Discussion Paper (DP8): Eastside Climate Change, Forest Health, Fire and Carbon Accounting

Elaine Oneil, Bruce Lippke, Larry Mason

Table of Contents

List of Figures

Introduction

Since 2000 Eastern Washington has been experiencing warmer summer temperatures than at any time in the last 100 years. Precipitation shortfalls have also been severe, culminating in near record shortfalls by 2005. These extreme weather conditions are consistent with predicted climate change impacts for the Columbia basin as a whole (Mote 2004). Research into the impact of climate change on forested ecosystems suggest that extreme disturbance events such as fires (Gedalof et al. 2005, McKenzie et al. 2004) and insect outbreaks (Carroll et al 2003, Logan et al. 2003) will be primary drivers of ecosystem change in the inland forests of the Pacific Northwest (PNW) Region. The loss of mature forest cover to insects and fire has implications far beyond the forest because of the linkages between climate change, carbon storage potential, sustainable ecosystems, living forests, renewable energy sources, and the forest product stream.

This paper integrates the issues emerging from current climate trends in Eastern Washington. It considers forest health and ecosystem sustainability, identifies current climate change impacts, and explores mitigation and adaptation strategies that go beyond the boundaries of forest landscapes to integrate clean energy initiatives into an integrated carbon framework. Removing high fuel loads in the forest effectively transfers carbon storage to long-lived product pools or biofuels that displace fossil fuels and energy intensive building materials while at the same time breaking the climate feedback of accelerated emissions. The combined effect of reducing wildfire risks and reducing reliance on fossil fuel intensive products and services provides the simultaneous benefits of reducing the carbon footprint and creating forest conditions that are more likely to be sustainable under changing climatic conditions.

Changing Weather and Climate Change Forecasts

We can gauge how Eastern Washington forests are responding to this climate trend by examining recent trends in fire, and insect activity. The most alarming trend is a substantial spike in pine mortality from mountain pine beetle (*Dendroctonus ponderosae*) (MPB). There were 20 times more pine trees killed from 2000-2004, than in the prior 20 years combined. Mortality trends from MPB were shown to be highly correlated with increasing summer temperatures and record low moisture levels (Oneil 2006). The record fires of 2006 were at least partially attributable to the presence of these recently killed pine stands, but extremely hot, dry conditions were necessary to carry the fires across approximately 400,000 acres. The increase in insect outbreaks and the extent of fires we have seen in the past five years suggests that climate change is already dramatically affecting our forested ecosystems.

While forests can decrease atmospheric CO_2 by storing Carbon in trees and wood products, the established link between elevated atmospheric concentrations of CO_2 (IPCC 2001) and temperature increases provide feedback inducing ever more emissions from the forest. Forests can increase or decrease atmospheric carbon dioxide (CO_2) through release (mortality and decay) or storage of carbon (growth). Pacala et al. (2001) estimated that 20-40% of all terrestrial carbon sequestration in the United States occurred in western forests. There is a feedback between forests, carbon uptake and climate change. Warmer climatic conditions are implicated in the increase in the magnitude and intensity of wildfires (Gedalof et al. 2005, McKenzie et al. 2004). Increases in wildfire frequency and intensity that release stored forest carbon could result in western forests becoming a source of carbon rather than a sink (Westerling et al. 2006). This positive feedback mechanism between warming climatic conditions and wildfire is also exacerbated by the presence of extensive mortality from insect and disease outbreaks. Lynch et al. (2006) found that the extent and severity of the Yellowstone fires were largely correlated with prior mountain pine beetle (MPB) (*Dendroctonus ponderosae*) outbreaks. Anecdotal evidence from Washington State's 2006 Tripod Complex fire in the Okanogan National Forest concurs with the Yellowstone evidence. Areas within the fire perimeter that burned most intensely had high density forests with extensive mortality from mountain pine beetle and Douglas-fir bark beetle (*Dendroctonus psuedotsugae*) attacks as well as substantial defoliation from spruce budworm *Choristoneura occidentalis* (Ketcham 2007).

Research elsewhere has found that MPB epidemics are highly correlated with winter warming trends which allow the insect to expand its range into previously unsuitable environments (Carroll et al 2003, Logan et al. 2003). Research on MPB infestations in Eastern Washington found that the MPB epidemic over the past five years is largely a consequence of increasing summer temperatures and record low moisture levels in the region (Oneil 2006). The MPB outbreaks in Washington State are in areas where MPB had previously existed in dynamic equilibrium with its hosts, though more high elevation forests are now affected. Whether the MPB has historically been excluded from a particular elevation or latitudinal band or part of the historical disturbance ecology, climate change is accelerating the impacts of this insect. While disturbances are a natural part of Washington's eastside forest ecology, the magnitude and severity of recent disturbance events are unusual. These disturbances are resulting in the loss of mature forest cover across the entire range of elevations where pine is present and in every forest type from high elevation National Forests to small private land owners near the forest/steppe boundary.

The loss of mature forest cover to insects and fire has implications far beyond the forest because of the linkages between climate change, carbon storage potential, sustainable ecosystems, living forests, renewable energy sources, and the forest product stream.

To mitigate the magnitude of the feedback mechanism between climate change and forest disturbance events, we can use targeted management activities to build resilience into the forests. Developing an approach that builds resilience into our forests requires that we explore current and predicted climate trends, and then link them to forest health conditions, tree physiological response, and forest stand dynamics to inform adaptive management approaches based on the future rather than the past.

Temperature and precipitation data for Eastern Washington were obtained from the Western Regional Climate Center (www.wrcc.org) to generate five-year running averages for the last 110 years in Figure DP8.1 (Oneil 2006). Since 1989, summer temperatures have been generally higher than any other time in the century and, since 1999, the warming trajectory has been beyond the historic range entirely. Summer temperature has not been below the 110-year average since 1984. The five-year running average of pre-growing season precipitation (September through June) has been below the 110-year average for 14 of the past 18 years. These temperature increases and precipitation shortfalls in Eastern Washington are consistent with predicted climate change impacts in the Columbia basin (Mote 2004).

The five-year running average of summer month temperatures (Figure DP8.1) is correlated to the annual acreage of MPB outbreaks in lodgepole pine (*Pinus contorta*) and ponderosa pine (*Pinus ponderosa*) (Figure DP8.2) (Oneil, 2006). The MPB outbreak data (PNWFHP 2006) were collated from aerial surveys flown from 1954-2005. The large increases in acreage attacked by MPB that appear in the mid-1980's and since 2000, positively correlate with five-year average temperature trends and negatively correlate with precipitation trends for the same periods.

