack the vertical and horizontal diversity historically developed on dry mixed-conifer sites. Today, all species in dry-mixed conifer forests of Oregon have higher densities of stems <41cm DBH and lower densities of larger diameter trees (Merschel 2010). The ecological effects of these changes are becoming apparent as our forests are valued for ecosystem services beyond just timber products.

Less obvious, perhaps, are the effects of these changes on CWD. Harvest regimes facilitated live tree growth and removed the standing timber before competitive stresses, biotic agents and/or irregular mortality (e.g., lightning) created CWD. In the event of moderate to large disturbances, salvage logging would remove much of the newly created CWD, further reducing the presence of this material within the landscape. The loss of these biological legacies and the reduction of large trees that develop into large CWD have reduced the presence of this material on landscapes.

Increasingly, forested landscapes are being valued for their ability to provide multiple ecosystem values, including clean water, clean air, wildlife habitat,

biodiversity, recreational opportunities and carbon based forest products.

Incorporating non-economic, ecosystem values into management regimes differs from forest management with the primary objective of timber production, the dominant management paradigm on public lands since creation of the Forest Reserves and Forest Service early in the 20<sup>th</sup> century. In western forests, contemporary management strategies tend towards ecological restoration and/or ecologically sustainable management practices that include management of dead wood resources as important structural components of forest ecosystems. The retention of snags and CWD are increasingly blended with fuels reduction treatments in dry forests in order to meet ecological and management needs (Brown et al. 2003).

Historically, salvage logging and replanting following fire was a common management practice used to take advantage of the timber resource and facilitate the development of regulated forests (Sessions et al. 2004). Public values have changed and new policies were created in the early 1990's altering decisions about appropriate post-fire management practices. These changes have resulted in many post-fire landscapes remaining unsalvaged and unplanted, although salvage and reforestation remains a management option. Most agencies cite the reduction of future fire severity by removal of large quantities of fire killed woody detritus as an ecological justification for salvage logging following high-severity fire disturbance (Sessions et al. 2004). The effectiveness of this management strategy at reducing future fire severity during the next burning event is still poorly understood; some research suggests salvage logging exacerbates the fuel hazard (Donato et al. 2006), improves the conditions (Monsanto and Agee 2008) or remains relatively neutral in its effect (McGinnis et al. 2010).

The objective of this study was to model succession of wildland fire fuels following high-severity fire and in order to assess the effects of salvage logging on these fuels. To meet these objectives, we explored the following question: Does salvage logging reduce future fire severity by removing CWD following a high-severity fire disturbance?

## Methods

Dry-mixed conifer forests of Oregon's eastern Cascades consist of multiple tree species in varying abundances, dependent primarily on site productivity and stand history. Dominant species include *Pinus ponderosa*, *Abies grandis/concolor* (hybrids occur in central Oregon and will be referred to as *Abies sp.*), *Pinus contorta*, *Pseudotsuga menziesii* and *Calocedrus decurrens*. Physiological characteristics of trees (diameter, height, wood density, weight distribution, presence of decay resistant heartwood, crown biomass) lead to different patterns of fall, breakage, and decay for each species. To understand the long-term CWD dynamics of stands with this level of complexity and the effects of salvage logging, an empirical model incorporating individual fall, breakage, and decomposition rates was developed using R statistical package. The basic design of our necromass dynamics model ("NecroDynamics") is depicted in Figure 4.1.

"NecroDynamics" predicts the condition of an individual snag each year during the simulation period. Snag conditions include: standing intact, falling intact, standing with top breakage, and falling after top breakage. Aerial or surface decomposition occurs annually for individual pieces of necromass dependent on their position. Decomposition is modeled as a negative exponential function and all pieces of necromass are tracked individually (i.e. snag, 1<sup>st</sup>, 2<sup>nd</sup> and, 3<sup>rd</sup> broken piece, final fallen snag) to capture total mass loss through time.

### *Necromass Calculation*

Species specific allometric equations are used to estimate stem wood, crown and bark biomass for all snags at the time of the fire (Brown 1978, Means et al. 1994, Jenkins et al. 2004). Whenever feasible, "NecroDynamics" uses equations developed from sites in Oregon but allometric equations for *P. menziesii* are from the Rocky Mountain region since allometric equations from Oregon for this species were developed from western Oregon sites.



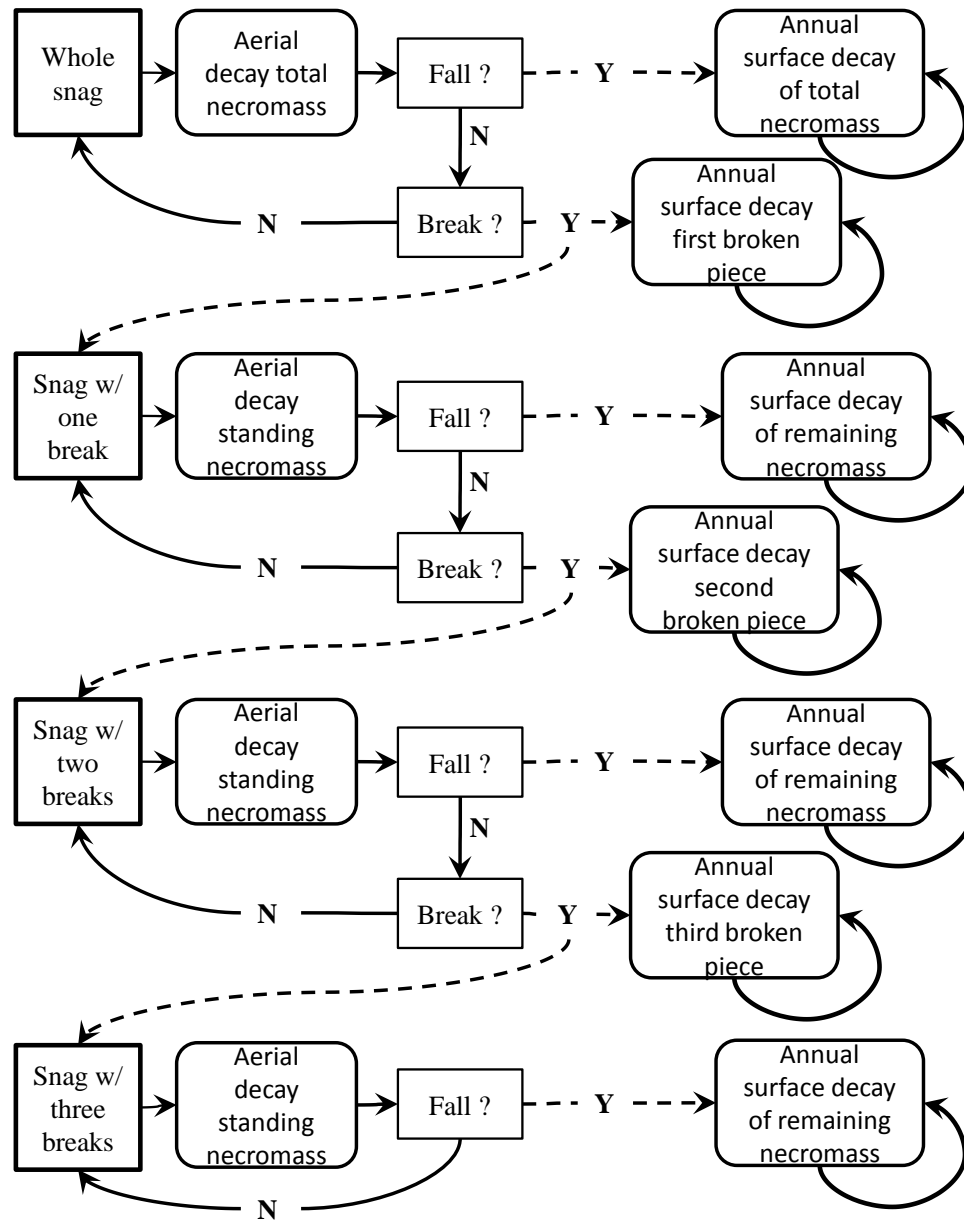


Figure 4.1: “NecroDynamics” individual snag fragmentation and decomposition process. Fragmentation and decomposition of individual snag and log pieces are tracked annually through time. Whole and broken snags decay at species specific aerial decay rates prior to fragmentation. Snags may or may not fragment each year depending on predicted fall and breakage years. Dashed lines represent transfer of necromass from snag to logs. Logs decay at species specific surface decay rates beginning the year following transfer. Stand level summaries are calculated for each year of the simulation period.

Live biomass estimates are calculated for total stem wood, bark and crown. Table 4.1 provides a summary of allometric equations used and their sources. Total crown biomass estimates are partitioned into size classes based on the diameter of the twigs and branches using equations developed by Brown (1978). The individual components estimated are: foliage, 1-hr (0-.62 cm), 10-hr (.62-2.54 cm), 100-hr (2.54-7.62 cm), and 1000-hr (> 7.62 cm), consistent with timelag fuel moisture classes used in fuel models for predicting fire behavior (Scott and Burgan 2005). Post-fire necromass is calculated as live biomass estimates reduced by combustion factors developed following high-severity fire in southwest Oregon (Campbell et al. 2007).

*P. ponderosa* heartwood contains decay resistant resins, tannins, etc. causing slower decomposition of this material relative to its sapwood (Harmon et al. 1986). This variation is supported by decay estimates obtained from downed logs sampled in this study. Allometric equations for estimating heartwood biomass have not been developed for *P. ponderosa* but we felt they were necessary to accurately estimate the decomposition rate of this species. Therefore, cross-section samples removed from snags at 25%, 50% and 75% of total snag height were separated into heartwood and sapwood components for decomposition analysis. Measurements of diameter inside bark and heartwood diameter were collected for each sample.

Heartwood biomass estimates were obtained for *P. ponderosa* snags and logs. Only samples collected at 25% total snag height were used to predict heartwood diameter in order to meet the assumption of independent samples. The DBH of snags and logs sampled in this study ranges from 33 to 129 cm. Statistical analysis consisted of simple linear regression using PROC Reg in SAS 9.2 (SAS Institute Inc. 2008).

Table 4.1: Allometric equations used to estimate live biomass. Crown biomass is partitioned into timelag fuel size classes using equations developed by Brown (1978) and reduced by combustion factors (Campbell 2007).

