Litter Decomposition under Logging Residues in a Thinned Inland Northwest Forest

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Abstract

As interest in wood bioenergy increases, the demand for woody residues also increases leading to ecological concerns about soil productivity and site sustainability. Removal of woody residue after harvesting can reduce litter decomposition rates by altering physical conditions of forest surface and N input. Thus, I studied the effects of biomass (residue) retention levels from thinning forests on changes in site physical conditions (e.g., temperature and moisture) and changes in litter decomposition and nutrients dynamics. In addition, I examined the compensating effect of fertilizer N additions. Decomposition was measured using a 172-day litterbag experiment in two forest stands which were thinned in 2013 and fertilized in 2014. Woody residue retention levels were: normal (1x), doubled (2x), or removed (0x), and there were adjacent unthinned control plots. The litterbags were placed on the forest floor in 2016.

The 0x retention level had litterbags with the lowest moisture and respiration, but the highest decomposition rate (k) and mass loss as compared to the control. Litterbag nitrogen (N) had the greatest losses in the 0x retention plots. There was no evidence that fertilization increased litter decomposition rates of litterbags. This litterbag study indicates that removal of woody residues from conifer forests in northern Idaho will not alter litterbag decomposition rates.

Keywords: Woody bioenergy, Bioenergy feedstock, Residue removal, Fertilization, Nutrients, Carbon, Nitrogen, Northwestern forests

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Dedication

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Chapter 1. Literature Review

Globally, forests provide numerous ecosystem services such as forest products (e.g. timber, paper, food), wildlife habitat, greenhouse gas reduction (carbon sink), and climate change mitigation (Maser and Trappe, 1984; Daily, 1997; Klein et al., 2013). One additional ecosystem service that is gaining more interest is bioenergy. Bioenergy is an alternative to fossil fuels that avoid increasing atmospheric carbon (C) (Mann and Tolbert, 2000; Demirbas, 2009). Although bioenergy may help reduce atmospheric C concentrations, there is concern that increased biomass removal will decrease forest productivity (Stupak et al. 2008; Page-Dumroese et al., 2010; Jang et al., 2015). The retained biomass residues after harvest operations will support the decomposition in the forest by modifying physical conditions on the forest floor. However, if too much biomass is removed, OM decomposition process may be altered (Finér et al. 2016), subsequently resulting in decreases in the sustainability of forest which mean maintaining both past productivity and services of forests for future generations and ecological integrity and healthy forests (Ranger and Turpault, 1999).

The limits for biomass retention in various ecosystems are not known. Therefore, it is critical to establish site-specific criteria for managing forest residues to achieve sustainable forestry during the production of bioenergy. A detailed understanding of the mechanisms that control decomposition after harvesting will help determine limits for biomass removals, the effects of residue removal on OM decomposition, and how decay rate changes may impact sustainable forestry.

1.1 Sustainable forest management

To attain continuous provision of forest products, land managers must practice sustainable forestry. Sustainable forestry means maintaining past productivity and services for future generations while maintaining ecological integrity and healthy forests. Maintaining ecological integrity and healthy forests can be realized through forest management. Well-managed forests are an opportunity to

improve ecosystem functions and forest productivity: they improve water quality or conserve watershed function, mitigate climate change, and increase forest tree growth by supporting nutrient dynamics (Sedell et al., 2000; Klein et al., 2013; Lundmark et al., 2014; Purahong et al., 2014). However, some managed forests do not show any improvement or, instead, show degradation after forest management such as forest fertilization and thinning. Forest thinning and fertilization practices are examples of forest management practices that can improve forest productivity. Forest thinning is a common silvicultural practice that reduces stand density, increases tree-level wood production, reduce fire risk, and increases resistance to pests (Drew and Flewelling, 1979; Mitchell et al., 1983; Baldwin et al., 2000; Agee and Skinner 2005; Tappeiner et al., 2007). Tree growth is increased by thinning because it alleviates high stand density and, in turn, competition for site nutrients and sunlight (Brix, 1981, 1983; Chase et al., 2016). In addition to tree growth, adequate thinning can improve the economic value of forests by increasing the residual tree physical qualities (e.g., form or growth) and stand genetic quality (e.g., removing undesirable trees; Zeide, 2001; Finkeldey and Ziehe, 2004). Also, proper thinning can improve the forest physical and chemical condition since thinning can increase the amount of sunlight and precipitation that reaches the forest floor, which are essential resources for plant growth (Prescott, 2002; Ma et al., 2010). However, inadequate or excessive thinning can reduce biodiversity (Chaudhary et al., 2016) and can lead forest to destruction (Piticar, 2016). To be specific, a heavily thinned spruce stand (after the stand age of 30) showed decreased stand stability and resulted in lower residual tree productivity (Piticar, 2016). Residue produced by thinning operations can be removed from the site for bioenergy production, but too much or repeated biomass removals can negatively affect decomposition and nutrient cycling (Stupak et al. 2008; Page-Dumroese et al., 2010; Jang et al., 2015).

