Nez Perce–Clearwater National Forests
Forest Plan Assessment

4.0 Baseline Assessment of Carbon Stocks

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4.0 Baseline Assessment of Carbon Stocks

4.1 Executive Summary

Forests substantially mitigate the climate effects of increasing atmospheric CO_2 concentrations by removing carbon from the atmosphere and storing it as biomass. Worldwide, forests offset about one-third of global CO_2 emissions from fossil fuel combustion. U.S. forests offset about 10%–15% of U.S. fossil fuel emissions. The Nez Perce–Clearwater National Forests store approximately 660 teragrams (Tg) of carbon and contain approximately 1.5% of total U.S. forest carbon stocks. Available information suggests that carbon stocks of the Nez Perce–Clearwater National Forests (Forests) have been increasing over the last several decades as they recover from extensive fires in the late 19th and early 20th centuries.

The future trajectory of carbon stocks on the Forests is uncertain and will depend on the spread of root diseases, the extent and severity of future fires, tree mortality caused by bark beetles and other forest insects, the rate of tree regeneration after disturbances, forest management practices, and potential changes in forest productivity. Projected changes in regional climate may exacerbate many of these change agents and thus reduce the carbon stocks on Forests. Forest management activities that reduce the potential for uncharacteristically large and severe natural disturbances and promote rapid forest regeneration after disturbances may reduce some of these potential risks to forest carbon stocks.

4.1.1 *Introduction*

From 1990 to 2006, forest ecosystems absorbed approximately one-third of the annual global carbon emissions from fossil fuel combustion and land use change (Bonan 2008; Canadell, Le Quere et al. 2007; Denman et al. 2007). The feedback of carbon between the atmosphere and terrestrial ecosystems has a significant impact on rates of climate change.

Carbon accrues in forest ecosystems through photosynthesis, and cycles within the system until it is lost through respiration or disturbance. Forests remove carbon from the atmosphere through photosynthesis and convert it into sugars used to grow leaves, wood, and roots. Forests also release carbon dioxide to the atmosphere through respiration and decay of dead wood, litter, and organic matter in soils. In addition, forest fires release some stored carbon to the atmosphere. Fires, insect outbreaks, pathogens, drought stress, and wind storms kill trees and increase the amount of biomass available for decomposition by microorganisms. Timber harvesting also removes carbon from the forest, although some of it is stored in wood products or used to produce energy, displacing fossil fuel use (Ryan et al. 2010) (Figure 4-1).

Large quantities of carbon are stored in the soil and forest floor; and soil usually represents a larger carbon pool than above-ground biomass on forest and woodland sites. Estimates prepared by the U.S. Geological Survey, for the conterminous United States, indicate that total soil organic carbon (SOC) storage is 73 petagrams (Pg) or billion metric tons, and total forest biomass carbon is 17 Pg (Sundquist et al. 2009). Soil organic matter (SOM) is made up of carbon-based molecules derived from dead plant and animal materials in various stages of decomposition, as well as living roots and soil organisms. Organic matter is essential in maintaining site productivity as required by the National Forest Management Act of 1976, and retaining SOM on site also helps mitigate climate change by reducing carbon dioxide (CO_2) inputs to the atmosphere.

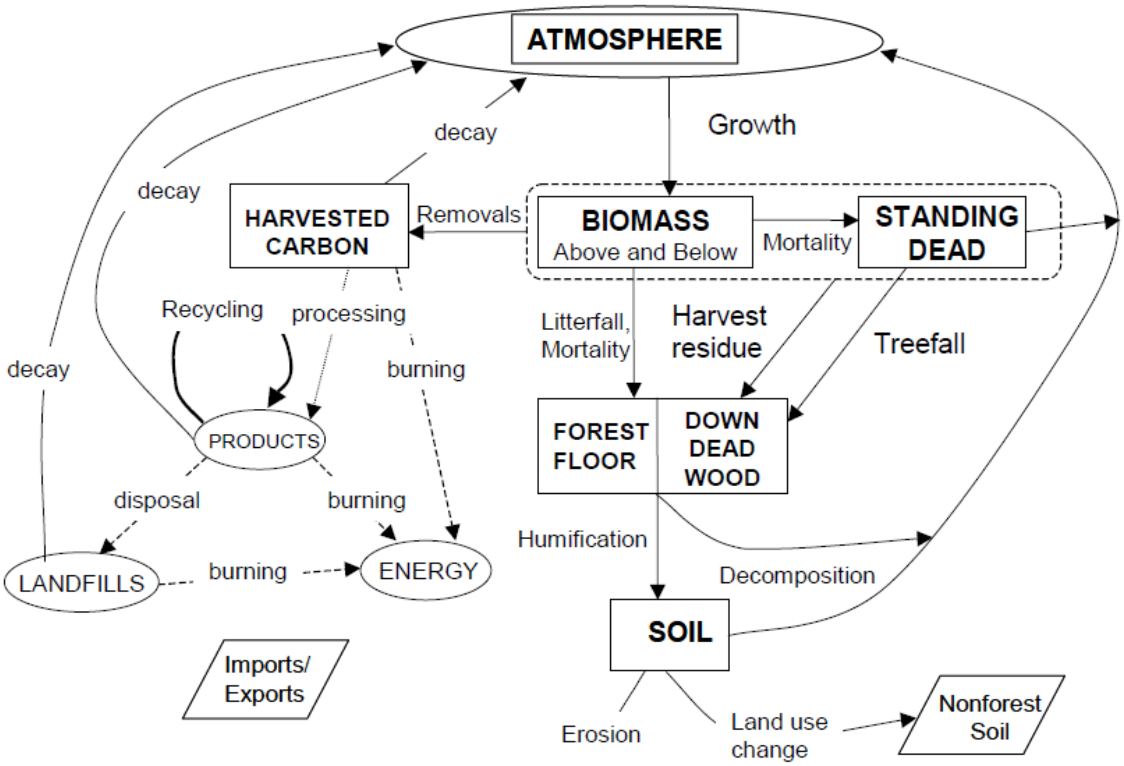

Figure 4-1. Forest sector carbon pools and flows

Carbon storage and sequestration rates are more stable over large forest areas that comprise a multitude of stands of different ages. With multiple stands in different stages of recovery from disturbance, some stands provide a carbon "sink," while others act as net "sources," releasing more greenhouse gases than they sequester (Ryan et al. 2010). Changes in the frequency or severity of disturbance regimes over large areas, compared to the historical baseline, can increase or lower the average carbon stocks in forests (Kashian et al. 2006; Smithwick et al. 2007; McKinley et al. 2011). Over time, these processes can significantly affect the amount of atmospheric CO_2 and, thus, the global climate (Sabine et al. 2004; Canadell, Pataki et al. 2007; Denman et al. 2007; Bonan 2008; McKinley et al. 2011).

Keeping carbon stored in soils and the litter layer can limit CO_2 release to the atmosphere. The size of stored forest carbon pools is affected by successional status, rate of net ecosystem production, and time since disturbance. Stored forest carbon generally increases with forest age (Pregitzer and Euskirchen 2004). The phase of aboveground carbon accrual can continue beyond 300 years in some forest types, depending on the frequency of disturbances (Keeton 2008). Carbon storage is generally higher in stands that are structurally complex, with snags (standing dead trees) and down woody debris (Nunery and Keeton 2010). Stands that are managed for maximum timber volume production typically have lower amounts of carbon, while stands managed for sawtimber and older forest habitat store larger amounts.

Temporal aspects of carbon capture and release are important. When older forests with a large quantity of stored carbon are harvested, or trees are removed by natural disturbance, carbon removed from the site can take more than a hundred years to be replaced (Harmon et al. 1990).

Carbon is released to the atmosphere through decomposition or incineration of wood products, and by accelerated decomposition of forest litter on the harvested site. Establishing young forests on harvested sites is considered "carbon neutral" in some carbon accounting schemes. A young forest has a more rapid initial rate of carbon capture, but a lengthy time period is required for the carbon pool to be replenished to pre-harvest levels. One strategy proposed for limiting peak concentrations of atmospheric CO_2 is to maintain stored pools of carbon in forests until after CO_2 has peaked, and then release it slowly. The rate of vegetative forest carbon gains and losses and vegetative carbon stocks varies over a forest's life cycle. When forests are disturbed by fire, harvest, insect outbreaks, and other perturbations, vegetative carbon stocks will usually recover fully over the forest's the life cycle (Kashian et al. 2006). Thus, over time, the net vegetative carbon change is often zero (Figure 4-2) (McKinley et al. 2011).

