Fire and Fire Suppression Impacts on Forest Soil Carbon

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Introduction

The potential of forest soils to sequester carbon (C) depends on many biotic and abiotic variables, such as: forest type, stand age and structure, root activity and turnover, temperature and moisture conditions, soil physical, chemical, and biological properties (Birdsey – Chapter 2, Johnson and Kern – Chapter 4, Pregitzer – Chapter 6, Morris and Paul – Chapter 7). Of increasing interest to U.S. and global soil C sequestration scenarios is the impact various forest management practices, such as harvesting, site preparation, reforestation, drainage, and fertilization, have on soil C pools and cycling (Post – Chapter 12, Hoover – Chapter 14). A subject of many recent studies is the possible effect of higher atmospheric CO_2 levels on forest soil C accumulations caused by increased tree growth, changing internal C allocations, and alteration of climate temperature and precipitation patterns (e.g. Caspersen et al., 2000). Projected climate change may also increase the incidence and severity of wildfires in some forest regions, which could have a major impact on soil C pools (Flannigan et al., 2000; Stocks et al., 2000).

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Confounding the possible impact of climate change on soil C pools is the widespread interest in using fire to restore perceived and real changes in forest productivity caused by fire suppression activities (Kimmins, 1977; Ballard and Gessel, 1983; Vose and Swank, 1993; Monleon et al., 1997). Much of the concern is focused on the relationships of fire, fire suppression, and nutrient losses and gains (Freedman, 1981). In particular, active fire suppression in forest stands that historically supported a regular fire return interval has led to C accumulation in many forest stands, especially in the western U.S.A. (Oliver et al., 1994). These accumulations are likely undesirable because of slower decomposition rates and the risk of increased fire severity (Covington and Sackett, 1984). Consequently, in this chapter we will discuss the impact of fire on soil C pools, recovery after fire, the effects of a fire suppression policy on soil C, methods to estimate C losses from fire, and the implications of fire management on soil C cycling and sequestration.

Soil Carbon Losses from Fire

Any fire alters the amount and distribution of C pools in forest soil (Wells et al. 1979). The amount of soil C lost due to fire depends on: (1) temperature of the burn, (2) amount and distribution of surface soil organic matter (OM), and (3) decomposition rates of residual OM after fire.

Fire temperature

Fire temperature, expressed as maximum ground temperature, is an important variable in soil C loss. Fire severity is usually described in three classes: light – where surface temperatures are around 250^{o} C, moderate – surface temperatures reach 400^{o} C, and high – surface temperatures in excess of 675^{o} C

(Neary, et al., 1999). Surface OM and C are consumed when fire temperatures reach 450 °C. Fires occurring in areas with heavy slash build-ups may produce ground temperatures of 500-700 °C, but can reach 1500 °C (DeBano et al., 1998). At these temperatures surface and subsurface organic matter (to a depth of 5 cm) can range from partially scorched (light) to totally consumed (high).

Most soil C losses occur when fires burn in one area (e.g. around downed logs) for a long period of time and transfer heat into the mineral soil (Neary et al., 2000). However, heating the forest floor at temperatures ranging from 175° C and 205° C for ≈20 minutes or longer can result in the formation of volitiles, which diffuse downward into the surface mineral soil. These organic compounds cool, condense on soil particles, and form water-repellant soil layers when mineral soil moisture content is less than 12-25% (Robichaud and Hungerford, 2000). Although extremely variable, measurements of hydrophobicity persistence in soil have shown that this layer weakens after the first year, but can persist for at least 2 years (Huffman et al. 2001; Pierson et al. 2001). It is unclear how much forest floor C moves into the mineral soil by this process, but could be appreciable in heavily burned areas.