Another important example of forest management is fertilization. Proper fertilization provides forest ecosystems with the limiting nutrients (Peterson & Peterson 1995) and stimulates microbial activity (Van Cleve & Moore 1978). As a result, fertilization increases tree growth and enhances the economic

value of stands (Peterson & Peterson 1995). However, excessive or improper fertilization can lead to an imbalance of forest nutrients, depression of soil mesofauna or microbial community, or pollution of groundwater or streams (Coleman et al., 2014; Berch et al., 2006; Fox et al., 2007; Erisman et al., 2013). Also, excessive or improper N fertilizer cause increased tree mortality. For example, a high rate of N fertilization without other nutrient decreased forest productivity and increased tree mortality while a balanced fertilizer formulation increased tree growth and forest productivity (Coleman et al., 2014). These examples of forest management suggest us that the appropriate fertilization should be identified to provide continuous forest products (Fox, 2000).

1.2 Biomass removal following thinning and its impact on decomposition

Wood bioenergy is an alternative to fossil fuels and is known to benefit atmospheric C. Compared to petroleum resources, woody bioenergy can mitigate greenhouse gas (GHG) emissions and balance atmospheric CO_2. For example, when petroleum resources are combusted to create energy, C is released into the atmosphere which is previously sequestered deep in the earth (International Energy Agency (IEA), 2005). The emitted C from petroleum resources increases atmospheric C. Similar to petroleum resources, wood bioenergy produces energy from combustion. During combustion of woody biomass, large amounts of C are also released into the atmosphere. However, unlike petroleum resources, bioenergy from woody biomass does not increase overall atmospheric C. This is because the released C from the combustion of woody biomass originates from the atmosphere. More specifically, trees can capture C in the atmosphere through photosynthesis and the captured C is kept in the cell of trees. Therefore, when the woody biomass is burned, C returns to where it came from. Thus, bioenergy may not alter the overall balance of atmospheric CO_2. In addition, the released CO_2 can be used by trees for production because trees can recapture more CO_2 during periods of rapid growth than what is emitted from woody biomass combustion (Union, 2009). Therefore, bioenergy can have a positive effect on the mitigation of GHG emissions.

In addition to benefits of atmospheric C, bioenergy also decreases risks. For example, after thinning or harvesting, non-merchantable tops, stumps, snags, and woody debris are generally left on site or burned for the nutrient supplement, decrease runoff or erosions, or to protect the soil surface from raindrop splash (Henderson, 1995; Powers et al., 2005). However, adding too much harvest residues can increase the possibility of wildfire, disease, or insects. Using wood that would be left on-site or burned for bioenergy requires more site-specific data to determine changes in forest sustainability (Fcllin, 1979; Thies & Russell 1983; Harvey, 1994; Mead, 2005; Perlack and Stokes, 2011).