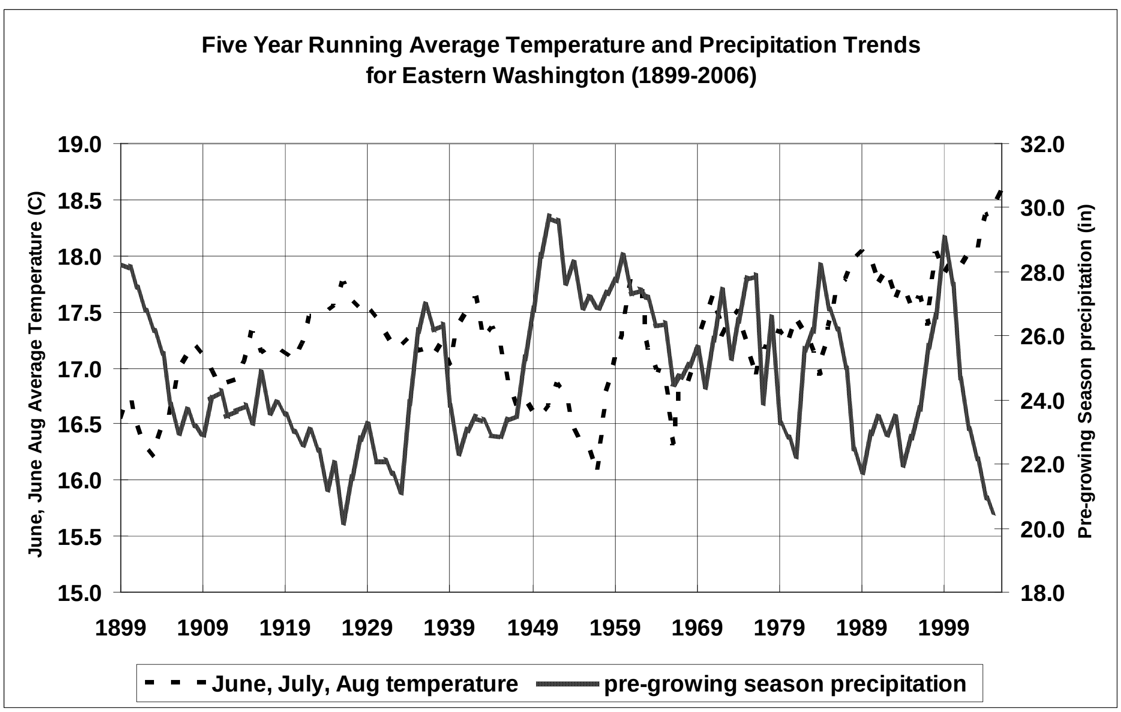

Figure DP8.1: Temperature and precipitation trends for Eastern Washington

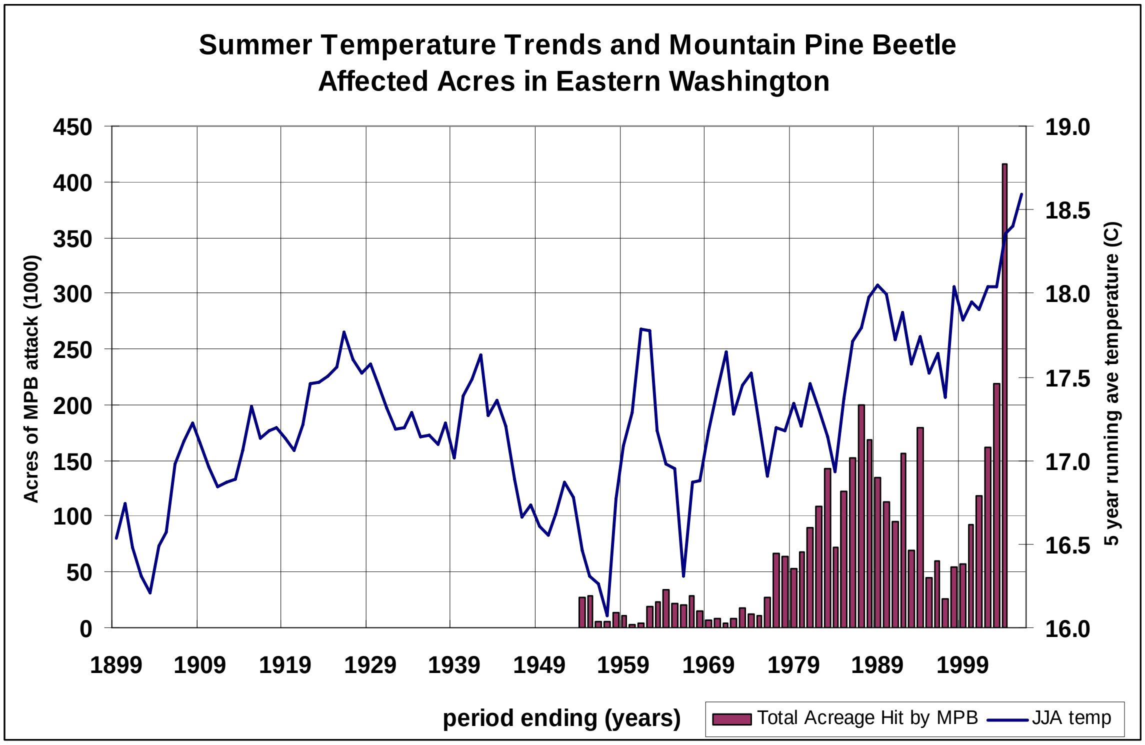

Figure DP8.2: Temperature trends and MPB activity in Eastern Washington on PP and LP forests.

The same aerial survey data (PNWFHP 2006) were used in Figure DP8.3 to depict a time series of mortality and mortality/acre from MPB infestations in ponderosa pine (PP) and lodgepole pine (LP) across all of Eastern Washington for 1979 to 2004 (Oneil, 2006). The average mortality rate for the 1980-2000 reporting period was relatively stable at approximately 2.2 trees per <u>infected</u> acre. Coincident with the rise in substantial increase in average summer temperature from 2000 onward, for the 2001-2005 reporting period the average MPB induced mortality rate increased to 8.4 trees per acre. In 2004, MPB affected over 415,000 acres in Eastern Washington, resulting in mortality to over 4 million pine trees, (DNR FH 2006) or about 20 times the average for the previous 20 years. Lodgepole pine mortality from MPB attacks since 2000 is more than 70% of total mortality for the entire 25-year period from 1979-2004 (Oneil, 2006b). Elevated mortality rates for pine species have the greatest impact on the 30% of Eastern Washington forest types that are dominated by pine. However, the percentage of non-Federal acres that would be potentially impacted is much higher as a pine component is found on almost 87% (3.3 million acres) of productive timber producing forests in Eastern Washington.

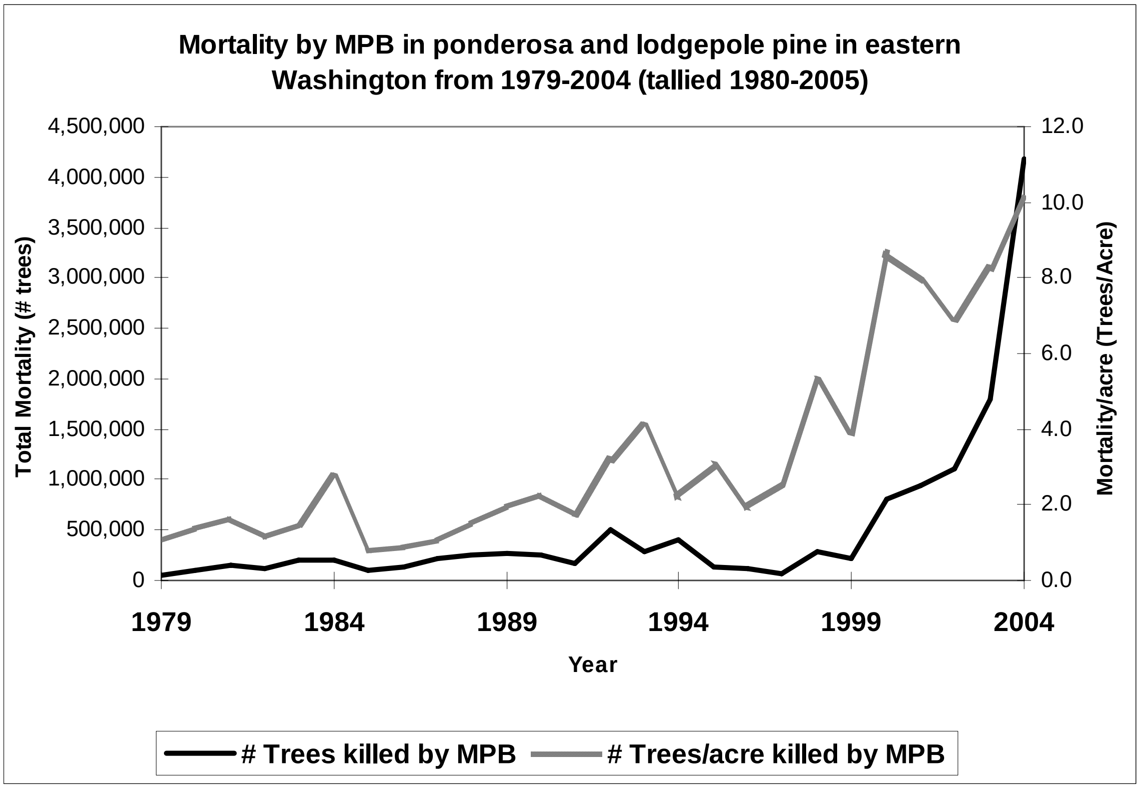

Figure DP8.3: Tree mortality from MPB in Eastern Washington 1979-2004.

The increases in MPB attacks are most strongly correlated with changes in average summer vapor pressure deficit (VPD) (Oneil 2006) reflecting a regional climate change impact on one of our more vulnerable tree species. Pine's role as the "canary in the coal mine" for eastside forests arises for several reasons. In comparison to other tree species, pines are host to the most aggressive native insect predator of eastside forests (USFS 2007). Pines are more sensitive to shifts in atmospheric dryness as measured by VPD than other tree species (Delucia et al 2000) and VPD increases exponentially relative to temperature increases (Waring and Running 1998), therefore even small shifts in temperature can generate substantial stress. Pines tolerate fewer years of stress than more shade tolerant species before succumbing to mortality (Keane et al, 1996). And finally, it is likely that the average VPD for the growing season months of June, July and August has reached a threshold value at which most tree species begin to exponentially decrease their stomatal conductance and shut down respiration to maintain water status (Waring and Schlesinger 1985).

In Figures DP8.4 and DP8.5, MPB attack data is segregated by species (PP and LPP) and location (National Forest and non-federal forests) to illustrate the substantial differences by species (about 3 TPA for PP and 11 TPA for LPP in 2005 for example) and by location (about 11 TPA affected in both species in 2005 on National Forests as compared to 8 TPA on non-federal forests).

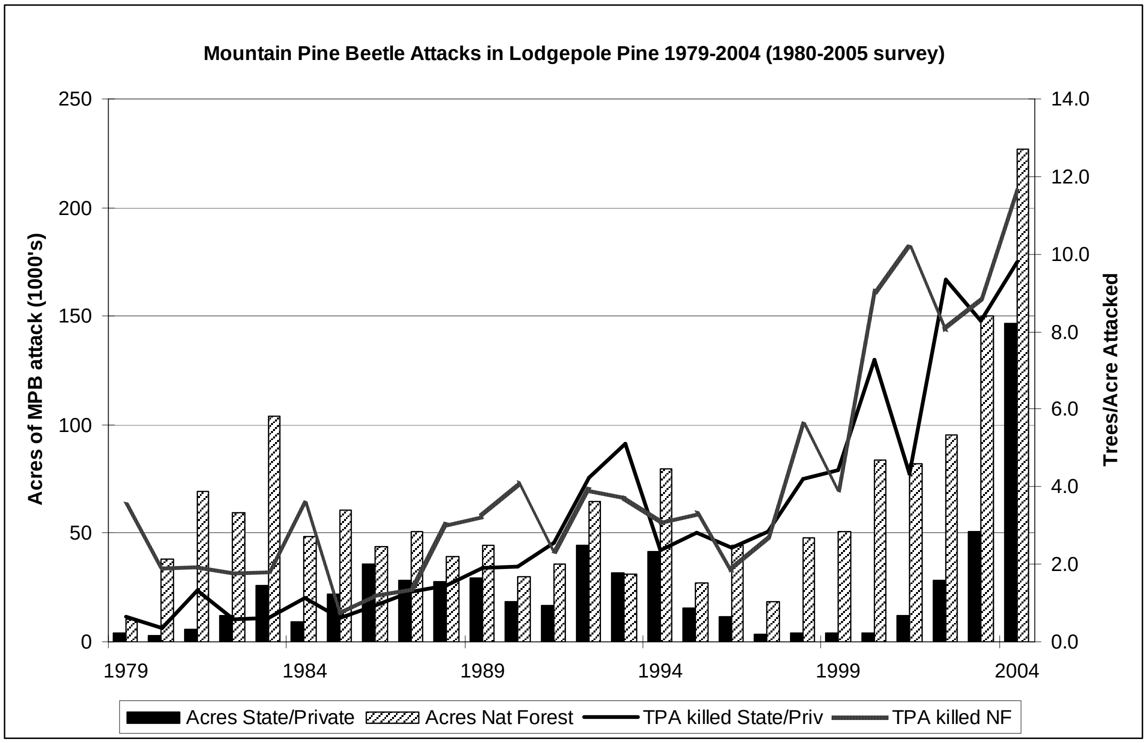

Figure DP8.4: Mountain pine beetle mortality in lodgepole pine for State, Private, and Tribal forest ownerships (SP) and National Forests (NF) 1979-2004