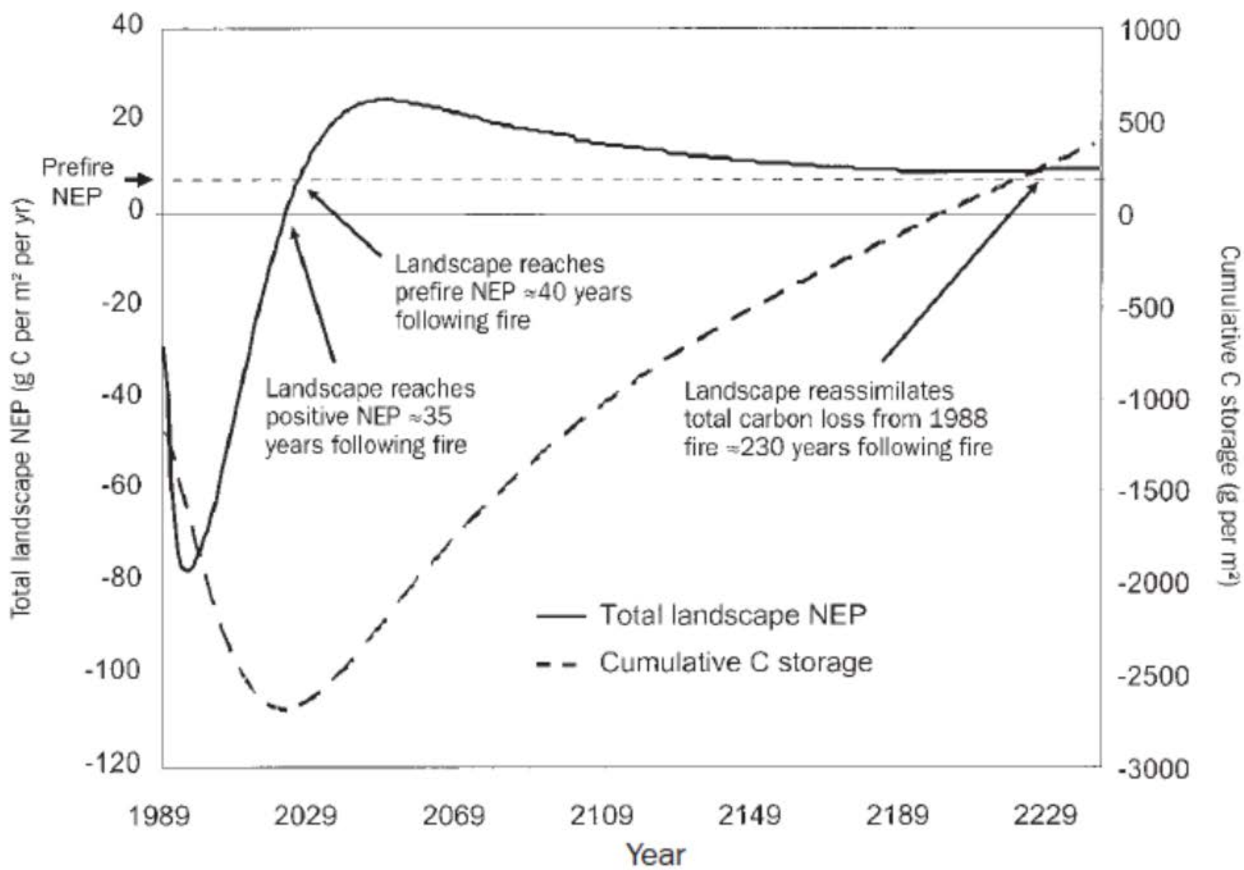

Figure 4-2. Post-fire forest carbon (C) and net ecosystem productivity (NEP) recovery over time, showing total carbon that includes the decomposition of trees killed by fire (dead wood) and tree regeneration (trees) in a case study from the 1988 Yellowstone Fires. If a forest regenerates after a fire and the recovery is long enough, the forest will recover the carbon lost in the fire and in the decomposition of trees killed by the fire. Figure is published in Kashian et al. (2006)

Most studies estimate that the terrestrial biosphere is currently a net sink, removing more carbon from the atmosphere than it is emitting; it is therefore mitigating the effects of CO_2 emissions from fossil fuel combustion and land use change (Denman et al. 2007; Le Quéré et al. 2009). Forests are the dominant contributors to the terrestrial ecosystem carbon sink, removing about

2.4 billion metric tons of CO_2 per year from the atmosphere from 1990 to 2007, offsetting roughly one-third of global CO_2 emissions from fossil fuel combustion (Pan et al. 2011).

It is clear that forests play a key role in mitigating global CO_2 emissions and, in turn, the rate of climate change (Nabuurs et al. 2007). However, the future of this ecosystem service is uncertain. Converting forests to non-forest, particularly in the tropics, and the potential effects of climate changes on forests raise questions about the future strength of the global forest carbon sink, and whether it may convert to an additional source of carbon to the atmosphere. Ultimately, the answers to these questions will have a significant impact on the global climate.

This section of the Assessment summarizes the best available scientific information on the carbon stocks and fluxes of the Forests. It provides estimates of existing carbon pools of the forest sector (live and dead aboveground biomass, soil carbon, and harvested wood products). These estimates are derived from local data collected during soil surveys, systematic forest inventory (the Forest inventory and Analysis Program), and forest harvest records.

4.1.2 *Current Carbon Stocks of the Nez Perce–Clearwater National Forests*

4.1.2.1 Current Carbon Stocks

The reservoir of carbon stored in U.S. forests is approximately 42,444 Tg (EPA 2013). Public forestlands contain approximately 42% of this carbon reservoir with National Forest System lands storing an estimated 26%, or approximately 11,000 Tg, of all forest carbon of the United States (Heath et al. 2011). The Forests store an estimated 660.4 Tg of carbon (Table 4-1), which represents about 1.5% of the total of approximately 44,907 Tg of carbon in forests of the coterminous United States (EPA 2013). The average density of forest carbon is about 319 Mg of carbon per hectare (Mg C/ha). The average carbon density of the Forests is among the highest in the Northern Rockies and interior western United States (Hicke et al. 2007; Potter et al. 2008).

For estimating carbon stocks or stock change (flux), carbon in forest ecosystems can be divided into the following five storage pools (IPCC 2003):

- Aboveground biomass, which includes all living biomass above the soil including stem, stump, branches, bark, seeds, and foliage; this category includes live understory
- Belowground biomass, which includes all living biomass of coarse living roots greater than 2 mm diameter
- Dead wood, which includes all non-living woody biomass either standing, lying on the ground (but not including litter), or in the soil
- Litter, which includes the litter, fumic, and humic layers, and all non-living biomass with a diameter less than 7.5 cm at transect intersection, lying on the ground
- Soil organic carbon (SOC), including all organic material in soil to a depth of 1 meter but excluding the coarse roots of the aboveground pools

For use in this assessment the five pools were categorized into two main groups: aboveground and soil carbon. The aboveground biomass includes live tree carbon (including coarse roots), understory vegetation (including coarse roots), and dead wood (standing and down). The soil carbon includes the forest floor litter layer and SOC (Table 4-1).

In addition, two harvested wood pools are necessary for estimating carbon flux:

- Harvested wood products (HWP) in use
- HWP in solid waste disposal sites (SWDS)

Table 4-1. Carbon stocking of the Nez Perce–Clearwater National Forests

Ecosystem Component	Mg C/ha	Tg C
Aboveground	**121.3**	**248.6**
Live trees (including coarse roots)	97.6	199.9
Understory vegetation (including coarse roots)	2.3	4.6
Dead vegetation (standing and down)	21.5	44.0
Soil	**198.0**	**405.8**
Forest floor	68.3	140.0
Mineral soil to 1 meter	129.7	265.8
Harvested Wood Products		**6.0**
Products in use		3.8
Products in SWDS		2.2
Total	**319.3**	**660.4**

Note: Total vegetation carbon estimate is data from Heath et al. (2011). Estimates of carbon stored in harvested wood products in use and in solid waste disposal systems data are from Anderson et al. (2013). Soil and forest floor calculations are based on National Cooperative Soil Survey Data, Van Dechert (1982), Page-Dumroese and Jurgensen (2006) and Smith and Heath (2002).

4.1.2.2 Soil Carbon

The persistence of carbon in soil is an ecosystem property. Carbon compounds are inherently unstable and owe their abundance in soil to biological and physical environmental influences that protect carbon and limit the rate of decomposition (Schmidt et al. 2011). Soil organic matter (SOM) is formed by the biological, chemical, and physical decay of organic materials that enter the soil system from sources aboveground (e.g., leaf fall, crop residues, animal wastes and remains) or belowground (e.g., roots, soil biota). The elemental composition of SOM varies, with values in the order of 50% carbon, 40% oxygen and 3% nitrogen, as well as smaller amounts of phosphorus, potassium, calcium, magnesium, and other elements as micronutrients. Large quantities of SOM accumulate in environments such as wetlands, where the rate of decomposition is limited by a lack of oxygen, and high-altitude sites where temperatures are limiting. SOM includes carbon compounds in the forest floor litter layer and the mineral soil to a depth of 1 meter (or depth to bedrock if the soil is shallower than 1 meter). Most carbon in mineral soil comes from root turnover (Schmidt et al. 2011), although some is moved from the forest floor into upper mineral soil layers (Qualls et al. 1991). The soil carbon stock on the Forests is 405.8 Tg C or 198 Mg C/ha.

Forest floor carbon numbers were generated using data from the National Cooperative Soil Survey, research data as analyzed by Smith and Heath (2002), and regional data collected by Page-Dumroese and Jurgensen (2006). SOC was estimated for the Forests using data from the National Cooperative Soil Survey and a graduate research project at the University of Idaho. A modified equation following the methods of Batjes (1996) was used to calculate the total SOC to a depth of one meter for the mineral soil. The estimated amount of SOC on the Nez Perce National Forest is 113.8 Tg SOC and on the Clearwater National Forest is 150.2 Tg SOC. This results in a total SOC stock of 265.8 Tg SOC on the Forests with an average carbon density of 129.7 Mg C/ha. This carbon density is within the ranges found in several research projects (Table 4-2). The greater amount of SOC on the Clearwater National Forest is expected as the forest, on average, receives more precipitation and is colder, which decreases organic carbon decomposition. Soil inorganic carbon (SIC) was not assessed for the Forests due to lack of data.

Table 4-2. Soil carbon densities from research

Environment	Soil Organic Carbon (Mg C/ha)	Source
Cool, conifer forests of U.S.	403–494	Kern (1994)
Cool, temperate forests of Maine	130	Davidson and Lefebvre (1993)
Global temperate forests	118	Schlesinger (1977)
Cool, temperate forest of north central U.S.	84–152	Franzmeier et al. (1985)

4.1.2.3 Harvested Wood Products

In addition to the ecosystem carbon stocks described above, wood products produced with timber from the Forests store approximately 6 Tg of carbon. HWP are products made from wood including lumber, panels, paper, paperboard, and wood used for fuel that are in use or have been discarded to solid waste disposal systems (SWDS), or landfills and dumps. Of this amount, an estimated 3.8 Tg are held by products currently in use and 2.2 Tg are stored in SWDS (Table 4-1). An in depth accounting of the HWD for the Forests can be found in Anderson et al. 2013.

4.1.3 *Recent Carbon Stock Trends of the Nez Perce–Clearwater National Forests*

4.1.3.1 Aboveground Forest Ecosystem Carbon

The principal drivers of aboveground forest carbon stocks are forest growth and mortality. Forest Inventory and Assessment (FIA) surveys completed in 2001 on the Nez Perce National Forest, estimated that net annual growth (annual growth minus losses due to mortality) is 306 million tons of biomass (Disney 2010). FIA surveys completed in 1999 on the Clearwater National Forest estimated net annual growth is 2.3 Mg of biomass (Hughes 2011). The primary agents for decadal and longer-scale carbon changes on the Forests are root diseases, wildland fire, bark beetles, and timber harvest. Root disease is the leading cause of tree mortality on both Forests (49% of all mortality on the Clearwater National Forest and 22% on the Nez Perce National Forest). Mountain pine beetle and other forest insects are the second leading cause of tree mortality on the Clearwater National Forest and Nez Perce National Forest (20% and 22%, respectively), followed by wildland fire (3% and 20%, respectively) (Disney 2010, Hughes 2011). Longer-term trends reflecting these change agents can be inferred from 20th century trends in forest age and structure classes.