Amounts and Distribution of Soil OM

Fuel amounts and distribution vary greatly by forest type and stand age (Johnson and Kerm – Chapter 4). Many variables contribute to the range of C estimates, such as vegetation type, stand age and structure, organic matter decomposition rate, and whether forest floor and surface woody residues are included with the mineral soil pool (Oliver and Larson, 1990). Total profile C pools are generally the highest in late seral/climax stages, when tree mortality becomes an important part of the soil C cycle (Harvey et al., 1999a). For old-growth stands in the Northwest USA, mineral soil C ranges from 17-151 Mg C ha^{-1} and surface organic horizons range from 48-246 Mg C ha^{-1} (Table 1). In many of these stands, the proportion of C in the surface organic horizons is greater than 50% of the total pool.

Generally, surface fuels are high right after a stand has been harvested, experienced severe blowdowns, or ice damage, and slowly decrease as these residues decompose with little additional organic matter input (20 – 70 years). In contrast, surface fuel accumulation after a severe wildfire is usually the opposite, with little surface organic material remaining after the fire. However, fuel levels increase 10–30 years later as snags of trees killed by the fire finally fall and litter inputs from the reestablished understory communities greatly increase (Covington and Sackett, 1992; Baird et al., 1999). Surface accumulations of dead and partially decayed coarse woody debris greatly increase the risk of another severe fire, which would deplete soil C pools even further (Oliver et al., 1994).

Organic Matter Decomposition

While most soil C is lost as a direct result of the combustion process, changes in the decomposition rate of soil organic matter can also play a key role in C storage. A number of studies have shown that decomposition rates of fresh litter placed on burned soil can range from nearly the same to 50% less, compared to unburned soil (Monleon and Cromack, 1996). However, organic matter remaining in the forest floor after a fire has been shown to decompose very rapidly, even though needles and leaves from both residual dead and live trees added considerable mass (Covington and Sackett, 1984). Krankina and Harmon (1994) estimate that it takes 0.5-1 year to decay the amount of organic matter lost when burning destroys 10% of the forest floor.

The increase in decomposition of residual soil organic matter is likely from increased microbial activity caused by warmer temperatures and more soil moisture in the newly opened stand (Harvey, 1994). There is also an increase in surface soil pH following fire as a result of basic cations released through the combustion process. Jurgensen et al. (1981) found soil acidity of the organic horizons had decreased over 2 pH units immediately following a fire in Montana, and had not returned to pre-fire levels after 4 years. The pH change varies depending on vegetation type, soil organic matter content, and precipitation (Grier, 1975), but the more severe the burn, the greater the soil pH increase. Both soil bacteria and fungi populations change in response to lower soil acidity levels, and are likely responsible for much of the increased organic matter decomposition (Pietikäinen and Fritze, 1995; Visser and Parkinson, 1999), Greater available N after fire would also contribute to greater decay rates (Raison, 1979).

Soil Carbon Changes

Pools after Fire

Many wildfires burn hotter than most prescribed fires and are associated with greater losses of soil C (Johnson, 1992; Table 1). Wildfires, which burn hundreds to thousands of hectares, have a larger potential for seriously affecting soil productivity than prescribed burns because soil and fuel moisture is usually lower and the weather is drier (McNabb and Cromack, 1990). However, frequent low-intensity prescribed burns can also cause significant surface and mineral soil

C losses (McKee and Lewis, 1983; Monleon and Cromack, 1996). Fire burning over a landscape often creates variable C losses depending on vegetation types (Table 2), but these estimates often do not include C losses in downed, intact logs.

In undisturbed forests, coarse woody debris can contribute another 5-30% to the C pool (Laiho and Prescott, 1999). Covington and Sackett (1984) noted that woody material was reduced by 63% after a prescribed burn in Arizona. On the eastern slopes of the Cascade Mountains in Washington, coarse woody debris mass declined 33% in a ponderosa pine (*Pinus ponderosa*) and Douglas-fir *(Pseudotsuga menziesii*) forest after burning (Baird et al. 1999). Similarly, after a burn in a high-elevation lodgepole pine (*Pinus contorta*) forest, coarse woody debris was reduced 80% (Lopushinsky et al., 1992). Variation in the amount of wood, decay stage, and intensity of fire will influence C storage in this forest component (Baird et al., 1999). Additionally, formation of charcoal in the surface soil will also enhance long-term C storage, and affect soil microbial populations (McKee, 1982; Pietikäinen et al., 2000).

