

Impact of Biomass Harvesting on Forest Soil Productivity in the Northern Rocky Mountains

Woongsoon Jang, Christopher R. Keyes, Deborah Page-Dumroese



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Abstract

Biomass harvesting extracts an increased amount of organic matter from forest ecosystems over conventional harvesting. Since organic matter plays a critical role in forest productivity, concerns of potential negative long-term impacts of biomass harvesting on forest productivity (i.e., changing nutrient/water cycling, aggravating soil properties, and compaction) have emerged. There is abundant prediction of long-term impacts of intensive biomass removal on forest productivity. However, the empirical knowledge and comprehensive understanding, especially on western forests, are limited thus far. Therefore, we utilize the available findings to evaluate potential impacts of increased biomass extraction on western forests. We compare biomass harvesting with natural disturbance regimes or conventional harvesting systems in terms of organic matter redistribution in order to evaluate the possible consequences of biomass harvesting on forest productivity. We review the role of organic matter on forest productivity and compare the organic matter redistribution or removal through biomass harvesting and natural disturbances or conventional harvesting to assess potential impacts. The summarized findings are: (1) the long-term impacts of intensive biomass harvesting will be mitigated by protection of the belowground organic matter; (2) biomass harvesting could result in the accelerated leaching of nutrients; and (3) immediate understory vegetation recovery can minimize potential negative impacts. Finally, sites sensitive to harvesting impacts (e.g., fine-textured soil and steep slopes) should be approached with caution and prior planning to minimize undesirable responses.

Keywords: biomass harvesting, site productivity, soil productivity, organic matter, ecological forestry

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Introduction

Timber harvesting involves the redistribution and exportation of forest biomass (i.e., organic matter). Due to emerging attention on using forest biomass as an alternative energy source, it is likely that the quantity and variety of forest biomass removal will increase (Janowiak and Webster 2010). Forest residues that had previously been considered non-merchantable, such as tops, branches, slash, cull, snags, coarse woody debris, stumps, and bark, are now being harvested for use (Benson and Schlieter 1980; Barger 1981; Berger and others 2013). Although intensive (whole-tree) harvesting has been conducted for decades in many temperate and boreal forests across the world (Thiffault and others 2011), most western U.S. operations and infrastructure have not been operating at that level. However, many western U.S. forests will likely be managed more intensively in the future, thereby increasing the level of biomass utilization.

As interest in biomass harvesting increases so do resulting ecological concerns. The diverse concerns for ecosystem functions include: wildlife habitat, water quality, biodiversity, air, and forest productivity. Among these, the primary concern for foresters and silviculturists is the potential negative impact on forest productivity. Therefore, the key question in this study is whether increased biomass removal causes undesirable long-term consequences to forest stand and soil productivity.

Forest (site) productivity can be defined as: “The relative capacity of an area to sustain a supply of goods or services in the long run” (Society of American Foresters 1998), and “The capacity of a forest to produce specific products (i.e. biomass, lumber) over time as influenced by the interaction of vegetative manipulation and abiotic factors (i.e. soil, climate, physiography)” (Soil Science Society of America 2008). Therefore, forest productivity is the integration of all environmental factors encompassing soil productivity, climate, topography, geology, vegetation, and the history of natural disturbances and anthropogenic interventions (Morris and Miller 1994; Grigal 2000). These interrelated factors can affect one another directly and indirectly, making the prediction of potential impacts on forest productivity complex. Biomass harvesting has few long-term impacts on most ecosystem processes, except soil productivity. Changes in soil nutrient cycling, compaction, and water holding are critical determinants of future forest productivity, which is why many biomass harvesting studies have placed more emphasis on the alteration of soil productivity (Thiffault and others 2011). Later, we discuss the potential negative effects of biomass harvesting on soil productivity as well as ways to mitigate those effects.

A fundamental change in forest management philosophy has occurred in recent decades. Emulating natural disturbance regimes has become a primary strategy to ensure a resilient ecosystem (Drever and others 2006; Franklin and others 2007; Berger and others 2013). This strategy maintains that natural disturbances and processes should be the foundation of the silviculture (Attiwill 1994; Rogers 1996; Seymour and Hunter 1999; Seymour and others 2002; Franklin and others 2007), and resource managers should use harvest methods that maintain complex structure, composition, and function of forest ecosystems (Long and others 2004) within their historical range and variability (*sensu* Keane and others 2009). A key assumption of this paradigm is that the indigenous vegetation community has evolved within a full range of environmental conditions; therefore, maintaining the ecosystem within this range is the best

method to prevent undesirable consequences (Seymour and Hunter 1999). Ecological concerns can be assessed by asking if the ecological consequences (impact) of biomass harvesting on forest productivity exceed the range of consequences created by natural disturbances.

As previously noted, biomass harvesting increases organic matter removal and results in a different configuration of the remaining organic matter within a forest. Therefore, comparing the amount of transferred organic matter through biomass harvesting to natural disturbances can provide a basis for predicting possible impacts to forest productivity. The objectives of this report are to: (1) review the role of the various organic matter types in forest productivity; (2) compare changes in organic matter resulting from natural disturbances and from biomass harvesting regimes; and (3) discuss the potential effects of biomass harvesting on forest productivity.

Role of Organic Matter in Forest Productivity

Classification and Distribution of Organic Matter in the Forest Ecosystem

Forest biomass refers to any organic matter, including living and non-living material. Roughly, carbon (C) pools in forest ecosystems can be grouped as: living (above and belowground) organisms, coarse woody debris (standing and fallen), forest floor (litter, duff, and humus, combined), and mineral soil. However, classification of organic matter pools often depends on a research objective. Moreover, the focus of a study may include only a subset of total C pools (Page-Dumroese and others 2006).

The simplest classification of organic matter pools is living and non-living. In this case, live biomass commonly refers to aboveground vegetation, including roots. Non-living biomass, also called detritus (e.g., Schlesinger 1977) or necromass (e.g., Palace and others 2007), represents non-living organic material from the canopy layer (i.e., snags), including the mineral soil to bedrock. Similarly, organic matter pools can be classified according to strata in the aboveground or belowground pool (e.g., Attiwill and Adams 1993). Living roots are pooled with aboveground biomass. Beyond these simple classification schemes, classification of forest detritus (non-living organic matter above the mineral soil layer) versus organic material in the mineral soil layer (e.g., Wang and others 2003) can be further separated into coarse or fine woody debris, litter fall, humus, duff, or soil wood (e.g., Jurgensen and others 1997; Rice and others 2004).

Site organic matter is commonly expressed as C stock, since C is a major element of organic matter. The C stock is defined as the amount of C in a pool, representing a system or reservoir's capacity to accumulate or release C (Forest Resources Assessment Programme 2004). For example, live biomass comprises 42% of total C stock of forest ecosystems in global scale (Pan and others 2013; Table 1). However, in boreal forests, only about 20% of total biomass is comprised of living biomass. C stock distributions are different for different biomes, but, generally, the most abundant organic matter pools are located in the soil (Goodale and others 2002; Rumpel and Kögel-Knabner 2011; Schmidt and others 2011).

Table 1—Distribution of carbon stock (Mg C/ha) by biome in 2007 (from Pan and others 2013).