Recent scientific literature documents the general pattern of changes in carbon stocks and net ecosystem productivity (NEP)[1] over the period of stand development in coniferous forests of the interior western United States (Smithwick et al. 2008; Bradford et al. 2008; Dore et al. 2008;

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[1] Net ecosystem productivity, or NEP, is defined as gross primary productivity (GPP) minus ecosystem respiration (ER) (Chapin et al. 2006). It reflects the balance between (1) absorbing CO_2 from the atmosphere through photosynthesis (GPP) and (2) the release of carbon into the atmosphere through respiration by live plants, decomposition of dead organic matter, and burning of biomass (ER). When NEP is positive, carbon accumulates in biomass. Ecosystems with positive NEP are referred to as a carbon sink. When NEP is negative, ecosystems emit more carbon than they absorb. An ecosystem with negative NEP is referred to as a carbon source.

Luyssaert et al. 2008; Irvine et al. 2007; Hall et al. 2006; Kashian et al. 2006; Law et al. 2001; Carey et al. 2001). Total carbon stocks decline from disturbance, and then increase rapidly during intermediate years. They then continue to increase, but at declining rate over time until another significant disturbance (regeneration timber harvest or tree mortality resulting from drought, fire, insects, disease, or other causes) kills large numbers of trees (Figure 4-2) (Pregitzer and Euskirchen 2004; Canadell, Pataki et al. 2007). Carbon flux and NEP are lowest, and usually positive (a carbon source to the atmosphere) in young stands (0–30 years old) following disturbance because carbon emissions (from decay of dead biomass) exceed the amount of carbon removed from the atmosphere by photosynthesis within the stand. As the stand develops, NEP increases and the stand becomes a carbon sink. NEP and carbon sink strength generally peak at the intermediate stage of stand development (40–100 years old), and then decline with age but often remain negative (Pregitzer and Euskirchen 2004; Canadell, Pataki et al. 2007) (Figure 4-2). Over the long term (centuries), net carbon storage is often zero, if stands regenerate after disturbance, because regrowth of trees recovers the carbon lost in the disturbance and subsequent decomposition of trees killed by the disturbance (Kashian et al. 2006).

On the Forests, the distribution of forest age and structure classes has changed substantially since the early 20th century (USDA Forest Service 2004). Intermediate age classes have increased in area while the amount of young stands has decreased. In most forest types, the abundance of older, late successional stands has declined. The cause of these changes varies by forest type and geographic location, but the most widespread agents of change are root disease, white pine blister rust, timber harvest, and the substantial decline in acres burned since 1935. The increase in intermediate age classes is primarily due to the forest regrowth that followed the large stand-replacing fires in the late 19th and early 20th centuries. Figure 4-3 displays the current age class distributions of the Forests.

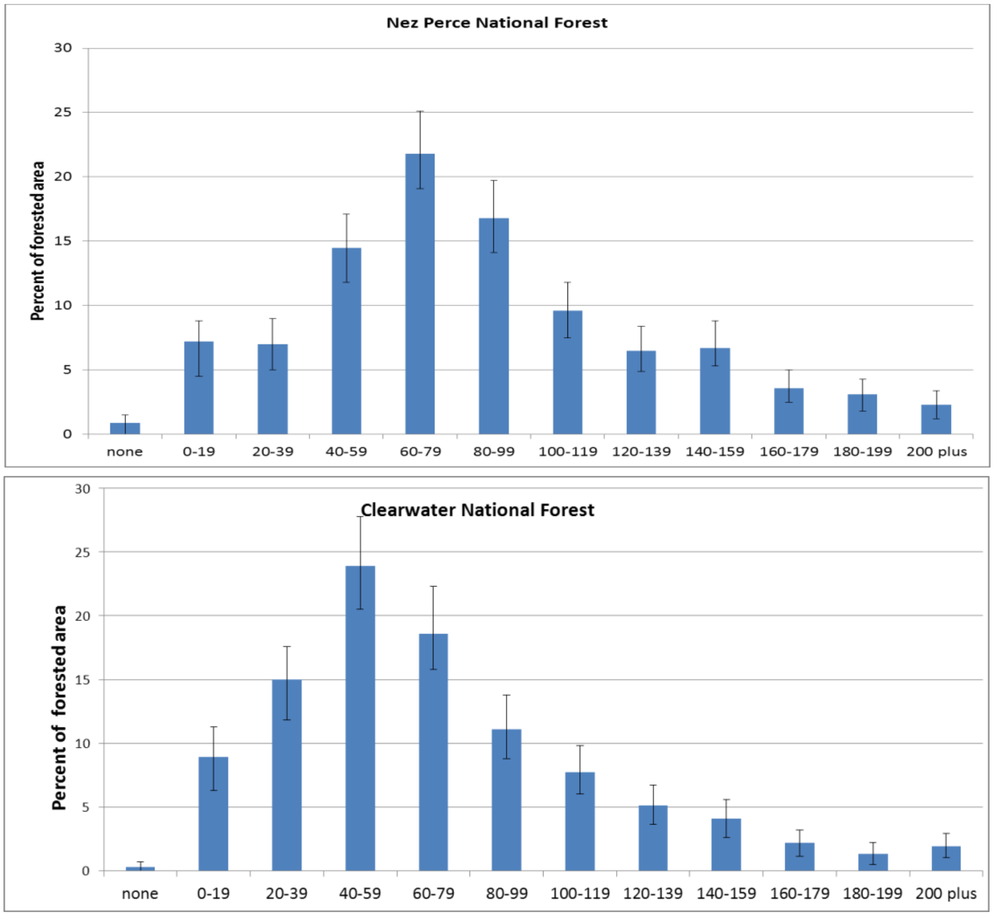

Figure 4-3. Estimated current age class distribution of the Nez Perce-Clearwater National Forests. Calculated values represent conditions at the time of inventory: 2000–2007 for the Nez Perce National Forest and 1998–2007 for the Clearwater National Forest. Narrow vertical lines represent the 90% confidence interval. The estimated means and confidence intervals are based on 324 Forest Inventory and Assessment field unit locations (1,246 subplots) from the Nez Perce National Forest and 293 Forest Inventory and Assessment field locations (1,127 subplots) from the Clearwater National Forest

These observed trends in age and structure classes on the Forests generally mirror those identified for much of the Inland Northwest (Hessburg and Agee 2003). Hessburg et al. (2000) constructed vegetation maps from 1932 to 1966 and 1981 to 1993 with aerial photographs of sample subbasins within the interior Columbia River basin. Comparing historic and current vegetation maps, Hessburg et al. (2000) found that the forests of central Idaho experienced a significant area increase of intermediate structural classes. Stand initiation structures (new forests) declined significantly due to fire exclusion, despite timber harvest activity. This analysis found no significant change in the amount of old forest structures (both single and multi-storied) in the central Idaho mountains “ecological reporting unit” (ERU) that contains most of the Forests (Hessburg et al. 1999, 2000). However, they noted that timber harvest activities reduced the abundance of medium- and large-sized trees distributed in other forest structures as remnants of stand-replacing fires.

These observations are supported by data on annual acres burned on the Forest over the last 110–140 years (Figure 4-4). These records indicate that a relatively high number of acres burned in the late 19th and early 20th centuries, followed by an extended period (4 to 5 decades) of comparatively few acres burned. Over the last 25 years (1985–2010), acres burned have increased over the mid-20th century, but less than the early 20th century.