Species	Component	Equation (Mg/ha)	Geographic Region	Source
<i>P. ponderosa</i>	stem wood	$\exp(-11.8236+2.9260*\log(\text{dbh}))$	Oregon	Harmon 1994, BIOPAK
	bark	$\exp(-9.93105+2.1677*\log(\text{dbh}))$	Oregon	Harmon 1994, BIOPAK
	total crown		Rocky Mts.	Brown 1978
	intermediate	$\exp(-0.7572+2.2160*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		
	dominant	$\exp(0.2680+2.0740*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		
<i>A. grandis</i>	stem wood	$\exp(-11.2634+2.7856*\ln(\text{dbh}))$	Oregon	Harmon 1994, BIOPAK
	bark	$\exp(-11.7086+2.7271*\ln(\text{dbh}))$	Oregon	Harmon 1994, BIOPAK
	total crown		Rocky Mts.	Brown 1978
	intermediate	$\exp(1.0144+1.6156*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		
	dominant	$\exp(1.3094+1.6076*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		
<i>P. contorta</i>	stem wood	$\exp(-9.24342+2.3438*\log(\text{dbh}))$	Oregon	Harmon 1994, BIOPAK
	bark	$(\exp(-12.8027+2.0676*\log(\text{dbh})))$	Oregon	Harmon 1994, BIOPAK
	total crown		Rocky Mts.	Brown 1978
	all	$(\exp(0.1224+1.8820*\log(\text{dbh}/2.54)))*453.592378*(10^{-6})$		
<i>P. menziesii</i>	stem wood	$\exp(3.8682+2.5951*\log(\text{dbh}))* (10^{-6})$	Rocky Mts.	Gholz 1979
	bark	$\exp(2.5975+2.4300*\log(\text{dbh}))* (10^{-6})$	Rocky Mts.	Gholz 1979
	total crown		Rocky Mts.	Brown 1978
	suppressed	$\exp(0.1508+1.862*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		
	intermediate	$\exp(1.1368+1.5819*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		
	dominant	$(1.0237*(\text{dbh}/2.54)^2-20.74)*453.592378*(10^{-6})$		
<i>C. decurrens</i>	stem wood	$\exp(1.992026+2.7334*\log(\text{dbh}))* (10^{-6})$	Oregon	Harmon 1994, BIOPAK
	bark	$\exp(0.500948+2.8594*\log(\text{dbh}))* (10^{-6})$	Oregon	Harmon 1994, BIOPAK
	total crown		Rocky Mts.	Brown 1978
	intermediate	$\exp(0.5743+1.7960*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		
	dominant	$\exp(0.8815+1.6389*\log(\text{dbh}/2.54))*453.592378*(10^{-6})$		

### ***Snag Fall Rates***

Snag fall rates were determined by sampling a chronosequence of burned dry-mixed conifer forests in Oregon's eastern Cascades. The chronosequence began one year post-fire and included sites at 1, 2, 5, 7, 15 and 24 years. All sampled sites were mature to old-growth stands dominated by *P. ponderosa* and *Abies sp.* without evidence of past harvest or recent salvage operations. A total of 5103 snags of these species were measured in the field and fall rates were estimated for *P. ponderosa*, *P. contorta*, and *Abies sp.* Sample sizes were not large enough to estimate fall rates for *P. menziesii* or *C. decurrens*, other co-dominant species found in this forest type. See Chapter 2 for a more in depth description of the statistical analysis for estimating snag fall rates.

Back-transformed predicted mean values from logistic regression equations represent the proportion of snags fallen at a particular year. Table 4.2 provides the equations for estimating the proportion of fallen snags. Mean values from these equations do not represent an individual snags probability of falling; rather the cumulative proportion of all snags fallen by that year. To estimate the year an individual snag falls, a conditional probability equation is used to convert total fallen snags into individual snag probabilities.

$$conditional\ probability = \frac{(proportion(t) - proportion(t - 1))}{(1 - proportion(t - 1))}$$

t = year since fire

There was an observed temporal lag for most species and size classes before snags began falling. If a lag was observed for a particular species and size class, the model was constrained so that snags begin falling after the lag year has been reached. Lag estimates range from 0-4 years and are provided in Table 4.2.

Table 4.2: Back-transformed logistic regression equations for estimating the proportion of snags fallen by species, diameter class and years since fire. <sup>1</sup> x = number of years since fire. Lag years represent the number of years before snags begin falling, as observed in this study.

Species	DBH Class	Lag (Yrs)	Equation
<i>Abies spp.</i>	<23 cm	1	$\frac{\exp(-3.5046 + 0.197x^1 - 0.5624 + 0.7122 + 0.1976x)}{(1 + \exp(-3.5046 + 0.197x - 0.5624 + 0.7122 + 0.1976x))}$
	23 - 41 cm	1	$\frac{\exp(-3.5046 + 0.197x - 0.5624 - 0.2529 + 0.1024x)}{(1 + \exp(-3.5046 + 0.197x - 0.5624 - 0.2529 + 0.1024x))}$
	>41 cm	4	$\frac{\exp(-3.5046 + 0.197x - 0.5624)}{(1 + \exp(-3.5046 + 0.197x - 0.5624))}$
<i>P. ponderosa</i>	<23 cm	0	$\frac{\exp(-3.5046 + 0.197x + 0.7122 + 0.1976x)}{(1 + \exp(-3.5046 + 0.197x + 0.7122 + 0.1976x))}$
	23 - 41 cm	2	$\frac{\exp(-3.5046 + 0.197x - 0.2529 + 0.1024x)}{(1 + \exp(-3.5046 + 0.197x - 0.2529 + 0.1024x))}$
	>41 cm	2	$\frac{\exp(-3.5046 + 0.197x)}{(1 + \exp(-3.5046 + 0.197x))}$
<i>P. contorta</i>	<23 cm	0	$\frac{\exp(-3.5046 + 0.197x - 0.06696 + 0.7122 + 0.1976x)}{(1 + \exp(-3.5046 + 0.197x - 0.06696 + 0.7122 + 0.1976x))}$
	23 - 41 cm	1	$\frac{\exp(-3.5046 + 0.197x - 0.06696 - 0.2529 + 0.1024x)}{(1 + \exp(-3.5046 + 0.197x - 0.06696 - 0.2529 + 0.1024x))}$
	>41 cm	0	$\frac{\exp(-3.5046 + 0.197x - 0.06696)}{(1 + \exp(-3.5046 + 0.197x - 0.06696))}$

### *Snag Breakage*

Top breakage is an important component of the fragmentation of snags into logs (Harmon et al. 1986). Fragmentation rates from breakage are highly variable and may be correlated with stochastic environmental events such as high winds, particularly those accompanied by snow and ice loads on the snags. A limitation of the chronosequence approach used in this study is the inability to know exactly which snags broke prior to falling; rather we tracked the distribution of snags still standing at the sampled years following a fire. This leads to increased uncertainty in the total proportion of snags that break at any given year. Tracking individual snags through time would reduce this uncertainty but requires prohibitively long observations not generally supported.

Logistic regression analysis was performed using a generalized linear model with a logit link function in SAS 9.2 (SAS Institute Inc. 2008). No significant linear trends were found for the proportion of snags broken by species, DBH, or time since fire (snag age). Although a statistically significant linear trend was not found using logistic regression, it is evident that top breakage begins the first year following a fire and the total proportion of snags with broken tops increases with time. The proportion of snags that break within a given year after a fire is included in the model because of the importance of this process in snag fragmentation and necromass transfer.

The annual increase in proportion of snags with top breakage is estimated by dividing the total number of snags broken at a fire site by the number of years since the fire, and then averaging this value across all fire sites. Multiplying the average breakage rate by the number of years since fire estimates the total proportion of standing snags broken by that year. As with snag fall, this estimate does not represent the probability of an individual snag breaking. A conditional probability, calculated the same as for fall rates, is used to estimate an individual snag's probability of breaking. This value is compared to a random number generated from a uniform distribution. For additional information see Chapter 2 (this document).

### ***Multiple Snag Breakage Events***

Statistical analysis showed time since fire was negatively correlated with snag break height. Two mechanisms can cause a decrease in average broken snag height over time; snags break at lower heights in later years and/or multiple breakage events occur. Estimates of timing and actual height reduction by multiple breakage events are beyond the scope of this study and would require monitoring individual snags over long periods of time.

The first breakage mechanism was accounted for since break height calculations predict lower break heights as time since fire increases. The second mechanism may occur as snag height slowly decreases when small pieces fragment off or by additional discrete breakage events occurring at longer temporal intervals with greater reductions in height. “NecroDynamics” assumes multiple discrete breakage events occur if snags remain standing long enough.

“NecroDynamics” predicts the occurrence of a second and third breakage event after the initial break year. These events occur at a randomly selected year 5-15 years after the previous event. The likelihood that a second or third breakage event occurs diminishes over time since snags continue to fall throughout this period. Snags that persist for >20 years are highly likely to experience at least one additional breakage event, ultimately accounting for the decrease in average snag height apparent in my results.

### ***Transfer of Necromass from Breakage***

A portion of standing necromass is transferred from snags to logs each time breakage occurs. The amount transferred is dependent on tree form and height at the point of breakage. Since volume is not distributed uniformly along the length of a tree stem, additional calculations are required to determine the amount of stem wood necromass transferred when a snag breaks.

The simplest approach to estimating necromass transfer is to use taper equations to determine the break point diameter in conjunction with the use of generalized volumetric shapes to estimate either the standing or the broken proportion of necromass. Trees are generally modeled using a combination of three volumetric shapes. The lowest portion of the tree stem is often modeled as a frustum of a neiloid, the central stem as a frustum of a paraboloid and the top portion of the tree as a cone or paraboloid (Husch, Miller and Beers 1993). Modeling the standing or broken portion using these shapes assumes total stem necromass from allometric equations is equivalent to necromass estimates using these volumetric shapes and published wood density values.

This assumption was tested by estimating necromass using a general tree shape model and published wood density values (Means et al. 1994). Necromass estimates (via the general tree shape model) were compared to those developed from species specific allometric equations. Total snag volume below breast height was modeled as a frustum of a neiloid and the remaining portion of the tree was modeled as a paraboloid. When breakage occurs, breast height to break height was modeled as a frustum of a paraboloid and the portion above break height is modeled as a paraboloid (Husch, Miller and Beers 1993).

Volume estimates using the general tree shape model requires estimates of diameter inside bark at DBH, diameter inside bark at break height and total tree height. Species specific taper and bark thickness equations are used to estimate diameter inside bark at breast and break height. Equations for *P. ponderosa*, *P. contorta*, and *A. grandis* were developed from samples obtained at Pringle Falls Experimental Forest (Garber and Maguire 2003). *P. menziesii* and *C. decurrens* taper equations were developed from samples taken in SW Oregon (Walters and Hann 1986). Average tree heights are estimated using equations obtained from simple linear regression analysis to predict tree height from DBH (using all intact snags measured in this study). Heights were only used for calculating the proportion of necromass transferred during the first break.



Total biomass estimates from allometric equations were consistent with biomass estimates using taper equations and the general tree form model for snags less than 70cm DBH, but do not correlate well for larger snags. To account for this disparity, the general tree shape model is used to estimate the total and broken volume of a snag. The proportion of total volume within the broken section of the snag, as calculated from the general tree shape model, is applied to the standing necromass (estimated from allometric equations) to determine the amount of necromass transferred to logs when breakage occurs.

Break height was recorded for all snags in field measurements. Equations to estimate break height were obtained from linear regression using PROC Reg in SAS 9.2 (SAS Institute Inc. 2008). Broken heights were log transformed to meet assumptions of normality (Ramsey and Shafer 2002). Details on statistical analysis are presented in Chapter 2.

### ***Bark Transfer***

In natural systems, crown and bark fragmentation occurs dependent and independent of snag fall or breakage (Harmon et al. 1986). “NecroDynamics” does not estimate transfer of bark or branches independently of snag fall or breakage. The proportion of bark necromass that transfers with breakage is equivalent to the proportion of stem wood transfer.

### ***Fine Woody Fuel Transfer***

FWD transfers from tree crowns to surface fuels with snag fall and breakage. The proportion of branches that transfer is dependent on the height of the break and total crown length. When snag break height is less than the assumed canopy base height, total branch biomass is transferred to surface pools. If break height is higher than canopy base height, then a proportion of branch biomass is transferred. For *Abies sp.*, *P. menziesii*, and *C. decurrens* canopy base height is assumed to be at 40%

of total height. *P. ponderosa* and *P. contorta* canopy base height is assumed to be at 50% of total height.