Fire Suppression

Frequent, low-intensity fires that occur in fire-dependent ecosystems, especially in the western USA, limit the amount of OM accumulation in the forest floor, and have little impact on overall soil productivity (Neary et al., 1999). However, large amounts of fine and coarse fuels accumulate on the soil surface of fire suppressed stands. These stands are at risk for substantial C losses and decreased site productivity when an infrequent, high-intensity fire occurs (Harvey et al., 1994; Oliver et al., 1994; Harvey et al., 1999b). For example, a ponderosa pine stand in Idaho with no fire for 87 years had 57% of the total profile C pool in the forest floor as compared to 20% in a stand where fires are frequent (10-15 yr) (Table 2; Figure 1; Neary et al., 2000). In the Pacific Northwest, USA many forested stands have a majority of the total profile C pool associated with these organic horizons because of fire suppression activities dating back to the early 1900's (Oliver et al, 1994). Grigal and Ohmann (1992) found an increase in both forest floor and mineral soil C pools on fire-suppressed upland sites in the upper Great Lakes region. However, Tilman et al. (2000) reported no soil C gains in a Minnesota oak-savana ecosystem, even though fire suppression cause nearly twice as much C to accumulate in above ground biomass than in stands which had a moderate-high fire frequency.

Fire suppression policies have also allowed many forest types in the Intermountain western U.S. to gradually shift from more open stands of fire-dependent ponderosa pine to closed stands comprised of a mixture fire-intolerant climax species, such and Douglas-fir and grand fir (Harvey et al., 1999a and b). Pine stands generally have a greater proportion of soil C in the surface mineral soil, while mixed pine/fir stands have more C in the forest floor and in coarse woody debris (Table 2). Higher C contents in surface organic materials resulting from fire suppression, increases the potential for greater C losses when a fire finally does occur. The development of a fir understory ladder-fuel complex in pine stands also increases the possibility of a stand-replacing fire, and consequently greater C losses (Harvey et al., 1994). Site productivity and recovery is often dependent on the amount of surface organic matter remaining after stand disturbance (Jurgensen et al., 1997).

Predicting Soil C Loss

As seen in Table 2, there many reports on soil C losses from site-specific fires. However, in order to incorporate C losses from fire into soil C sequestration predictions and models, estimates are needed of soil C losses from many forest types, age classes, etc. for which no fire-loss information is available. Without such estimates, prediction of which stands are at greatest risk from catastrophic fires, changes in soil C pools over larger scales, and forest ecosystem responses to fires of various intensities are difficult to formulate. One method would be to use published soil survey information and fire criteria formulated in the USDA Forest Service Regional Soil Quality Standards and Guidelines. These standards and guides were designed to measure the possible impact of management on soil productivity, site resiliency, and long-term productivity (Jurgensen et al., 1997; Johnson and Todd, 1998). Although specific thresholds were written for each USDA Forest Service Region and for several different disturbance regimes, these fire standards can be used to estimate the range of possible soil C losses from fires of different severities (Page-Dumroese et al., 2000).

Several examples of this type of calculation are shown for representative forest stands in the Pacific Northwest (Table 1). The amounts and distribution of soil C in western old-growth stands shows the wide variability in soil C pools among different forest ecosystems, and can be used as an upper limit of C losses after a severe, stand-replacing fire. We followed the methods outlined in Page-Dumroese et al. (2000) and used two possible burning scenarios: (1) 15% of the area was burned severely enough to reach the USDA Forest Service detrimental fire threshold, and (2) 100% of the soil surface was burned severely enough to reach the USDA Forest Service detrimental fire threshold for that forest.

As shown in Table 1, most of the C in the forest floor would be destroyed in the event of a severe, stand-replacing fire, while mineral soil losses likely would be relatively small. Even though sitka spruce stands on the Oregon Coast have large accumulations of soil C, their fire frequency is very low (>900 yrs) and hot, stand-replacing fires rarely occur (Agee, 1993). Consequently, the probability of losing large amounts of soil C to fire in this ecosystem is small. However, other forests with higher fire-return intervals, such as ponderosa pine, have much smaller C pools, but would sustain much greater short-term ecosystem losses of C. Stands with more C in the mineral soil (hemlock in Montana – 60%) are at a much lower risk for catastrophic C losses than those with a larger proportion in the surface organic matter (grand fir in Idaho with 33%).