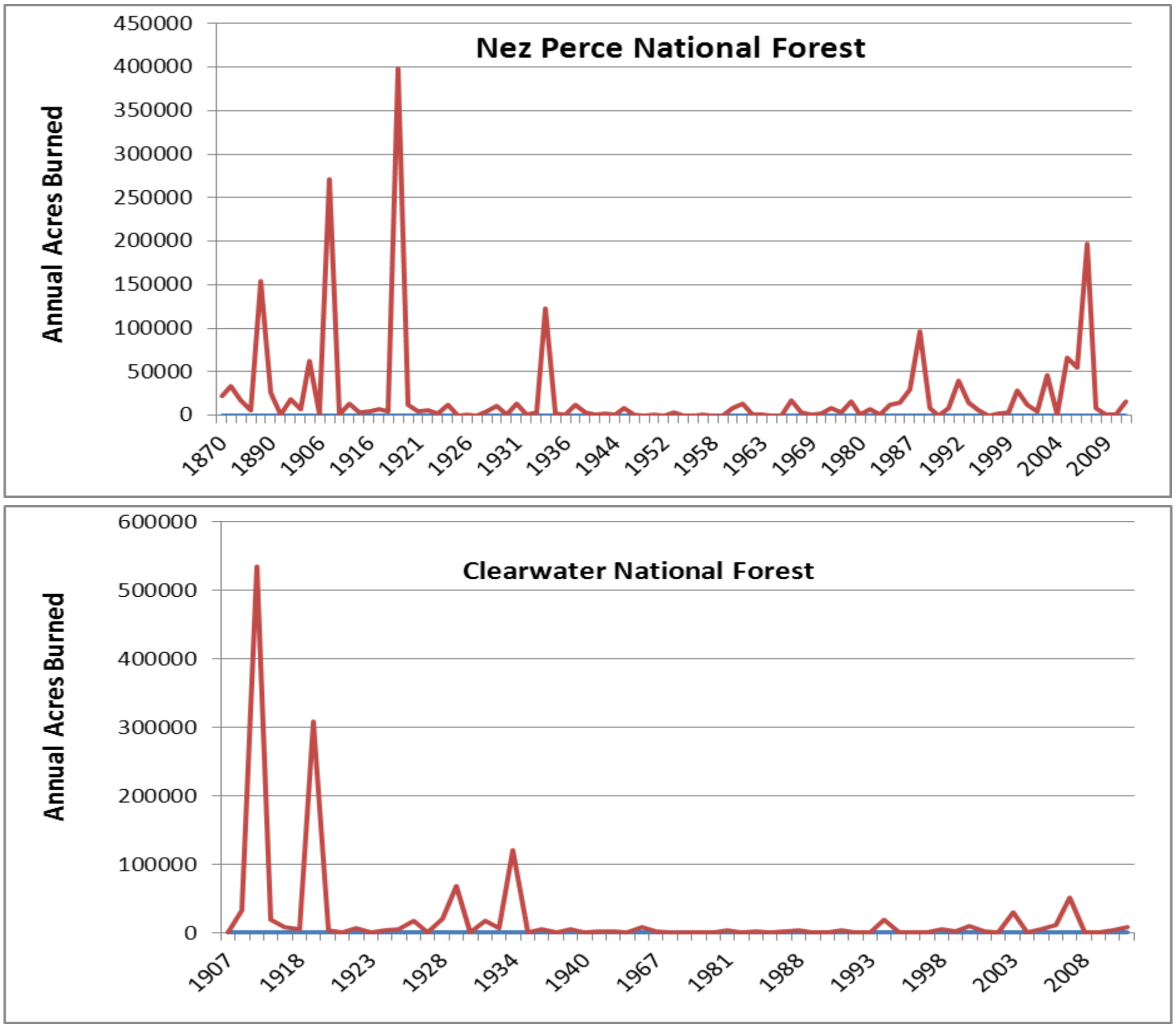

Figure 4-4. Annual acres burned on Nez Perce-Clearwater National Forests

Recently, bark beetle populations and resulting tree mortality have increased substantially in western North America. On the Forests, beetle-caused tree mortality has also been substantial, although less severe than some other areas (Table 4-3 and Figure 4-5).

Table 4-3. Acres of bark beetle mortality 2001–2011

Trees Per Acre	1–5	6–15	15+
Nez Perce National Forest (acres)	292,236	138,748	61,194
Clearwater National Forest (acres)	217,740	42,175	5,276
Total (acres)	509,976	180,923	66,740

Note: Data are compiled from aerial detection surveys. The following bark beetles are included in these data: Mountain Pine Beetle, Douglas-fir Beetle, Spruce Beetle, Western Pine Beetle, Western Balsam Bark Beetle, Pine Engraver, Douglas-fir Engraver, and Fir Engraver.

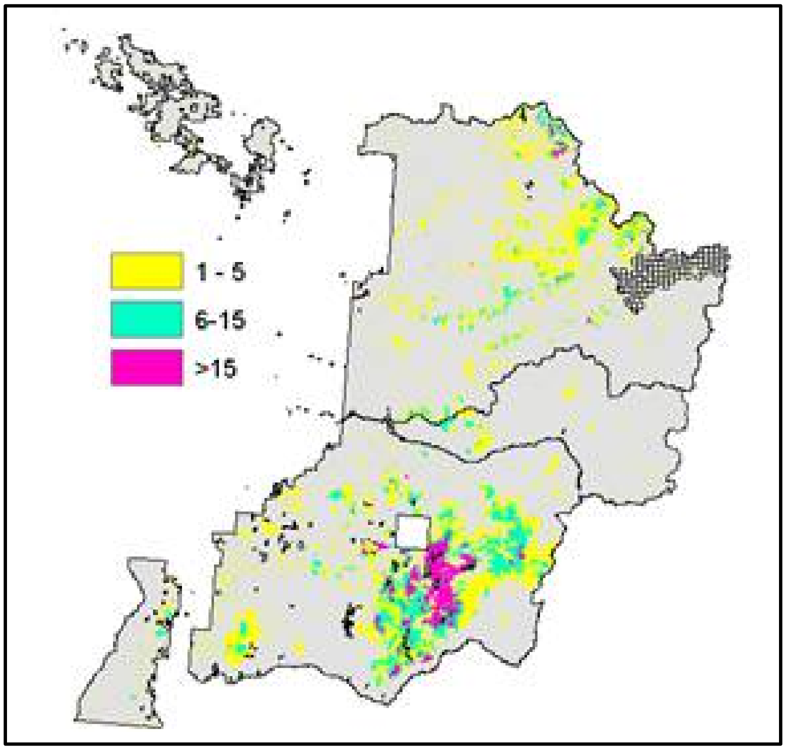

Figure 4-5. Areas of the Nez Perce-Clearwater National Forests with bark beetle-caused tree mortality 2001–2011. Tree per acre classes represent the number of trees per acre killed by bark beetles.

Root diseases have a substantial, and perhaps the most significant, effect on forest carbon stocks and flux on the Forests. Root disease-caused tree mortality has long been common in Idaho forests. However, over the past century, disease-tolerant species such as western white pine, western larch, and ponderosa pine have decreased significantly in abundance due to white pine blister rust, wildfire suppression, and historical harvesting practices. These species have been replaced with Douglas-fir, grand fir, and subalpine fir, which are most susceptible to root diseases, resulting in substantially increased tree mortality and productivity losses in today's forests (Byler et al. 2000). Root diseases reduce stand densities, stall forest succession, result in smaller trees, and substantially decrease forest productivity (Figure 4-6). Thus, moderate and high-severity root disease centers are a major source of forest mortality and a long-term constraint on forest carbon sequestration rates.

Figure 4-6. Typical root disease centers of moderate severity. Note the dead standing trees, reduced stand density, loss of crown cover, and substantially lower productivity. Because these effects persist for long periods until disease resistant tree species are able to occupy the site, root diseases limit forest carbon stocks and sequestration rates for longer periods than other disturbances.

Root disease is the leading cause of tree mortality on the Nez Perce National Forest (22% of all mortality) (Disney 2010) and Clearwater National Forest (49% of all mortality) (Hughes 2011) (Table 4-4). Root diseases affect more acres on these National Forests than wildland fire, bark beetles, and timber harvest combined. Because root diseases can reduce tree growth and stocking densities for many decades, their effects on forest carbon stocks and flux are more persistent than the effects from other disturbance agents.

Table 4-4. Estimated acres of low, moderate, and high root disease effects on the Nez Perce-Clearwater National Forests

Root Disease Severity	Low (1–20 ft^2 basal area loss)	Moderate (21–80 ft^2 basal area loss)	High (>80 ft^2 basal area loss)
Nez Perce National Forest (acres)	864,119	338,639	15,976
Clearwater National Forest (acres)	609,280	469,787	21,795
Total (acres)	925,099	808,429	37,771

Source: National Insect and Disease Risk Map (USDA Forest Service 2006)

4.1.3.2 Soil Carbon

Soil carbon is easily lost but difficult to rebuild. In the past 25 years, one-quarter of the global land area has suffered a decline in productivity and in the ability to provide ecosystem services because of soil carbon losses (Bai et al. 2008). Natural processes and human activities can result in the loss of organic matter and stored carbon from a site. Forest harvesting can affect the stored soil carbon pool through changes in environmental conditions (e.g., temperature, light) that stimulate decomposition, while reducing inputs of litter and woody debris so the carbon pool is not as rapidly replenished (Grigal and Vance 2000).

Disturbance and Soil Carbon

The amount of carbon loss varies considerably after each timber harvest. Factors include the silvicultural prescription, harvesting method, tree species, amount of debris left after harvest, site preparation techniques, soil characteristics, and climate. Harmon and Marks (2002) modeled carbon pools in Pacific Northwest forests, finding that the factors critical to optimizing carbon storage were rotation length, amount of wood harvested, and amount of detritus removed by slash burning. In the years following major disturbances, the losses from decay of residual dead organic matter exceed the carbon uptake by regrowth (Nabuurs et al. 2007).

A review on harvesting techniques suggested that their effect on soil carbon is rather small (Johnson and Curtis, 2001, Nave et al. 2010), but response of the forest litter layer is more variable and difficult to quantify; it is affected by site conditions, amount of woody residue, and site preparation practices (Grigal and Vance 2000, Gower et al. 2006). Differences in intermediate and regeneration harvests have been observed but are not fully understood in relation to soil carbon. The effects of thinning are unclear (Schils et al. 2008). Thinning affects the distribution of biomass in a forest stand and changes the microclimate. Decomposition of forest floor carbon is temporarily stimulated (Aussenac 1987; Piene and Van Cleve 1978). The stand microclimate returns to previous conditions unless the thinning intervals are short and intensities are high. Litterfall is temporarily lowered in strongly thinned stands. This reduces forest floor accumulation, but the input of thinning residues into the soil may compensate for that (de Wit and Kvindesland 1999). Regeneration harvesting removes biomass, disturbs the soil, and changes the microclimate more than a thinning operation (Schils et al. 2008). In the years following harvesting and replanting, soil carbon losses may exceed carbon gains in the aboveground biomass. The long-term balance depends on the extent of soil disturbance. Continuous cover forestry with selective harvesting is linked with reduced soil disturbance compared with clear-cut harvesting, which may decrease soil carbon losses (ECCP-Working group on forest sinks 2003). Olsson et al. (1996) found that clear-cutting conifers in Sweden with bole-only removal led to a large reduction in soil carbon and nitrogen; additional removal of logging debris after clear-cutting had a minor effect. Similar results were noted for microbial biomass and soil nitrogen mineralization at aspen North American Long Term Soil Productivity (LTSP) study sites in Michigan; the initial bole clear-cut had a greater effect than additional removals (Hassett and Zak 2005).

Johnson and Curtis (2001) suggest that the amount of woody residue left after harvest is a “dominant control” of the forest floor. The litter layer and woody debris can be partially mixed into mineral soils during some harvests, where decomposition proceeds more slowly (Yanai et al. 2003). Harvest residues left on the soil surface increase the carbon stock of the forest floor, disturbance of the soil structure leads to soil carbon loss. Nave et al. (2010), in a meta-analysis of 432 harvests in temperate forests worldwide, report a statistically significant 8% reduction in overall soil carbon (including both forest floor and mineral soil). The forest floor lost an average of 30% of carbon following harvest, but coniferous and mixed forests lost less carbon than hardwood forests. Whole-tree harvest caused a small decrease in A-horizon carbon stocks, whereas conventional harvesting, leaving the harvest residues on the soil, resulted in a small increase.