	Boreal	Temperate	Tropical intact	Tropical regrowth	Global
Live biomass	47.5	60.7	163.9	60.8	94.2
Non-living biomass	191.7	94.0	118.6	78.6	129.4
Total	239.2	154.7	282.5	139.4	223.6

Organic matter in woody residues, the forest floor, and mineral soil are essential for maintaining ecosystem function by supporting soil C cycling and sequestration, nitrogen (N) availability, gas exchange, water availability, and biological diversity (Jurgensen and others 1997). Loss of organic matter resulting from stand disturbance can drastically change long-term soil productivity (Grigal and Vance 2000; Page-Dumroese and Jurgensen 2006). For example, undisturbed subalpine fir/queen’s cup (*Abies lasiocarpa* [Hook.] Nutt./*Clintonia uniflora* Menzies ex Schult. & Schult. f. [Kunth]) stands in Montana and Idaho can have vastly different total amounts of organic matter in and on the soil (Montana total organic matter pool 431 Mg/ha; Idaho total organic matter pool 593 Mg/ha; Page-Dumroese and Jurgensen 2006). However, in Montana 40% of the organic matter pool was on the soil surface, whereas in Idaho, the surface organic matter pool comprised only 27%. The distribution of organic matter in and on the soil depends on various factors, such as soil type and texture, stand age, species composition, geographic region, and stand history. However, the majority of soil organic matter (SOM; in this report, collectively refers to non-living organic matter in the understory) is concentrated in the organic layer near the forest surface (Jurgensen and others 1997). In the mineral soil layer, a large amount of SOM is distributed at a shallow depth (Harvey and others 1994). Between 39 and 70% of total organic matter in the top 1 m of soil is concentrated in the first 30 cm (Batjes 1996). More than 90% of total root biomass of temperate coniferous forest is concentrated in the top 1 m of soil, and other forest types have even shallower root distribution (e.g., 83% of root biomass is concentrated within only the top 30 cm in boreal forests) (Jackson and others 1996). It is now known that considerable amounts of organic matter are allocated in deeper soil layers, the importance of which is emphasized in specific regions, but there is very little work on its distribution (Rumpel and Kögel-Knabner 2011). Understanding the distribution of organic matter within a given stand is crucial for determining the impacts of harvesting for saw timber or bioenergy, prescribed fire, or wildfire.

Living organic matter

Living organic matter (i.e., vegetation) plays an essential role in nutrient and energy cycling (Figure 1) and, therefore, in productivity of forest ecosystems. Nutrients enter the forest ecosystem in the form of rain and dust and by biological fixation (nutrient conversion to available form by living organisms), and they exit in stream water (drainage) and in gaseous form (e.g., volatilization by fire). Vegetation captures (“uptake” in Figure 1) the inorganic nutrients, holds and synthesizes organic matter (“internal redistribution”), and redistributes (“return”) organic matter through litter fall and leaching (Attiwill and Adams 1993; Farve and Napper 2009). In other words,

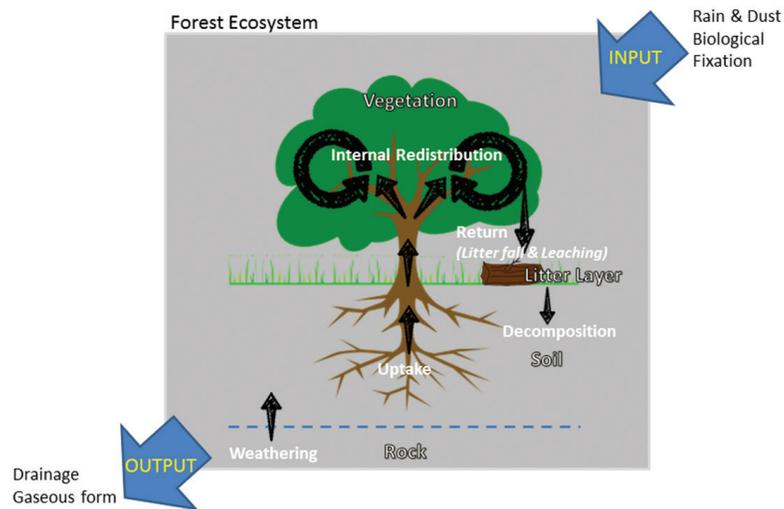


Figure 1—Schematic illustration of general nutrient cycle in forest ecosystems (modified from Attwill and Adams 1993).

vegetation consumes available (but mobile) nutrients, transforms the nutrients into stationary forms, and releases the formed nutrients simultaneously.

In addition to nutrient and energy cycle, forest vegetation is a core component to determining forest productivity. As shown in Figure 1, living organic matter can produce non-living organic matter such as dead needles, leaves, or branches, and cones. Needles and leaves (litter) from vegetation is a primary source of SOM (Lorenz and Lal 2005) that is utilized and decomposed (“decomposition” in Figure 1) by various organisms. Annual production of litter fall in Sitka spruce (*Picea sitchensis* [Bong.] Carrière) was calculated as 3.151 Mg/ha in Scotland and Northern England, indicating that the annual litter fall production was equivalent to approximately 11% of total accumulated organic matter contents in the forest floor (Miller and others 1996).

Mortality of fine roots can provide essential resources to soil microbes. Decaying fine roots are regarded as more important input to soil layer than surface residues (Powers and others 2005). Annual SOM derived from fine roots was estimated at about 4.1 Mg/ha from a mixed hardwood forest in Massachusetts (McClagherty and others 1982). Annual fine root production was measured as 3.76 Mg/ha on multiple stands in Wisconsin and Massachusetts (Aber and others 1985). Aber and others (1985) reported that turnover rate of fine roots ranged from 48 to 82% (1.8 to 3.1 Mg/ha/yr).

On the other hand, living vegetation can influence understory productivity through modifying microclimate. First, solar radiation can be regulated by upper-layer vegetation. The amount of available light for photosynthesis generally decreases closer to the forest floor (Wright and others 2006). According to the Beer-Lambert Law, the amount of photosynthetically available radiation in the understory decreases exponentially as the leaf area aboveground increases (e.g., Pierce and Running 1988).

Only 0.5 to 5% of the full solar radiation can reach the understory in many closed forests (Chazdon and Pearcy 1991). Since diminished light availability limits growth and, therefore, determines species composition in the understory (Montgomery and Chazdon 2002; Neufeld and Young 2003), the amount of vegetation in the canopy layer also affects forest productivity. In addition, attenuated solar energy reaching the understory helps regulate soil temperature. Since soil temperature is involved in various belowground processes such as root growth, decomposition, and N mineralization (Waring and Running 2007), change of soil temperature can affect soil productivity significantly. In the inland Pacific Northwest, Jurgensen and others (1992) found a general trend that removing the overstory increased the temperature of the soil layer. In the northern Rocky Mountains, Hungerford and Babbitt (1987) suggested that understory vegetation removal can increase the ground surface temperature up to 6.1 °C (11.0 °F) in some months. If soil respiration is measured to quantify the belowground processes, a temperature increase of 10 °C (18 °F) can result in 3.4 to 5.6 times increase in soil respiration in mixed hardwood forests of the northeastern United States (Davidson and others 1998).

Aboveground vegetation can affect the soil moisture content. Aboveground vegetation biomass influences soil water content directly through two kinds of hydrological processes: evapotranspiration and interception of rainfall. In the northern Rocky Mountain region, about 30 to 40% of total annual precipitation is lost through evapotranspiration (Running and others 1989). Vegetation can intercept from 5 to 26% of total annual precipitation (Helvey and Patric 1965; Waring and Running 2007). Therefore, aboveground vegetation determines the amount of water in the soil layer that plants can utilize. Since soil moisture is utilized not only by plants but also by diverse soil microorganisms pertaining to nutrient cycling (Harvey and others 1980b), soil water content can also be an influential factor for forest productivity.

Non-living organic matter

Non-living organic matter occupies the majority of total organic matter in most biomes (Table 1). These materials are distributed among the coarse woody debris, forest floor (all organic horizons), soil wood (woody residue in the mineral soil), and mineral soil layers (Table 2). In western-montane forests, non-living organic matter averages 26%, 9%, 10%, and 55%, respectively (Page-Dumroese and others 1990). SOM content is closely bound to soil productivity unless an environmental factor (e.g., extremely low temperature and drought) limits microbial activity. In general, organic matter depth reflects forest productivity (Jurgensen and others 1997); the deeper the organic matter, the more productive the site. This highlights the importance of belowground processes and the amount of soil organic matter for soil productivity. The majority of nutrients in organic matter exist as forms that cannot be instantly utilized by plants. Therefore, organic materials must be disassembled (i.e., decomposition) and transformed (i.e., mineralization) into inorganic forms by belowground processes. SOM is involved in these processes and can influence forest productivity indirectly, but crucially, through altering physical, chemical, and biological properties of soil (Grigal 2000; Hatten and Zabowski 2009).