Soil C Recovery after Fire

We have shown that fire can reduce soil C pools depending, on fire severity. Equally important to forest ecosystem health and productivity is how quickly soil C pools recover after fire. Severe fires often result in long-lasting C declines, while low-intensity fires may result in little C change, and sometimes an increase in site productivity (Baird et al., 1999). For example, 24 yrs after an intense burn in a radiata pine stand in Australia, mineral soil C was still 40-50% lower than unburned controls (Sands, 1983). However, surface organic matter can recover much more rapidly. Visser and Parkinson (1999) reported forest floor recovery 41 years after wildfire in Alberta jack pine stands.

A study established in western Montana in the late 1960's to investigate the impacts of fire on soil and vegetation dynamics has given an insight on how a subalpine fir, Douglas-fir, and Engelmann spruce ecosystem can respond to both prescribed fire and a wildfire. Study plots on different slope aspects (north, east, south, and west) were clearcut in 1966 and prescribed burned during 1967 and 1968. In August 1967 a wildfire burned (or reburned) several of the plots (DeByle, 1981; Shearer, 1989). In 1996 we resampled representative plots along with adjacent undisturbed stands to evaluate C recovery over this 29-year period (Table 3).

Thirty years after site treatment (prescribed fire, wildfire, or clearcut and not burned) all disturbed stands have lower levels of both C and organic matter content in the forest floor. Particularly outstanding are the low levels of C and organic matter in the dry phase stand that had a wildfire after clearcut (5 Mg ha^{-1} C and 9 Mg ha^{-1} organic matter compared to 36 Mg ha^{-1} C and 67 Mg ha^{-1} organic matter in the old-growth stand). In the moist phase of the forest type (north and east slopes), the thinned and prescribed burn stand were similar in both forest floor C and organic matter after 30 years. However, the old-growth stand had more than twice as much forest floor C and organic matter.

Five years after logging and fire, surface mineral soil organic matter content had recovered to 5.2% and was similar to the undisturbed old-growth stand levels (DeByle, 1981). At the time of our resampling, mineral soil OM averaged 7.4% for all plots and was similar to the undisturbed areas indicating mineral soil organic matter (and likely C) continues to accumulate (Page-Dumroese, unpublished data). The striking differences among this forest floor and mineral soil data illustrate the importance of evaluating both mineral soil and surface organic matter pools before and after fire.

Management Perspective

Many ecosystems, particularly in the western U. S. A., are now overloaded with surface fuels that have accumulated from fire suppression. This type of stand condition, with large amounts of surface fuel, is conducive to wildfire and may trigger catastrophic changes in soil productivity if fire severity is high (Sands, 1983; Harvey et al., 1999a). Much of a forest stands C storage likely occurs aboveground or in the deeper mineral soil horizons (Grigal and Ohmann, 1992; Tilman et al., 2000). Therefore, changes in mineral soil C (or lack of change) may not be an indicator of total site C losses, since most C losses from fire occur in the forest floor material. Maintaining total soil profile C levels and soil productivity while reducing wildfire incidence in fire suppressed stands will likely be achieved through a variety of strategies aimed at developing stands consisting of multiple species and multiple ages rather than managing for one species or age (Tiedemann et al., 2000). This change in structure and age will also affect biological decomposition. Increased biological decomposition, along with prescribed fire, thinning, and salvage logging can all be used to reduce fuel loads to help protect the soil resource.