The proportion of branches transferred due to breakage is based on the assumption that tree crowns form a cone. The proportion of total branch necromass transferred is equal to the proportion of the cone volume above break height. Calculation of the cone volume requires the radius of the crown base and crown length. Crown radius is predicted from equations developed by Gill et al (2000). Crown length estimates are calculated from tree height estimates and the assumed crown base height. The proportion is applied uniformly to all branch sizes and only applies to the first breakage event. Total remaining branch necromass is transferred to surface pools in the second breakage event if any remains on the standing snag.

“NecroDynamics” was developed independently of surface fuel estimates providing us an opportunity to validate the model results. We compared model estimates with fuel transect estimates by including modeled “point” estimates as an independent variable in regression analysis. The point-to-point comparison provides us with a method to evaluate the accuracy of our model relative to estimates from fuels transects.

### ***Decomposition***

Decomposition rates vary dependent on moisture and temperature regimes as well as substrate quality. Decomposition rates of standing necromass are hypothesized to be less than surface decomposition rates in dry-climate forest ecosystems (Harmon et al. 1986). In this study, *A. grandis* and *P. ponderosa* snags were sampled along a chronosequence to estimate snag decay rates. *P. ponderosa* logs, felled as hazard trees during fire suppression activities, were sampled to estimate log decay rates. Twelve *A. grandis* and *P. ponderosa* snags were felled from 5 sites along a 7 year chronosequence at 1, 2, 5, 5, and 7 years post-fire. Target snag diameter was 41cm but ranged from 35cm – 50cm. *P. ponderosa* logs were sampled at the same sites except the site that was 1 year post-fire. Three cross-sections were

removed from each snag and log at 25%, 50%, and 75% of total height/length.

Samples were transported back to the College of Forestry at Oregon State University and analyzed for changes in wood density.

Several decomposition models have been proposed for woody material, but the negative exponential model is the most widely used and accepted model (Harmon et al. 1986). “NecroDynamics” assumes a negative exponential decay model for all coarse and fine woody detritus. Decomposition loss rate-constants were not sampled for all tree species found in the dry-mixed conifer forests of Oregon’s eastern Cascades. Wood density and k-constants for tree species not sampled in this study were obtained from published sources. Details on the statistical analysis for sampled species are presented in Chapter 2 (this document). Table 4.3 summarizes all k-constants and their sources used in the model.

Although no significant reduction in wood density was found for *P. ponderosa* snag sapwood or heartwood, decomposition is assumed to occur along the first 1.37 meters of the snag stem. This portion of the snag is assumed to have increased moisture content due to “wicking” and decreased exposure to solar radiation and wind, relative other portions of the snag. “NecroDynamics” assumes snag biomass below 1.37 meters decays at the same rate as logs. Biomass below 1.37 meters is modeled as a frustum of a neiloid (Husch, Miller and Beers 1993). The surface k-constant is multiplied by the proportion of total stem biomass present below 1.37 meters, to estimate the effective snag k-constant. The effective decay rate is re-calculated after any top breakage event and applied to the entire snag. We assume all species except *Abies sp.* have limited decomposition while standing and calculate their standing k-constant using this method. *Abies sp.* decay at the sampled decay rate from this study. Figure 4.2 provides an example of the changing effective decay rate for *P. ponderosa* snags (assuming a combined heartwood and sapwood decay rate of 0.024).

Table 4.3: A summary of all k-constants used in the “NecroDynamics” model. Size classes are equivalent to timelag fuel moisture size classes. All size classes decay at the same rate regardless of aerial or surface orientation.

Substrate	k-constant	Source
<i>P. ponderosa</i> snag sapwood	prop. of surface rate	Current study
<i>P. ponderosa</i> snag heartwood	prop. of surface rate	Current study
<i>P. ponderosa</i> log sapwood	0.0362	Current study
<i>P. ponderosa</i> log heartwood	0.01642	Current study
<i>A. grandis</i> snag	0.01488	Current study
<i>A. grandis</i> log	0.038	Harmon et al. 2007
<i>P. contorta</i> snag	prop. of surface rate	Current study
<i>P. contorta</i> log	0.027	Busse 1994
<i>P. menziesii</i> snag	prop. of surface rate	Current study
<i>P. menziesii</i> log	0.016	Harmon et al. 2007
<i>C. decurrens</i> snag	prop. of surface rate	Current study
<i>C. decurrens</i> log	0.016	Harmon et al. 2007
<i>Abies</i> sp. 1-hr size class	0.05	Harmon, unpublished data
<i>Abies</i> sp. 10-hr size class	0.04	Harmon, unpublished data
<i>Abies</i> sp. 100-hr size class	0.03	Harmon, unpublished data
<i>P. ponderosa</i> 1-hr size class	0.04	Harmon, unpublished data
<i>P. ponderosa</i> 10-hr size class	0.03	Harmon, unpublished data
<i>P. ponderosa</i> 100-hr size class	0.03	Harmon, unpublished data
<i>P. contorta</i> 1-hr size class	0.05	Harmon, unpublished data
<i>P. contorta</i> 10-hr size class	0.045	Harmon, unpublished data
<i>P. contorta</i> 100-hr size class	0.03	Harmon, unpublished data
Plant Litter	0.28	Monleon and Cromack 1996

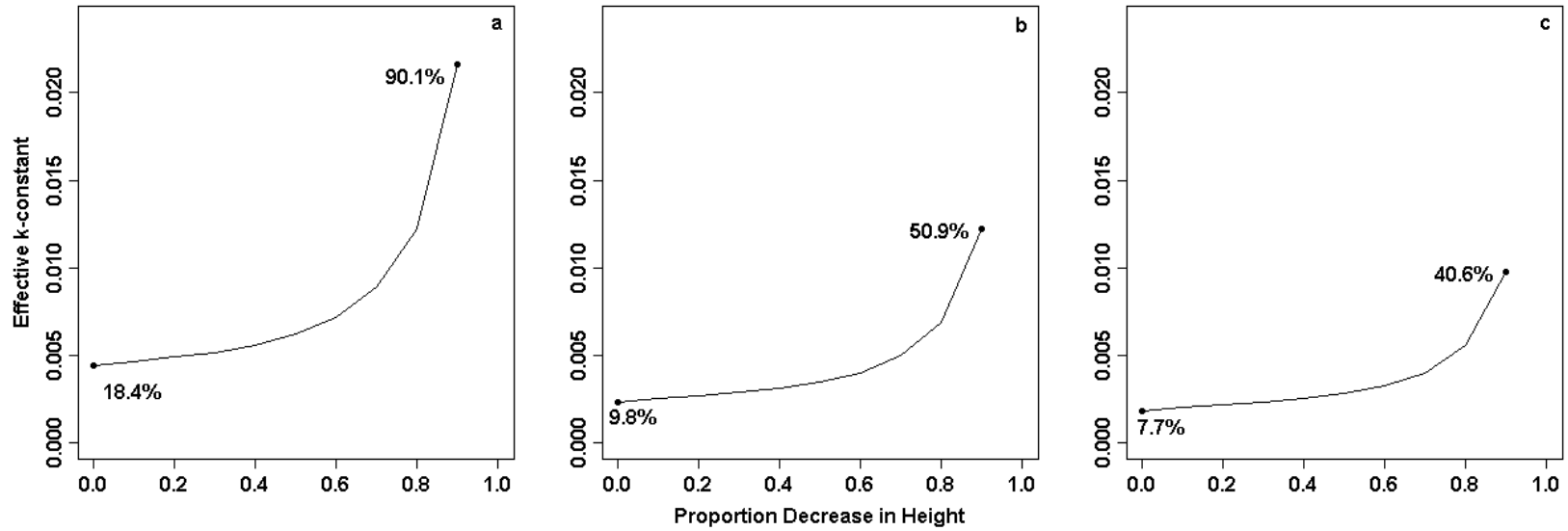


Figure 4.2: Example of the effective decay rate for snags of varying sizes that did not exhibit, or are assumed to not have, statistically significant decomposition throughout the stem. The effective decay rate is calculated as the proportion of surface decay rate equal to the proportion of aboveground tree volume found below 1.37 meters and is applied to the entire stem. In this example, the decay rate is based on *P. ponderosa*'s combined decay rate constant ( $k = 0.024 \text{ yr}^{-1}$ ). Percent values indicate the proportion of effective decay rate to surface decay rate. a = 20 cm DBH snag, b = 50 cm DBH snag, c = 100 cm DBH snag.

### ***Combustion of CWD by Decay State***

The amount of CWD available for combustion varies by decay state, position and interaction with other combustible plant material (Albini and Reinhardt 1997). To date, no published studies have quantified the proportion of log biomass combusted by decay class, but rotten logs have been reported to combust with greater efficiency and heat than sound logs (Albini and Reinhardt 1997, Monsanto and Agee 2008). Tinker and Knight (2000) reported that only 8% of log biomass was combusted in fires in Yellowstone National Park and another 8% was converted to charcoal. Although this is an average across an entire stand with varying abundances of log biomass by decay class, we used 8% as the combustion efficiency of sound wood, defined as woody detritus with a density  $>0.30 \text{ g cm}^{-3}$ . Brown (1974) assumed rotten woody detritus had a density of  $0.30 \text{ g cm}^{-3}$ , which does not correlate directly to studies of wood density by decay class but roughly equates to decay class 2-3 for species found in dry-mixed conifer forest (Harmon et al. 2008). Decay class 2 logs are relatively sound and decay class 3 logs have friable sapwood and little bark remaining attached (Cline et al. 1988). By the time logs reach decay class 4, density values tend to converge around  $0.20 \text{ g cm}^{-3}$ , although variation does occur within and among species (Harmon et al. 2008). We believe decay class 4 logs represent rotten material and assume that wood densities between  $0.30$  and  $0.20 \text{ g cm}^{-3}$  represent partially rotten material. “NecroDynamics” assumes partially rotten logs combusts with a total efficiency of 0.50 and fully rotten logs (i.e.  $<0.20 \text{ g cm}^{-3}$ ) combust with an efficiency of 0.95.

### ***Salvage Logging***

We used “NecroDynamics” to compare the effects of salvage logging on the succession of FWD and CWD. Live vegetation is not modeled by “NecroDynamics”, so the modeled successional trajectories show the effects of salvage logging on legacy woody detritus only. Our salvage scenario assumes 80% of snag biomass  $>23\text{cm}$  is removed from the system to be processed offsite, and 50% of snag biomass  $<23\text{cm}$  is

immediately felled during the salvage operation. Individual trees are randomly selected for removal. Crown fuels are transferred immediately to the forest floor if a tree is salvaged, with a 10% offsite removal due to burial or transfer to landings. All remaining snags fragment using the same modeling parameters for natural stand development.

## Results

### *Heartwood Biomass and Effective Decay Rate*

“NecroDynamics” calculates heartwood biomass using a snag’s DBH, height and stem taper. The internal heartwood structure is assumed to develop in the shape of a cone. Diameter inside bark at the base of the snag is predicted from DBH using *P. ponderosa* taper equations (Garber and Maguire 2003). The diameter of heartwood at the base of the tree is calculated using the regression coefficients from Table 4.4.

Heartwood height is modeled as the point along the tree stem where diameter inside bark is 17.5 cm (i.e. the stem height where heartwood would not be present as estimated by the regression equation in Table 4.4). This model assumes minimal decrease in sapwood radial thickness with height increase along the tree stem. At 100cm diameter inside bark, sapwood radial thickness is estimated to be 10.08cm, decreasing to 8.75cm at the point when heartwood is assumed to be absent.

Table 4.4: Linear regression coefficients for estimating *P. ponderosa*’s heartwood diameter from diameter inside bark (Adjusted R-squared = 0.9298).