Table 2—The distribution of non-living organic matter (Mg/ha) in western montane forests (from Page-Dumroese and others 1990).

Habitat type	Residue	Forest floor	Soil wood	Mineral soil	Total
Cedar/hemlock (Montana)	84	50	51	145	330
Cedar/hemlock (Idaho)	154	23	48	201	426
Subalpine fir	146	36	36	153	371
Douglas-fir	45	26	26	133	230
Ponderosa pine	20	7	2	160	189
Average (%)	26	9	10	55	100

Physical Properties

Soil water retention

Organic matter can enhance soil water conditions in various ways. First, organic matter on the soil surface can lower soil temperature and increase soil moisture retention to prevent evaporative loss (Powers and others 2005). In addition, large amounts of moisture may be captured by detritus, especially fallen and decaying coarse wood. Page-Dumroese and others (1990) reported that more than five times more available water was stored in woody residue than in mineral soil in a Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) stand in northern Idaho. Abundant soil water is crucial for seedling survival, establishment, and growth and is essential on sites that experience a summer drought (Padilla and Pugnaire 2007). In addition, as climate changes, SOM may provide available water to keep trees healthy longer in times of drought (Allen and others 2010).

Soil structure

SOM can also increase porosity, thereby decreasing bulk density, improving soil structure, and enhancing soil water holding capacity (Shepherd and others 2002). SOM is a major binding agent, cementing individual soil particles together into more stable soil aggregates (Jastrow 1996). Aggregates alter pore size distribution and can enhance water infiltration into the soil. Increased porosity improves soil gas exchange, which is required for respiration (Bronick and Lal 2005). Moreover, porosity enhances root expansion by providing lower-density soil that is easily penetrated by roots.

Chemical Properties

Nutrients

Plants require various essential nutrients to grow. Some elements can be absorbed from the atmosphere through stomata (e.g., C, hydrogen, and oxygen), but the majority of other essential nutrients must be acquired from the soil layer through the root system. Non-living organic matter consists of these essential elements primarily since it is made up of material that was once alive. Thus, the amount of organic matter reflects the quantity and quality of essential nutrients in soil. The majority of available nutrients are concentrated in the soil organic layer. In Idaho batholith, the soil organic layer contained 88% of total potassium (K), calcium (Ca), magnesium (Mg), N, and phosphorous (P) (Megahan 1990). In Minnesota, the forest floor and mineral soil layers contained 4 to 20 times more N, Ca, and Mg than aboveground vegetation (Alban and others 1978).

Cation exchange capacity (CEC)

CEC refers to the ability of soil particles to hold and exchange metallic nutrients such as Ca, Mg, and K. Many essential nutrients are only supplied from soil, existing as a form of cations to be utilized by plants. However, due to their electrical characteristics, the cations may easily be leached by water. Since SOM and clay particles are negatively charged, they can hold these base cations. Therefore, soils with high CEC have many sites to capture nutrients for the vegetation, preventing cations from leaving the ecosystem (DeByle 1980). For this reason, soil CEC can be used as a barometer of soil productivity.

Buffering of soil pH change

The availability of soil nutrients is affected by soil acidity. For example, P is highly sensitive to soil pH; it converts easily into a less available form both in alkaline and acidic conditions. In addition, nitrification rates are slower in acidic soils (De Boer and Kowalchuk 2001). Since it is the process by which unavailable N converts to available form, slower rates can limit plant growth. Other micronutrients, such as manganese, iron, and copper, generally tend to be less available as soil pH increases.

Soil acidity can also impact the soil microbial activities. Nicol and others (2008) found that the abundance and diversity of nitrifying bacteria differs according to soil pH, which indicates that nitrification might change in accordance with soil pH gradient. In general, the richness of soil bacteria is highest in neutral soils and decreases as the soil becomes more acidic (Fierer and Jackson 2006). Similarly, mycorrhizal abundance and nutrient uptake ability are also affected by soil acidity. Erland and Söderström (1990) found that the number of mycorrhizal root tips of Scots pine (*Pinus sylvestris* L.) seedlings was maximized around pH 5 and decreased with decreasing pH (increased acidity). Chalot and others (1995) reported that nutrient (amino acid) uptake through ectomycorrhizae was optimized around pH 4. Therefore, rapid changes in soil acidity alter soil microbial activity.

However, abundant SOM can ameliorate the potential adverse impacts of sudden changes in soil pH. Here, CEC plays a key role as a buffer. Soil acidity is determined by the concentration of hydrogen ion (H^+) in the soil. As a cation, the hydrogen ion competes with other cations. If soil acidity increases (i.e., higher hydrogen ion concentration), more hydrogen ions occupy the exchangeable sites, thereby increasing the leaching risk of base cations. Therefore, soils with higher CEC (and higher SOM) can be more resistant to sudden changes of soil acidity.

Biological Properties

Soil organic matter provides food and habitat resources to various soil mesofauna (soil invertebrates) and microfauna (e.g., bacteria and fungus). These organisms are the backbone of decomposition and mineralization processes, breaking down and converting organic matter to forms that plants can use. The nutrients are then re-absorbed by roots and recycled by vegetation. Thus, soil organism activity positively impacts forest productivity through the creation of soil porosity and increases in air and water movement. Also, some soil microbes secrete chemical compounds to alter and stabilize soil structure (Tan and others 1978; Tisdall and Oades 1979). Finally, predation

(grazing) upon microbes by soil invertebrates has been found to release considerable amounts of available nutrients to plants (Molina and Amaranthus 1990).

Soil microbe activity is often determined entirely by the amount of organic matter in the mineral soil (Harvey and others 1980a). In addition, N-fixing microbes use decaying wood as a major energy resource (Jurgensen and others 1980). Since N is generally known as a major limiting nutrient of forest soil productivity (Binkley 1991; Vitousek and Howarth 1991), there is a strong positive relationship between the amount of SOM and forest productivity. Jurgensen and others (1980) found this to be true when comparing forest productivity and the amount of N fixation in western larch (*Larix occidentalis* Nutt.) forests of Montana.

Additionally, organic matter in mineral soil performs a crucial function to support ectomycorrhizae. Ectomycorrhizae are root-based fungal symbionts that help determine tree performance (Kropp and Langlois 1990) of most commercial coniferous species in northwestern U.S. forests (Wiensczyk and others 2002). Ectomycorrhizae enhance water and nutrient uptake through a symbiotic relationship that supplies the fungus with carbohydrates. Therefore, the abundance and diversity of ectomycorrhizae are also closely related to soil productivity (Perry and others 1987). In northern Rocky Mountain forests, Harvey and others (1980b) reported that more than 60% of total ectomycorrhizae were distributed in soil wood and humus. In other parts of the world, the highest concentrations of mycorrhizal activity are found in organic layer and mineral soils close to the surface (Neary and others 1999). Thus, the distribution and abundance of SOM determine the mycorrhizae abundance and, in turn, affect site productivity.

Organic Matter Redistribution Following Natural Disturbances and Biomass Harvesting

There are very few data in the western United States on the distribution of organic material on and in the soil. This information is critical to gauge the impacts of bioenergy harvesting on the stand (Page-Dumroese and Jurgensen 2006; Page-Dumroese and others 2010). We stress the importance of maintaining organic detritus because it is important for the ecological functions of mammals and birds (Maser and Trappe 1984) as well as plants (Harmon and Franklin 1989). Decaying logs store nutrients and water (Sollins and others 1987) and provide for humus formation (McFee and Stone 1966). Forest detritus is also an important nutrient cycling pool (Franklin and Waring 1980). However, there are few links that show the importance of organic matter pools to tree (or other vegetation) growth. For example, the North American Long-Term Soil Productivity (LTSP) study found no effect following removal of large quantities of organic matter from a loblolly pine (*Pinus taeda* L.) stand after 10 years, even though organic matter removal would be expected to lower N mineralization rates and hinder tree growth (Sanchez and others 2006). Further, in a meta-analysis of several LTSP study sites, forest floor removal had no overall significant effect on seedling diameter, except in ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) ecosystems (Fleming and others 2006). The authors attributed growth differences to warm-humid conditions and rapid growth demands for available soil nutrients. Organic matter removal may promote early season growth by increasing

soil temperatures, but it may suppress summer growth by reducing soil water content (Fleming and others 2006). However, maintaining ecosystem processes from one rotation to the next will likely depend on maintaining a minimal amount of surface and soil organic matter.