While it seems desirable to accelerate burning frequency in fire-dependent ecosystems that have not experienced recent fires, this could lead to changes in the cycling of soil nutrients (e.g. N, P, S), loss of soil water-holding capacity, increased soil hydrophobicity, alteration of microbial communities, and impaired long-term soil productivity through loss of organic matter on some sites. A lowering of soil productivity after fire would also reduce future soil C sequestration, since biomass production in the subsequent stands will be less (Tiedemann et al., 2000). However, the extent and impact of such soil changes under controlled burning conditions are largely unknown, and need to be considered as part of any large-scale ecosystem fire management plan.

References

Agee, J. K. 1993. Fire Ecology of Pacific Northwest forests. Island Press, Washington, D. C. 493 p.

Baird, M., D. Zabowski, and R. L. Everett. 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. *Plant and Soil* 209: 233-243.

Ballard, R. and S. P. Gessel (Eds.). 1983. IUFRO Symposium on Forest Site and Continuos Productivity, Seattle, WA. USDA Forest Service. Gen. Tech. Rep. PNW-GTR-163. 406 p.

Caspersen, J. P., S. W. Pacala, J. C. Jenkins, G. C. Hurtt, P. R. Moorcroft, R. A. Birdsey. 2000. Contributions of land-use history to carbon accumulation in U. S. forests. *Science* 290: 1148-1151.

Covington, W. W. and Sackett S. S. 1984. The effect of a prescribed burn in southwestern ponderosa pine on organic matter and nutrients in woody debris and forest floor. *For Sci*. 30: 183-192.

Covington, W. W. and Sackett, S. S. 1992. Soil mineral nitrogen changes following prescribed burning in ponderosa pine. *For. Ecol. Manage.* 54: 175-191.

DeBano, L. F., D. G. Neary, and P. F. Ffolliott. 1998. Fire's effects on ecosystems. John Wiley and Sons, New York. 333 p.

DeByle, N.V. 1981. Clearcutting and fire in the larch/Douglas-fir forests of western Montana-a multifaceted research summary. USDA Forest Service, Intermountain Forest and Range Experiment Station. Gen. Tech. Rep. INT-GTR-99. 73 p.

Dyrness, C. T., K. VanCleve, and J. D. Levison. 1989. The effect of wildfire on soil chemistry in four forest types in interior Alaska. *Can. J. For. Res.* 19: 1389-1396.

Fernandez, I. J., J. Logan, and C. J. Spenser. 1989. The effects of disturbance on the mobilization and distribution of nutrients and trace metals in forest soils. Environmental Studies Center, Univ. of Maine, Orono, ME. 15 p.

Flannigan, M. D., B. J. Stocks, and B.M. Wotton. 2000. Climate change and forest fires. *The Sci. of the Total Environ.* 262: 221-229

Freedman, B. 1981. Intensive forest harvest: a review of nutrient budget considerations. Information report M-X-121. Canadian Forest Service. Fredericton, N. B. 77 p.

Grier, C. C. 1975. Wildfire effects on nutrients distribution and leaching in a coniferous ecosystem. *Can. J. For. Res.* 5: 599-607.

Grigal, D. F. and L.F. Ohmann. 1992. Carbon storage in upland forests of the Lake States. *Soil Sci. Soc. Am. J.* 56: 273-296.

Harvey, A. E. 1994. Integrated roles for insects, diseases and decomposers in fire dominated forests of the Inland Western United States: Past, present, and future health. *J. Sustain*. 2:211-220.

Harvey, A. E., J. M Geist, G. I. McDonald, M. F. Jurgensen, P. H. Cochran, D. Zabowski, and R. T. Meurisse. 1994. Biotic and abiotic processes in eastside ecosystems: the effects of management on soil properties, processes, and productivity. USDA Forest Service, Pacific Northwest Station. Gen. Tech. Rep., PNW-GTR-23. 21 pp.

Harvey, A.E., R. T. Graham and G. I. McDonald. 1999a. Tree species composition changes-soil organism interaction: Potential effects on nutrient cycling and conservation in interior forests. *In* Proceedings: Pacific Northwest Forest & Rangeland Soil Organism Symposium: organism functions and processes, management effects on organisms and processes, and role of soil organisms in restoration. USDA Forest Service, Pacific Northwest Research Station. Gen. Tech. Rep. PNW-GTR-461. p. 137-145.