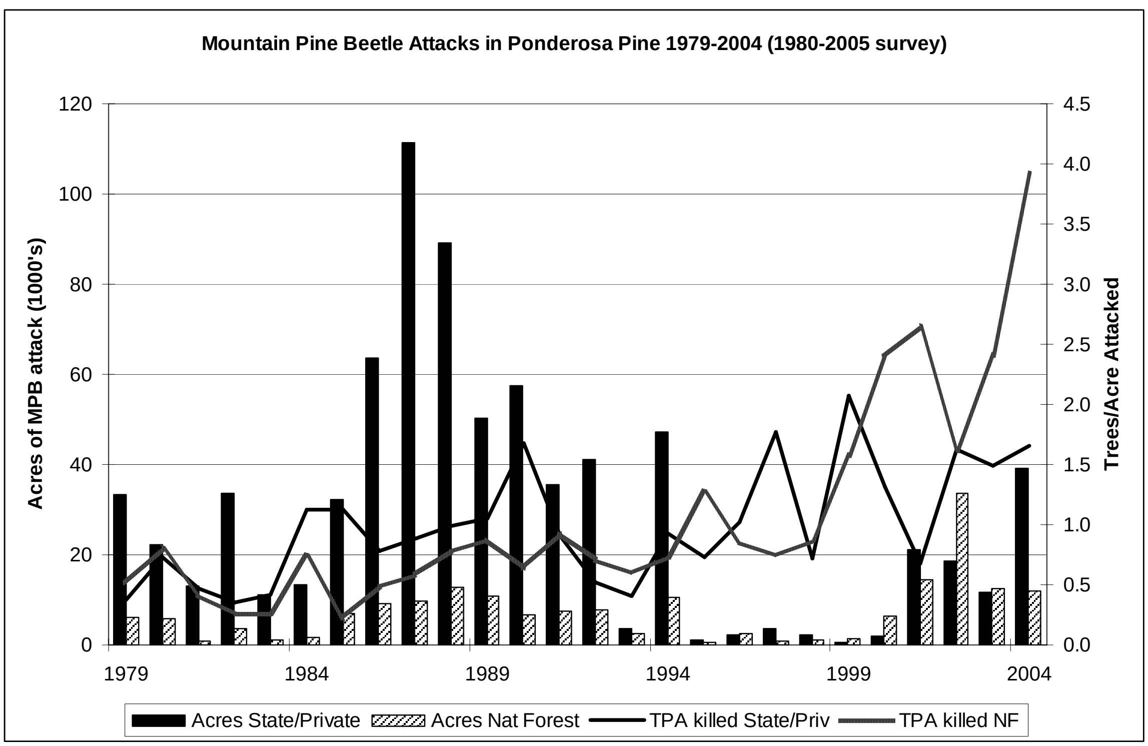

Figure DP8.5: Mountain pine beetle mortality in ponderosa pine for State, Private, and Tribal forest ownerships (SP) and National Forests (NF) 1979-2004

Figure DP8.4 illustrates similar trends in MPB activity in lodgepole pine on both National Forest and non-federal forests which include State, Private and Tribal lands. State lands refer to forests managed by the Washington Department of Natural Resources (DNR) for the benefit of trust beneficiaries. In contrast, Figure DP8.5 indicates a poor correlation between MPB activity on National Forests and other ownerships. The differences may be attributable to differences in management activities, age and size classes of the PP resource, or some other variable that has yet to be determined.

Just as baseline mortality equations in growth models vary by habitat type and species, so too does the expected mortality rate from insect attack. There has been extensive research documenting insect risk as being dependent upon stand density and stress (Sartwell 1971, Sartwell and Stevens 1975, Schenk et al. 1980, Schmid and Mata 1992, Shore and Safranyik 1992, and others). Hazard or risk rating systems developed from this research fail in predictive capacity when expanded outside the region where they were derived (Shore et al 1989 and 2000) and individual predictor variables commonly used in risk rating systems varied widely in their predictive power across the region (Amman and Anhold 1989). In addition the non-linear trends in recent insect outbreak conditions suggest that estimating likelihood of attack and predicting long-term outcomes based on recent insect trend data requires a high resolution model that incorporates differences between habitat types and stand characteristics as well as incorporating stresses associated with extreme weather and predicted climate change. The approach taken for this analysis uses historical weather, stand density, site carrying capacity and insect outbreak variables as predictors for future trends in insect attack. This approach allows for the testing of impacts of various density thresholds in order to predict outcomes and provide management options that reduce insect outbreaks and risk across a wide range of disparate stand conditions.

Figure DP8.6 displays a change in the character of MPB attacks – that is, prior to 2000 when there was an increase in MPB mortality it was concentrated in a few large patches so average polygon size was high when outbreaks occurred (as in 1984, 1993, 1994) but in the more recent years, though mortality is up, the average polygon size is low or even stable. This trend suggests a more widespread ecological footprint which will influence long-term trends in forest stand dynamics. A confounding factor is that improved technology has become available starting in 2003 which improves the accuracy at capturing small polygons which tends to reduce the overall average polygon size. However, information from the data survey crew (Moore, 2006) concurs with this assumption of a broader ecological footprint for this insect in particular.

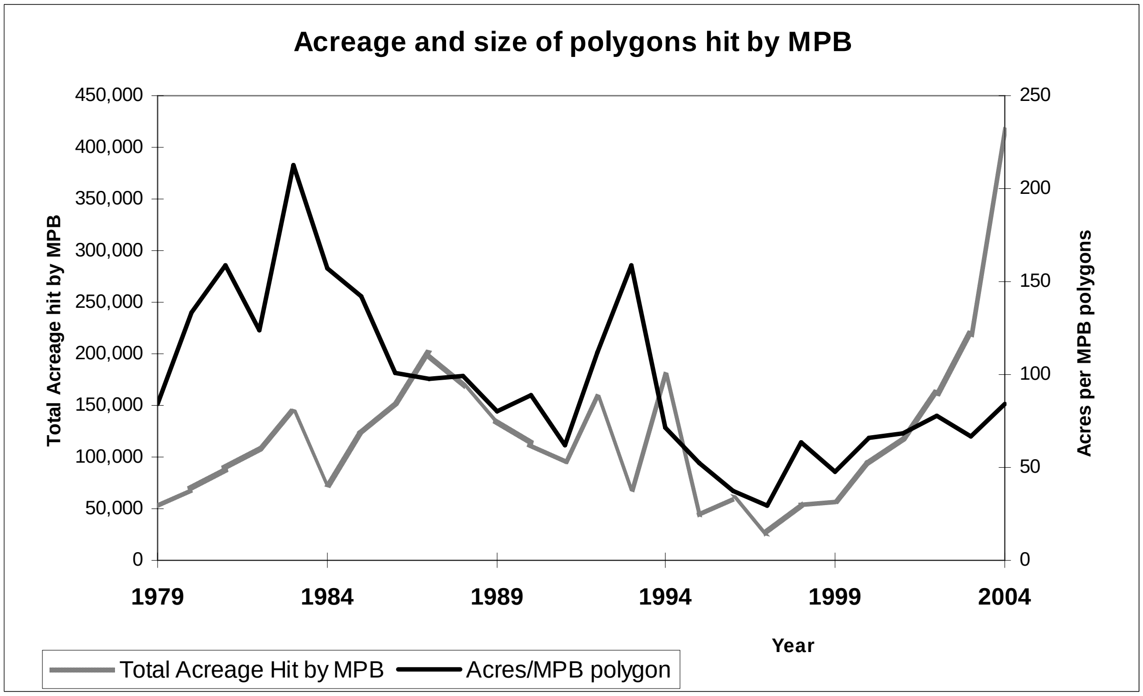

Figure DP8.6: Acreage and size of MPB outbreaks from aerial surveys of Eastern Washington 1979-2004

While the mortality increases can be built into projections and treatments can be designed to reduce the consequences of insect attacks, there remains a large uncertainty as to whether insect losses will continue to accelerate or perhaps respond to natural events such as population collapse once suitable host species have been eliminated. An alarming scenario can be drawn by comparison to the problems being experienced in British Columbia where MPB outbreaks have destroyed millions of acres of mature pine (Eng et al. 2006) and are now attacking non-host species and young, vigorous trees that would never have been attacked under prior conditions.

The climate trend data would indicate that other tree species and forest types are also at risk as they are experiencing the same gradient of environmental stress as that of the pines. While mortality impacts are not currently as substantial for Douglas-fir as they are for the pines, there are significant outbreaks of spruce budworm and Douglas-fir bark beetle. While more research is required to elucidate likely impacts, elevated mortality trends on pine species and Douglas-fir combined would affect 80% of our forest types in Eastern Washington.

Addressing Insect Outbreaks

Controlling insect outbreaks involves reducing forest density, removing at risk species or stand cohorts, or harvesting the forest while the insects are present to remove pest populations. All of these measures have provided control under historical climate conditions. Under the current climatic conditions and presumably future conditions, the equilibrium state between the host trees and their insect predators is no longer clearly defined. Old treatment regimes may not afford effective control over insect populations, thus leading to larger epidemics and higher mortality overall.