Land management paradigms have shifted from standard silvicultural practices (e.g., clearcut, selection harvests, and shelterwoods) to emulating natural disturbances. With this shift in ecosystem management philosophies, the view of natural disturbance has changed. Now, forest scientists emphasize the complementary function of natural disturbances to ecosystem (Rogers 1996). Further, many forest scientists assert that silviculture should be founded on the ecosystem processes (Seymour and others 2002). As a result, scientists are endeavoring to understand and integrate diverse ecosystem processes, including natural disturbance, into silvicultural knowledge and skills (e.g., Seymour and Hunter 1999; Noss and others 2006; Franklin and others 2007; Berger and others 2013).

Disturbance can be defined as a discrete event causing a change in an ecosystem's structure, composition, or physical environment (Pickett and White 1985). Forest response to disturbance is dynamic (McClougherty and others 1982; Oliver and Larson 1996) and can be described by the disturbance agent (e.g., fire, disease, and insects), frequency (interval), size, magnitude (intensity and severity), predictability, and synergism (Pickett and White 1985). The primary disturbance agents in northern Rocky Mountain forests are: fire, insects/pests, wind, and debris avalanches. These disturbances can be considered mechanisms for redistributing organic matter from one pool to another and/or exporting organic matter out of the ecosystem. For example, fuel combustion by fire results in the loss of organic materials, whereas tree mortality by bark beetles transfers organic matter from the living to non-living pool.

In this section, we discuss the characteristics of biomass harvesting in western forests, particularly Inland Northwest forest ecosystems. We compared the ecological consequences of biomass harvesting with those of natural disturbance agents and high-light changes in the organic matter pools.

Biomass Harvesting in Western Forests

On a large-scale, thus far in the northern Rocky Mountain Region, only merchantable stems (e.g., pole timber: 12.7-22.6 cm diameter, saw timber: larger than pole size; Simmons and others 2014) are extracted, and residues (non-merchantable material) are piled-and-burned or broadcast-burned (Figure 2). More intensive biomass harvesting can be roughly classified into two methods: whole-tree and complete-tree harvesting (sensu Hakkila and Parikka 2002). Non-merchantable tops and crowns are harvested in whole-tree harvesting. Complete-tree harvesting is more intensive and extracts both stumps and roots (e.g., Walmsley and Godbold 2010). Whole-tree harvesting has been studied in northern Europe and northeastern America for decades. Recently, foresters from those regions are investigating the feasibility of complete-tree harvesting for bioenergy (Benjamin and others 2010; Berger and others 2013). However, because complete-tree harvesting requires additional operations and processes, further economic feasibility assessments are needed. In addition, impacts on societal values for more intensive harvesting should be investigated. Biomass harvesting in the northern Rocky Mountains is more likely to be whole-tree harvesting rather than complete-tree.

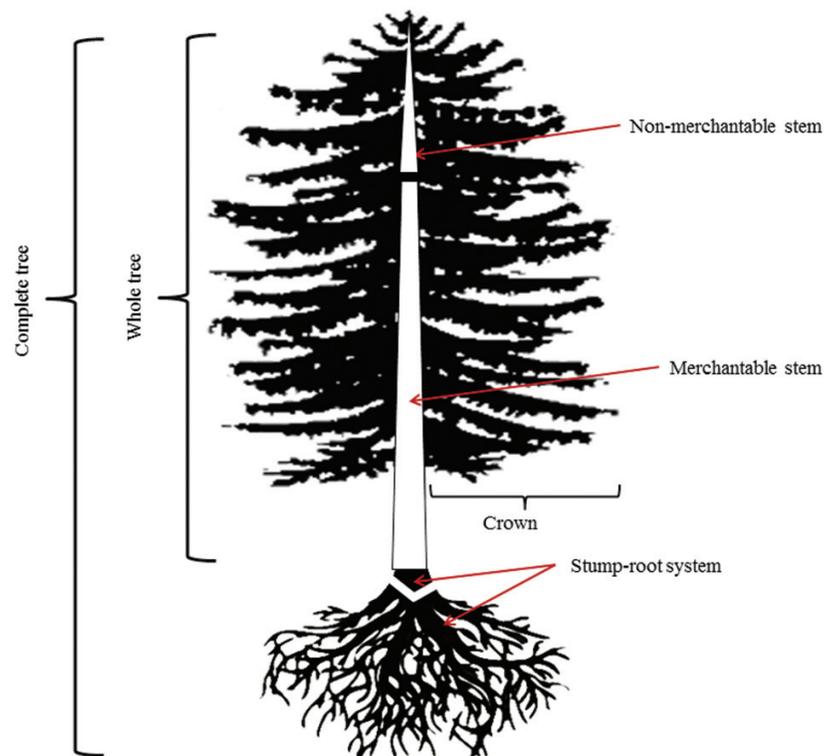


Figure 2—Biomass components of a tree (redrawn from Hakkila and Parikka 2002).

Whole-tree harvesting has a substantial impact on live vegetation (Berger and others 2013). Usually, every tree over a certain diameter size class is cut and a significant proportion of the (living) organic matter is permanently removed from forest. According to Smith and others (1986), about 90% of the total above-stump biomass was harvested in a red spruce (*Picea rubens* Sarg.)-balsam fir (*Abies balsamea* [L.] Mill.) forest in Maine. This information is confounded by the fact that allometric equations used to determine forest residues remaining after whole tree or cable yarding systems may overestimate by 30 to 40% (Anil Kizha, Humboldt State University, personal communication). Therefore, understanding the biomass estimates pre-harvest should help determine how much residue remains after logging.

The residual stumps and roots transfer from the living organic matter pool to the detritus pool. If the root:shoot ratio for biomass is 0.26 (Cairns and others 1997), then 26% of total removed biomass transfers to SOM pool. Changes in coarse and fine woody debris pools are minimal. Mortality of smaller sized trees and understory vegetation can occur during operation but the impact may be negligible. Forest floor displacement can also occur during harvest operations, leaving the mineral soil exposed to erosion, rainfall impact, and localized nutrient removals (Ballard 2000). In addition, slash piling and burning of logged areas can remove a significant portion of N over the affected area (Hickling 1997).

Natural Disturbance Agents in Northwestern Forests

Wildfire

Fire is a primary disturbance agent, not only in northern Rocky Mountains but also in most forest regions throughout the world. Wildfires alter various ecosystem components, including vegetation, soil, water, and air. Wildfire also changes the distribution of organic matter pools in many ecosystems (Page-Dumroese and others 2000; Page-Dumroese and Jurgensen 2006). Fire consumes organic matter through combustion (Hatten and Zabowski 2009) and, depending on the severity of the fire, organic matter is distilled, charred, or completely oxidized (Neary and others 1999; Certini 2005). Substantial organic matter consumption begins around 220 °C (428 °F), and beyond this consumption increases rapidly. At this temperature, many nutrients (e.g., N and P) are volatilized or transformed into unavailable forms (Giovannini and others 1990; Neary and others 1999).