Site preparation techniques include manual, mechanical, and chemical methods and prescribed burning, most of which lead to the exposure of the mineral soil by removal or mixing of the organic layer. The soil disturbance changes the microclimate and stimulates the decomposition of SOM, thereby releasing nutrients (Palmgren 1984; Johansson 1994). A review on the effects of site preparation showed a net loss of soil carbon and an increase in stand productivity (Johnson 1992). The effects varied with site and treatment. Several studies that compared different site preparation methods found that the loss of soil carbon increased with the intensity of the soil disturbance (Johansson 1994; Örlander et al. 1996; Schmidt et al. 1996). At scarified sites, organic matter in logging residues and humus, mixed with or buried beneath the mineral soil, is exposed to different conditions for decomposition and mineralization compared to conditions existing on the soil surface of clear-cut areas. The soil moisture status of a site has great importance for the response to soil scarification. The increase in decomposition was more pronounced at poor, coarsely textured dry sites than on richer, fresh to wet sites (Johansson 1994). Sandy soils are particularly sensitive to management practices, which result in significant losses of carbon and nitrogen (Carlyle 1993). Intensive site preparation methods might result in increased nutrient losses and decreased long-term productivity.

Longer rotation periods have been proposed to foster carbon sequestration in forests. The effect of increased rotation lengths is mainly determined by the current management practice. Longer rotation lengths with more old forests lead to higher carbon pools than short rotations with only young plantations. Old-growth forests have the highest carbon density, whereas younger stands have a larger carbon sink capacity. The theoretical maximum carbon storage (saturation) in a forested landscape is attained when all stands are in old-growth state, but this rarely occurs as natural or human disturbances maintain stands of various ages within the forest (Nabuurs et al. 2007). After harvest operations, soil carbon pools in managed forests recover to the previous level. Short rotation lengths where the time of harvest is close to the age of maximum mean annual increment will maximize aboveground biomass production, but not carbon storage. Longer rotation periods imply that the disturbance frequency due to forest operations is reduced and soils can accumulate carbon (Schulze et al. 1999). Growth and yield tables suggest that stand productivity declines significantly in mature forest stands. However, a mature Siberian Scots pine forest and old-growth forests in the United States transferred a higher proportion of carbon into the soil than in the early stages of stand development and continuously increased the soil carbon stock (Harmon et al. 1990; Schulze et al. 2000). The accumulation of carbon continues until the carbon gain from photosynthesis is larger than respiration losses. Late-successional species (e.g., beech, Norway spruce) are able to maintain high carbon sequestration rates for longer than pioneer tree species. Over mature forest stands are not able to close canopy gaps created by natural mortality or thinning. Consequently the decomposition of SOM is enhanced and decreases the soil carbon pool. Several modelling studies suggest that very long rotation lengths do not necessarily maximize the total C balance of managed forests (Cannell 1999; Liski et al. 2001; Harmon and Marks 2002). In general, ageing of forests results in increasing carbon densities in management systems with longer rotation lengths, provided the harvest age is not beyond the age where the forest stand turns from a net sink to a source of carbon (Jandl et al. 2007). The magnitude of the effect of increased rotation lengths depends on the current management practice. At the landscape level, longer rotation lengths with more old forests lead to higher carbon pools than short rotations with only young plantations.

The forest floor plays a major role in site productivity. Forest floor processes are variable and not fully understood; however, research has consistently shown that the loss or alteration of the litter layer has implications for soil fertility and populations of forest organisms. Powers et al. (2005) note its function as a reservoir for available nitrogen and phosphorous. Findings from the LTSP study suggest that under moderate and warmer climates, carbon from harvest residues is mainly respired as CO_2, and very little carbon is incorporated into the soil (Powers et al. 2005). However, the most drastic management scenario of the LTSP study shows that the removal of the forest floor (in addition to harvest residue removal) also led to reduced nitrogen availability and significant reductions in soil carbon concentrations down to a depth of 20 cm (Powers et al. 2005). Jurgensen et al. (1997) noted that “a number of studies have linked substantial reduction in mycorrhizae development and tree growth to high levels of soil disturbance or removal of organic horizons” and that “timber harvesting and extensive site preparation… reduces the amount of surface organic material.” Microbial communities and other soil organisms are affected by the loss of organic material. Belleau et al. (2006) associated the amount of woody debris retained in boreal aspen stands with favorable soil nutrient levels, and concluded that the amount of slash left after harvest was “the main factor found to affect soil microbial community characteristics and soil nutrient availability.” Battigelli et al. (2004) reported 1-year findings for a LTSP site in British Columbia, focusing on Oribatid mite species (beetle mites that feed on living and dead plant material and fungi, and are active in decomposition). They found that “soil compaction and organic matter removal significantly reduced the density and diversity of soil mesofauna,” with the loss of organic material being of greater concern than compaction.

Results from the LTSP suggest that decomposition and carbon release from woody debris after forest harvest varies by climate zones. In warmer and dryer climates, carbon is mainly respired and released to the atmosphere rather than being incorporated into the soil. In wetter and cooler climates, much of the carbon may be moved into the soil over time (Powers et al. 2005). Soil taxonomic order predicted much of the variation in carbon storage in mineral soils in the meta-analysis by Nave et al. (2010), which may be related to typical management practices on certain soils, or differences in residue quality. Nave et al. (2010) found that Inceptisols had significant declines in carbon storage after harvests. The rate of woody debris input needed to maintain soil carbon pools is unclear. Jurgensen et al. (1997), in a summary paper on productivity considerations for the Intermountain West, discuss how much woody residue is needed to maintain SOM levels. They note that various guidelines in the region ranged from retaining 4.5 tons/acre after harvest (for all woody debris, including coarse and fine wood) up to 56 tons/acre. The lower values applied to dry conifer sites; based on slow cycling rates for these systems, it was thought that the lower levels of residue retention would be sufficient to maintain SOM. Fire risk was also a consideration. Higher levels of woody residue retention were suggested for mixed conifer forests in the northern Rocky Mountains.

At a global scale, it is uncertain whether forest harvesting has a positive or negative effect on carbon balances. Removing biomass from a forest removes carbon and decreases the site carbon pool; carbon re-accrues on the site as the forest regrows, and the site may or may not eventually store the same amount of carbon as it did prior to harvest. If harvested materials are utilized in a process that retains carbon in forest products for a long time period, the activity can delay the release of some CO_2 to the atmosphere and be a net benefit to atmospheric carbon balances. Whether these benefits are realized depends on the type of product and its lifespan, energy utilization in harvesting and processing, the fate of waste wood, and potential for accelerated organic matter decomposition on the harvested site.

Fire and Soil Carbon

The role of fire in ecosystem carbon changes is not straightforward. Several experiments showed that wildfire had caused increases in soil carbon, which may be driven by the incorporation of charcoal into soils and new carbon inputs via post-fire N2 fixation (Hirsch et al. 2001; Johnson and Curtis 2001; Johnson et al. 2004; Schulze et al. 1999). Fire oxidizes organic forms of carbon to CO_2, which is rapidly lost to the atmosphere. The degree to which soil and forest floor carbon is affected is variable and site-specific, depending on fire intensity and soil burning temperatures, amount of organic matter on the site, and other environmental conditions during and after the fire. Carbon losses can be large in catastrophic wildfires. Severely burned plots on the 2002 Biscuit Fire lost an average of 22% of soil carbon (Erickson and White 2008). Most carbon losses are from the forest floor and surface horizons to about 10 cm; fire has little impact on deeper mineral soil layers.

Prescribed fires are designed to be cooler, and generally have less impact on soil carbon. Studies in the southeastern U.S. noted decreases in forest floor carbon, but nearly equivalent carbon increases in the upper 5–10 cm of soil (Johnson 1992).

Fire suppression can lead to fuel buildup and increase the risk of catastrophic fires, often leading to large losses of carbon when burns do occur. For this reason, it is advantageous over the long term to reduce the risk of catastrophic fire by removing fuels from the site or conducting prescribed burns, even though they result in short-term carbon losses.

4.1.3.3 Harvested Wood Products Carbon

Timber harvesting and other types of silvicultural practices remove biomass from the ecosystem, reducing carbon stocks. The Forests are one of the top timber producing forests in the Northern Region. Timber production has varied greatly over the past 100 years with a peak in the mid 1960s and has since been on a declining trend (Figure 4-7). Cummulative carbon storage in HWP shows a similar trend on a delay. Cumulative carbon stocks in wood products harvested from the Forests peaked in the late 20th century and have declined since 1995 (Figure 4-8). Annual change in carbon stored generally mirrors the trend in annual acres harvested and shows a decline since the mid-1960s (Figure 4-9). A negative annual change, as seen since the year 2000, results in the HWP becoming a source of atmospheric carbon rather than a sink.

As of 2013, approximately 3.8 Tg of carbon are stored in harvested wood products currently in use, and another 2.2 Tg are stored in solid waste disposal systems (e.g., landfills).