Variable	Estimate	Std Error	p-value	LCL <sup>1</sup>	UCL <sup>1</sup>
Intercept	-16.68	1.35812	<0.0001	-19.374	-13.986
Diameter inside bark	0.96501	0.02624	<0.0001	0.91297	1.01706

The regression equation suggests that snags < 20cm DBH have negligible heartwood biomass, supported by observation of snags in this study. At 40cm DBH, average heartwood biomass is < 20% of total volume of a cross section sample.

Figure 4.3 depicts the percent of total volume of the cross section samples that were heartwood.

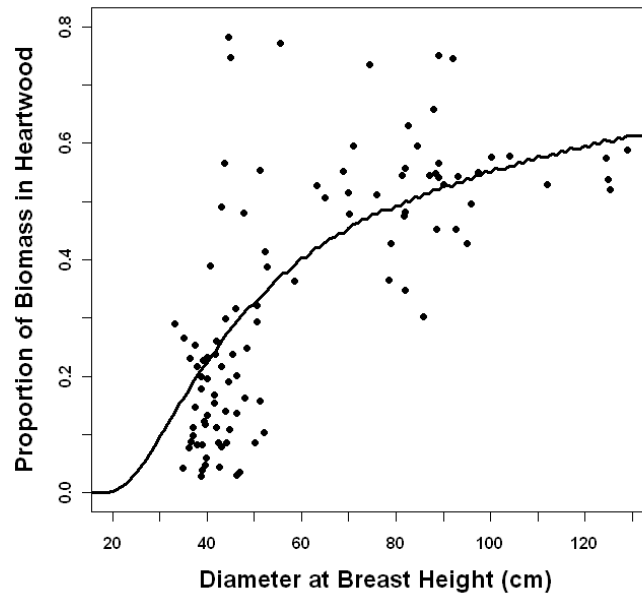


Figure 4.3: Percentage of total *P. ponderosa* biomass made up of heartwood. Trees <20 cm are assumed to have no decay resistant heartwood. Points are observed proportions from cross-section samples and the line is the modeled proportion.

The points represent actual proportions of total biomass made of heartwood for the cross-sections taken from 25% of total tree height, not the entire tree. The line represents the modeled proportion of total biomass made of heartwood (not a statistically derived linear relationship). Considerable variability exists in the proportion of total biomass made of heartwood, especially at smaller diameters, since total leaf area, water stress and age of the tree all contribute to the development of heartwood. In general, the model appears to capture the average proportion of volume present. The effective decay rate for the entire tree of this species is a combination of heartwood and sapwood decomposition loss rate-constants as depicted in Figure 4.4.



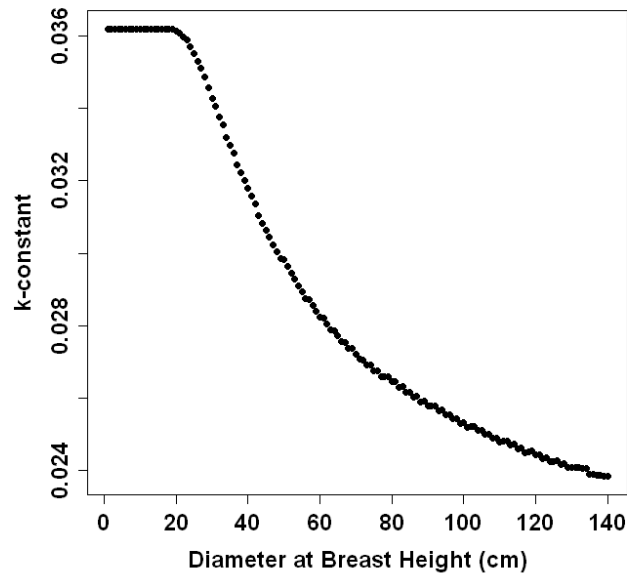


Figure 4.4: Effective decomposition loss rate-constant for *P. ponderosa* logs. The rate is calculated as a combination of sapwood and heartwood k-constants and the ratio of heartwood to sapwood volume.

### ***Woody Fuel Succession***

“NecroDynamics” was developed independently of our fuel transect samples, providing independent samples for comparison to predicted model values and an assessment of model accuracy. We used individual predicted point values from modeled stands for each plot, at their representative year. These values were used as a substitute for time since fire (number of fragmented snags for 10-hr fuel) in each fuel size class. Table 4.5 provides linear regression estimates comparing model output (“Model” variable) with fuel transect estimates for fine and course woody fuels. Evaluation of general modeling trends suggests that the transfer mechanism in the model accurately captured the deposition of FWD and CWD from biological legacies.

Table 4.5: Comparison of predicted values from “NecroDynamics” and independently sampled fuels transect data. The variable “Model” is the modeled fuel biomass value for each plot at the time since fire of that plot. The model variable substitutes for time since fire in all regression analyses. A significant correlation suggests the model is capable of accurately predicting fuel biomass at a given year since fire.

Dependent		Independent	95% Confidence Intervals				
Fuel Type	Variable	Variable	Estimate	Std Error	p-value	Lower	Upper
1-hr (0-0.62cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	0.36355	0.07499	<0.0001	0.20911	0.51799
		Model	0.19836	0.17257	0.2613	-0.15706	0.55378
		avg. litter biomass (Mg ha <sup>-1</sup> )	-0.03063	0.00793	0.0007	-0.04696	-0.0143
		% shrub crown dead	0.06251	0.00793	< 0.0001	0.04618	0.07883
Model Adj. R <sup>2</sup> = 0.7341							
10-hr (0.63-2.54cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	1.14668	0.36578	0.0044	0.39334	1.90003
		Model	1.033	0.16866	< 0.0001	0.68563	1.38036
		avg. litter biomass (Mg ha <sup>-1</sup> )	-0.18205	0.04838	0.0009	-0.2817	-0.08241
		% shrub crown dead	0.12998	0.04302	0.0057	0.04139	0.21857
Model Adj. R <sup>2</sup> = 0.5865							
100-hr (2.55 - 7.62cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	0.99853	0.79325	0.2185	-0.62637	2.62342
		Model	1.38982	0.22408	< 0.0001	0.93081	1.84882
		Model Adj. R <sup>2</sup> = 0.5637					
Total FWD (< = 7.6cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	2.37798	0.90037	0.0138	0.52723	4.22873
		Model	0.77007	0.16462	<0.0001	0.43169	1.10846
		% shrub crown dead	0.23913	0.0756	0.0039	0.08373	0.39453
Model Adj. R <sup>2</sup> = 0.6752							
1000-hr (> 7.62cm)	Biomass (Mg ha <sup>-1</sup> )	Intercept	7.70673	7.60742	0.3197	-7.87635	23.28982
		Model	1.13593	0.14407	< 0.0001	0.84081	1.43105
Model Adj. R <sup>2</sup> = 0.6784							

The model variable was not a statistically significant predictor variable for estimating 1-hr fuel biomass (Table 4.5). This lack of correlation further indicates the potential non-linearity of 1-hr fuel inputs from varying proportions of crown scorch/combustion and inputs from live vegetation. Assuming the transfer function in “NecroDynamics” is a potential trajectory of 1-hr fuel biomass, we see the importance of incomplete crown combustion (year 1) and inputs from live vegetation (beginning in year 15) on post-fire 1-hr fuel biomass (Figure 4.5).

“NecroDynamics” estimated peak 1-hr fuel accumulation from biological legacies occurred 12-13 years post-fire, after which decay losses were greater than input rates and 1-hr biomass decreased (Figure 4.5). This estimate represents transfer of 1-hr biomass with snag fall and breakage which may underestimate the rate of 1-hr fuel deposition from biological legacies if this material fragments independently of snag fall or breakage. If the rate of 1-hr fuel deposition increased, peak surface accumulation would occur earlier than estimated by “NecroDynamics”.

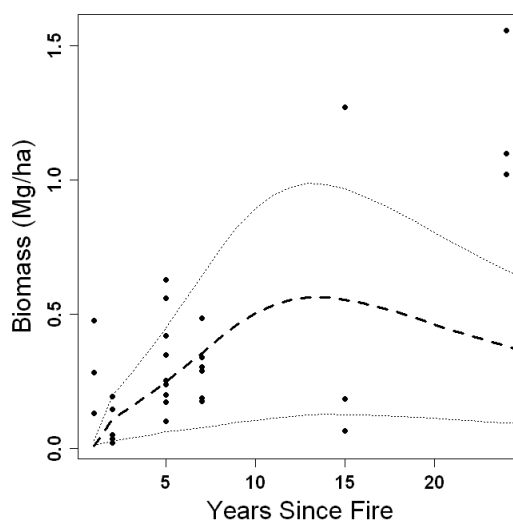


Figure 4.5: 1-hr fuel accumulation during 24 years of succession following high-severity fire. Points represent fuels transect estimates of 1-hr biomass. Dashed line is an average stand prediction from “NecroDynamics”. Dotted lines are stands with predicted high and low values. Fuel transect estimates were not used to develop the model so the points represent independent samples.

Since 10-hr biomass accumulation is positively correlated with snag fall and breakage, substituting “NecroDynamics” model output offered an opportunity to comprehensively account for snag dynamics and decomposition. When model output was included as a predictor variable for 10-hr fuel biomass, it was significantly correlated with 10-hr fuel biomass. The correlation resulted in a model explaining more variation in our data than simple linear regression analysis in Chapter 3.

The amount of variation explained in 10-hr fuel biomass increased by 7%, creating a richer model that captured additional variables not originally found in regression analysis (Table 4.5). 10-hr fuel transect estimates peaked in year 15, coinciding with predicted peak accumulations at 14 – 16 years from “NecroDynamics” (Figure 4.6). The “new” predictor variables are the same statistically significant variables for predicting 1-hr fuel biomass but with varying magnitude (Table 3.2).

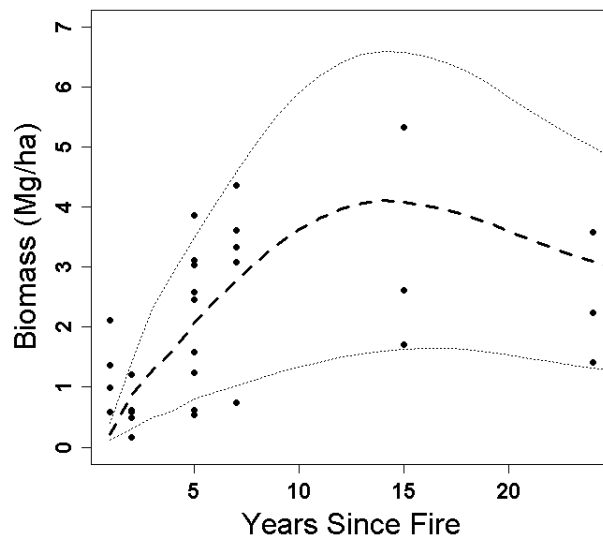


Figure 4.6: 10-hr fuel accumulation during 24 years of succession following high-severity fire. Points represent fuels transect estimates. The dashed line represents a plot with average 10-Hr fuel loading modeled using “NecroDynamics”. Dotted lines are modeled trajectories for a stand with high and low 10-hr biomass.

“NecroDynamics” predicted that maximum accumulation of 100-hr biomass occurred around 15 years post-fire, the point when crown biomass transfer from snag fall and breakage was complete and/or was less than decomposition losses (Figure 4.7). Model output accounts for 56.4% of the variation found in the fuels data, a reduction of 12% in the amount of variation explained by regression analysis (compare Tables 3.2 and 4.5).