During fire, organic C in live vegetation and SOM is converted to C monoxide, C dioxide and methane, which are immediately released into the atmosphere (Czimeczik and others 2005). C loss during a wildfire in Alaska was calculated to be as much as 20.1 Mg C/ha (French and others 2002). In the Canadian boreal forest, the average amount of direct C lost during a fire was estimated to be 13 Mg C/ha (Amiro and others 2001). On a global scale, it has been estimated that wildfire in boreal forests consumes one-third of net primary productivity (Preston and Schmidt 2006) while van der Werf and others (2010) calculated that in the United States, approximately 6.3 Mg C/ha is consumed during wildfire. A small fraction (0.7 to 8%, Czimeczik and others 2005) of C is converted into charred organic C (black C), which is highly resistant to decomposition in many ecosystems (DeLuca and Aplet 2008) and influences soil processes by increasing CEC (Preston and Schmidt 2006). In many fire-prone ecosystems, black C accounts for a considerable proportion of the total soil C pool (Schmidt and others 1999).

Fire affects living plant tissue in several ways. For example, surface fires kill understory vegetation and result in dead forbs, grasses, and shrubs, as well as the loss of some forest floor material; whereas crown fires damage overstory crown and bole and can kill the tree. Fire severity will determine the loss or decomposition of organic matter above and belowground. In addition, the heat pulse into the soil will contribute to changes in surface or belowground C pools. The threshold temperature for killing plant roots is 48 °C (118 °F; DeBano and others 1998), but disruptions in the biological function of soil begin at 40 to 70 °C (104 to 158 °F). Microbial mortality occurs between temperatures of 50 and 121 °C (122 to 250 °F; Neary and others 1999).

Consequences of wildfire regime and biomass harvesting are summarized in Table 3. Wildfire results in the loss of organic matter, the magnitude of which depends on the fire intensity and severity (Keeley 2009). In the aspect of tree mortality, biomass harvesting can have a similar impact as stand-replacing wildfires. However, even though trees are killed by wildfire and some loss of site C results, a considerable amount of matter will likely remain on-site. Estimates of residual biomass vary from 15% (Fahnestock and Agee 1983; Agee 1996) to 60% (Van der Werf and others 2010). Abundant coarse woody debris is produced as snags begin to fall to the ground (Oliver and Larson 1996; Tappeiner and others 2007). According to Berger and others (2013),

Table 3—Impacts of fire and biomass harvesting on organic matter and stand structure, with comparison to silvicultural treatments.

	Fire			Biomass harvesting
	Stand replacement	Mixed severity	Low severity	
Organic matter extraction	High	Moderate	Low	Very high
Overstory tree mortality	Most	Selective	None	Most
Understory mortality	High	High	High	Low
CWD production ^a	High	Moderate	Rare	Low
Impact on SOM ^b	High (–)	High (–)	High (–)	Moderate (+)
Heterogeneity creation	Low	High	Moderate	Low
Interval (years) ^c	100-400	30-100	5-30	40-160 ^d
Similarity to biomass harvesting	Moderate	Low	Rare	—
Similarity to other silvicultural treatments	Even-aged management	Uneven-aged management	Fuel reduction treatment	—

^a Snags and fallen CWD; CWD = coarse woody debris

^b Both organic layer and soil layer were lumped together; SOM = soil organic matter

^c Arno and others 2000

^d Rotation cycle of ponderosa pine (source: Tappeiner and others 2007)

fire produces 18 to 31 Mg/ha of coarse deadwood on the soil surface. In contrast to wildfires, biomass harvesting provides little coarse woody debris except through the remaining stumps and any snags left as wildlife trees unless it is a requirement of the timber sale contract.

The dissimilarity between biomass harvesting and fire regimes is apparent in what remains in the understory and belowground layers. Even low severity fires kill most understory vegetation (Oliver and Larson 1996), whereas biomass harvesting can result in minor changes to understory biomass and diversity. In addition, wildfire burning on the soil surface removes the forest floor and can result in loss of organic matter within the mineral soil (Neary and others 1999). About 75% of total surface organic materials (e.g., forest floor, twigs, leaves, and coarse wood) are depleted after fire in U.S. forests (van der Werf and others 2010). After a moderate-to-severe wildfire in Arizona, approximately 38% of the understory vegetation and 23% of the forest floor remained in a ponderosa pine forest (Campbell and others 1977) and, as noted above, heat that transfers into the mineral soil can disrupt soil microfauna and fine root biomass (Agee 1996). During biomass harvesting, broken branches and twigs as a result of tree falls contribute to fine (<7.5 cm [3 in]) woody material on the soil surface and usually result in only small patches of exposed mineral soil. Such differences in surface organic matter affect mineral soil water balance and chemistry, stream flow, and sedimentation differently (Long 2009). The temporal redistribution pattern of organic matter is also worth considering. The rotation age of ponderosa pine (biomass harvesting) can be calculated at 40 to 160 years based simply on culmination of mean annual increment (Tappeiner and others 2007). This rotation age is comparable to the interval of the mixed severity fire regime (Table 3). However, fire-return intervals of natural ponderosa pine are known to be less than 20 years (Weaver 1959; Agee 1996), which is more similar to the low severity fire regime. This means there is conflict between

the ecological interval to maintain a ponderosa pine stand and the rotation age for timber harvesting. Presumably, these discords can be found in many different species and will probably cause a transition of species composition, resulting in different biomass productions, even from the same site.

Similarity between fire regimes and other silvicultural treatments can be found in terms of structural change after disturbance. Stand-replacement fire regimes resemble even-aged silvicultural systems (e.g., clearcut, shelterwood, and seed tree harvesting) in terms of stand initiation and consequent single cohort generation. From this standpoint, biomass harvesting might be comparable to the stand-replacement fire regime. Correspondingly, mixed severity fire regimes can be linked to uneven-aged silvicultural systems such as group or single tree selection harvest methods. These silvicultural systems can create patches and mimic species-specific mortality. Lastly, various fuel reduction treatments such as prescribed burning, thinning, or releasing treatments (in understory) can emulate low severity fire regime in terms of intensive understory removal. In particular, prescribed fire has been used to mimic this natural disturbance regime (Long 2009). While harvesting and wildfire may be able to produce similar looking stands, it is difficult to equate subsequent stand dynamics as a result of differing soil properties after disturbance.

Insect/Pest attack

The extent of damage from insect or pathogen attacks varies depending on the condition of insect, pathogen, host, and environment. Insect or pathogens may cause immediate mortality or temporary weakness such as defoliation or stunted growth. Even when damage to one tree is initially limited, it can later be killed by successive attacks (e.g., Långström and Hellqvist 1993). Moreover, infected stands are more likely to be disturbed by other agents (e.g., windthrow and wildfire). Understanding these complex interactions and consequences is quite challenging. The following discussion focuses mainly on fatal insect or pest disturbance agents for simplicity.

The insect or pathogen disturbance regimes can be grouped into two classes depending on severity: stand-replacing and gap-scale (Table 4). The most distinctive characteristic of insect or pathogen attacks is the host specificity. Therefore, the severity of stand-replacement by insects or pathogens can occur either on single species (monoculture) or close to single species stands. An example of stand-replacing insect disturbance is the recent mountain pine beetle (*Dendroctonus ponderosae* Hopkins) outbreak in North America. During the last decade, more than 10 Mha of pine stands were severely damaged in the western United States and British Columbia (Meddens and others 2012). The scale of this outbreak is an order of magnitude larger in area and severity than all previously recorded outbreaks, and it resulted in converting forests from a small net C sink to a large C source during and immediately following the outbreak (Kurz and others 2008).

Insect or pathogen attack regimes are distinctly different from fire. Non-fire disturbances scarcely cause a direct loss of organic matter. Due to its host specificity, mortality rates vary by species. Host specificity also results in erratic patch patterns and can increase complexity of the landscape. Unless the disturbance agent kills the seedlings of its host, mortality of understory is negligible. In addition, non-fire disturbances generate a pulse of fine woody debris input due to the loss of foliage

Table 4—Impacts of beetle attack, windthrow, and debris avalanche flow on organic matter and stand structure, with comparison to silvicultural treatments. CWD = coarse woody debris and SOM = soil organic matter.