Harvey, A. E., R. T. Graham, G. I. McDonald, and M. J. Larsen. 1999b. Fire/Decay: Managing codependent forest processes across the landscape. *In* L. F. Neuenschwander and K. C. Ryan (eds.) Proceedings of the Joint Fire Sciences Conference and Workshop, Vol II. University of Idaho Press. pp. 179-189.

Huffman, E. L., L. H. McDonald, and J. D. Stednick. 2001. Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. *Hydrol. Process.* 15: 2877-2892.

Johnson, D. W. 1992. Effects of forest management on soil carbon storage. *Water, Air, and Soil Pollution* 64: 83-120.

Johnson, D. W. and D. E. Todd, Jr. 1998. Harvest effects on long-term changes in nutrient pools of mixed oak forests. *Soil Sci. Soc. Am. J.* 62: 1725-1735.

Jurgensen, M. F., A. E. Harvey, and M. J. Larsen. 1981. Effects of prescribed fire on soil nitrogen levels in a cutover Douglas-fir/western larch forest. USDA Forest Service, Intermountain Research Station, Research Pap. INT-RP-275. 5 p.

Jurgensen, M. F., A. E. Harvey, R. T. Graham, D. S. Page-Dumroese, J. R. Tonn, M. J. Larsen, and T. B. Jain. 1997. Impacts of timber harvesting on soil organic matter, nitrogen, productivity, and health of Inland Northwest forests. *For. Sci.* 43: 234-251.

Kimmins, J. P. 1977. Evaluation of the consequences for future tree productivity of the loss of nutrient in whole-tree harvesting. *For. Ecol. Manage.* 1: 169-183.

Kraemer, J. F. and R. K. Hermann. 1979. Broadcast burning: 25-year effects on forest soils in the western flanks of the Cascade Mountains. *For. Sci*. 25: 427-239.

Krankina, O. N. and M. E. Harmon. 1994. The impact of intensive forest management on carbon stores in forest ecosystems. *World Res. Rev.* 6:161-177.

Laiho, R. and C. E. Prescott. 1999. The contribution of coarse woody debris to carbon, nitrogen, and phosphorus cycles in three Rocky Mountain coniferous forests. *Can. J. For. Res.* 29: 1592-1603

Lopushinsky, W., D. Zabowski, and T. D. Anderson. 1992. Early survival and height growth of Douglas-fir and lodgepole pine seedlings and variations in site factors following treatment of logging residues. USDA Forest Service, Pacific Northwest Research Station. Res. Pap. PNW-RP-451.

McKee, W. H. 1982. Changes in soil fertility following prescribed burning on coastal plain pine sites. USDA Forest Service, Southeastern Experiment Station. Res. Pap. SE-RP-234.

McKee, W. H. and C. E. Lewis. 1983. Influence of burning and grazing on soil nutrient properties and tree growth on a Georgia coastal plain site after 40 years. *In* E. P. Jones, Jr. (ed.) Proceedings of the 2nd biennial Southern silviculture research conference, USDA Forest Service Southeastern Experiment Station. Gen. Tech. Rep. SE-GTR-24.pp, 79-87

McNabb, D. H. and K. Cromack, Jr. 1990. Effects of prescribed fire on nutrients and soil productivity. *In* J. D. Walstad, S. R. Radosevich, D. V. Sandberg (eds.) Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR. Oregon State University Press. pp. 125-142.

Monleon, V. J. and Kermit Cromack, Jr. 1996. Long-term effects of prescribed underburning on litter decomposition and nutrient release in ponderosa pine stands in central Oregon. *For. Ecol. and Manage.* 81:143-152.

Monleon, V. J., K. Cromack, Jr., and J. D. Landsberg. 1997. Short- and long-term effect of prescribed underburning on nitrogen availability in ponderosa pine stands in central Oregon. *Can. J. For. Res.* 27: 369-378.

Neary, D. G., C. C. Klopatek, L. F. DeBano, and P. F. Ffolliott. 1999. Fire effects on belowground sustainability: a review and synthesis. *For. Ecol. and Manage* 122: 51-71.