Much like the situation with fire, once insect and disease populations reach epidemic proportions, control options become limited or non-existent. Such is the case in British Columbia where the MPB epidemic has impacted 9.2 million hectares in 2006 for a cumulative impact of 14 million hectares (34.5 million acres) (Carroll, 2006). Impacts from the MPB epidemic in central British Columbia have defied all control efforts and provided many surprises to managers. Whereas MPB typically hit stressed trees over 80 years of age, it is now killing 30-year old plantations of widely spaced, vigorous trees (Grainger, 2006). Whereas stands in close proximity to an existing outbreak were historically considered at high risk for infestation, now stands 2-400 km (125-250 miles) distant are succumbing to attack (Carroll, 2006). And finally, though MPB was known to prefer large diameter trees and especially target them during an epidemic, now small diameter trees are succumbing to attack (Grainger, 2006). In British Columbia's epidemic, management agencies are no longer aiming for control, but for mop-up and mitigation of cascading environmental, economic, and social impacts (Shultz, 2006). Avoiding such high cost, high impact outcomes such as those presented by the British Columbia example requires a concerted effort on several fronts.

Prior research suggests that thinning Washington's eastside forest types that have a pine component to control density, reduce tree stress, and remove insects is a useful approach for controlling insect outbreaks. However, recent anecdotal evidence from eastside forest managers provides numerous examples where recently thinned stands have been overcome by MPB in the past five years of hot dry conditions. Oneil (2006) found that since 2000, climate conditions were the only significant predictors of MPB attack, regardless of stand age, density, diameter, basal area, or percent of susceptible species in the stand. This current evidence suggests that thinning to historical target densities may not be sufficient to control insect epidemics. Historical thinning regimes were based on research from areas as far away as South Dakota and California. Those thinning regimes have been updated to reflect site specific conditions in southeastern Washington State (Powell, 1999). However, Powell's recommendations are confined to southeastern Washington State and were based on carrying capacity defined by habitat types under historic climatic conditions. Not only are climate conditions changing, but the habitat types are also changing in response to climate forcing. In this state of flux, prior thinning and stocking recommendations can serve as a baseline to work forward from, but likely will not be sufficient to build the necessary resilience to climate change into eastside forests.

In order to effectively respond to insect epidemics under a changing climate regime, it will likely be necessary to manage our forests for greatly reduced density and basal area relative to current conditions. What those densities should be will depend on how climate influences future temperature and moisture conditions. In order to be successful at producing forests capable of sustaining and overcoming insect attacks, research is needed to elucidate the precursors of outbreaks at the tree and stand level, rather than focusing on insect dynamics. By understanding how the dynamic equilibrium between the tree host and its insects is altered during altered climate conditions we will be able to define the most applicable and effective treatment options.

Predictions of Increasing Fire Risk

Increases in the incidence, magnitude, and intensity of forest fires in inland forests have been attributed to a combination of changes in forest composition and structure from fire suppression, grazing, and past harvest practices (Sampson and Adams 1994, Pyne 1997, Arno 2000) and shifts in summer weather conditions that make more and hotter fires more likely (Gedalof et al. 2005, Westerling et al. 2006). Overly dense forests become drought-stressed and susceptible to insect mortality (Halloin 2003, WSU 2007). Westerling et al. (2006) found that spring snow melt is occurring earlier as a result of warming trends and that this adds to summer moisture shortfalls which increase wildfire hazard. Their analysis found that the number of large wildfires in the West was four times greater since 1986 than in the 16 years prior and that these fires affected 6.5 times more area across the entire region. The correlation between spring snow melt and warmer summer temperature has been shown to be statistically robust, suggesting that continued increases in spring temperatures will be accompanied by dry summers and extreme wildfire hazard conditions.

In an analysis that examines specific sub-regions of the west, McKenzie et al. (2004) concluded that summer temperature was the major driver in producing extreme wildfire events in Washington. This research further concluded that even in the 'best case' scenario predicted by climate models, the extent of wildfire in inland west forests will increase 2 to 3 times over the next century. Since 1995, approximately 1.1 million acres of forests have burned in Eastern Washington. Approximately 1/3 of that acreage burned in 2006 in the most severe fire season since 1994, producing the largest fires since the 1903 Yacolt Burn (Christiansen 2007). The amount of dead and dry timber greatly exacerbated the effect of the extremely hot summer conditions, rendering the fires largely uncontrollable by conventional response (Christiansen 2007).

Historically, fire likely impacted approximately 82% of the forests (Camp et al. 1996) over any given 100-year period in the East Cascades region and would have impacted virtually all of the forests in the southeast corner of the state (Olson, 2000). Similar fire impacts are likely for northeastern Washington, but detailed fire histories have not been compiled for that region. Since 1995, approximately 1.7%/year of the area in National Forests and 0.4%/year of the area under DNR fire protection have burned in Eastern Washington (DNR fire statistics). At this 12-year rate, an area equivalent to that of the entire Eastern Washington National Forest system would burn within 58 years or 2.1 times more area than the historical extent found by Camp et al. (1996). McKenzie et al. (2004) and Gedalof et al. (2005) suggest that the extent and severity of wildfires are predicted to increase in size by 2-3 times even under the most optimistic climate change scenarios. To understand what a doubling of extent and severity might entail in terms of costs, ecological impact, and community and social disruption, we can start by examining the trends in these variables since 1995. Because wildfire activity is correlated with forest condition and our forests are under increasing stress from climate change and insect and disease outbreaks, the impacts we can estimate from the 1995-2006 data may be substantially lower than the impacts we may experience in the future.

Addressing Fire Risk

Historical studies and demonstrations have shown that treatments to remove surplus fuel loads and reduce forest density are successful at minimizing fire impacts. Reducing forest densities has the additional benefit of increasing the resiliency of live trees in the face of climate change. Improving the resilience of live trees can help to reduce the positive feedback that occurs when insect mortality increases the amount of dead biomass available when a fire is ignited.

Thinning forests to reduce their vulnerability to fire requires that the cut biomass be removed to effectively reduce fuel loads (Raymond and Peterson, 2005). The removal costs can be substantial if the thinned material is too small for the existing product stream. In the short term, one way to justify the costs of treatments that can mitigate fire impacts is to use the avoided costs approach identified by Mason et al. (2003) and recommended by the Washington Department of Natural Resources Forest Health Strategy Work Group convened by the Governor in 2004. The avoided cost approach (Discussion Paper 10-E) compares the costs that would be saved by thinning forests now in order to reduce the risks and impacts of future fire events against the cost of thinning. The costs that can be at least partially be avoided include fire fighting, fatalities, facility losses, timber losses, carbon emissions, regeneration, and rehabilitation costs. And there are also value benefits including the community value placed on fire risk reduction, increased water yield through stand density reduction, carbon stored in products while displacing fossil fuel intensive substitutes and regional economic benefits. Considering the high cost of fighting fires, $167 million in 2006, assessing the benefit of thinning in comparison to fire fighting costs is the first place to start. If even a fraction of the areas thinned became defensible space in which to control fire spread, there may be a substantial monetary benefits from applying treatments as insurance against the expected larger losses from fighting fires on overstocked stands.

In the longer term, approaches and policies that promote forest biomass utilization such as conversions to liquid fuels or electrical energy may create sufficient market demand to make removal of small diameter wood more economically feasible. Whether wood removal investments are justified by recognition of avoided costs or by positive market sales, biomass use as a renewal energy source and the reduction in fires could substantially improve the carbon footprint of Eastern Washington forests.

Carbon Sequestration in Forests, Forest Products, and Substitution Pools

Forests increase or decrease atmospheric CO_2 through growth and mortality (Sampson and Hair 1992). One ton of CO_2 contains 0.27 tons of carbon. One ton of carbon is equivalent to 3.7 tons of CO_2. There are approximately 0.47 tons of carbon per cubic meter of wood and combustion of one cubic meter of wood results in release to the atmosphere of 1.7 tons of CO_2 (Innes and Peterson 2004). Dead and dying forests are subject to higher risks of fire which accelerates the release of carbon to the atmosphere (Lynch et al. 2006) and warmer summer temperatures are linked to increased wildfire activity (Gedalof et al. 2005, McKenzie et al. 2004).

Pacala et al. (2001) found that 20-40% of all terrestrial carbon sequestration in the United States occurred in western forests. Increases in wildfire frequency and intensity that release stored forest carbon could reverse that relationship such that forests would become a carbon source rather than a sink (Westerling et al. 2006). For example, in 2006, wildfires in Eastern Washington consumed close to 400,000 acres of forestland generating smoke plumes that released volumes of CO_2 to the atmosphere equivalent to the annual emissions of 1 million Sport Utility Vehicles (SUV's) (Mason 2006). Much of the area burned in the 2006 fires contained dense forests with elevated mortality levels caused by MPB attacks. Since 1995, approximately 1.1 million acres of state and federal forestlands have burned in Eastern Washington. Average annual volumes of CO_2 released by forest fires in Eastern Washington have equaled the emissions equivalent of 297,000 additional SUV's every year since 1995. Forest management strategies that remove surplus fuel loads in overly-dense Eastern Washington forests can help to reduce annual CO_2 emissions with the additional benefit of restoring forest health and resiliency and avoiding catastrophic forest fires.

Life-Cycle Inventory/Life-Cycle Analysis

Forest ecosystems are known to absorb large quantities of carbon which is stored in solid wood, vegetation, litter, and soils thereby reducing atmospheric CO_2. Young healthy forests take up carbon at high rates, while the net carbon uptake in older forests ultimately slows with age followed by release from mortality, decay, and/or wildfire. In addition to the forests, the end-use of timber harvested from forests is a factor in evaluating the net consequences of forestry to the global carbon cycle (Johnson et al, 2005). Forest products that are durable goods, such as building materials or furniture, store embodied carbon for the life of the product. Short-term products, such as paper and cardboard, once used and allowed to decay or burn, release stored carbon to the atmosphere. Carbon embodied in milling residuals and discarded wood products may be sequestered in landfills for long periods of time. When forest products are used in place of non-wood products such as steel and concrete, that are much more energy-intensive in their manufacture, releases of atmospheric carbon are avoided (Lippke et al., 2005) When forest biomass is used to generate energy as a substitute for fossil fuels, releases of fossil carbon are avoided (Birdsey 1992).