	Insect/pest attack		Windthrow ^a	Debris avalanche ^b
	Stand-replacing	Gap-scale		
Organic matter extraction	Rare	Rare	Rare	Very High
Overstory tree mortality	Selective/high	Selective/moderate	Selective/low	Most
Understory mortality	Rare	Rare	Rare	High
CWD production	High	Moderate	Moderate	None
Impact on SOM	High (+)	Rare (+)	Rare (+)	High (–)
Heterogeneity creation	Low ^c	High	High	Low
Interval (years)	40-100 ^d	24-46 ^e	100 ^f	2-25 ^g
Similarity to biomass harvesting	Moderate	Low	Low	High
Similarity to other silvicultural treatments	Even-aged management	Uneven-aged management	Uneven-aged management	Biomass harvesting

^a Gap-scale disturbances were considered

^b Swanson and Swanson 1976

^c In case of aggressive epidemic in homogenous stand

^d Spruce bark beetle (Zhang and others 1999)

^e 2-year-cycle budworm (Wong and others 2003)

^f White and others 1985

^g Debris flows (Parrett and others 2004)

(a common symptom of weakened by trees) and the dead trees become snags and ultimately coarse woody debris on the soil surface. These can be interpreted as organic matter transfers from living vegetation to the SOM pool.

As shown in Table 4, gap-scale insect or pathogen outbreak does not seem to be similar to biomass harvesting. Like the mixed severity fire regime, gap-scale insect and pathogen attacks are similar to single or group selection harvest systems. In terms of understory impact, insect/pest attack is more similar to silvicultural systems than to the fire disturbance regime. For stand-replacing, non-fire disturbances, overstory mortality is the one common denominator when compared to biomass harvesting. Even if insect or pathogen outbreaks occur in single species stands, the residual stand complexity generated after disturbance is likely to be greater than after biomass harvesting unless specified during harvest operations.

Wind

Catastrophic wind damage can be an important agent for structural loss in a forest (Everham and Brokaw 1996). Structural and compositional loss within a forest and the formation of various sized gaps are all dependent on wind severity. Both biotic and abiotic factors can be altered during wind events. Tree size, species, and stand condition (canopy, structure, and density), or the presence of insect or pathogen damage are

all important factors that determine wind impacts. In addition, topography and soil conditions are also important site characteristics influencing the type and amount of damage (Everham and Brokaw 1995). This implies that wind disturbance interacts frequently with other agents and often plays the role of “secondary” disturbance (Franklin and others 2007; Sibold and others 2007). Small-scale disturbances play a pivotal role in forming and maintaining the forest stand structure, especially in forests where stand-replacing disturbances are scarce (Lertzman and others 1996).

Severity of wind disturbance can vary from the magnitude of single tree mortality to stand-replacement (e.g., hurricane, typhoon, and tornado). However, stand-replacing wind events are rare in the northern Rocky Mountain region, so we primarily discuss the gap-scale wind disturbance agent here.

Winds cause mortality by either uprooting or breaking the stem. Neither case results in site organic matter losses; however, uprooting can damage understory vegetation, expose the mineral soil, and bring subsurface rocks closer to the surface. Unless there is a large wind event, the affected area is restricted since nearby soil horizons and understory vegetation can remain intact (Franklin and others 2007). Fallen and broken trees simultaneously increase the coarse woody debris organic matter pool and decrease the living organic matter pool. Since gap-scale wind disturbance produces relatively small sized patches in general, spatial heterogeneity of the stand can increase (<200 m²; White and others 1985).

Unless the spatial pattern of damage is large-scale, there are not many similarities between wind disturbances and biomass harvesting. Common traits exist only in terms of intact understory and soil layer. Rather, uneven-aged management (single tree or group selection) likely has more site similarities to the gap-scale wind disturbance regime. However, single tree selection cannot emulate wind disturbance perfectly since, in most cases, it cannot generate coarse woody debris and localized soil perturbation (Franklin and others 2007).

Debris avalanche

Soil erosion processes are one of the major disturbance agents in the northern Rocky Mountain region. The term “debris avalanches” refers to rapid soil mass-water (plus vegetation) movement from hillslopes, such as landslides, avalanches, and debris flows (Swanston and Swanson 1976). Although these disturbance agents may play a pivotal function in a specific area (e.g., determination of timberline by avalanche), very few studies pertain to the impacts of these disturbances on ecosystem function (Attiwill 1994).

Similar to biomass harvesting, debris avalanches remove a sizable amount of living organic matter and generate high mortality regardless of canopy strata. Since almost all aboveground material (and even some mineral soil close to the surface) is swept to a lower hillslope position, generation of on-site coarse woody debris is limited.

Regardless of these similarities, debris avalanches are different from biomass harvesting in that they result in detrimental changes to the understory, forest floor, and mineral soil. These soil and site changes substantially hinder regeneration and understory vegetation recovery, leading to reduced productivity (Grigal 2000). Given the nature of avalanches, these disturbance agents are quite dissimilar to the other silvicultural treatments.

Comparing Impacts of Biomass Harvesting and Natural Disturbance Agents on Forest Productivity

If nutrient losses are compensated by nutrient influx, then the impact of a natural disturbance on productivity can be determined by its influence on the SOM pool (Tables 3, 4). Following natural disturbances, the amount of newly produced coarse woody debris and the magnitude of perturbation on understory vegetation can affect productivity as well. In this manner, we can compare the potential impact of natural disturbances on productivity.

As previously mentioned, debris avalanches are the most devastating natural disturbance on forest sites. They remove aboveground vegetation and can also remove most of the forest floor and some mineral soil, thereby having a highly detrimental effect on forest productivity. Moreover, since debris avalanches are more likely to occur on productive sites (e.g., soil-water accumulated zone), the magnitude of negative impacts might be greater than on lower productivity sites. In a conifer forest of British Columbia, a landslide site had a 70% reduction of wood volume production compared to adjacent harvested stands in the first 60 years (Smith and others 1986). In addition, Megahan (1990) noted that the site impacts from landslides in the northern Rocky Mountain region are similar to those in British Columbia.

Fire can also adversely impact forest productivity. Potential detrimental impacts include: interruption of litter fall from the overstory, consumption of organic matter in the forest floor and soil layer, disruption of belowground biota, and increased leaching due to lack of aboveground vegetation. However, fire can also create several beneficial conditions to promote vegetation production. These impacts, whether detrimental or beneficial, are directly related to and determined by the extent, duration, and severity of the fire.

Wind, insects, or pathogens can have somewhat similar consequences on forest productivity (Table 4). For these disturbance agents, little organic matter is lost; dead overstory trees stay on-site as either snags or fallen logs, and detrimental impacts on understory and belowground layers are rare.

In summary, biomass harvesting may have a stronger negative impact on productivity than insect or pathogen outbreaks and wind, but less severe impacts as compared to fire since biomass harvesting can conserve the understory vegetation, forest floor, and soil layer. Therefore, the rank of detrimental impacts among the disturbance agents can be seen as: debris avalanches>fire>biomass harvesting>winds \approx insect/pest attack, but the temporal and spatial distribution of the disturbance agent must be considered. Note that adverse consequences depend on the intensity of disturbances, and other site conditions can substantially ameliorate or aggravate the negative impacts.