Neary, D. G., L. F. DeBano, and P. F. Ffolliott. 2000. Fire impacts on forest soils: a comparison to mechanical and chemical site preparation. *In* E. K. Moser and C. F. Moser (eds.). Fire and forest ecology: innovative silviculture and vegetation management. Tall Timbers Ecology Conf. Proceedings No. 21. pp. 85-94.

Oliver, C. D. and B. C. Larson. 1990. Forest Stand Dynamics. McGraw-Hill, New York. 467 p.

Oliver, C. D., D. E. Ferguson, A.E. Harvey, H.S. Malany, J. M. Mandzak, and R. W. Mutch. 1994. Managing ecosystems for forest health: An approach and the effects on uses and values. *J. Sustainable For.* 2: 113-133.

Page-Dumroese, D., M. Jurgensen, W. Elliot, T. Rice, J. Nesser, T. Collins, R. Meurisse. 2000. Soil quality standards and guidelines for forest sustainability in northwestern North America. *For. Ecol. and Manage* 138: 445-462.

Pierson, F. B., P. R. Robichaud, and K. E. Sapeth. 2001. Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed. *Hydrol. Process.* 15: 2905-2916.

Pietikäinen, J. and H. Fritze, 1995. Clear-cutting and prescribed burning in coniferous forest: comparison of effects on soil fungal and total microbial biomass, respiration activity, and nitrification. *Soil Biol. and Biochem.* 27: 101-109.

Pietikäinen, J. O. Kiikkilä and H. Fritze. 2000. Charcoal as a habitat for microbes and its effect on the microbial community of the underlying humus. *Oikos* 89: 231-242.

Raison, R. B. 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. *Plant and Soil* 41: 73-108.

Robichaud, P. R. and R. D. Hungerford. 2000. Water repellency by laboratory burning of four northern Rocky Mountain forest soils. *J. Hydrology* 231-232: 207-219.

Sands, R. 1983. Physical changes to sandy soils planted to radiata pine *In* R. Ballard and S. P. Gessel (eds.) IUFRO symposium on forest site and continuous productivity. USDA Forest Service Pacific Northwest Station Gen. Tech. Rep. PNW-GTR-163.

Shearer, R. C. 1989. Fire effects on natural conifer regeneration in western Montana. *In* D. M. Baumgartner, et al. (eds). Proceedings of prescribed fire in the Intermountain region: a forest site preparation and range improvement symposium. Washington State Univ. p. 19-33.

Stocks, B. J., M. A. Fosberg, M. B. Wotton, T. J. Lynham, and K. C. Ryan. 2000. Climate change and forest fire activity in North American Boreal Forests. *In* E. S. Sasischke and B. J. Stocks (eds.). Fire, climate changes, and carbon cycling in the Boreal Forest. New York: Springer-Verlag. pp. 368-376.

Tiedemann, A. R., J. O. Klemmedson, E.L. Bull. 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk? *For. Ecol. and Manage*. 127: 1-18.

Tilman, D., P. Reich, H. Phillips, M. Menton, A. Patel, E. Vos, D. Peterson, and J. Knops. 2000. Fire suppression and ecosystem carbon storage *Ecology* 81: 2680-2685.

Visser, S. and D. Parkinson. 1999.Wildfire vs. clearcutting: impacts on ectomycorrhizal and decomposer fungi. *In* Meurisse, R., W. Ypsilantis, and C. Seybold (Technical Eds.). Proceedings of the Pacific Northwest forest and rangeland soil organism symposium. USDA Forest Service, Pacific Northwest Station. PNW-GTR-461. pp.114-123

Vose, J. M. and W. T. Swank. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands: aboveground biomass, forest floor mass, and nitrogen and carbon pools. *Can. J. For. Res.* 23: 2255-2262.

Wells, C. G., R. E. Campbell, L. F. DeBano, C. E. Lewis, R. L. Fredricksen, E. C. Franklin, R. C. Froelich, and P.H. Dunn. 1979. Effects of fire on soil. A state-of-the-art review. USDA Forest Service. Washington, DC. WO-GTR-7. 34 p.