“NecroDynamics” supports the non-linear successional trajectory of 100-hr fuels following high-severity fire observed in field data but generally under predicts biomass accumulation. We believe the discrepancy is a function the line intercept method not accounting for decomposition. The similarity in fuel trajectories between fuels transect estimates and model outputs suggest “NecroDynamics” process of fuel transfer is adequate at capturing deposition rates (Figure 4.7).

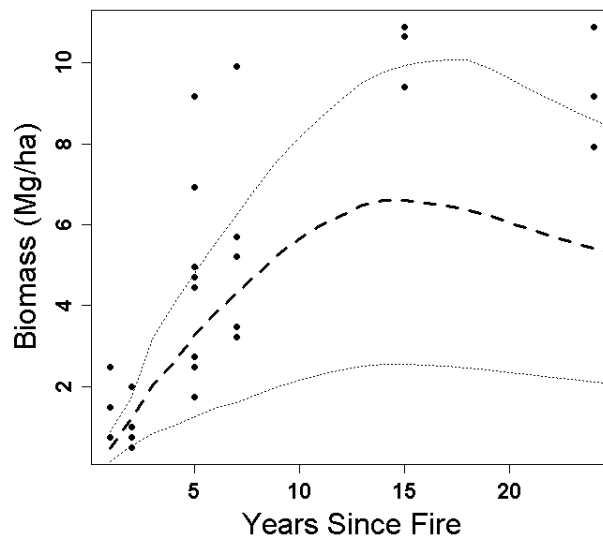


Figure 4.7: 100-hr fuel accumulation during 24 years of succession following high-severity fire. Dashed line represents “NecroDynamics” modeled output of a stand with an average amount of 100-hr biomass and the dotted lines represent high and low values.

Field observations and modeled outputs correlate well when estimating 1000-hr fuel accumulation. Substituting the model variable for time since fire in regression analysis increased the amount of variation in the data explained by the model by 5% (Table 4.5). Modeled CWD biomass accumulation indicated peak accumulation occurred 23 - 26 years following high-severity fire, increasing with increasing stand biomass. The accumulation trajectories were consistent with trends observed from field data (Figure 4.8). This peak does not represent the end of CWD input from snags; instead it represents the point when biomass loss from decomposition is equal to biomass input from snag fragmentation.

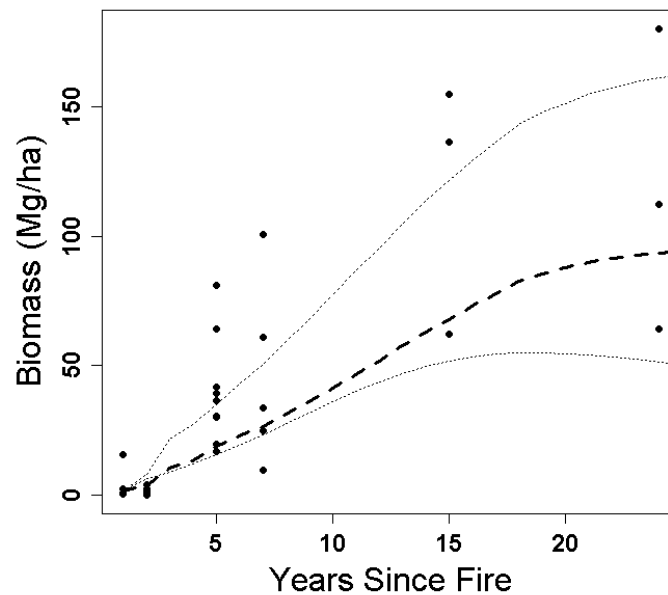


Figure 4.8: CWD fuel accumulation during 24 years of succession following high-severity fire. Dashed line represents “NecroDynamics” modeled output of a stand with an average amount of CWD biomass and the dotted lines represent high and low values.

## Discussion

### *Fine Woody Detritus Succession*

Accumulation of 1-hr fuel biomass following high-severity fire is dominated by inputs from live vegetation. For 1-hr fuels, there was no improvement to linear regression fit using modeled outputs, although time since fire doesn't explain a lot of the variation in 1-hr biomass. We believe this occurs for two reasons. First, the initial pulse of 1-hr fuels from scorched but not consumed canopies are not predicted by "NecroDynamics" since we assume combustion of overstory crowns equals average values obtained from broad, high-severity fire conditions. Additionally, after year 7 the ability of the model to predict observed biomass diminishes because 1-hr fuels continuously assimilate into the litter layer and new inputs accrue from *de novo* shrub vegetation, parameters not included in "NecroDynamics".

Peak 10-hr biomass estimates are dependent on total amounts transferred with snag fall and breakage as well as persistent decomposition. FWD fuel transects do not estimate decay classes so the biomass reduction observed in our data might be a function of particle fragmentation and assimilation into the litter layer. Therefore, the 10-hr biomass reduction exhibited after 14 years in both modeled and sampled trajectories may be a result of different mechanisms which cannot be discerned from the current study.

"NecroDynamics" captures non-linear trends in 100-hr fuels succession but estimates are consistently lower than transect estimates (Figure 4.7). Underestimation of 100-hr biomass by "NecroDynamics" is partially explained by decomposition. Decay classes were not assigned to 100-hr fuel particles so mass loss due to decomposition is not fully accounted for in the line intercept method. For example, when a stand having average 100-hr fuel biomass is modeled without decomposition, we see increased correlation between model results and fuels transect estimates (Figure 4.9).

The lines in Figure 4.9 appear highly correlated suggesting much of the discrepancy between model output and linear regression can be attributed to decomposition. During field sampling, noting decay classes for 100-hr fuel particles could account for this discrepancy and improve overall estimates of biomass obtained from line-intercept or fixed area sampling.

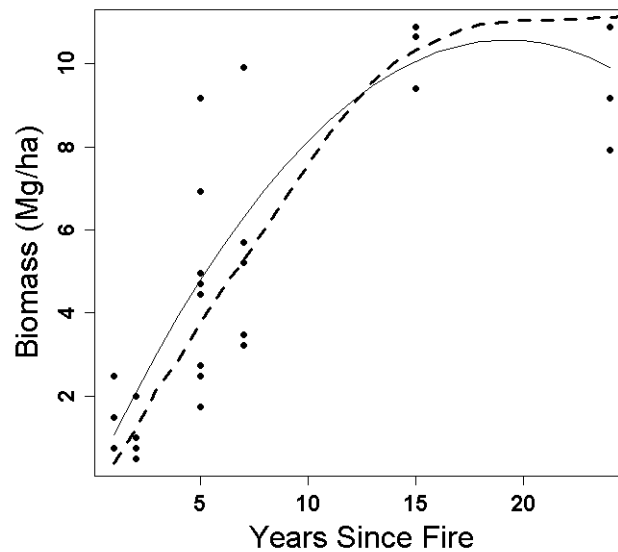


Figure 4.9: Average 100-hr biomass estimates from regression analysis and “NecroDynamics” without decomposition. The solid line is the least squared means fit of 100-hr biomass as predicted from time since fire and its quadratic term. The dashed line is a stand sampled in this study with an average amount of 100-hr biomass.

Although it appears decomposition could account for the majority of the difference, other processes occur that complicate this relationship. Fragmentation of larger fuel particles into smaller pieces is not accounted for in “NecroDynamics”, although some of this is represented in decomposition loss rate-constants. Assimilation of this material into the litter layer isn’t accounted for either, but likely plays a smaller role than it does for smaller fuel particles. Lastly, the entire tree stem biomass was assumed to be part of CWD but the tops of trees consist of some 100-hr biomass. We considered the effects of this to be minimal compared to the strong



relationships we have found. Overall, model results for 100-hr fuel biomass explain 12% less variation than linear regression analysis. We believe that predictions from “NecroDynamics” are likely more accurate since biomass reductions from decomposition are not accounted for in fuels transects, which may account for the majority of the discrepancy found between “NecroDynamics” and transect estimates. Additional modeling and field sampling exercises are needed to determine if this conclusion is correct.

Transfer of snag crown biomass from fall and breakage develops consistent patterns of FWD deposition. Peak accumulation across all FWD size classes occurs between 15 and 20 years, consistent with linear regression analysis. Inputs may occur beyond these years but decomposition loss is greater than inputs, effectively reducing the amount of biomass on site. Figure 4.10 depicts FWD succession in three stands that represent high, medium and low total biomass stands sampled from our sites (as described in Chapter 2, Figure 2.5).

Individual crown biomass is directly related to DBH, but the amount of crown biomass found in individual fuel size classes at the hectare scale is dependent on the total number of trees, their species composition and the diameter distribution of those trees. Figure 4.10 shows that the medium biomass stand has higher amounts of 100-hr fuel. This occurs because of the relatively large number of *P. ponderosa* and *P. menziesii* snags found in the plot.

“NecroDynamics” uses average fall and breakage rates so stochastic environmental events (i.e. wind, snow loads) that increase deposition beyond average conditions will increase observed surface biomass. This occurred in sites we sampled, most notably in sampled sites at years 5 and 7 (Figures 4.5 – 4.8). We expect that stands experiencing increased fall or breakage in earlier years will converge with modeled outputs in future years as biomass decays quicker and inputs from snags decrease. Tracking these plots over time would likely yield individual pathways of succession similar to the vector pathways reported for woody decomposition by Harmon et al. (2000).

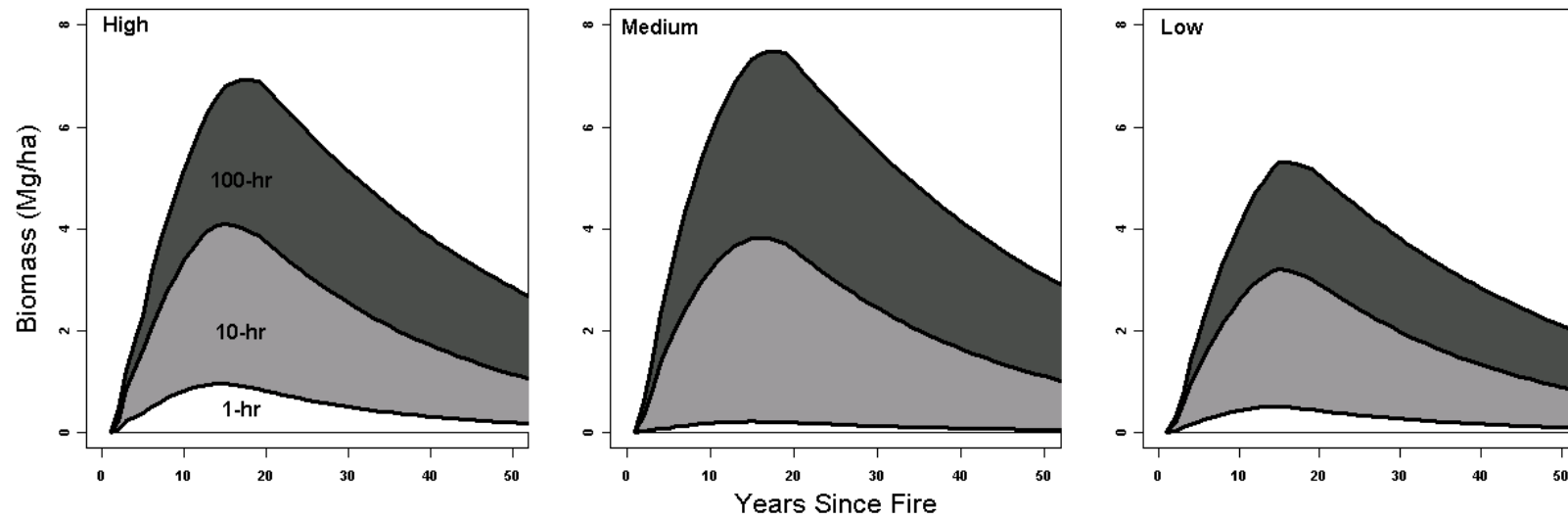


Figure 4.10: Modeled FWD fuel succession for 3 different stands. a = Davis Lake Plot 2 (high biomass), b = Cherry Creek Plot 3 (medium biomass), c = GW Plot 3 (low biomass). Biomass in each fuel size class varies by trees/hectare, diameter distribution and species.