Removal of small-diameter woody biomass after thinning has become more common in many forest as the demand for woody biomass energy has increased (Benson and Schlieter, 1980; Barger, 1979; Janowiak and Webster, 2010; Perlack and Stokes, 2011; Berger et al., 2013). However, as interest in biomass removal increases so do ecological concerns. Among the many ecological concerns with biomass removal are changes in ecosystem functions and forest productivity or sustainability (Jang et al., 2015). Forest productivity is supported by natural nutrient cycles. These nutrient cycles are largely maintained by decomposition of organic matter (OM). Organic matter includes dead animal and plants, leaves, and woody debris and each OM was a component of plants, trees, and animals when they were alive. Once plant and tree die naturally or through thinning or harvesting, they decompose into soil OM. Organic matter contains many of the essential nutrients in the original living organisms from which they were derived. Organic matter decomposition can continuously supply essential nutrients for the plants and trees. (Alban et al., 1978; Klockow, 2012; Wall, 2008). Although some leached organic nutrients can be directly absorbed by the plant, other nutrients must be transformed into plant-available forms through a transformation of OM into soluble forms (Didham, 1998; Wardle et al., 2003; Chapin et al., 2011). The decomposition processes can be classified into 3 types: leaching, fragmentation, and chemical alteration. Leaching is the process that removes soluble nutrients in OM and mineral soil during precipitation events. Conversely, nutrients can be added when dissolved nutrients in precipitation are directly absorbed by plants after they reach the soil. OM fragmentation is

the process that turns large OM into smaller particles or removal of the cuticle or bark. Fragmentation is accelerated by soil animals or an active freeze and thaw cycle (Chapin et al., 2011). Chemical alterations of OM are conducted by microbial decomposers such as bacteria and fungi. Microbial decomposers secrete enzymes that break down OM into plant-available forms of nutrients. Through these 3 types of decomposition process, in many biomes, about 90% of total amount of the essential nutrients (nitrogen (N), phosphorus (P), potassium (K), and calcium (Ca)) absorbed by plants are supplied by nutrients recycled (Chapin et al., 2011). The remaining 10 % of essential nutrients in forest ecosystems are from other sources such as deposition/fixation or mineral weathering (Chapin et al., 2011). Therefore, OM decomposition plays a critical role in sustaining forest nutrient cycling (McGill and Cole, 1981; Harvey et al., 1987; Powers et al., 1990).

Decomposition may be partially supported by woody biomass retention during thinning or harvesting operations. When the residues are left on-site, they may alter physical environments such as temperature and moisture on the forest surface and in the soil (Henderson, 1995; Powers et al., 2005). More specifically, logging residue creates shade on the forest floor which can lower temperature and prevent evaporative losses (Powers et al. 2005). In addition, biomass retention can provide available water, since woody residues hold five times more available water than mineral soil in a northern Idaho Douglas-fir (*Pseudotsuga menziesii* Mirb. Franco) stand (Page-Dumroese et al., 1991). Changes in the physical condition of the forest floor resulting from biomass retention may increase OM decomposition because climate factors (temperature and moisture) have critical controls over OM decomposition (Swift et al., 1979; Wickland et al., 2008). Rising temperature with an appropriate range of moisture content increases OM decomposition rate; however, without proper moisture content, the increased temperature does not increase OM decomposition (Murphy et al. 1998; Chapin et al., 2011). This is mainly because temperature and moisture play an important role in regulating the metabolism of microbial decomposers (Swift et al., 1979; Lavelle et al., 1993; Lloyd and Taylor, 1994; Chapin et al., 2011). Therefore, abundant moisture is crucial for the establishment of forest

productivity especially on sites that experience a summer drought which can impede OM decomposition (Padilla and Pugnaire 2007). In other words, removal residue on the site reduces moisture content and the activity of microbial decomposers (Hicks, 2000; Hicks et al., 2003; Chapin et al., 2011), consequently, reducing OM decomposition rate.

Inland Northwestern forests in the USA are considered as a potential location for the woody biomass energy industry which may result in increased use of woody residues. However, there is potential on some sites where all woody biomass is removed for bioenergy to have negative impacts on forest sustainability due to altered microclimate and nutrient cycling (Grigal, 2000; Thiffault et al., 2011; Lewandowski et al., 2016). Although biomass removal after thinning seems to negatively impact site OM and ultimately decomposition, there is scant evidence in the literature that point to the adverse effect of biomass removal on OM decomposition (Grigal, 2000; Prescott et al., 2002; Thiffault et al., 2011). Therefore, it is necessary to investigate the impact of small woody biomass removal after thinning on OM decomposition. In addition, an alternative treatment to compensate lost nutrients due to the residue removal should be considered to reduce negative impacts when woody residue should be removed.