The Consortium for Research on Renewable Industrial Materials (CORRIM), developed techniques for determining the life cycle inventory/life cycle analysis (LCI/LCA) of wood and wood products as reported in Wood and Fiber Science Special Issue #37. In that issue, Lippke et al. (2005) use LCI/LCA techniques to trace the carbon in forests, through the product stream, to eventual release back into the atmosphere. The research identified and accounted for four major carbon pools: 1) carbon in the forest; 2) carbon in products that leave the forest; 3) carbon associated with the use of forest biomass and product residuals as an energy source; and 4) the carbon offsets from the substitution that occurs when wood building materials displace products like steel or concrete. Lippke et al. (2004, 2005) found that forests that are periodically harvested, planted, and re-grown to produce a continuing series of short- and long-lived products and energy feedstocks, sequester and offset more cumulative carbon than forests that are left unharvested. This finding is illustrated by the graphs below that depict comparative examples of carbon accounting associated with an even-aged managed forest (Figure DP8.7) and an unmanaged forest (Figure DP8.8) in western Washington. Figure DP8.7 characterizes the time dynamic nature of carbon storage as quantified in metric tons per hectare for a 45-year commercial rotation as a cumulative sequence of carbon storage and release in the forest, in products, and the impact of product substitution for non-wood alternatives. Figure DP8.8 shows the accumulation over time of carbon for the same beginning forest inventory, but with no treatments, no disturbances, and no products and hence no substitution for fossil fuels or energy intensive product alternatives.

While the carbon in the forest in Figure DP8.7 is shown to cycle with each rotation around a steady state trend line, the carbon in product pools, net of energy used in harvesting, processing and construction, gradually increases over time. When the avoided carbon emissions from the displacement of fossil fuels and fossil fuel intensive building products are included, there is a substantial increase in total stored and offset carbon that can be seen to surpass the cumulative carbon storage in forest biomass when there is no harvest activity as displayed in Figure DP8.8. While carbon stored in the forest reaches a steady state, the use of wood in construction displaces fossil fuel intensive products, thereby storing carbon while also reducing carbon emissions. Increasing the acreage under forest provides a one-time increase in forest carbon. If the forests are harvested and reforested, additional carbon storage is provided by the periodic production of long-lived products, and by displacement of fossil fuels for energy and substitution of energy intensive building materials with carbon neutral wood products.

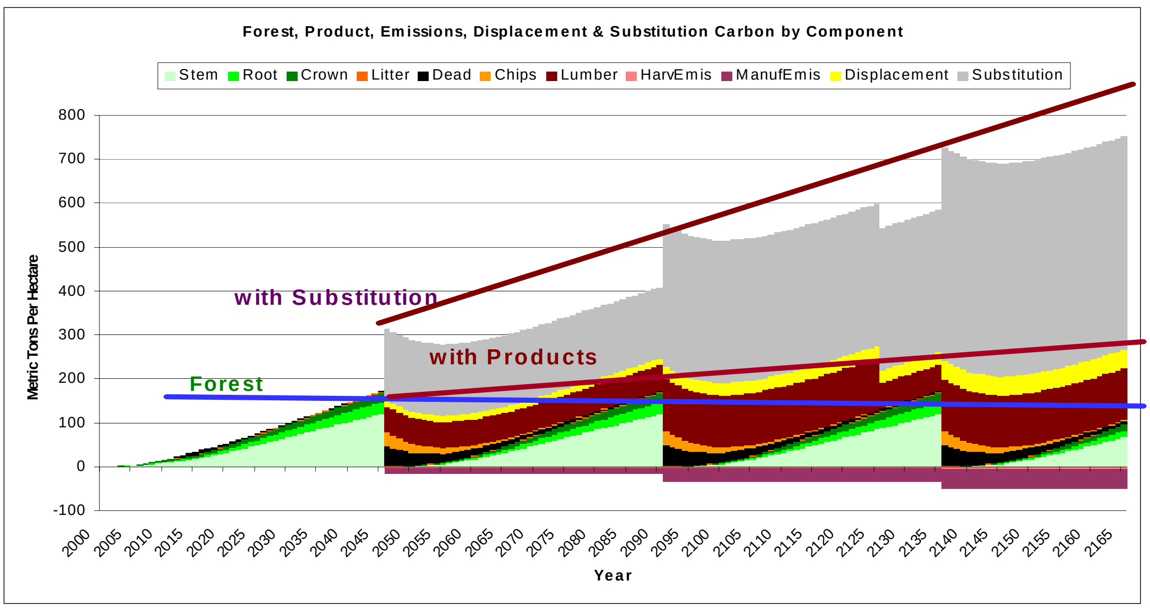

Figure DP8.7: Carbon pools for a single acre of commercial forest under a 45-year rotation.

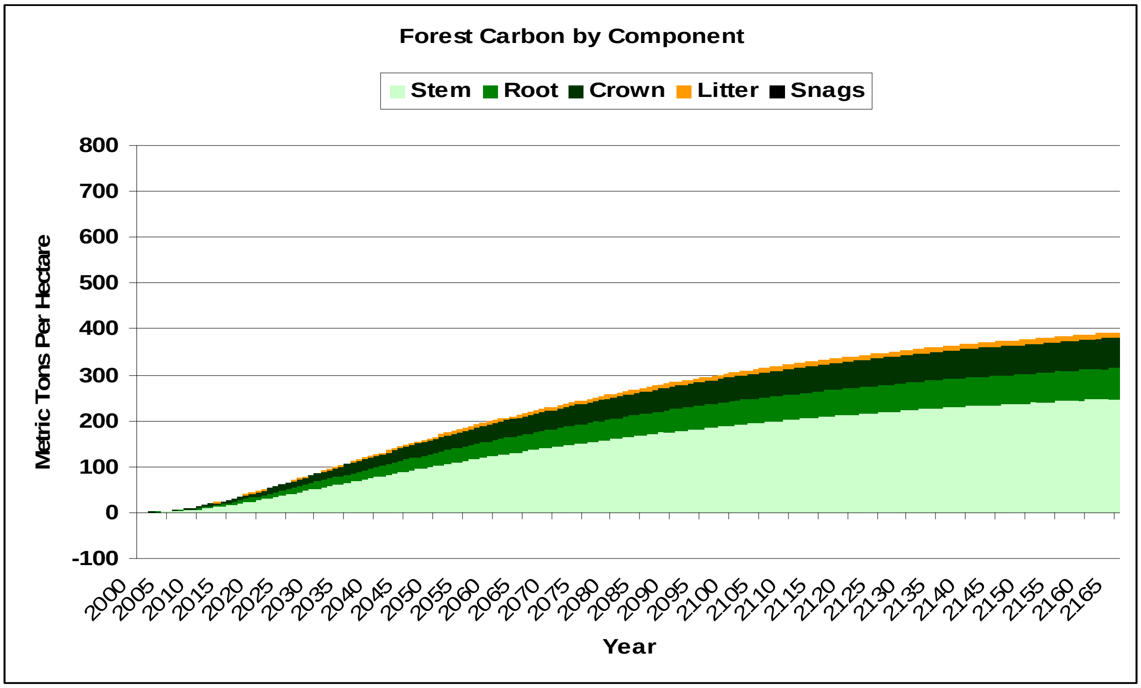

Figure DP8.8: Carbon pools for a single acre of forest with no harvest and no disturbance.

Carbon Impacts of Eastside Management Alternatives

As part of the timber supply study we developed a range of management alternatives that reflect current and probable activities across all commercially available forests of Eastern Washington. We used the outcomes from the management study to estimate carbon impacts by region and owner group. That carbon study is based on the LCI/LCA techniques developed by CORRIM. We report on the results here with the cautionary note that management intensities and treatment regimes applied on Eastside forests are in flux because of changing markets, infrastructure, and ownership pattern. Adding climate change with its impacts on fire and insect outbreaks creates a substantial level of uncertainty as to the simulated outcomes.

To accommodate variability in Eastern Washington forests, summaries of cumulative carbon pools through time have been developed for the entire landscape rather than as a single-stand example as is given in Figure DP8.7. Figure DP8.9 displays simulated estimates of the weighted-average amount of carbon produced per year per acre in the forest, product, displacement, and substitution pools for all private forests in Eastern Washington. These simulations assume that these forests will continue to produce timber volumes approximately equal to the volumes removed from 1980-2002 as a base case scenario. The simulation also assumes that the forest products and co-products include lumber, chips, and hogfuel that are used for products in similar ways as they are used today. The product streams do not account for anticipated increases in biomass removal associated with the current focus on using forest residuals as bioenergy and biofuel. How clean energy policy initiatives alter the relative mix of energy, co-product and product outputs from industrial production will alter the relative importance of the product, displacement, and substitution carbon pools depicted in these graphs, but will not alter the relative importance of these pools relative to the forest pool.

Figure DP8.9 indicates that after 100 years the average carbon per acre stored in the forest is only one third of the total carbon benefit accrued on private Eastern Washington forests under the base case scenario. The remaining carbon benefit accrues in product, displacement and substitution pools that are off the forest and thus largely beyond the risk of burning in a wildfire.