### *Coarse Woody Detritus Succession*

Dry-mixed conifer forests are dominated by multiple tree species that fragment and decay at varying rates (Bull 1983, Everett et al. 1999, Harmon et al. 2007). Initial snag biomass must be estimated, for example, using species specific allometric equations predicting total stem biomass from diameter at breast height (Jenkins et al. 2004). Total stem biomass is exponentially related to diameter at breast height so larger diameter trees have increasingly more biomass than smaller diameter trees. For example, a *Pinus ponderosa* tree 100cm DBH is equivalent to 13.58, 41cm DBH trees. The non-linear relationship of biomass and DBH complicates drawing simple relationships between trees per hectare, basal area per hectare and total stem biomass. In general, higher stem biomass sites have more trees with DBH >70 cm leading to slower snag fragmentation and, in the case of species with decay resistant heartwood, slower decomposition with increasing heartwood biomass. Additionally, the higher biomass sites we sampled supported a larger population of *P. menziesii*, a more decay resistant species (Harmon et al. 2007).

Snag fragmentation trends do not vary greatly by initial biomass in dry-mixed conifer forests of Oregon's eastern Cascades. The cumulative deposition of CWD biomass from snag breakage and fall develop the same pattern of snag biomass over time, although the total amount of biomass at any given time will vary by initial snag biomass (Figure 4.11). The abundance of large diameter trees (which have a disproportionate amount of biomass and slower fall rates) in high biomass sites results in significantly more snag biomass present past 35 years relative to lower biomass sites.

Additionally, the slower deposition rates of high biomass stands result in peak log accumulation occurring later relative low biomass stands. Log biomass in the high and medium biomass stands peaks 26 years post-fire, although biomass values  $\pm 4$  years are within 5 Mg ha<sup>-1</sup> of the peak.

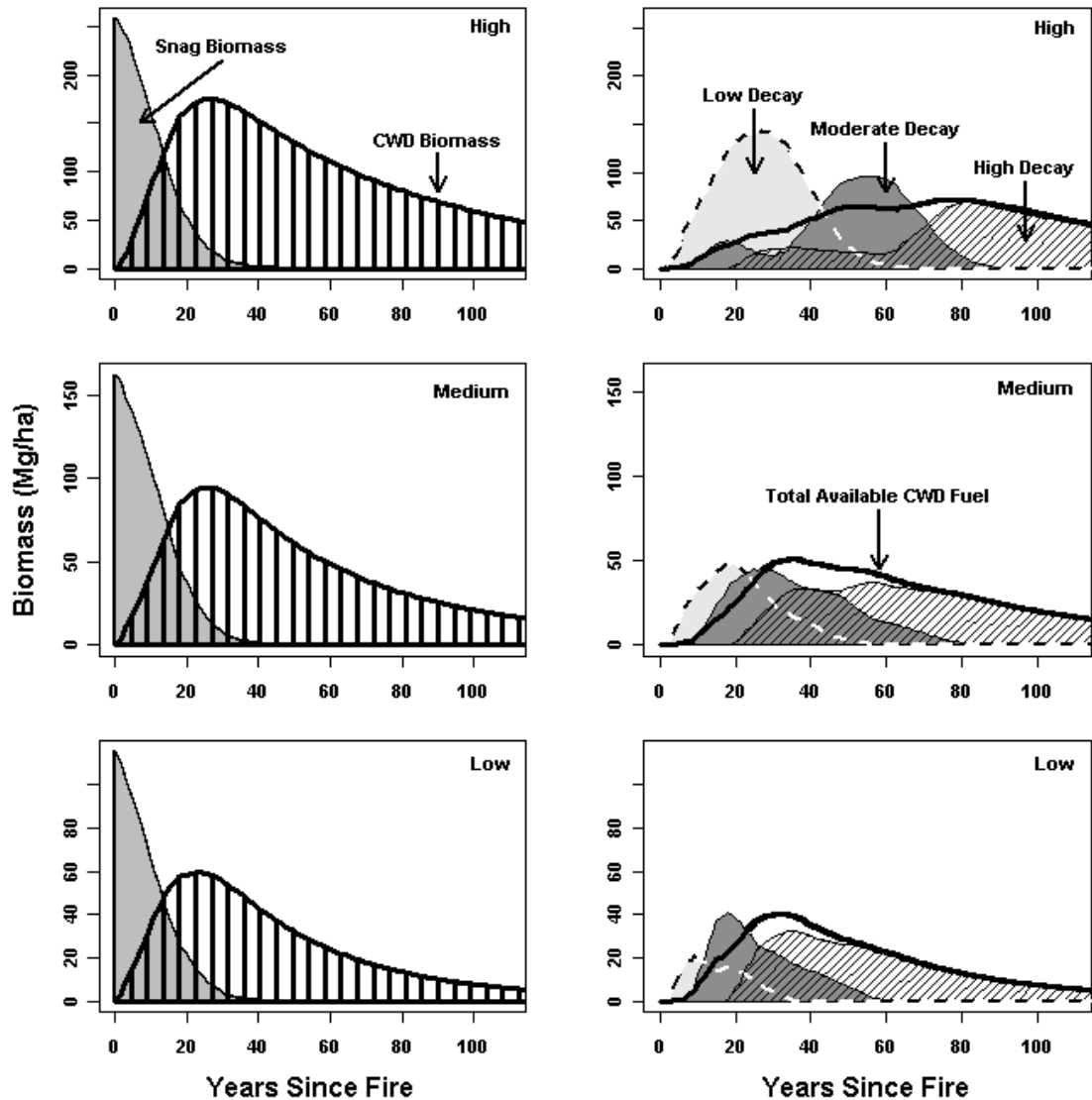


Figure 4.11: CWD succession and fuel availability. The y-axis scale varies by stand condition. Graphs on the left (high, medium and low biomass) depict snag fragmentation and total log accumulation following high-severity fire. Shaded region is snag biomass and striped is log biomass. Graphs on the right depict the decay state of logs and total available surface CWD fuel (solid black line) for the same stands. Low decay (light gray) is all wood biomass with a density  $>0.30 \text{ g/cm}^3$ , moderate decay (dark gray) is all wood between  $0.20$  and  $0.30 \text{ g/cm}^3$ , and high decay (striped) is all wood  $<0.20 \text{ g/cm}^3$ . Consumption amount for low decay state is assumed to be  $0.08$ , moderate is  $0.50$  and high is  $0.95$ .

Peak log biomass in low biomass stands occur at 23 years post-fire with negligible biomass difference for several years before and after the actual peak year. Table 4.6 summarizes peak log accumulation and available fuel for each stand condition used in this example.

Table 4.6: Peak surface CWD and available fuel biomass for three stand conditions. Davis Lake #2 represents a high biomass stand, Cherry Creek #3 a medium stand and GW#3 a low biomass stand. Values in parentheses are percent of total snag biomass. Year = number of years post-fire.

Plot	Total Snag Biomass (Mg ha <sup>-1</sup> )	Peak CWD Biomass		Peak Available Fuel Biomass	
		(Mg ha <sup>-1</sup> )	Year	(Mg ha <sup>-1</sup> )	Year
Davis Lake #2	258.3	174.9 (67.7%)	26	72.2 (28%)	75
Cherry Crk #3	161.9	94.7 (58.5%)	26	51.1 (31.6%)	35
GW #3	115	59.3 (51.6%)	23	40.2 (35%)	32

Snag breakage and fall are the primary mechanism controlling the timing of log accumulation, but continued decomposition reduces the total amount of log biomass remaining at the time of peak accumulation. Decomposition reduces total stand biomass by 32-48% before peak accumulation occurs (Table 3.5). Legacy CWD biomass will continue to decrease until all legacy material is lost to heterotrophic respiration, leaching, or assimilated into soil horizons (Harmon et al. 1986). Future CWD will develop as tree death occurs from the regenerating stand (Harmon 2009)

Variation in the combustion efficiency of logs by decay state develops unique patterns in available surface CWD fuel by initial stand biomass. Total available surface CWD fuel is the sum of log biomass in each decay state (i.e. low, moderate or high decay), multiplied by their respective combustion efficiencies. Available surface CWD fuel in the low and medium biomass stands peaks at 32 and 35 years post-fire, respectively (Table 4.6). Approximately 1/3 of the initial snag biomass is available for combustion, assuming our combustion efficiency values are representative. These stands follow a similar trajectory with an early and pronounced peak in available surface CWD fuel, although the medium biomass stand has a less pronounced peak.

The primary cause of this trend is their compositional dominance by *Abies sp.* snags which decay relatively rapidly as snags and logs (Chapter 2). Lower decay rates indicative of *P. ponderosa* heartwood, *P. menziesii* and *C. decurrens* cause delayed inputs and lower carbon releases, reducing their transition to highly combustible logs.

The effect of decay resistant species on available surface CWD fuel is more evident in the high biomass stand. A large proportion of biomass in these stands consist of *P. menziesii* and *P. ponderosa* snags >60 cm DBH, decreasing the average stand decomposition rate. “NecroDynamics” tracks each log piece individually through time which is evident by the distinct peaks of each decay state. *P. menziesii* and *P. ponderosa* have relatively high wood densities for conifer in dry-mixed conifer forests (0.45 and 0.38 g cm<sup>-3</sup> respectively), and low decay rates so they contribute to the prolonged transition through the medium density decay class and late accumulation in the high density decay state. The peak accumulation of available surface CWD fuel occurs after 72 years of succession, although the stand does have greater than 60 Mg ha<sup>-1</sup> of available fuel from 44-96 years post-fire.

### **Modeled Fuel Succession Following Salvage Logging**

Dry-mixed conifer forests in Oregon’s eastern Cascades have experienced structural changes due to a reduction in fire for greater than a century, first noted in 1943 (Weaver 1943). Although high-severity fire did occur in these forest systems, the extent has increased recently due to the lack of fire across the landscape (Spies et al. 2006). Pre and post-fire management strategies are being conducted to reduce the impacts of fire today and in the future. Management strategies must maintain the ecological functions supported by snags and logs, while mitigating future fire hazard (Brown et al. 2003). Post-fire management strategies, therefore, should be made based on long-term impacts of today’s actions on the availability of CWD to meet ecological needs and the potential for this material to increase future fire severity.

Future fire severity may be elevated by an increased abundance of many fuel types, including fine woody detritus, coarse woody detritus, and dense live vegetation

(Donato et al. 2006, Thompson et al. 2007, Monsanto and Agee 2008). Each of these fuel types has the ability to cause future high-severity fire independently or through complex interactions among the fuel types. Salvage logging and reforestation will affect these fuel types in various interacting ways (McGinnis et al. 2010).