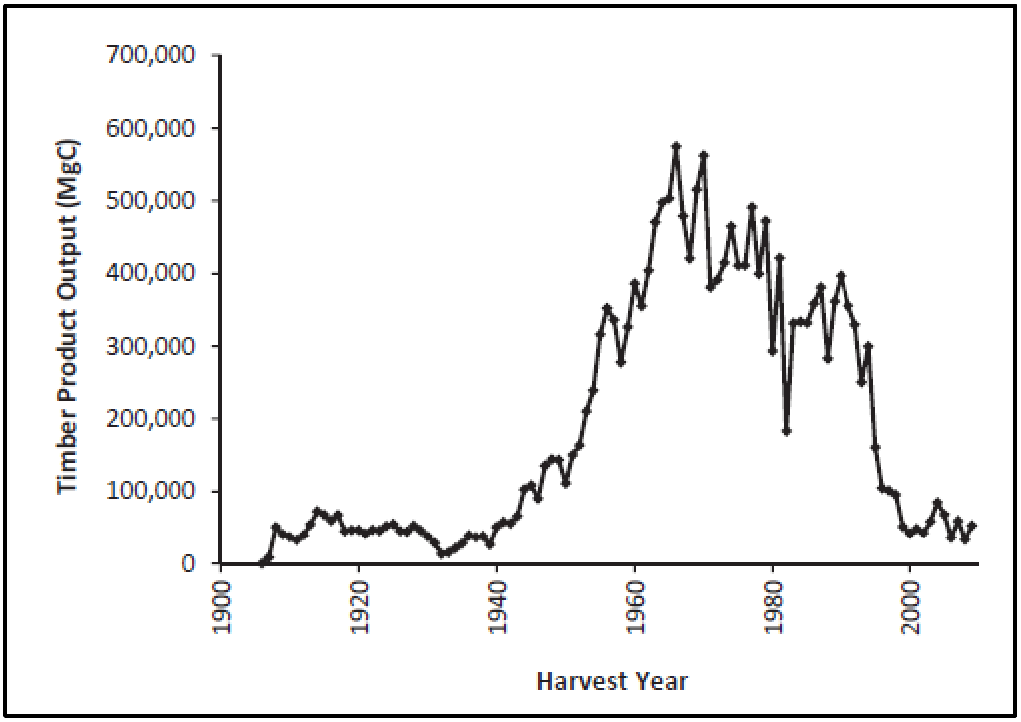

Figure 4-7. Annual timber production from 1906 to 2010 (from Anderson et al. 2013)

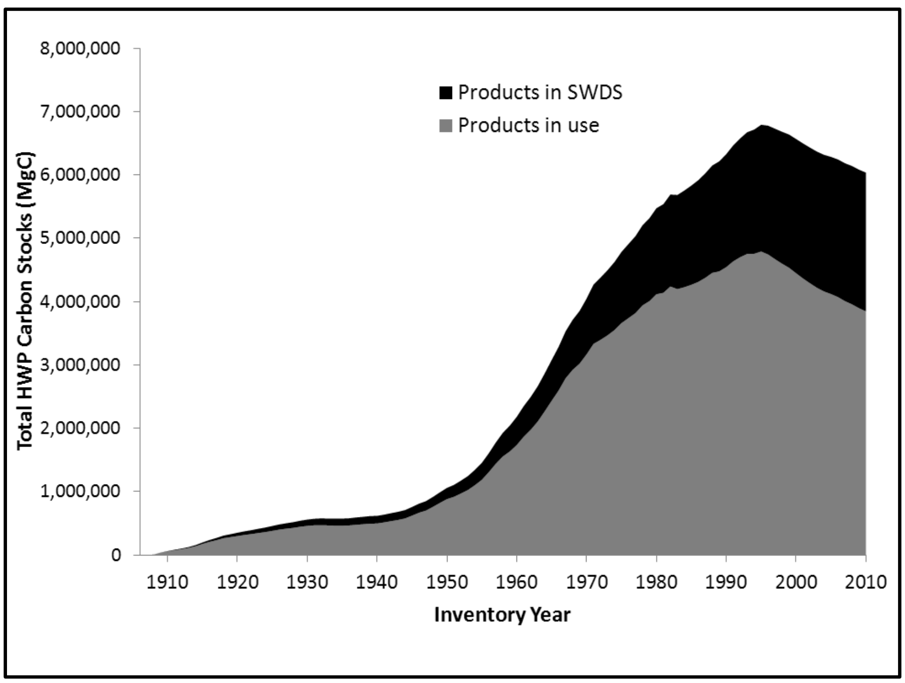

Figure 4-8. Cumulative total carbon stored in harvested wood products (HWP) manufactured from Nez Perce-Clearwater National Forests. Figure from Anderson et al. (2013).

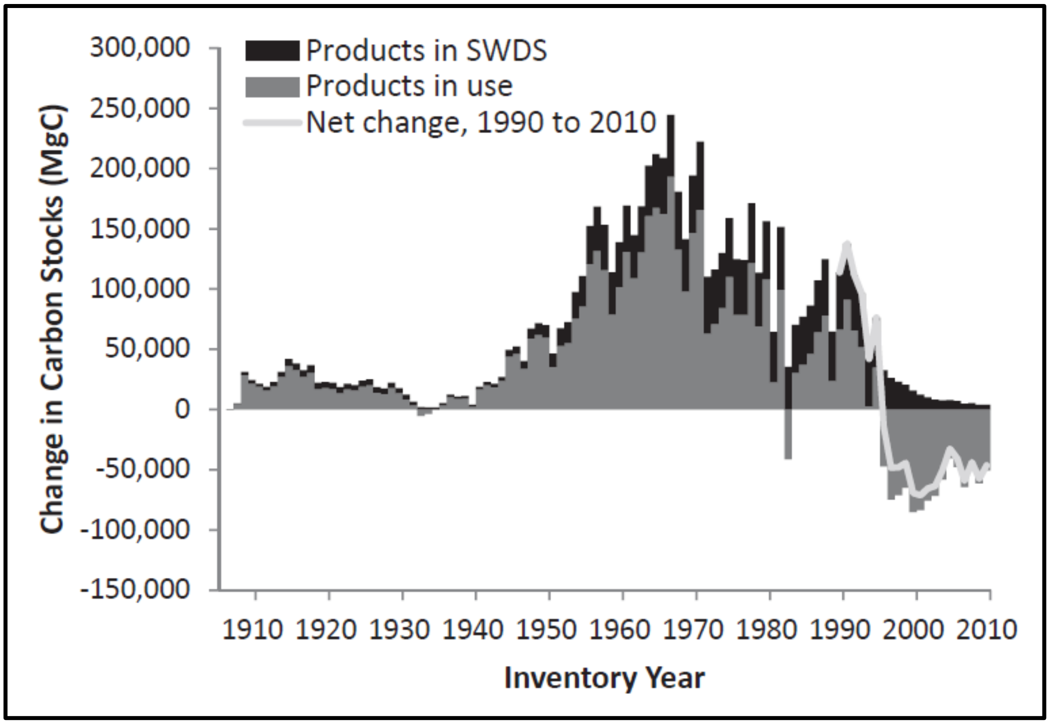

Figure 4-9. Net change in harvested wood products (HWP) carbon stocks from the previous year. (from Anderson et al. 2013)

Projected Trends in Forest Carbon Stocks and Flux

The future of the terrestrial carbon sink of western U.S. forests is uncertain due to the multiple interacting factors that influence carbon stocks and fluxes (Lenihan et al. 2008a; Ryan et al. 2008; Birdsey et al. 2007). These factors include climate variability and change; potential positive effects of increased atmospheric CO_2 concentrations on plant productivity; frequency, duration, and severity of moisture stress; changes in the rate and severity of natural and human disturbances; and land management practices (Canadell, Pataki et al. 2007).

Projections of the future of the U.S. carbon sink, based on national trends in land-use change and fire suppression, indicate that the U.S. carbon sink will decline over the 21st century due to a slowing of ecosystem recovery from 19th century land use and vegetation response to 20th century fire suppression (Hurtt et al. 2002). This analysis, which does not include projected climate changes, also concluded that U.S. forests would convert to a large carbon source if fire suppression is ineffective in the 21st century.

Modeling experiments based on projected changes in climate, but not land use, suggest that the future strength of the U.S. carbon sink is very sensitive to the degree of change in climate, particularly precipitation, and fire regimes (Bachelet et al. 2001; Lenihan et al. 2008a,b). If precipitation and temperature increases are small or moderate, net ecosystem productivity and carbon stocks are expected to increase. Conversely, if climate changes result in decreased precipitation and soil moisture during the growing season, net ecosystem productivity is expected to decline due to drought stress and may result in a net carbon source to the atmosphere (Lenihan et al. 2008a,b). Increasing concentrations of atmospheric CO_2 may moderate these impacts by enhancing vegetation productivity and water use efficiency (Bachelet et al. 2001;

Joyce and Nungesser 2000; Lenihan 2008a,b), at least up to a point where nutrient limitations and increasing temperatures overwhelm the beneficial effects of CO_2 concentrations (Fishlin et al. 2007). Increases in annual area burned may further reduce net ecosystem productivity and carbon stocks despite the potentially positive effects of increasing CO_2 concentrations (Lenihan et al. 2008a,b).

Empirical analyses of the growth rates of trees in the Pacific Northwest demonstrate the potential impacts of climate change on forest productivity and reveal that high- and low-elevation forests respond differently to climate variability. Seasonal photosynthesis ("carbon uptake period") and annual growth rates of high-elevation forests (e.g., subalpine fir, mountain hemlock, and high-elevation lodgepole pine and Douglas-fir) are commonly limited by a relatively short growing season, low soil temperatures, and long periods of snowcover (Littell et al. 2008; Chin et al. 2008; Case and Peterson 2007; Case and Peterson 2005; Peterson et al. 2002). Growth rates increase in these high-elevation forests during years with earlier spring snowmelt, abnormally warm annual temperatures, and longer growing seasons. These results suggest that projected changes in regional climate will likely lead to increased productivity and carbon stocks of high-elevation forests.

Conversely, growth rates of lower- and mid-elevation ponderosa pine, Douglas-fir, and lodgepole pine forests of the Pacific Northwest and Northern Rockies tend to be limited by low growing-season precipitation and high growing-season temperatures (Littell et al. 2008; Case and Peterson 2007; Case and Peterson 2005; Watson and Luckman 2002). During these conditions, the rate of water loss from evapotranspiration is greater than the rate of water absorption by roots, resulting in water stress (Case and Peterson 2007). Prolonged periods of water stress significantly reduce a tree's ability to photosynthesize (Kozlowskie and Pallardy 1997). As a result, climate projections with increased frequency of reduced snowpack, earlier spring snowmelt, increased temperatures during the growing season, and little or no significant increase in summer precipitation likely will result in reduced forest productivity and carbon sequestration in low- and mid-elevation forests of the Pacific Northwest and Northern Rockies (Boisvenue 2007; Boisvenue and Running 2010). Recent research suggests that regional warming and water balance deficit trends over the late 20th century are contributing to rapid and widespread increases in mortality rates and slight decreases in forest density and basal area in old-growth forests throughout the western United States (van Mantgem et al. 2009).

In addition to the gradual changes in forest productivity and carbon stocks resulting from directional climate change, episodic events such as large high-severity fires and large-scale insect outbreaks can significantly affect carbon stocks and flux of forest ecosystems. In the short term (decades), disturbances can convert regional carbon sinks to a carbon source (Kurz, Stinson, and Rampley 2008; Kurz, Stinson, Rampley et al. 2008; Kurz, Dymon et al. 2008). Over the long term (centuries), the effects of disturbances on the regional carbon balance are neutral assuming similar vegetation regrows on the disturbed area and the long-term frequency and severity of disturbances does not change (Kashian et al. 2006; Canadell, Pataki et al. 2007). The potential fertilization effect of atmospheric CO_2 concentrations may influence the rate of terrestrial carbon recovery (Lenihan et al. 2008b; Balshi et al. 2009). One recent study of ponderosa pine stands in western Montana and eastern Idaho concluded that recent increases in atmospheric CO_2 concentrations increased growth rates in older trees (Knapp and Soulé 2010).