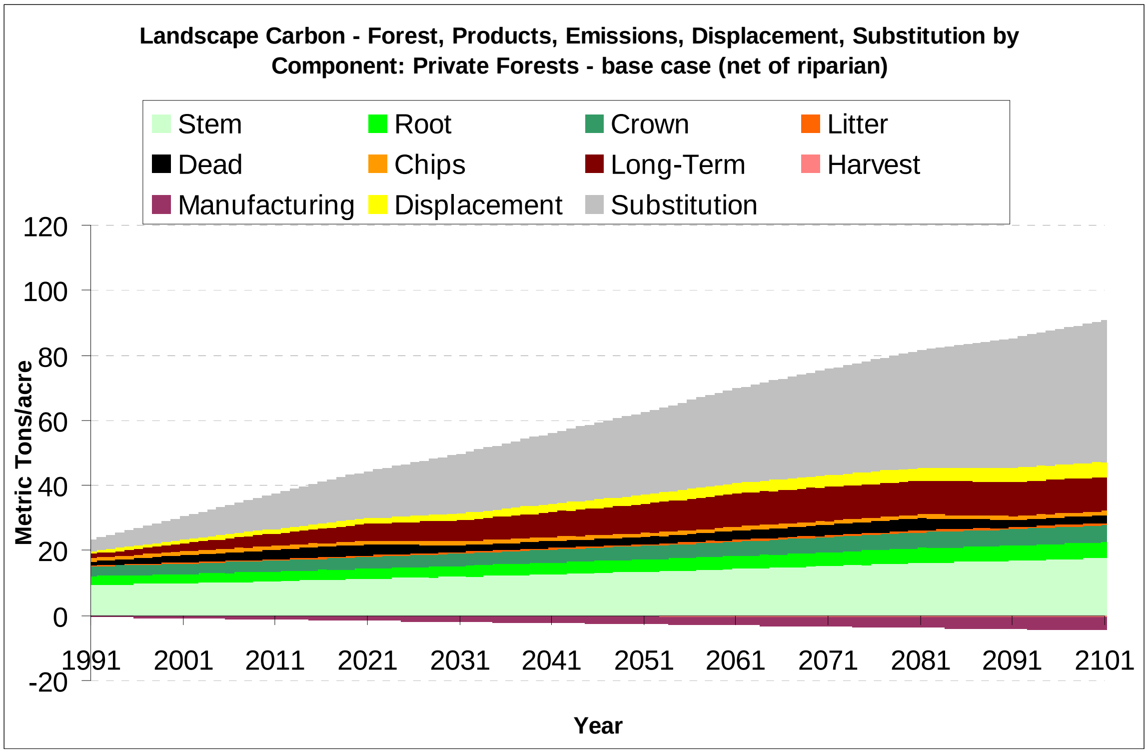

Figure DP8.9: Metric Tons per acre in carbon pools for commercially available acres on private forests in Eastern Washington

Carbon storage potential on federal forests produces a different set of outcomes that highlight the issues around fire and insect risks. If we assume no harvest, no fire and no insect and disease impacts on National Forests in Eastern Washington, the carbon sequestration potential of these forests is approximated by Figure DP8.10. Figure DP8.11 represents the approximate impact of current harvest rates on National Forests in Eastern Washington by simulating thin-from below treatments on approximately 12,000 acres per year of the Wildland Urban Interface (WUI). Figure DP8.11 illustrates how removal of small diameter material generates fewer carbon benefits because relatively few long-lived products or substitution benefits accrue under such management scenarios. This does not suggest that thinning from below is a poor choice for federal forests because the thinning from below is anticipated to reduce fire risk substantially (Discussion Papers 10 and 11).

McKenzie et al. (2004) indicates that we can expect at least a doubling of fire frequency and extent in Eastern Washington. Linking this research to work done by Camp on the historic levels of fire refugia (i.e. the area that didn't burn under historical fire conditions) suggests that under the most optimistic climate change scenarios approximately 1.7% of the acres of National Forest in Eastern Washington would burn in each decade. A 1.7% burn rate per decade would generate the forest carbon footprint given in Figure DP8.12, assuming that all acres are mature timber and only burn once. Figure DP8.12 provides what is likely an upper bound estimate of the carbon release potential if these forests burn at rates predicted by recent climate change research. If the forests burn at rates higher than anticipated under climate change scenarios, then there would be more emissions from these forests. In this rough approximation, regeneration is not estimated as regeneration delays and failure rates would need to be more accurately determined. The 'Dead' component is the residual burned wood that decays, and thus releases carbon, at a rate of approximately 0.5 tons/carbon/acre/year. The 'Emissions' component is the carbon equivalent from emissions released from the burned forest based on 6 tons/acre emitted for every acre burned (Mason et al. 2003).

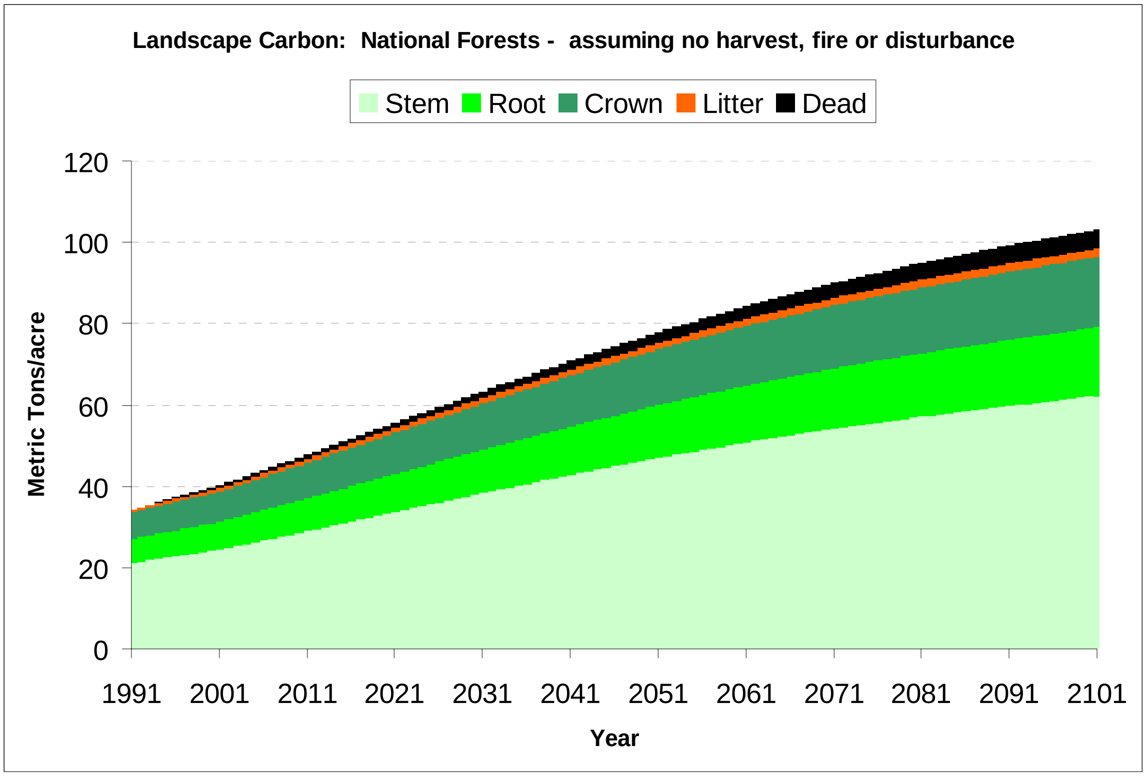

Figure DP8.10: Tons per acre carbon pools for National Forests in Eastern Washington assuming no management

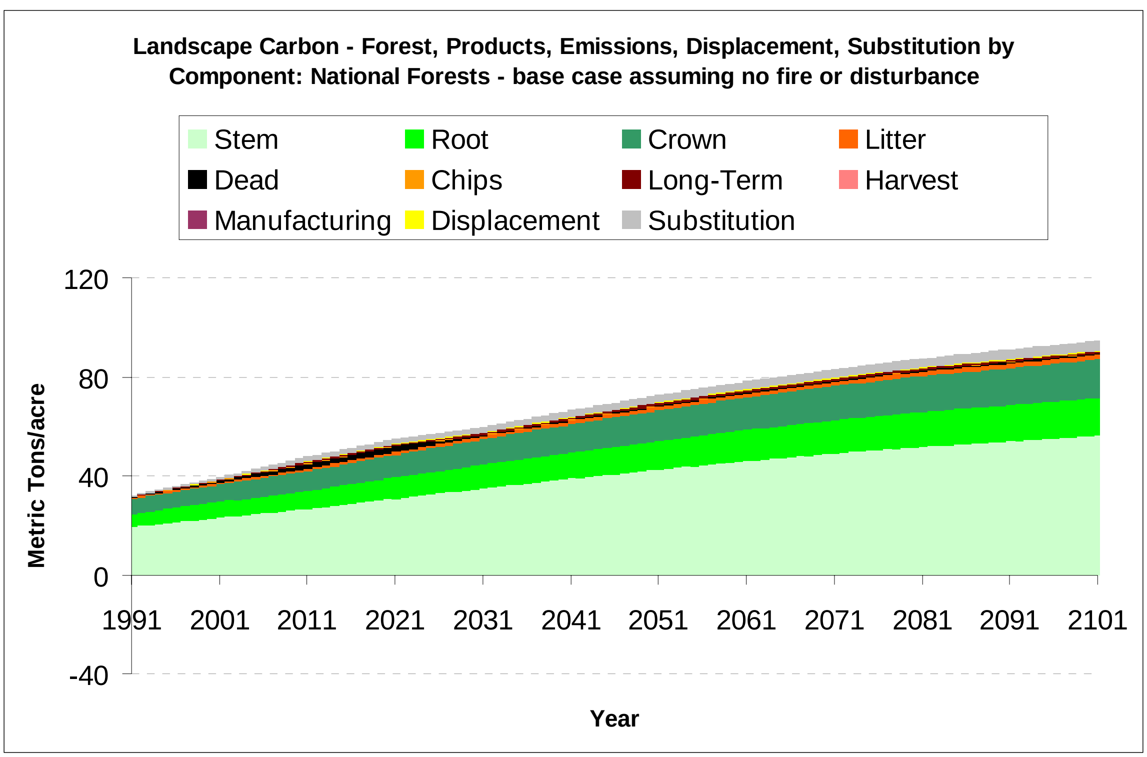

Figure DP8.11: Tons per acre carbon pools for National Forests in Eastern Washington under current management intensities