### ***Role of Fine Woody Detritus***

Fine woody detritus is the primary fuel component that affects fire rate of spread, flame length and fireline intensity (Byram 1959, Rothermel 1972). FWD fuels, along with ladder fuels, are targeted for mechanical reduction to minimize the effects of a fire event on residual trees, but are only effective if FWD fuels are treated by prescribed fire (Raymond and Peterson 2005, Prichard et al. 2010). Elevated levels of fine fuels and the resulting fire behavior suggest fine fuels are of critical concern when attempting to mitigate post-treatment fire behavior.

Salvage logging causes elevated levels of FWD immediately following the operation. Donato et al. (2006) reported salvage logging following high-severity fire in southern Oregon increased median FWD biomass by >300% compared to non-salvaged units (totaling 6.4 Mg ha<sup>-1</sup>). McGinnis et al. (2010) reported 9.8 to 18.7 Mg ha<sup>-1</sup> of FWD (53-62% higher than unsalvaged sites 4-6 years post-fire) occurred in salvage logged sites following high-severity fire in dry-mixed conifer forests of California's Sierra Nevada Mts. By 13-20 years post-fire, they report no significant differences between salvaged and unsalvaged sites. Our estimates of FWD following salvage logging are similar to these studies, even under varying initial biomass. According to "NecroDynamics" results and our assumed salvage logging scenario, salvage logging increases surface FWD fuels for about 15 years, although differences are negligible after approximately 10 years. Figure 4.12 compares FWD succession for a salvage logged stand relative to a naturally developing stand. Stand conditions simulated for this figure are GW #3, the low biomass stand depicted in Figure 2.5. General trends are similar across all stand conditions but the magnitude of FWD by size class will vary by  $\pm 1$  Mg ha<sup>-1</sup>. Increased levels of FWD can increase fire

behavior and effects, reducing salvage loggings ability to mitigate future fire behavior by elevating fire hazard for up to 20 years (Donato et al. 2006, McGinnis et al. 2010).

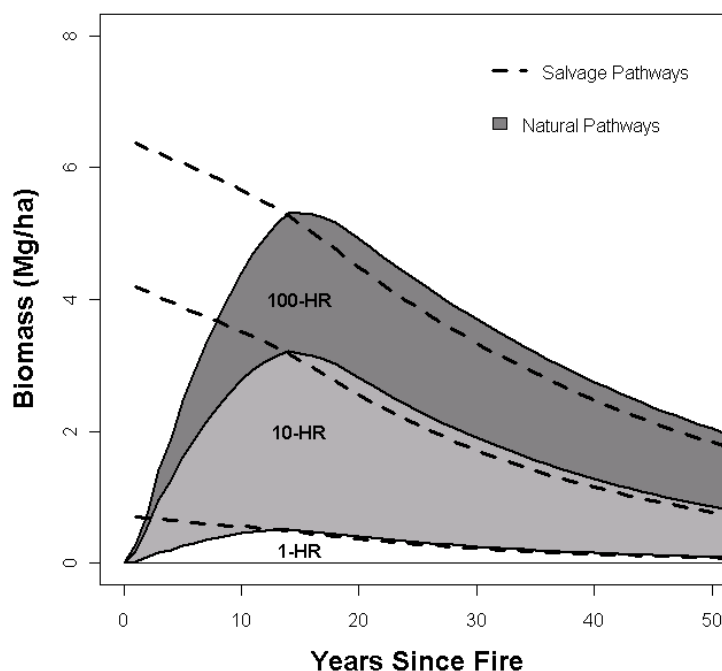


Figure 4.12: FWD succession for a salvaged and unsalvaged stand following high-severity fire. Salvage logging elevates FWD for 15-20 years before the values converge. Only legacy biomass is considered in this model.

Although elevated FWD fuel biomass is increased following salvage operations, fire hazard and behavior may not be exacerbated by this condition due to a lack of fuel continuity. A fuel particle must be within close enough proximity to transfer heat from combustion to a neighboring fuel particle (Albini and Reinhardt 1997, Rothermel 1972). Without the development of a continuous litter layer or other fine fuels, we do not expect a flaming front to carry through this material. For example, brush disposal following logging operations occurs during “red needle” conditions, usually after 1 year of curing. Heat generated by combustion of the “red” foliage carries flaming fronts through the fuel bed and generated enough heat to combust other FWD, and to some extent, CWD. Without fine fuel particles with



sufficient continuity, fire spread is not rapid and enough heat may not be generated to combust larger fuels (Albini and Reinhardt 1997).

Following salvage logging, it is possible that herbaceous fuels accumulate rapidly enough to substitute for the foliage biomass in slash disposal treatments. In the eastern Cascades, herbaceous loadings are generally low following high-severity fire with a few exceptions. In these cases, the increase in FWD following salvage logging will increase fire effects, making salvage logging an ineffective means of fuels reduction in the first 10 - 15 years following a fire.

Inputs from live vegetation are important in developing a litter layer, creating fuel continuity, and increasing 1 and 10-hr fuel loadings. Salvage logging alone does not reduce shrub growth so the effect of inputs from live vegetation will be relatively the same between salvaged and unsalvaged sites (Lopez Ortiz 2007). However, shrub control associated with planting will reduce shrub biomass and alter inputs from live vegetation but can increase the level of herbaceous fuels on site (McGinnis et al. 2010). Rapid reforestation of conifer trees through shrub control will likely delay, but ultimately substitute for fuel inputs and effectively negate any benefits received from shrub control, although reduced shrub biomass can decrease fire behavior if this fuel layer sustains combustion (McGinnis et al. 2010, Thompson et al. 2007).

### ***Role of Coarse Woody Detritus***

Combustion of high levels of CWD may have the same effect on aboveground vegetation as combustion of areas with high loadings of FWD, although the mortality mechanism may be different. Tree mortality occurs from CWD fuel through canopy mortality as a result of high levels of energy release and/or tree mortality by root necrosis. In eastern Washington dry-mixed conifer forests, CWD combustion was estimated to cause tree mortality across ~25% of a stand through root necrosis (Monsanto and Agee 2008). The impact of CWD on root necrosis is a function of total area covered by CWD and the ability of this material to ignite and sustain combustion. Reduction of total CWD numbers and total CWD loadings through

salvage logging will reduce the impact of CWD on fire severity through this mechanism. The area affected by CWD is beyond the scope of this study but is an important factor in determining the potential impacts of the combustion of CWD on future fire effects. Figure 4.13 shows the succession of CWD after salvage logging for the same three stand conditions depicted in Figure 4.10.

In addition to the immediate effects of the combustion of CWD, long-term impacts on site productivity can occur (Debano et al. 1998). Hebel et al. (2009) reported a decrease in site productivity following complete combustion of logs in dry-mixed conifer forests of Oregon's eastern Cascades. In these forests, reduced productivity is likely ephemeral since the sites are being colonized by microbial communities, shrub and herbaceous plants as well as being encompassed within a larger environment of nitrogen fixing *Ceanothus velutinus* (Hebel et al. 2009).

## Conclusions

Dry-mixed conifer forests of Oregon's eastern Cascades consist of multiple tree species in varying abundances, creating complex snag and decomposition dynamics. To understand and account for variation in fall, breakage and decay rates, a model had to be developed. "NecroDynamics" incorporates these variables so that stand level CWD dynamics can be better understood and incorporated into decisions relevant to management of post-fire environments.

Salvage logging to reduce future fire severity has been proposed as an ecological justification for conducting the operation, without long-term investigations regarding the benefits and consequences of these operations on ecosystem functioning (Beschta et al. 2004, Sessions et al. 2004). Our study focused on understanding the successional dynamics of post-fire fuels.

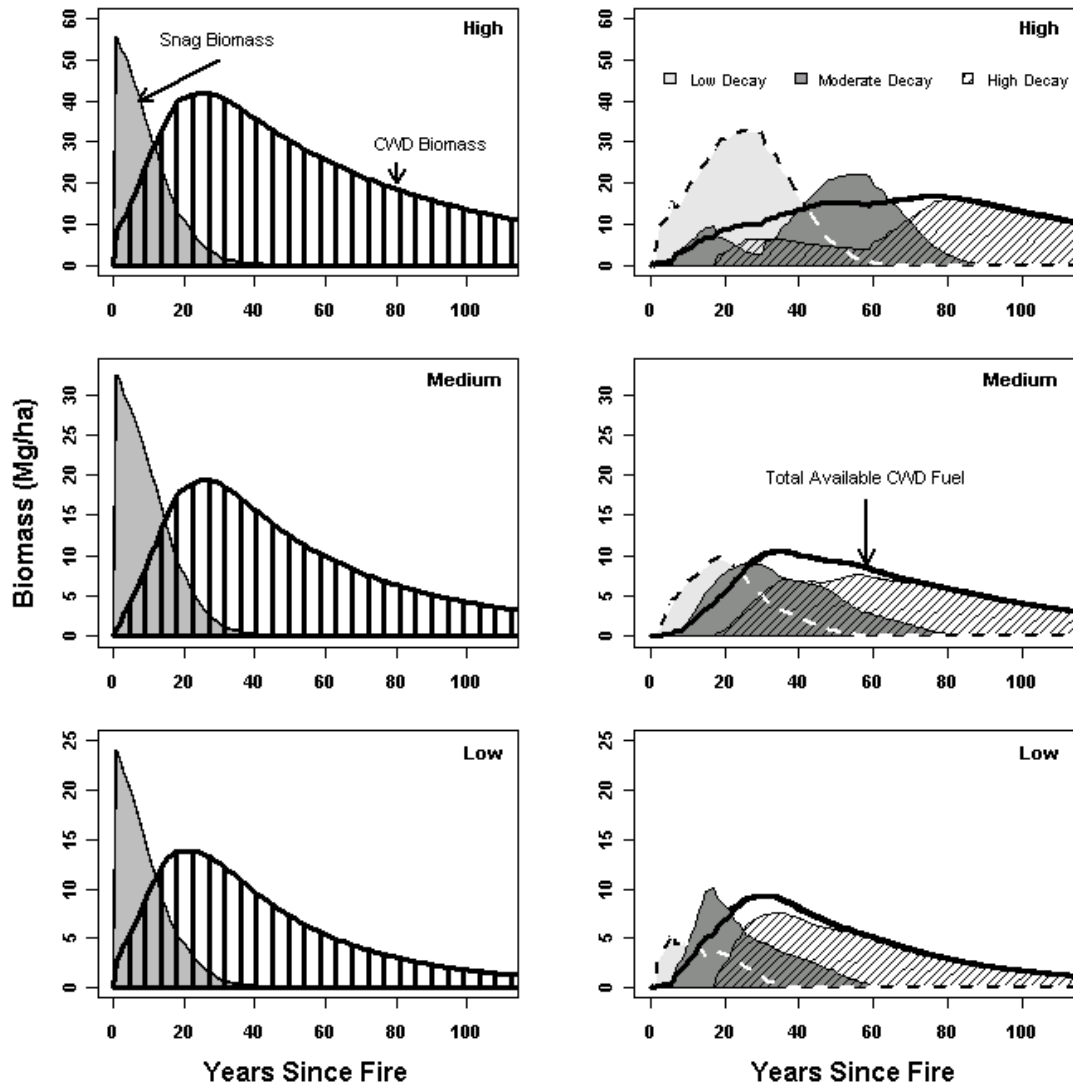


Figure 4.13: Post-fire necromass dynamics after salvage logging in high-severity stands. Graphs on the left (high, medium and low biomass) depict snag fragmentation and total log accumulation following high-severity fire. Shaded region is snag biomass and striped is log biomass. Graphs on the right show decay state and total available CWD fuel (solid black line) for the same stands adjacent on the left. Low decay (light gray) is all wood biomass with a density  $>0.30 \text{ g/cm}^3$ , moderate decay (dark gray) is all wood between  $0.20$  and  $0.30 \text{ g/cm}^3$ , and high decay (striped) is all wood  $< 0.20 \text{ g/cm}^3$ . Combustion amount for low is assumed to be  $0.08$ , moderate is  $0.50$  and high is  $0.95$ .