Biomass Harvesting and Forest Productivity: Key Findings

Impact of Biomass Exportation

Nutrient removals during timber harvesting can be substantial, especially where whole-tree harvesting is practiced (Ballard 2000). Most western forests are considered to be N deficient (Binkley 1991), so removal of a large proportion of N (or other

nutrients) during harvesting may be a cause for concern on many forest sites in the Inland Northwest (Weetman and Webber 1972; Foster and Morrison 1976). On sites around the country, other macronutrients, such as P (e.g., Yanai 1998) and K (e.g., Goulding and Stevens 1988) have decreased after biomass harvesting. Usually loss of macro- or micronutrients is associated with the removal of tops (leaves and branches) during harvesting. These portions of the tree contain much higher nutrient concentrations than boles. Generally, plants allocate the highest concentration of nutrients into foliage and branches and the lowest concentration into the stem (Table 5; Farve and Napper 2009). This has been reported in various studies from mixed hardwood forests in Wisconsin (Pastor and Bockheim 1984) and New Hampshire (Whittaker and others 1979), to Douglas-fir in British Columbia (Pang and others 1987). Therefore, biomass harvesting that removes whole-trees or operations that leave tops in slash piles likely remove more nutrients than bole-only harvesting and scattering slash back on the harvest site. Alban and others (1978) asserted that the whole-tree harvesting removes 2 to 11 times more nutrients than conventional harvesting in northeastern forests. In an upland mixed oak forest in Tennessee and coniferous forests in Maine, whole-tree harvesting resulted in removal of three times more nutrients than sawlog (bole only) harvesting (Johnson and others 1982; Smith and others 1986). This was also a consistent response for site nutrient changes in northern Europe (Mälkönen 1976).

Calcium (Ca) is one nutrient that is vulnerable to whole-tree harvesting throughout the United States (Boyle and others 1973; Johnson 1982; Federer and others 1989). This is because the amount of Ca in the soil pool is often relatively low, and, in contrast, the accumulation of Ca in aboveground vegetation for several species (e.g., aspen [*Populus* spp.], sugar maple [*Acer saccharum* Marshall], and white spruce [*Picea glauca* (Moench) Voss]) is significant. Various studies have indicated that Ca is the nutrient most likely to be lost even though it seldom limits plant growth in natural condition (Farve and Napper 2009).

The key question in determining whether biomass harvesting would deplete nutrients in the long term is whether the amount of nutrient influx through precipitation and fixation can balance the nutrient requirement for vegetation growth throughout the rotation. Annual accumulation of nutrients in vegetation shows similar magnitudes

Table 5—Nutrient distribution in aboveground tree components of 34-year-old Douglas-fir (from Pang and others 1987; Farve and Napper 2009). (Values are rounded.)

Tree component	Nutrient concentration (%)				
	N	P	Ca	K	Mg
Current foliage	29	26	14	29	28
Old foliage	26	39	29	25	27
Current twigs	21	17	14	19	20
Branches	10	7	17	11	10
Bark	7	7	10	12	8
Dead branches	6	3	15	2	6
Wood	2	1	1	2	1

with nutrient influx through precipitation in general temperate forests (Alban and others 1978). In the northern Rocky Mountain region, Stark (1980) estimated that the amount of nutrients removed during biomass harvesting did not exceed the amount that would likely be recovered within the next rotation (70-100 years) of Douglas-fir stands. However, this depends on the nutrient pools within the mineral soil and forest floor (Alban and others 1978; Farve and Napper 2009). Therefore, on soils that are nutrient limited branches and leaves or needles should remain on-site until the nutrients leach into the soil.

Impact of Living Organic Matter Removal

Openings in the forest canopy after harvesting will alter site microclimate and affect understory growth. Increased solar radiation results in increased soil temperature. In addition, once the forest canopy is removed there is decreased transpiration and rainfall interception, which can increase soil moisture. Consequently, decomposition and mineralization processes will be accelerated. Unless the soil available nutrients are taken up and stored immediately by vegetation, it is likely they will be leached deeper into the soil profile or into groundwater. Therefore, prompt recovery of vegetation either by natural regeneration or planting after harvesting is critical to prevent leaching. Removal of the overstory reduces fresh litter inputs and combined with accelerated decomposition for a warmer, wetter soil condition, the forest floor also decreases. In the mineral soil layer, fine root turnover will be temporarily interrupted; however, a pulse of non-living organic matter (roots) input will occur immediately after harvesting.

Impact on Soil Physical Properties

One of the most adverse impacts of harvesting operations on soil productivity is compaction caused by heavy machinery trafficking (Janowiak and Webster 2010; Page-Dumroese and others 2010). More intensive harvest operations that use numerous skid trails or drive to each tree will exacerbate soil compaction under many soil conditions. Compaction increases soil bulk density, hampering air movement and water permeability (Thibodeau and others 2000). In compacted soil, soil fauna activity and fine root development are restricted considerably. Moreover, as previously mentioned, since the majority of fine root distribution and microbial activity are concentrated within the shallow range of topsoil, the negative impact of compaction on forest productivity is substantial (Page-Dumroese and others 2010).

Although diverse soil factors affect the susceptibility to compaction (Page-Dumroese and others 2010), soil texture is a primary determinant (Powers and others 2005). In general, soils with a high clay (<0.002 mm) content are easily compacted but fine-textured ash-cap soils are also at risk (Johnson and others 2007). Increased soil moisture also makes soil particularly susceptible to compaction and forest residues may have to be used on skid trails when soil moisture is high (Han and others 2006). In addition, SOM can play an important role in ameliorating compaction susceptibility since it can improve the soil structure, aeration, permeability, and activity of soil organisms. In mixed conifer stands of British Columbia, Hope (2007) found that soil compaction was recovered in 10 years. Initial differences in soil bulk density caused by harvesting operation disappeared, and planted lodgepole pine (*Pinus*

contorta Douglas ex Loudon) and hybrid spruce (*Picea glauca* [Moench] Voss × *Picea engelmannii* Parry) seedling growth was no different after 10 years. The author hypothesized the amelioration of soil compaction is related to the amount of SOM contributed by abundant fine roots. A similar result was observed in the lodgepole pine forest of British Columbia (Simard and others 2003). Harvest operations should be conducted with consideration of compaction susceptibility and sufficient amounts of organic matter must be retained to maintain soil productivity.

Impact on Soil Chemical and Biological Properties

After harvesting, increased decomposition rates coupled with the interruption of litter inputs will decrease both forest floor and mineral soil SOM pools and alter mineral soil chemical properties. With a decrease in SOM comes a concomitant reduction in CEC resulting in nutrient leaching. In addition, whole-tree harvesting can increase the C:N ratio (Olsson and others 1996), making the immobilization of N more dominant (Farve and Napper 2009) and resulting in delayed aboveground vegetation recovery. The longer the delay in regeneration, the more nutrients are lost. Therefore, on many sites in the Inland Northwest (particularly poor soils) it may be critical to replant after harvesting to take advantage of mobile soil nutrients and mycorrhiza formation (Molina and Amaranthus 1987).

Timber harvesting can alter soil acidity in a way that affects soil productivity. Soil pH generally increases after harvest (Jurgensen and others 1997), explained by: (1) transformation of humus, (2) release of cations during decomposition of organic matter, and (3) utilization of hydrogen ions during the mineralization process (Nykqvist and Rosén 1985). In general, soil microbes prefer more alkaline conditions (Jurgensen and others 1997). However, intensive biomass harvesting can render a soil environment that is relatively more acidic than with conventional harvesting. Soil with low CEC will acidify more easily and, consequently, lose base cations (i.e., leaching). Furthermore, formation of soil aggregate structure can be hindered in acidic soil. Therefore, maintaining the soil organic pool by keeping SOM in the forest floor and mineral soil intact and aiding rapid reforestation are essential for maintaining forest productivity.

Decreases in the SOM pool can also affect the forest's biological properties. The abundance and activity of soil organisms would likely be limited in cases of deficient food and habitat resources. Bengtsson and others (1997) found that intensive harvesting decreased the abundance of soil arthropods in a Scots pine stand in Sweden 15 to 18 years after harvesting. Since soil fauna can contribute significantly to N mineralization (Anderson and others 1983; Verhoef and Brussaard 1990), the decrease of soil mesofauna might result in decreased productivity. Among those soil organisms, ectomycorrhizae are critical. Mycorrhizal fungi improve seedling survival and growth by enhancing the uptake of nutrients and water, and help protect against pathogens (Harley and Smith 1983). However, without a host, mycorrhizae can hardly survive more than 3 years (Wiensczyk and others 2002). Short-rotation, repeated intensive biomass harvesting may negatively impact ectomycorrhizal abundance (Mahmood and others 1999). Successful reforestation depends on the capacity of tree seedlings to capture site resources early, which also assures space to grow and adequate tree vigor necessary to survive insects, pathogens, and climatic stressor (Molina and Amaranthus 1987).