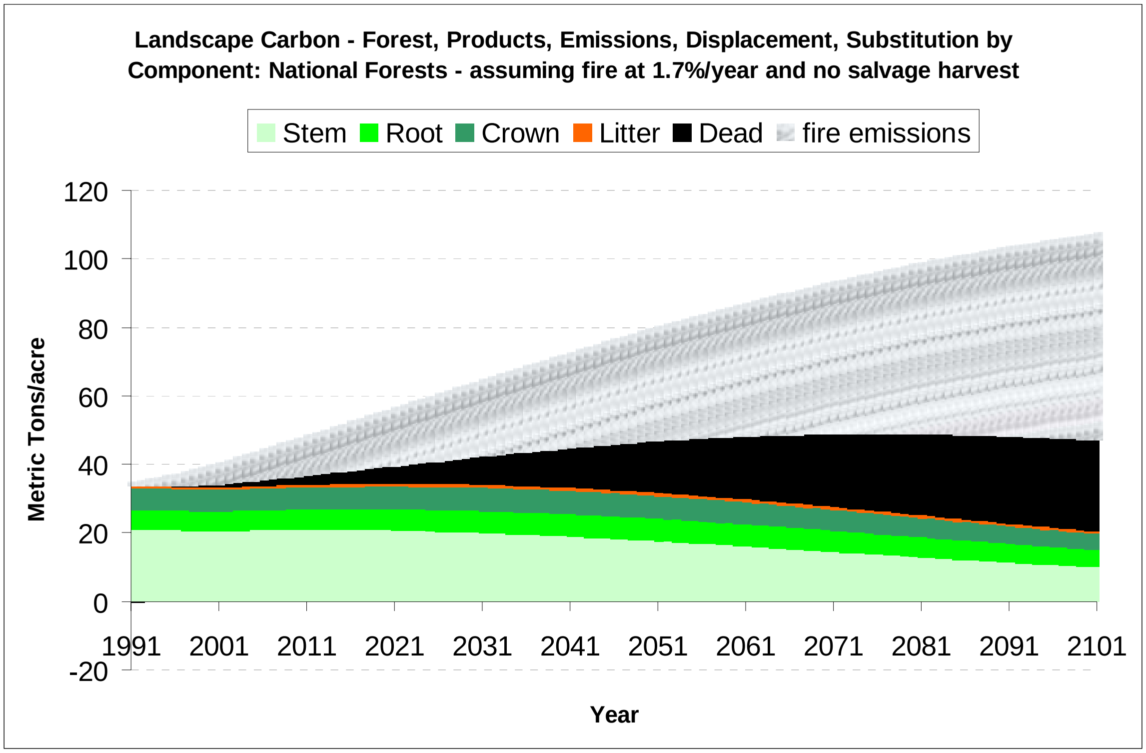

Figure DP8.12: Tons per acre carbon pools for National Forests in Eastern Washington assuming a 1.7% burn rate based on climate change estimates

While this is a coarse approximation of potential carbon impacts from fire under expected climate change scenarios, it does highlight how unmanaged forests are likely to become a source of carbon emissions rather than a sink. This analysis also illustrates the hazard associated with slow adoption of restoration activities designed to reduce fire risk on these forests.

An alternative treatment scenario that applied restoration thinnings to all dry and mesic forest types within the National Forests produces carbon impacts reflected in Figure DP8.13. Treatments on National Forests (Figures DP8.11 and DP8.13) produce fewer long-lived products than those on private lands because of the retention of large diameter trees on treated areas and the large number of acres left untreated. A comparison of Figures DP8.9 and DP8.11 (assuming no carbon releases from wildfire) indicates that, while by the end of the period, the carbon benefits/acre are approximately the same, on average 19 tons/acre more carbon is stored in all carbon pools (forest, substitution, displacement and product) on private acres than on National Forest acres. Furthermore, the risk of carbon release from wildfire on National Forest acres is substantially larger as they carry an average of 63 metric tons/acre versus 24 metric tons/acre in the forest pool on private forests. When more acres of National Forests are treated (Figure DP8.13), the overall average carbon stored is the same as under current treatment levels, but there are 11 tons/acre removed from the forest which reduces risk exposure to wildfire and carbon emissions.

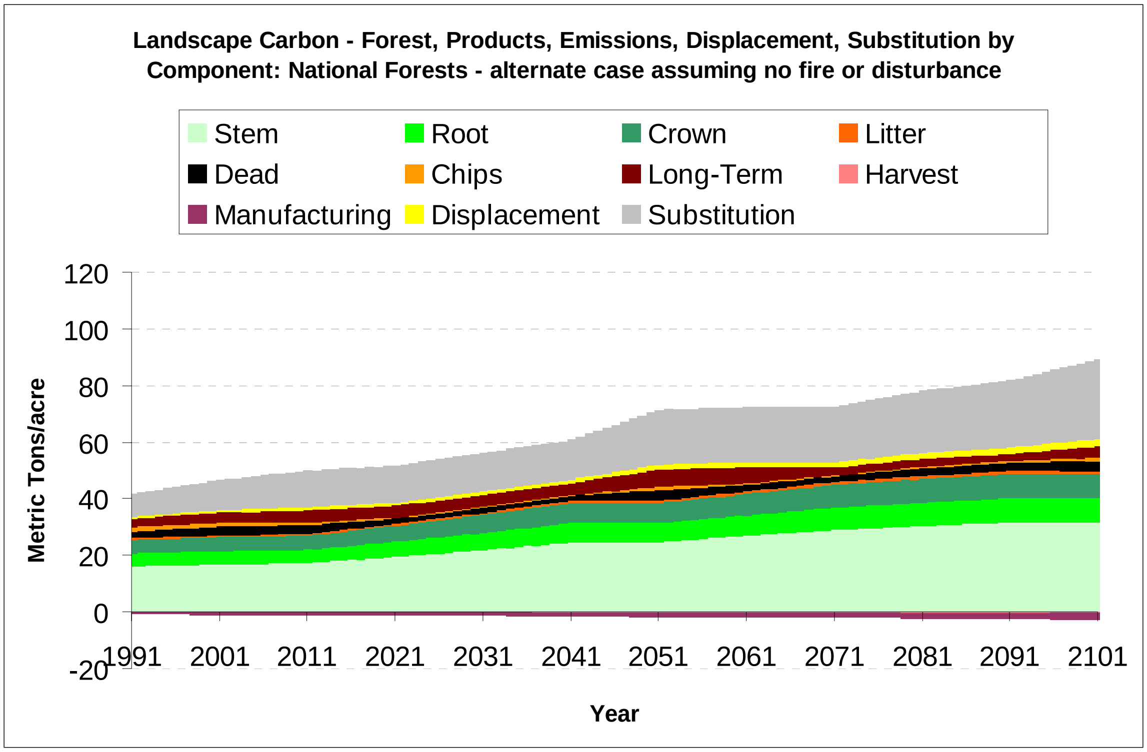

Figure DP8.13: Tons per acre carbon pools for National Forests in Eastern Washington with thinning treatments on dry and mesic forests

While Eastside forests produce less biomass carbon per acre than Westside forests, effective management of Eastside forests for products and for fire risk reduction has multiple carbon benefits. Management can reduce the amount, intensity, and duration of wildfire (Mason et al. 2003) and related carbon release (Lippke et al. 2006) as well as increase the storage of carbon in the product, displacement and substitution pools. Active management could also produce more acres with low density mature trees that may be better suited for adaptation to shifting climate conditions. Gains could include greater overall carbon sequestration accompanied by healthier forest conditions where fire, insects, and disease hazards have been reduced. Without such management actions, wildfire seasons such as those that occurred in Eastern Washington in 2006 result in CO_2 emissions that negate carbon emission reductions achieved elsewhere.

These scenarios are based on management simulations developed to obtain merchantable forest products for the private sector and reduce fire risk on public lands. However, the full extent of future wildfire risk reduction for either kind of treatment scenario is uncertain as currently available forest fire models are based on forest, fuel, and fire behavior conditions that existed prior to changing climate trends.

Summary

In 1997, the Kyoto Protocol to the United Nations Framework Convention on Climate Change (UNFCCC 1997) identified forests and forestry specifically as important to the global carbon cycle. However, climate change mitigation protocols established by the Kyoto Protocol assume that carbon storage associated with forestry is limited to afforestation, reforestation, and deforestation. This narrow view of forest carbon mitigation is not without controversy. The carbon benefit of product substitution associated with displacement of fossil fuel energy and energy-intensive non-wood building materials has been well-documented (Perlack et al. 2005, Perez-Garcia et al. 2004, Boman and Turnbull 1997, Schlamadinger and Marland 1996, Buchanan and Honey 1995). Naburrs et al. (2000) examined the importance of broadening the Kyoto Protocol and found that more than 50 percent of potential additions to forest carbon storage in the United States could accrue from pest and fire management. Lippke et al. (2006) demonstrated that, primarily as a result of reduced forest fire emissions and increased long-lived product production, 56 percent more carbon was stored over a 50-year period in a managed rather than an unmanaged Eastern Washington forest.

Fire-prone forests of Eastern Washington can be managed to reduce atmospheric carbon in four basic ways:

- Absorption of atmospheric carbon through photosynthesis to storage in vegetation.
- Extension of carbon storage in long-lived products.
- Reduction of fossil fuel emissions through substitution for energy and building products.
- Reduction of carbon releases associated with forest mortality, decay, and wildfire.

In 2001, the Intergovernmental Panel on Climate Change (IPCC) issued a report that acknowledged forest growth, products, substitution, and disturbance avoidance as integral components of managed forest ecosystems and the global carbon cycle. Recognition of carbon boundary conditions that include all forest flows represents a choice for comprehensive versus selective environmental accounting with implications for improvement in forest health and climate change adaptation effectiveness.

Factors contributing to climate change are likely to persist through the near future. Forest management responses must consider climate impacts in order to reduce undesirable consequences of insect attacks, increased mortality and uncharacteristic fires. Monitoring of changing forest health trends combined with research on the broader implication of treatment alternatives, given that the climate is already outside the range of past experience, will be essential for development of adaptive response. A new understanding of changing forest management options and societal priorities will be needed to align strategies for effective forest health and climate change mitigation.

References

Arno, S.F. 2000. Fire in Western Forest Ecosystems. In: Brown, J.K., Smith, J.K., eds. Wildland Fire in Ecosystems: Effects of Fire on Flora. RMRS-GTR-42: vol. 2. USDA For. Serv. Rocky Mtn. Res. Sta.: 97-120.