Salvage logging increases total FWD for 15-20 years post-fire, and then essentially converges with inputs from snag breakage and fall (Figure 4.12). We do not expect elevated hazard from this increase since other fuel bed layers have not developed sufficiently to sustain a flaming front. Studies have shown it requires about 10 years for fuels to develop significantly enough to sustain a flaming front even when an overstory canopy remains intact (Collins et al. 2008).

Coarse woody detritus has the potential to input large amounts of dead fuels as snags break and fall. The amount of biomass present in future years is dependent on initial stand biomass, snag dynamics and snag and log decay rates (Harmon 2009). Total log biomass peaks 23-26 years post-fire, but not all of this fuel is available for combustion. After accounting for the combustion efficiency of decayed CWD, we found that at its peak, available CWD is only 28-32% of initial biomass and reaches its peak 32-75 years post-fire, depending on species composition and abundance.

Decisions regarding salvage logging should account for the long-term trajectories of all fuel layers since reducing hazard in one fuel layer may exacerbate another (Donato et al. 2006). If fire hazard is a significant concern, the mitigation of this hazard may only be achieved by manipulating multiple fuel beds with repeated treatments. This hazard should be weighed against the ecological impacts imposed on the system from these operations, and evaluated in a landscape context (Beschta et al. 2004). Overall, salvage logging alone will not mitigate future fire hazard and thus should not be considered as an ecological justification for conducting these operations.

The potential for increased severity of future fires is a concern in post-fire environments. Currently, managers lack a comprehensive understanding of the hazard that fire killed snags may or may not present in the future. “NecroDynamics” can be used to estimate the amount of necromass available for combustion in the event of a future fire occurrence. Mass loss over time can only be estimated by incorporating all the dynamics present in this model. Estimates of the contribution of fire killed necromass to total available fuel will help managers make decisions on the appropriateness of post-fire management.

## CHAPTER 5: CONCLUSIONS

The availability of snags and logs to meet ecological needs is dependent on their creation, fall, fragmentation and decomposition (Harmon 2002). The ecological importance of snags and logs, and the potential hazard associated with future fire severity, require managers to make decisions today that have long-term implications (Brown et al. 2003). Snags and logs provide structural habitat valuable to multiple vertebrate and invertebrate species in dry-mixed conifer forest of Oregon (Thomas et al. 1979, Bull et al. 1997) and are important for saprophyte communities, ecosystem nutrient cycling, and the carbon balance of a forest (Franklin et al. 1987, Harmon et al. 1986, Harmon 2009, Triska and Cromack 1979). Reduce future fire severity has been proposed as an ecological justification for conducting salvage operations, without long-term or detailed investigations regarding the benefits and/or consequences of these operations. Decisions about the appropriateness of salvage logging in post-fire environments must consider the implications of snag removal on ecosystem functioning and weigh this against meeting management objectives (Beschta et al. 2004, Sessions et al. 2004).

### Snag Dynamics

One of the most evident effects of fire disturbance is a pulse of tree mortality, dramatically increasing the abundance of snags within an ecosystem for a brief period of time. Snags and logs provide structural habitat valuable to multiple vertebrate and invertebrate species in dry-mixed conifer forests of Oregon (Bull et al. 1997, Thomas et al. 1979). CWD is also important for saprophyte communities, ecosystem nutrient cycling, and the carbon balance of a forest (Harmon et al. 1986, Triska and Cromack 1979, Harmon 2009). The availability of snags and logs to meet ecological needs is dependent on their creation, fragmentation and decomposition (Harmon 2002).

Snag dynamics (fall and breakage rates) were estimated for *Abies sp.*, *Pinus ponderosa* and *Pinus contorta* in DBH classes <23 cm (small), 23-41 cm (medium)

and >41 cm (large). A total of 5,103 snags in thirty 0.25-ha plots were sampled at seven different fire sites, covering a 24 year chronosequence following high-severity fire disturbance. *Pinus ponderosa* and *Pinus contorta* snags had the quickest fall rates with estimated half-lives of 7-8 and 12-13 years for small and medium snags, respectively. Large *Pinus ponderosa* snags had an estimated half-life of 17-18 years. *Abies sp.* snags fall rates were slower, with half-life estimates of 8-9, 14-15 and 20-21 years for small, medium and large snags respectively. Breakage rates were variable but correlated with wood strength, crown and stem weight and crown position (exposure to wind). Combined, the physical transfer of snag necromass in this region follows a predictable trend that includes a lag period, followed by accelerated deposition, ultimately slowing after about 20 years.

Continued decomposition occurs in snags and logs during their existence. In dry forest environments, snags were hypothesized to decay at a reduced rate relative to logs. Decomposition loss rate-constants were obtained by removing three cross-sections from sixty fire-killed *Abies sp.* snags, sixty *Pinus ponderosa* snags, and forty *Pinus ponderosa* logs. *Abies sp.* snags exhibited significant decay with an estimated decomposition loss rate-constant of  $k = 0.0149 \text{ yr}^{-1}$ . *Pinus ponderosa* snags did not exhibit significant decay, but log did. Sapwood and heartwood decomposition loss rate-constants equaled  $k = 0.0362 \text{ yr}^{-1}$  and  $k = 0.0164 \text{ yr}^{-1}$  respectively. These values confirm hypothesized differences in decay rates among species and between snags and logs in dry forest environments. The availability of CWD to meet ecological needs is dependent on the species composition, DBH distribution and snag necromass fragmentation.

Snag dynamics result in a complex pattern of carbon emissions from post-fire coarse woody detritus. Maximum carbon emissions occur >20 years post-fire, thus carbon is conserved on the site longer than assumed when applying surface decay rates to all CWD immediately post-fire. If maximizing carbon storage with active management becomes the primary objective of a dry-mixed conifer forest, this study

provides a comprehensive basis for comparing the success of accomplishing this objective.

### **Fuel Succession**

Coarse woody detritus has the potential to input large amounts of dead fuels as snags break and fall. The amount of biomass present in future years is dependent on initial stand biomass, snag breakage and fall rates and surface and standing decay rates (Harmon 2009). Total log biomass peaks 23-26 years post-fire, but not all of this fuel is available for combustion. After accounting for the combustion efficiency of decayed surface CWD, we found that at its peak, available surface CWD is only 28-32% of initial biomass and reaches its peak 32-75 years post-fire, depending on species composition and abundance.

In dry-mixed conifer forests of Oregon's eastern Cascades, live fuels respond quickly to post-fire conditions and are dominated primarily by shrubs adapted to post-fire environments (Brown and Smith 2000). Herbaceous communities do not contribute significantly to fuel loadings, but shrub biomass increases linearly for the first several decades and have the potential to exacerbate fire behavior in the event of a subsequent fire event (Thompson et al. 2007). The risk posed by this fuel layer is only elevated during periods of high drought stress, transitioning this fuel layer from a heat sink to a heat source (Rothermel 1972). The main contributions of shrubs regarding post-fire fuels and fire hazard are increases in litter, duff and FWD biomass. After a few decades, the shrub stratum is beginning to show signs of competitive stress leading to increased mortality as overstory trees begin to occupy the site and out compete this vegetative layer (Oliver and Larsen 1996). The impact of this transition will be a pulse of FWD from shrub biomass. Salvage logging has little direct impact on shrub biomass unless additional management actions are used to control competing vegetation (Lopez Ortiz 2007).

Total fine woody detritus accumulation is controlled by snag breakage and fall during the first decade of succession. Although total FWD is well correlated to snag

crown deposition, 1-hr fuel biomass alone is not well correlated remaining in relatively low abundance until shrub mortality occurs. 10-hr fuels are directly related to snag crown breakage and fall, but also increase with shrub canopy mortality after 10-15 years of development. 100-hr fuels are controlled by snag breakage and fall with no inputs from *de novo* vegetation during the first couple of decades of succession.

Salvage logging increases total FWD for 15-20 years post-fire, and then essentially converges with inputs from snag breakage and fall (Figure 4.12). We do not expect elevated hazard from this increase since other fuel bed layers have not developed sufficiently to sustain a flaming front. Studies have shown it requires about 10 years for fuels to develop sufficiently to sustain a flaming front even when an overstory canopy remains intact (Collins et al. 2008).

Decisions regarding salvage logging should account for the long-term trajectories of all fuel layers since reducing hazard in one fuel layer may exacerbate another (Donato et al. 2006). If fire hazard is a significant concern, the mitigation of this hazard may only be achieved by manipulating multiple fuel beds with repeated treatments over time. This hazard should be weighed against the ecological impacts imposed on the system from these operations, and evaluated in a landscape context (Beschta et al. 2004). Overall, salvage logging alone will not mitigate future fire hazard and thus is not justified ecologically from a fuels perspective.

## **Future Considerations and Opportunities**

Fire disturbance is a natural ecosystem process integral to the development of dry forest environments found throughout the western United States (Agee 1993). Fire extent and severity are anticipated to increase in these environments due to altered stand conditions and climate fluctuations (Brown et al. 2004, Weaver 1943, Westerling et al. 2006). Understanding the long-term impacts of these fires is critical for understanding the effects of decisions today on long-term ecosystem properties and land management objectives. Expansion of this research into other dry forest



environments would provide necessary information for making more informed and appropriate management decisions.

In addition to expanding this research to other dry forest types and geographic locations, the prevalence of mixed-severity fire regimes in many dry forests warrants investigation of snag dynamics and fuel succession to moderate and low severity fire sites. Incorporating sites with moderate and low severities will allow for a more comprehensive evaluation of modeled snag and log availability, at landscape scales, following contemporary fire disturbance.

Currently, this research has focused on average rates of fragmentation and decomposition of woody detritus. “NecroDynamics” provides an adequate modeling framework for exploring uncertainty in our estimates of the dynamics of fire killed woody detritus. An important part of future research regarding CWD dynamics is quantifying the effects of this variability by conducting an uncertainty analysis on the parameters incorporated into this modeling framework. For example, considerable variability is present in snag fall and breakage rates as well as wood density, decomposition loss rate-constants, and initial stand conditions. Using boot strapping techniques with Monte Carlo simulations can provide quantitative assessments of model uncertainties on snag and log persistence, resultant carbon emissions and fuel contributions of this material.

Quantifying fuel succession across multiple severities and at a landscape scale is crucial information for making informed fire management decisions. If fuel accumulations occur at varying rates, reintroducing fire into recently (<20 years) burned sites through prescribed fire may accomplish ecological and management goals with reduced risk of fire escape. Fuels succession can also be extended to natural ignitions occurring within a recently burned landscape. Predictions of fire behavior and effects can inform fire ecology and management personnel on expected impacts of the fire and improve their decision making capacity.

“NecroDynamics” provides opportunities to explore fuel succession to other severities, although it does require expansion of the fuel accumulation portion to live

vegetation. “NecroDynamics” modeling capacity can be used to test additional hypotheses regarding application of prescribed fire in recently burned sites until empirical data can be obtained. Exploration of landscape patterns of fuel succession through modeling exercises can provide adequate levels of information to assist in management agencies decisions regarding the allocation of resources towards ecosystem restoration goals. Reestablishing and maintaining fire regimes in dry forests is a critical component of ecosystem restoration and continued function.

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