Table 1. Carbon pools and distribution in select soil components in undisturbed stands and potential loss (if 15-100% of the land area is affected) due to prescribed or wildland fire. (Data from Page-Dumroese, unpublished)

Species and Location	Stand Age	Pool	Carbon Distribution	Range of Potential C loss	
	(yrs)	(Mg ha^{-1})	(%)	(%)[1]	(Mg ha^{-1})
Hemlock, Idaho	120				
Surface organic matter[2]		75	56	11-100	4-75
Mineral soil (0-30 cm)		59	44	1-9	<1
Hemlock, Montana	100				
Surface organic matter		70	40	10-100	1-70
Mineral soil (0-30 cm)		102	60	2-9	2-9
Grand fir, Idaho	150				
Surface organic matter		75	67	5-31	3-25
Mineral soil (0-30 cm)		35	33	-[3]	-
Ponderosa pine, Idaho	87				
Surface organic matter		50	57	2-25	1-8
Mineral soil (0-30 cm)		38	43	-	-
Ponderosa pine, California	300				
Surface organic matter		48	44	2-23[4]	1-7
Mineral soil (0-30 cm)		62	56	-	-
Subalpine fir, Montana	100				
Surface organic matter		61	62	9-100	2-61
Mineral soil (0-30 cm)		37	38	1-3	<1
Douglas-fir, Oregon	200				
Surface organic matter		152	66	19-100	24-152
Mineral soil (0-30 cm)		81	34	<1-2	<1
Sitka spruce, Oregon	150				
Surface organic matter		246	66	29-100	56-246
Mineral soil (0-30 cm)		131	34	<1-3	<1

[1] Range of C lost if 15-100% of an area has detrimental burning

[2] Surface organic matter includes the forest floor and highly decayed coarse wood, not intact coarse woody debris

[3] No predicted C loss from the mineral horizon

[4] No detrimental burning standard for USDA Forest Service Region 5, we assumed similar standards as USDA Forest Service Region 4.

Table 2. Examples of fire impacts on soil C levels

Location	Species	Carbon changes	Reference
Wildfire			
Washington	Mixed conifer	40% loss of forest floor	Grier, 1975
Alaska	Spruce, birch and aspen	Increased loss of forest floor with increasing fire intensity	Dyrness et al., 1989
Maine	Mixed hardwoods	Large loss of forest floor, mineral soil recovery after 1 yr.	Fernandez et al., 1989
Washington	Ponderosa pine/Douglas-fir	C was 30% less after 1 yr	Baird et al., 1999
	Lodgepole pine/Englemann spruce	C was 10% less than control after 1 yr.	
Prescribed fire			
Washington	Mixed conifer	26% increase in OM in north Cascades, 2% decrease in south Cascades	Kraemer and Hermann, 1979
Alabama	Longleaf pine Burned every 2 years in winter	No difference from control	McKee, 1982
Florida	Slash pine Periodic winter burn and annual winter burn	After 20 yrs,periodic burn plots had 17% more C than control. Annual burn plots were not different than control	McKee, 1982
Australia	Radiata pine	40-50% loss to 60 cm in mineral soil.	Sands, 1983
Minnesota	Oak savana	38%-70% less C in forest floor.	Tilman et al., 2000

Table 3. Average forest floor carbon and organic matter pools in a western Montana subalpine fire forest (two habitat type phases) 30 yrs after treatment.

Habitat phase and Treatment	**Carbon**	**OM**
	(Mg ha^{-1})	
Dry phase		
Uncut old-growth	36	67
Thinned (not burned)	28	54
Prescribed burn	7	14
Prescribed and wildfire	13	26
Wildfire –uncut	15	30
Wildfire after clearcut	5	9
Moist phase		
Uncut old-growth	61	108
Thinned (not burned)	28	53
Prescribed burn	26	48

Figure Caption

Figure 1. Schematic representation of C accumulation in forest ecosystems with and without frequent fire (after Oliver et al, 1994).