Climate Change and Biomass Harvesting

Running (2009) expects future climate change in the northern Rocky Mountain region to be manifested by longer growing seasons due to increased temperatures, less snow, more rain, and longer summer drought. Furthermore, elevated CO₂ levels are expected to increase photosynthetic rates. In this scenario, nutrient cycling processes such as mineralization and nitrification would be stimulated by increased temperatures. At first glance, forest productivity in the northern Rocky Mountains would appear to increase; however, increased summer drought would limit forest productivity. Whether biomass harvesting would adversely affect forest productivity under changing climate depends on if retained soil and organic matter provide ample nutrients and water to vegetation.

If Running's prediction is realized, forest productivity decreases due to drought may constrain biomass harvesting levels. However, if vegetation production increases in response to elevated CO₂, then nutrient demands may increase. In both scenarios, retained organic matter on the forest floor and in the mineral soil enhances beneficial soil properties such as water holding capacity during summer drought. Thus, if biomass harvesting damages soil properties or decreases the soil nutrient pool, climate change will exacerbate the negative impacts on forest productivity.

Discussion

In Johnson and Curtis' (2001) meta-analysis, they noted that whole-tree harvesting decreases soil C and N by 6%, whereas conventional sawlog harvesting increases both C and N by 18%. Nitrogen reductions from harvesting may need to be offset by fertilization treatments (Himes and others 2014). Current empirical experiments have reported no loss of soil C with increasing biomass removal intensity (e.g., Powers and others 2005; Nave and others 2010; Thiffault and others 2011). Also, Olsson and others (1996) found no difference in soil C in relation to biomass removal intensity in Scots pine and Norway spruce (*Picea abies* [L.] Karst.) stands in Sweden 15 years after harvesting.

In European experiments, differences in stand production between whole-tree harvesting and conventional harvesting are more commonly detected. In a Sitka spruce plantation in England, Walmsley and others (2009) observed a 10% reduction in dbh increment after whole-tree harvesting. Proe and others (1996) found 32% tree volume reduction of planted Sitka spruce seedlings 14 years after whole tree harvesting in England. In Scandinavia, Egnell and Leijon (1999) and Egnell and Valinger (2003) also observed a consistent reduction of Norway spruce and Scots pine tree growth after whole-tree harvesting. In addition, Jacobson and others (2000) found a significant difference in tree volume growth in pine and spruce stands 10 years after harvesting. These authors speculate that increased N immobilization due to the presence of logging residues may counteract growth retardation due to N removal after whole-tree harvesting.

The North American Long-Term Soil Productivity (LTSP) study results often do not follow those of the European studies. The LTSP research network did not show

any significant impacts of intensive biomass removal on vegetation production after 10 years (Powers and others 2005). In addition, a western larch forest in Montana showed results consistent with LTSP 38 years after intensive biomass harvesting (Jang and others 2013).

Responses to intensive harvest operations differ by site and species. For example, within the LTSP network some loblolly pine stands showed a different response (i.e., consistent volume growth reduction) as compared to other stands (Powers and others 2005). Likewise, Scots pine stands were less consistent than Norway spruce stands in European studies (Egnell and Leijon 1999). In that sense, the difference of Johnson and Curtis' (2001) analysis might not be related to harvest intensity as much as species (i.e., conifer versus hardwood).

There are still numerous gaps in our understanding of intensive biomass utilization for bioenergy production, nutrient cycling, stand productivity, and soil quality. In addition, local climate regimes and long-term climatic changes that interact with biomass harvesting, thereby influencing SOM decomposition rate, is also important to consider. Climate change can impact both input and output sources of SOM pools. Specifically, it may increase primary production of vegetation which would increase input to SOM pools and increase the decomposition rate of belowground organic matter, resulting in simultaneously increased output of the SOM pool. Thus, if the contribution of increases in primary production to SOM pools exceeds the increase in output of organic matter through elevated decomposition rate, then climate change would not result in adverse consequences on forest productivity or vice versa. However, there remains controversy regarding the potential impact of climate change on the SOM cycle (Davidson and Janssens 2006). This is a key research need which is critical to prevent undesirable consequences of biomass harvesting on our ecosystems.

Management Implications

There is abundant information on the prediction of long-term impacts of intensive biomass removal on forest productivity. However, the empirical knowledge and comprehensive understanding, especially on western forests, are limited thus far. Therefore, we used the available findings to evaluate potential impacts of increased biomass extraction on western forests. In summary:

- 1. In general, the long-term impacts of intensive biomass harvesting for bioenergy production will likely cause few impacts on within-stand nutrient cycling if the forest floor and mineral soil are protected** (Page-Dumroese and others 2010). This implies that sufficient nutrients are stored in the mineral soil, and inputs through precipitation or dry deposition are adequate to support the nutrient requirements of the subsequent rotation.
- 2. There is a risk of nutrient losses from many forest ecosystems.** Excess nutrients from the addition of needles, leaves, and branches will likely be leached through the mineral soil unless rapid re-vegetation occurs (Boyle and others 1973). Increased leaching can result in one or more nutrient deficiencies, and subsequent seedling or shrub growth can be stunted (“nutrient shock” ;sensu Stark 1980).

- 3. Favorable conditions created after harvesting can accelerate immediate understory vegetation recovery.** In a clearcut western larch stand in Montana, 37% of pre-harvest shrub volume recovered within 4 years after harvesting. Remarkably, in the understory protected treatment, 62% of pre-harvest shrub volume recovered during the same period (Schmidt 1980). Such a rapid re-vegetation of understory has been reported to play an important role in maintaining forest productivity in the early stand developmental stage (e.g., Turner and Long 1975). However, shrub growth may compromise tree seedling regeneration and planting may be necessary to achieve the desired stand conditions and appropriate species to withstand a changing climate.
- 4. These general conclusions may not be applicable to every forest and site condition; some forest sites likely require further attention.** Although many sites in the Inland Northwest may respond similarly, other sites are at-risk or sensitive to harvest operations. While these sites and stands can still be managed, extra care may be needed to minimize impacts. For example, dry, low productivity sites may have large soil organic pools, but have less nutrient inputs through precipitation and dry deposition (Jurgensen and others 1997; Page-Dumroese and Jurgensen 2006). Moreover, if precipitation occurs when plants are dormant, the risk of nutrient leaching increases (Alban and others 1978). Conversely, wetter and warmer sites of higher productivity tend to have shallower organic layers because of higher decomposition rates. Reductions in the forest floor layer occur rapidly if harvesting accelerates those decomposition rates. On those sites, it is critical to retain the forest floor during harvest operations and to recruit organic horizon parent material (woody residue). For example, Jurgensen and others (1997) argued that larger amounts of woody residue should be retained on moist sites (22-36 Mg/ha) as opposed to dry sites (10 Mg/ha). In addition, certain forest soils should be harvested with a high degree of care, particularly those that: (1) are not resilient to ground-based harvest systems, (2) are relatively infertile, (3) are compaction prone, (4) are exposed to short fire return intervals, (5) have insufficient regeneration sources and understory vegetation, or (6) support tree species that demand high soil nutrient concentrations (Page-Dumroese and others 2010).
- 5. Conventional silvicultural treatments should be considered for biomass-harvested sites.** For successful regeneration, site preparation treatments such as prescribed burning, forest floor scalping, or planting are often required for some species (e.g., western larch) to achieve adequate regeneration. Although these treatments have the potential to remove a large portion of the forest floor, spot treatments or mosaics of forest floor and mineral soil may provide enough access to nutrient and water that the impacts can be minimized. On some sites, fuel reduction treatments, such as repeated thinnings, can result in insufficient nutrient cycling for healthy and productive forests (Page-Dumroese and others 2010) and this should be determined prior to harvest operations.

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