On the Forests, carbon stocks and flux rates will vary over coming decades in response to complex and uncertain interactions between climate variability and change, forest age class distribution, disturbance-recovery processes, and possible effects of CO_2 concentrations on forest productivity (Hyvönen et al. 2007; Smithwick et al. 2008). The contribution of forest regrowth from past disturbances is expected to decline as the maturing forests grow more slowly and take up less CO_2 from the atmosphere. Projected climate changes for the region suggest that relatively high-elevation forests may increase in productivity and carbon sequestration, whereas these processes may decline in low- and mid-elevation forests with south and southwesterly aspects. Potential increases in the frequency and size of high-severity fires, bark beetle outbreaks, and root disease occurrence could also significantly impact the carbon budgets of these forests over the 21st century. Extensive high-severity fires, large scale tree mortality from bark beetles, and productivity losses due to root diseases could convert the Forests from a net carbon sink to a carbon source for several decades (Bond-Lamberty et al. 2007; Kurz, Stinson et al. 2008; Kurz, Dymon et al. 2008). In addition, timber harvesting will affect the amount of ecosystem carbon stored and the short-term net flux of carbon within the atmosphere. However, the net contribution to atmospheric CO_2 concentrations resulting from fire, insect-caused tree mortality, and timber harvest is expected to be approximately zero over the long term as long as disturbed areas regenerate with similarly productive species and the disturbance frequency and intensity does not change (Kashian et al. 2006).

Using harvested forest biomass will continue to store carbon in wood products and landfills (EPA 2008; Skog 2008; Skog and Nicholson 2000; Skog and Nicholson 1998) and may reduce the demand for more fossil fuel-intensive products such as steel and cement (Pérez-Garcia et al. 2005; Malmsheimer et al. 2008). In addition, emerging markets in forest biomass for use in energy production could offset fossil fuel emissions (Malmsheimer et al. 2008; Nicholls et al. 2009).

Climate change is expected to have significant impacts on soil carbon dynamics (Schils et al. 2008; Conant et al. 2011). Rising atmospheric CO_2 levels could increase biomass production and inputs of organic materials into soils. However, increasing temperatures could reduce SOC by accelerating the microbial decomposition and oxidation of SOM (Victoria et al. 2012). Soil carbon has the potential to affect carbon sequestration in the forested ecosystem. Soil carbon can be broken into three pools based on storage time. The active pool contains litter and fine roots with an anticipated turnover time of days up to a year (Beedlow et al. 2004). The intermediate pool, with turnover times of years and decades, is the largest pool. A variety of carbon forms are found in the intermediate pool (Beedlow et al. 2004). The third pool is the passive pool. The passive pool includes carbon that persists in the soil for more than a century in forms such as humus (Beedlow et al. 2004). While the intermediate pool is the most likely pool to play into carbon sequestration, maintaining the passive pool may be the most important for long-term carbon sequestration (Beedlow et al. 2004). Over long periods of time carbon storage in the soil varies mainly as a result of climatic, geological, and soil-forming factors; while over shorter periods of time, it is mainly vegetation disturbances or succession, and changes in land use patterns that affect carbon storage (Batjes 1996).

4.1.4 *Key Sources of Uncertainty*

4.1.4.1 Changes in Climate

Net ecosystem productivity is very sensitive to changes in temperature, precipitation, soil moisture, and other climate characteristics (Angert et al. 2005; Piao et al. 2008; Piao et al. 2009). Climate change also has a significant impact on the extent and severity of wildland fires, population dynamics of bark beetles and other forest insects, moisture stress on trees, and other disturbance processes. All global climate models project surface temperature warming in the Northern Rockies. Average annual temperatures are expected to increase by +1.5 degrees Fahrenheit (°F) to 5.9 °F by the 2040s, depending on the rate of greenhouse gas emissions. These projected temperature increases exceed observed 20th century year-to-year variability. Annual mean temperature could change by –10% to +20% by the 2040s. Many climate models project increases in precipitation during the winter and decreases in summer; however, projected precipitation changes are comparable to 20th century variability. These regional climate projections suggest increasing water deficits for forests, which increases tree stress and mortality, tree vulnerability to insects, and fuel flammability. The severity of these potential climate change effects remains somewhat uncertain at local scales.

4.1.4.2 Disturbance Regimes

High-severity disturbance events have a substantial and rapid impact on forest carbon stocks and flux. Persistent changes in the frequency, extent, and severity of disturbances can alter long-term (decades or longer) regional net carbon balances. Yet, knowledge of the future trajectory of wildfires, insect outbreaks, drought severity and duration, and other major forest disturbances is limited. The available scientific evidence suggests the average annual area burned by wildfires is likely to increase in coming decades in the Pacific Northwest and Northern Rockies. Similarly, higher temperatures and water stress may increase the susceptibility of trees to bark beetles and other insect and pathogens. Available scientific information suggests that the risks of bark beetle mortality may increase in higher elevation forests. The uncertainty in these projections, however, is greater at finer spatial scales.

4.1.4.3 CO_2 Fertilization

CO_2 is a fundamental building block of photosynthesis. Trees and other plants grown in elevated CO_2 environments have increased growth rates, productivity, and water use efficiency compared to controls (Norby et al. 2005). Thus, some evidence indicates that increasing atmospheric concentrations of CO_2 may increase forest productivity. However, the results of these controlled experiments have not been widely confirmed in natural environments (Knapp and Soulé 2010). Additional studies have suggested that the potential CO_2 fertilization effect is limited to young plants and by water and nutrient availability (particularly nitrogen) (Norby et al. 2010). In addition, some evidence exists that trees and other plants acclimate to elevated CO_2 concentrations over time, thus reducing the duration of the potential fertilization effect. In sum, considerable uncertainty exists about the potential of elevated CO_2 concentrations to increase net ecosystem productivity, carbon storage, and the carbon sink strength of forests. Also models predict that many forests will be impacted by a combination of extreme weather events, insects, diseases, and ozone; therefore growth and carbon capture may not be accelerated in most instances (Karnosky et al. 2007; Scheller and Mladenoff 2005; Frelich and Reich 2009).

4.1.4.4 Potential Changes in Forest Composition

Long-term projections of regional net carbon balances depend on assumptions about the future vegetation composition of currently forested areas (Kashian et al. 2006; Canadell et al. 2007). In coming decades, climatically suitable habitat for many tree species may shift from their current locations (Rehfeldt et al. 2006). Some models suggest that changes in climatically suitable habitat combined with amplified disturbance regimes may result in some forests of the Northern Rockies converting to non-forest vegetation (Westerling et al. 2011). However, considerable uncertainty exists regarding the effects of climate change on the composition of forest vegetation. These uncertainties in future forest composition and structure contribute to the uncertainty in long-term projections of forest carbon stocks and flux and regional net carbon balances (Smithwick et al. 2008; Rhemtulla et al. 2009).

4.1.4.5 Biomass Utilization

Using woody biomass for energy production and as a substitute for more greenhouse gas intensive materials (e.g., steel and cement) has the potential to provide substantial global carbon benefits (Nabuurs et al. 2007). However, the capacity to realize these potential carbon benefits is uncertain due to current technological limitations, social and political issues, and the reliability of feedstock supplies. At regional and local scales, limited and declining capacity in the wood products industry adds further uncertainty to projections of the size of the carbon pool in harvested wood products and the use of woody biomass to displace fossil fuels. While many short-term studies showed no negative effect of harvest residue removal on growth (Roberts et al. 2005), it is possible that negative growth impacts occur in the long term. This has been shown in Northern Sweden for whole-tree harvesting in Scots pine stands on nutrient-poor sites, where growth declines were revealed only 12–24 years after harvesting (Egnell and Valinger 2003). Therefore, utilizing forest harvest residues on poor sites could be detrimental to site productivity and long-term soil carbon storage without compensatory fertilization (Sverdrup and Rosen 1998; Richardson et al. 2002; Raulund-Rasmussen et al. 2008). With a doubling of biomass removals in intensive biomass harvesting, the nutrient removal may increase by 6 or 7 times (Raulund-Rasmussen et al. 2008). Even on more fertile soil types, it is beneficial to retain foliage on the site (Samuelsson 2002). Thus it is beneficial to exclude small branches and foliage from the biomass removals by extracting dry residues in the case of coniferous species (to allow needles to drop before chipping) (Richardson et al. 2002). When foliage and roots are removed as well (i.e., in whole tree harvesting and stump extraction), detrimental impacts can occur, especially on nutrient-poor sites. More research is needed to reveal whether wood ash recycling or conventional fertilization will be sufficient to sustain long-term site productivity under such conditions by replenishing the exported nutrients (Raulund-Rasmussen et al. 2008).

4.1.5 *Potential Mitigation Options*

The recent Intergovernmental Panel on Climate Change report identifies four general categories of options to reduce emissions by sources and/or increase carbon sequestration by:
(1) maintaining or increasing forest area; (2) maintaining or increasing site-level carbon density; (3) maintaining or increasing landscape-level carbon density; and (4) increasing off-site carbon stocks in wood products and enhancing product and fuel substitution (Nabuurs et al. 2007).

4.1.5.1 Land Exchange

Occasionally, the Forests have the opportunity to exchange lands with willing landowners. Where land exchanges result in a net increase in forest productivity or net forested acres within the National Forest System, they may maintain or increase the area of productive forests.

4.1.5.2 Prompt Regeneration of Disturbed Areas

Rapid tree planting in areas severely disturbed by wildfire can accelerate carbon accumulation, and thus increase stand- and landscape-level carbon density over time. An evaluation of management options to modify the net carbon balance of Canadian forests found that the potential for increasing the forest carbon sink strength was largest when regeneration occurred quickly after natural disturbances (Chen et al. 2000). On the Forests, natural regeneration is often, but not always, successful over time. The interior of high-severity burn patches are most prone to long-delayed tree regeneration. In these areas, rapid post-fire tree planting may accelerate forest development and carbon accumulation. However, such treatments are costly and may be financially infeasible (Chen et al. 2000).