Birdsey, R.A. 1992. Carbon Storage and Accumulation in United States Forest Ecosystems. GTR-WO59. USDA For. Serv. NE For. Exp. Sta.

Boman, U.R., and J.H. Turnbull. 1997. *Integrated biomass energy systems and emissions of carbon dioxide.* Biomass and Bioenergy 13:333-343.

Bowyer, J., D. Briggs, B. Lippke, J. Perez-Garcia, J. Wilson. 2004. Life Cycle Environmental Performance of Renewable Materials in Context of Residential Building Construction: Phase I Research Report. Consortium for Research on Renewable Industrial Materials CORRIM Inc. Seattle WA. 60pp +15 chapter modules of approximately 600pp.

Buchanan, A.H. and B.G. Honey. 1995. *Energy and carbon dioxide implications of building construction.* Energy and Buildings 20: 205-217.

Christiansen, Vicki, 2007, DNR Testimony to Congressional Subcommittee on Forests and Forest Health - Wildfires and Their Aftermath: Protecting Communities, Watersheds and Wildlife, presented on October 4th, 2006 by the Executive Director of Regulatory Programs, Washington State Department of Natural Resources.

DeLucia, E. H., H. Maherali, et al. 2000. *Climate-driven changes in biomass allocation in pines*. Global Change Biology 6(5): 587-593.

DNR FH, 2006, Washington State Department of Natural Resources Forest Health Section, http://www.dnr.wa.gov/htdocs/rp/forhealth/

Frear, C., B. Zhao, G. Fu, M. Richardson, S. Chen, and M. Fuchs. 2005. Biomass Inventory and Bioenergy Assessment; An Evaluation of Organic Material Resources for Bioenergy Production in Washington

State. Publication No. 05-07-047. Washington Department of Ecology and Washington State University. 120pp. http://www.ecy.wa.gov/pubs/0507047.pdf

Gedalof, Z., D. L. Peterson, et al. 2005. *Atmospheric, climatic, and ecological controls on extreme wildfire years in the northwestern United States*. Ecological Applications 15(1): 154-174.

Halloin, L. 2003. Major Bark Beetles of the Intermountain West. WA. Dept of Nat'l Res.: Olympia. 18pp. http://www.dnr.wa.gov/htdocs/rp/forhealth/wadnrbarkbeetle.pdf

Halloin, L. 2003. Major Defoliating Insects of the Intermountain West. WA. Dept of Nat'l Res.: Olympia. 13pp. http://www.dnr.wa.gov/htdocs/rp/forhealth/wadnrdefoliators.pdf

Innes, J.L. and D.L. Peterson. 2004. Managing Forests in a Greenhouse World – Context and Challenges. p.1-9. In: Climate Change, Carbon, and Forestry in Northwestern North America: Proceedings of a Workshop November 14-15, 2001 Orcas Island, Washington, USDA For. Serv. PNW Res. Sta. GTR-614. 120pp.

International Panel on Climate Change (IPCC). 2001. Climate change 2001: the scientific basis. Contribution of Working Group I to the third assessment report of the Intergovernmental panel on Climate. Cambridge, UK: Cambridge University Press. 785pp. plus appendices.

Johnson, L.R., B.R. Lippke and E. Oneil, in prep, Forest Resources of the Inland West, CORRIM: Phase II Interim Report, Seattle WA.

Keane, R.E., P. Morgan and S.W. Running. 1996. FIRE-BGC, Res Pap INT-RP-484, USDA For Serv, RMRS, Ogden, UT, 122pp.

Ketcham, Tom, 2007, personal communication, Tom is a Forester with the Methow Valley District, Okanogan-Wenatchee National Forest and was part of the response team for the 2006 Tripod Fire.

Lippke, B. R., J. Comnick and C.L. Mason. 2006. Alternative Landscape Fuel Removal Scenarios: Impacts of Treatment Thinning Intensity and Implementation Schedules on Fire Hazard Reduction Effectiveness, Carbon Storage, and Economics. RTI/CORRIM Joint Working Paper No 6, Seattle WA, 36 pp.

Lippke, B., J. Wilson. J. Bowyer, J. Perez-Garcia, J. Bowyer, J. Meil. 2004. *CORRIM: Life Cycle Environmental Performance of Renewable Building Materials.* Forest Products Journal. 54(6):17.

Lynch, H. J., R. A. Renkin, et al. 2006. "The influence of previous mountain pine beetle (Dendroctonus ponderosae) activity on the 1988 yellowstone fires." Ecosystems 9(8): 1318-1327.

Mason, C.L. 2006. Forest Fuels Reductions and Biomass-to-Energy; What is the Connection? Presentation to the Idaho Wildland Fire Conference. Sept. 26-27, 2006. Boise, ID.

Mason, C.L., H.K. Rogers, B.L. Lippke, K. Ceder, T. Bloxton, J. Comnick, J. McCarter, K. Zobrist, 2003. *"Investigation of Alternative Strategies for Design, Layout, and Administration of Fuel Removal Projects", section on Market and Non-Market Values*, College of Forest Resources, Rural Technology Initiative, University of Washington, July 2003: www.ruraltech.org)

McKenzie, D., Z. Gedalof, D. Peterson, and P. Mote. 2004. Climate Change, Wildfire, and Conservation. Conservation Biology.18:4. pages 890-902.

Mote, P.W. 2004. How and Why is Northwest Climate Changing? Pp 11-22 In Climate Change, Carbon, and Forestry in Northwestern North America: Proceedings of a Workshop November 14-15, 2001 Orcas Island, Washington, USDA For. Serv. PNW Res. Sta. GTR -614. 120 pp.

Mote, P.W., E.A. Parson, A.F. Hamlet. 2003. Preparing for climatic change: the water, salmon, and forests of the Pacific Northwest. Climate Change. 61:45-88.

Nabuurs, G.J., A.V. Dolman, E. Verkaik, P.J. Kuikman, C.A. van Diepen, A. Whitmore, W. Daamen, O. Oenema, P. Kabat, and G.M.J. Mohren. 2000. *Article 3.3 and 3.4. of the Kyoto Protocol - consequences for industrialised countries' commitment, the monitoring needs and possible side effect.* Environmental Science and Policy, 3(2/3) 123-134.

Oneil, E. 2006. Developing Stand Density Thresholds to address Mountain Pine Beetle Susceptibility in Eastern Washington Forests, PhD Dissertation, University of Washington, Seattle, 99pp.

Oneil, E. 2006b. Update on Eastern Washington Mortality Trends with Implications for

Eastern Washington Timber Supply, Factsheet #40, Rural Technology Initiative, October 2006, www.ruraltech.org/pubs/fact_sheets/

Pacala, SW, et al. 2001. *Consistent Land- and Atmosphere-Based U.S. Carbon Sink Estimates*. Science 292:2316-2320.

Perez-Garcia, J., B. Lippke, J. Comnick, and C. Manriquez. 2004. Tracking carbon from sequestration in the forest to wood products and substitution. In: CORRIM: Phase I Final Report. Module N. Seattle, WA. 26pp.

Perlack, R., L. Wright, A. Turhollow, R. Graham, B. Stokes, and D. Erbach. 2005. Biomass as Feedstock for a Bioenergy and Bioproducts Industry: The Technical Feasibility of a Billion-Ton Annual Supply. U.S. Department of Energy. Oak Ridge National Laboratory. Oak Ridge, TN. 60pp.

PNWFHP. 2006. Pacific Northwest Region Forest Health Protection, USDA, Forest Service, http://www.fs.fed.us/r6/nr/fid/index.shtml

Powell, D. C. 1999. Suggested Stocking Levels for Forest Stands in Northeastern Oregon and SouthEastern Washington: An Implementation Guide for the Umatilla National Forest. Pendleton, OR, USDA, For. Serv. PNW Region, Umatilla National Forest: 300 pp.

Pyne, S.J. 1997. America's Fires: Management on Wildlands and Forests. Durham, NC: Forest History Society. 54 pp.

Raymond, C.L. and D.L. Peterson. 2005. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Can. J. For. Res. 35:2981-2995.

Sampson, N. and D.Adams, eds. 1994 Assessing Forest Ecosystem Health in the Inland West Journal of Sustainable Forestry, Vol.2 No 1-4.

Sampson, R.N., and D. Hair, eds. 1992. Forests and Global Change. Volume One: Opportunities for Increasing Forest Cover. Report prepared for the USDA For. Serv., US EPA, US DOE, and The American Forests Council. WA D.C. 245pp. plus appendices.

Schlamadinger, B. and G. Marland. 1996. The role of forest and bioenergy strategies in the global carbon cycle. Biomass and Bioenergy 10:275-300.

UNFCCC. 1997. Kyoto Protocol to the United Nations Framework on Climate Change. UNFCCC Document FCCC/CP/1997/7/Add.1.

USFS. 2007. Rocky Mountain Region, Mountain Pine Beetle Management Guide, http://www.fs.fed.us/r2/fhm/bugcrud/silvmpb.htm

Waring, R.H. and W.H. Schlesinger. 1985. Forest Ecosystems: Concepts and Management. Orlando, FL, Academic Press.

Waring, R.H. and S.W. Running 1998. Forest Ecosystems: Analysis at Multiple Scales, San Diego, CA, Academic Press.

Watson, Robert T., Ian R. Noble, Bert Bolin, N.H. Ravindranath, David J. Verardo, and David J. Dokken, 2001, IPCC Special Report on Land Use, Land-Use Change And Forestry, http://www.grida.no/climate/ipcc/land_use/309.htm#s6-3-3

WSU. 2007. Forest Health Issues. Washington State University Cooperative Extension. http://ext.nrs.wsu.edu/forestryext/foresthealth/

Westerling, A.L., H.G. Hidalgo, et al. 2006. Warming and earlier spring increase western US forest wildfire activity. Science 313(5789): 940-943.

Western Regional Climate Center. 2007. http://www.wrcc.dri.edu