1.3 Nitrogen fertilization effects on decomposition following biomass removal.

Removal of harvest residues may result in a shortage of N. This is because woody residue retention after thinning may provide a slow, but a steady supplement of N through long-term decomposition (Piatek et al., 1999; Blumfield and Xu, 2003; O'Connell et al., 2004). Therefore, biomass removal leads to less N inputs. The reduced N inputs can lead to decreased OM decomposition rates. This is because N is an essential nutrient for microbial decomposers to increase their biomass and to produce energy and enzymes which are required for breaking down components of organic matter (Boberg, 2009). Thus, the subsequent loss of N can negatively affect the activity of microbial decomposers. More specifically, since microbial decomposers have their own elementary C/N ratio (stoichiometry

ratio), when N becomes limited, microbial decomposers must find N from new sources to fulfill their own stoichiometry ratio of C/N (Smolander et al., 2008). The microbial stoichiometry of the C/N ratio refers to how much N is required for microbial decomposers to build microbial biomass. In general, C is high enough in both soil and OM; however, N is not. If microbes cannot fulfill their own stoichiometry C/N ratio, the decomposition can be slower. This is because they should acquire enough N from OM or the soil (N limited condition, high C/N ratio) otherwise they must find other sources of N until they have enough N to run their metabolism and decompose OM. Therefore, C/N ratio is one of the more important factors for OM decomposition because decomposition rates can increase or decrease as the ratio changes (Berg and McClaugherty, 2008). Specifically, when N is limited, C/N ratio increases and in turn, the rate of microbial decomposition can be slower (Smolander et al., 2008). In this regard, biomass removal can have the negative influence on the rate of decomposition and ultimately forest productivity (Piatek et al., 1999; Grigal, 2000; Helmisaari et al., 2011; Lewandowski et al., 2016).

This negative impact of residue removal can be more detrimental to sites with nutrient deficiencies (Ranger and Turpault, 1999). For example, N deficiency is common in the northwestern USA soils (Fenn et al., 1998) where forest soils are acid soils and usually poor in mineral nutrients, particularly N (Tate, 1992). As noted above, N deficiencies may limit litter decomposition and primary production (Binkley, 1991). As a result, woody residue removal during harvest operations can exacerbate N-limited condition in northwestern forests by decreasing decomposition rate leading to a negative effect on forest productivity.

One strategy to restore both reduced N inputs and decomposition rate associated with residue removal is fertilizers which can directly supply nutrients to the soils (Helmisaari et al., 2011). In forests, N deficiency can be solved by N addition since the application of fertilizer can increase N directly to the soil and indirectly to leaves after uptake. Both increased N in the leaves and endogenous N can alleviate N limited conditions as well as accelerate OM decomposition in forests by supporting

microbial decomposers (Safford and Czapowskyj, 1986; Perala and Laidly, 1989; Prescott et al., 1999; Brockley, 2006; Berg and McClaugherty, 2008). Based on these results, it is expected that N fertilization can counter the negative effects of biomass removal on decomposition rate and satisfy microbial decomposer N needs (Shafii et al., 1990; Mika and Vander Ploeg, 1991; Fan et al., 2002). Therefore, fertilization can maintain nutrient cycles even in the environment where decreased N results from biomass removal.

Recently, land managers seek to increase woody biomass fuel production in the western USA during thinning operations. While forest managers apply N fertilization for timber production, the application of N fertilization for timber production may help to support woody biomass fuel production by providing N which can be reduced by biomass removal. However, there is concern regarding the indiscreet application of N because reliance on N fertilizer on sites without woody biomass retention may result in soil acidification or N leaching into groundwater (Erisman et al., 2013). In addition, the impact of N fertilization on surface decomposition rates is not clear (Perala and Laidly, 1989; Prescott et al., 1999; Brais et al., 2002; Mariani et al., 2006; Jang et al., 2015). Therefore, more research is necessary to investigate the combined impacts of biomass removal and N fertilization on-site processes and productivity.