4.1.5.3 Extended Rotations

Several commentators have suggested that increasing timber harvest rotation length can produce global carbon benefits by increasing forest carbon storage (Birdsey et al. 2007; Nabuurs et al. 2007; Ingerson 2007; Leighty et al. 2006; Birdsey et al. 2000). In concept, increasing rotation ages can increase stand- and landscape-scale carbon storage by holding more carbon in forests and avoiding emissions from harvesting. However, several factors suggest that achieving carbon benefits from extended rotations may be problematic.

Extended harvest rotations focused on specific ownerships, forests, and regions will reduce annual timber harvest levels and wood products production in the affected area. Such local and regional reductions would likely be offset by market-driven harvest increases by other timberland owners in other regions. For example, more than 85% of the reductions in timber harvest levels on western federal forests in the late 1980s and 1990s were replaced by increased harvest by other timberland owners and regions, including international imports (Wear and Murray 2004; Murray et al. 2004). As a result of this "leakage," little or no net effect would be likely on the national or global terrestrial carbon balance, and no net effect on atmospheric concentrations of CO_2, as a result of increasing rotation lengths on the Forests. In addition, increased lumber prices resulting from timber sale reductions (Wear and Murray 2004) could lead to increased use of more energy-intensive materials (e.g., steel and cement), and net increases in greenhouse gas emissions from fossil fuel combustion.

Extending rotation ages also increases exposure of landscape-scale carbon stocks to high-severity disturbances, such as wildfires, (Kurz, Stinson, Rampley et al. 2008) and may even increase the probability of bark beetle outbreaks (Kurz, Dymond et al. 2008). In fire-prone areas, such as the Forests, the probability that the theoretical carbon storage benefits of extended rotations will be substantially reduced is increased. Thus, the carbon storage benefits may not persist or be sustainable for extended periods. Recent analysis indicates that the risk of carbon loss due to wildfire is higher on the Forests than most other forested areas of the United States (Hurteau et al. 2009).

4.1.5.4 Fire Suppression

Several authors have suggested that continued or increased fire suppression effort can help maintain or increase landscape-level carbon density and storage in U.S. forests (Birdsey et al. 2007; Nabuurs et al. 2007; Birdsey et al. 2000). However, fire management strategies to increase forest carbon storage must consider both the amount of carbon stored and the stability of that storage as climate and fire regimes change (Schimel 2004; Schimel and Braswell 2005).

Aggressive fire suppression can limit the number and size of large fires and, therefore, may increase forest carbon storage and sink strength, at least for the short term. However, these carbon storage gains are unlikely to be sustained over time. Since 1986, the number of large forest fires in the Northern Rockies increased more than tenfold (1,100%); and the area burned by large fires increased more than threefold (350%) compared to the period 1970 to 1985 (Westerling et al. 2008). Numerous simulations of the effects of projected climate change on wildfire in western North America all indicate a rising probability of increased annual area burned and increased frequency of high-severity fires (Westerling and Bryant 2008; Nitschke and Innes 2008; Bachelet et al. 2007; McKenzie et al. 2004; Brown et al. 2004). If observed trends continue or if the projected changes in fire regimes are even partially realized, aggressive fire suppression is likely to lead to most acres burning in fewer, more extreme, and unmanageable events with greater losses of forest carbon stocks (Hurteau et al. 2008). Thus, it is likely that, at best, the carbon benefits of aggressive fire suppression are temporary, not permanent, and may even result in greater greenhouse gas emissions from fires and loss of forest carbon stocks than would occur with less aggressive fire suppression (Kirschbaum 2006; Breshears and Allen 2002).

4.1.5.5 Using Biomass for Energy Production

According to the Intergovernmental Panel on Climate Change (IPCC), "When used to displace fossil fuels, woodfuels can provide sustained carbon benefits, and constitute a large mitigation option" (Nabuurs et al. 2007). A recent study estimates that U.S. forests are capable of sustainably producing 368 million dry tons of wood per year, with 41 million dry tons from currently unused logging residues and 60 million dry tons from hazardous fuel treatments (Perlack et al. 2005). If applied to bioenergy production, this wood residue could offset a substantial percentage of U.S. CO_2 emissions from fossil fuels (Richter et al. 2009).

In addition to ongoing energy production from milling byproducts at area wood-processing facilities, several opportunities exist to use wood residues from timber harvest, hazardous fuel reduction projects, and other silvicultural treatments on the Forests. These opportunities include Avista Corporation's Bioenergy Plant in Kettle Falls, Washington; several area pellet plants; and area schools and other facilities with high-efficiency wood heating systems. Potential exists for a substantial increase in wood energy production in central Idaho that could replace CO_2 emissions from fossil fuels while also reducing CO_2 emissions from pile burning and other forest residue treatments. However, while many short-term studies showed no negative effect of harvest residue removal on growth (Roberts et al. 2005), it is possible that negative growth impacts occur in the long term. Intensively managed plantations have nutrient demands that may affect soil fertility and soil properties, leading to higher erosion of the uncovered mineral soil surface (Pérez Bidegain et al. 2001; Carrasco-Letellier et al. 2004); they may also influence biological properties changes (Sicardi et al. 2004) if the choice of species is not properly matched with site conditions.

Biomass-derived, granular charcoal with high carbon content is known as "biochar". It can store carbon in soils, but may not necessarily be stable over long time periods; its longevity depends on combustion temperatures during pyrolysis and interactions with soil minerals (Schmidt et al. 2011). Application of charcoal to soils is hypothesized to increase bioavailable water, build soil organic matter, enhance nutrient cycling, lower bulk density, act as a liming agent. It also reduces leaching of pesticides and nutrients to surface and ground water. The half-life of carbon in soil charcoal is in excess of 1,000 yrs. Hence, soil-applied charcoal will make both a lasting contribution to soil quality; and carbon in the charcoal will be removed from the atmosphere and sequestered (Laird 2008).

4.1.5.6 Mitigation Options for Soil Carbon Sequestration

Several strategies can maintain or increase forest soil carbon storage: maintain site productivity, avoid soil disturbance, use forest management practices that store more carbon, and avoid catastrophic mortality by establishing species diversity. Maintaining site quality is essential to ensure that forests continue to capture and store carbon at their maximum capacity (Gough et al. 2007). Factors that impact productivity, such as decreased soil fertility, stressors, or loss of productive area to permanent roads and landings, will reduce the potential to store carbon. Avoiding soil disturbance will limit erosion losses, minimize conditions that lead to increased SOM decomposition, and allow stable organo-mineral complexes to form (Jandl et al. 2007). Recommended forest management practices to store more carbon include increasing average tree diameter and height; allowing forests to become older (i.e., use extended rotations); maintaining full stocking; decreasing the frequency of harvests; and retaining dead woody debris on site (Ray et al. 2009; Nunery and Keeton 2010; Ryan et al. 2010). Avoiding catastrophic mortality and establishing a mixed species forest go hand in hand (Jandl et al. 2007; Ray et al. 2009; Nunery and Keeton 2010). Fuel treatments exchange current carbon storage for the potential of avoiding large carbon losses in wildfire (Ryan et al. 2010). Forest management activities to increase stand-level forest carbon stocks include harvest systems that maintain partial forest cover. These harvest systems also minimize losses of dead organic matter (including slash) or soil carbon by reducing soil erosion, and by avoiding slash burning and other high-emission activities (Nabuurs et al. 2007).

Possible additional strategies, that require further study, are to manipulate forest species to favor those that allocate more carbon belowground (Jandl et al. 2007) and increase the growth rate of existing forests through silvicultural methods, fertilization, water management, and/or the use of different tree species (Jandl et al. 2007; Ryan et al. 2010).

4.1.5.7 Summary of Mitigation Options

At the global scale, preventing large-scale conversion of forests to other land uses (deforestation), primarily in the tropics, provides the greatest opportunity to mitigate the trend of increasing atmospheric concentrations of CO_2 (Nabuurs et al. 2007). In the United States, the largest and most effective mitigation opportunity has already been taken—creating State and federal public forests that share the common objective of “keeping forests as forests” in perpetuity.

Within the context of public forests, individual land management actions are unlikely to have significant long-term effects on the atmospheric concentrations of CO_2 and other greenhouse gases. Without a substantial reduction in fossil fuel emissions, the impacts of projected climate change on disturbance regimes and species composition will likely overwhelm the short-term effects of land management actions. As the IPCC concluded, “In the long term, a sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual sustained yield of timber, fiber or energy from the forest, will generate the largest sustained mitigation benefit” (Nabuurs et al. 2007). However, there is still limited insight regarding impacts on soils, lack of integrated views on the many site-specific studies, hardly any integration with climate impact studies, and limited views in relation to social issues and sustainable development. Little new effort was reported on the development of global baseline scenarios of land-use change and their associated carbon balance, against which mitigation options could be examined. Quantitative information on the cost-benefit ratios of mitigation interventions is limited. Finally, knowledge gaps still exist in terms of how forest mitigation activities may alter, for example, surface hydrology and albedo (IPCC 2007).

Forestry mitigation projections are expected to be regionally unique, while still linked across time and space by changes in global physical and economic forces. Overall, it is expected that boreal primary forests will either be sources or sinks depending on the net effect of some enhancement of growth due to climate change versus a loss of SOM and emissions from increased fires. The temperate forests in the United States, Europe, China, and Oceania, will probably continue to be net carbon sinks, favored also by enhanced forest growth due to climate change. In the long-term, carbon will only be one of the goals that drive land-use decisions. Within each region, local solutions have to be found that optimize all goals and aim at integrated and sustainable land use. Developing the optimum regional strategies for climate change mitigation involving forests will require complex analyses of the trade-offs (synergies and competition) in land use between forestry and other land uses; the trade-offs between forest conservation for carbon storage and other environmental services such as biodiversity and watershed conservation and sustainable forest harvesting to provide society with carbon-containing fiber, timber, and bioenergy resources; and the trade-offs among utilization strategies of harvested wood products aimed at maximizing storage in long-lived products, recycling, and use for bioenergy (Nabuurs et al. 2007).

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