To summarize, bioenergy from woody biomass can have a positive effect on atmospheric CO_2 but can have either positive or negative effect on forest ecosystem function, depending on site-specific properties and the amount of material removed. The negative effect of residue removal, especially on surface processes such as litter decomposition, is not clear. Therefore, there is a need to determine how woody residue removal for bioenergy affects litter decomposition and how much residue should be retained to maintain litter decomposition in both the short- and long-term. To assess short-term changes, I assessed the limits of residue retention in an Inland Northwest forest to establish specific criteria for managing forest residue to support excess residues being used for bioenergy production. In

addition, I will assess whether N fertilization can maintain litter decomposition by compensating for less N input due to residue removals.

References

Agee, J. K., & Skinner, C. N. (2005). Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*, *211*(1), 83-96.

Alban, D. H., Perala, D. A., & Schlaegel, B. E. (1978). Biomass and nutrient distribution in aspen, pine, and spruce stands on the same soil type in Minnesota. *Canadian Journal of Forest Research*, *8*(3), 290-299.

Baldwin, V. C., Peterson, K. D., Clark, A., Ferguson, R. B., Strub, M. R., & Bower, D. R. (2000). The effects of spacing and thinning on stand and tree characteristics of 38-year-old loblolly pine. *Forest Ecology and Management*, *137*(1), 91-102.

Barger, R. L. (1979). The forest residues utilization R&D program. In *Harvesting and utilization opportunities for forest residues in the northern Rocky Mountains. Symposium proceedings* (pp. 28-30).

Benson, R. E., & Schlieter, J. A. (1980). Logging residues in principal forest types of the Northern Rocky Mountains [Volume, estimates, forest inventory; Montana; Idaho; Wyoming]. *USDA Forest Service Research Paper INT (USA). no. 260*. PP.14

Berch, S. M., Brockley, R. P., Battigelli, J. P., Hagerman, S., & Holl, B. (2006). Impacts of repeated fertilization on components of the soil biota under a young lodgepole pine stand in the interior of British Columbia. *Canadian Journal of Forest Research*, *36*(6), 1415-1426.

Berg, B., & McClaugherty, C. (2008). Plant litter. *Decomposition, humus formation, carbon sequestration. 2nd Ed Springer*.

Berger, A. L., Palik, B., D'Amato, A. W., Fraver, S., Bradford, J. B., Nislow, K., ... & Brooks, R. T. (2013). Ecological impacts of energy-wood harvests: lessons from whole-tree harvesting and natural disturbance. *Journal of Forestry*, *111*(2), 139-153.

Binkley, D. (1991, August). CONNECTING SOILS WITH FORESTPRODUCTnnTY. In *Proceedings Management and Productivity of Western-Montane Forest Soils* (p. 66 - 69).

Blumfield, T. J., & Xu, Z. H. (2003). Impact of harvest residues on soil mineral nitrogen dynamics following clearfall harvesting of a hoop pine plantation in subtropical Australia. *Forest Ecology and Management*, *179*(1), 55-67.

Boberg, J. (2009). *Litter decomposing fungi in boreal forests*(Vol. 2009, No. 75).

Brais, S., Paré, D., Camiré, C., Rochon, P., & Vasseur, C. (2002). Nitrogen net mineralization and dynamics following whole-tree harvesting and winter windrowing on clayey sites of northwestern Quebec. *Forest Ecology and Management*, *157*(1), 119-130.

Brix, H. (1981). Effects of thinning and nitrogen fertilization on branch and foliage production in Douglas-fir. *Canadian Journal of Forest Research*, *11*(3), 502-511.

Brix, H. (1983). Effects of thinning and nitrogen fertilization on growth of Douglas-fir: relative contribution of foliage quantity and efficiency. *Canadian Journal of Forest Research*, *13*(1), 167-175.

Brockley, R. P. (2006). *Effects of fertilization on the growth and foliar nutrition of immature Douglas fir in the interior cedar-hemlock zone of British Columbia: six-year results*. British Columbia, Forest Science Program.

Chapin III, F. S., Matson, P. A., & Vitousek, P. (2011). *Principles of terrestrial ecosystem ecology*. Springer Science & Business Media.

Chase, C. W., Kimsey, M. J., Shaw, T. M., & Coleman, M. D. (2016). The response of light, water, and nutrient availability to pre-commercial thinning in dry inland Douglas-fir forests. *Forest Ecology and Management*, *363*, 98-109.

Chaudhary, A., Burivalova, Z., Koh, L. P., & Hellweg, S. (2016). Impact of forest management on species richness: global meta-analysis and economic trade-offs. *Scientific reports*, *6*.

Coleman, M. D., Shaw, T. M., Kimsey, M. J., & Moore, J. A. (2014). Nutrition of Douglas-fir in the Inland Northwest. *Soil Science Society of America Journal*, *78*(S1), S11-S22.

Daily, G. (1997). *Nature's services: societal dependence on natural ecosystems*. Island Press.

Demirbas, A. (2009). Progress and recent trends in biodiesel fuels. *Energy conversion and management*, *50*(1), 14-34.

Didham, R. K. (1998). Altered leaf-litter decomposition rates in tropical forest fragments. *Oecologia*, *116*(3), 397-406.

Drew, T. J., & Flewelling, J. W. (1979). Stand density management: an alternative approach and its application to Douglas-fir plantations. *Forest Science*, *25*(3), 518-532.

Erisman, J. W., Galloway, J. N., Seitzinger, S., Bleeker, A., Dise, N. B., Petrescu, A. R., ... & de Vries, W. (2013). Consequences of human modification of the global nitrogen cycle. *Phil. Trans. R. Soc. B*, *368*(1621), 20130116.

Fan, Z., Moore, J. A., Shafii, B., & Osborne, H. L. (2002). Three-year response of ponderosa pine seedlings to controlled-release fertilizer applied at planting. *Western Journal of Applied Forestry*, *17*(3), 154-164.

Fcllin, D. G. (1979). A review of some relationships of harvesting, residuc management, and fire to forest insects and disease. In *Procecdings of thc Symposium on Environmcntal Conscquenccs of Timber hiarvesting. Gen. Tech. Rep. INT-90. Ogden, Ul: US Dcpartment of Agriculture, Forest Service, Intermountain Forcst and Range Experiment Station* (pp. 335-416).

Fenn, M. E., Poth, M. A., Aber, J. D., Baron, J. S., Bormann, B. T., Johnson, D. W., ... & Stottlemyer, R. (1998). Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. *Ecological Applications*, *8*(3), 706-733.

Finér, L., Jurgensen, M., Palviainen, M., Piirainen, S., & Page-Dumroese, D. (2016). Does clear-cut harvesting accelerate initial wood decomposition? A five-year study with standard wood material. *Forest Ecology and Management*, *372*, 10-18.

Finkeldey, R., & Ziehe, M. (2004). Genetic implications of silvicultural regimes. *Forest Ecology and Management*, *197*(1), 231-244.

Fox, T. R. (2000). Sustained productivity in intensively managed forest plantations. *Forest Ecology and Management*, *138*(1), 187-202.

Fox, T. R., Allen, H. L., Albaugh, T. J., Rubilar, R., & Carlson, C. A. (2007). Forest fertilization and water quality in the United States. *Better Crops*, *91*(1), 1-9.

Grigal, D. F. (2000). Effects of extensive forest management on soil productivity. *Forest Ecology and Management*, *138*(1), 167-185.

Harvey, A. E., Geist, J. M., McDonald, G. L., Jurgensen, M. F., Cochran, P. H., Zabowski, D., & Meurisse, R. T. (1994). Biotic and abiotic processes in eastside ecosystems: the effects of management on soil properties, processes, and productivity.

Harvey, A. E., Jurgensen, M. F., Larsen, M. J., & Graham, R. T. (1987). Decaying organic materials and soil quality in the Inland Northwest: a management opportunity. USDA For. Serv. Gen. Tech. Rep. INT-225 15p.

Helmisaari, H. S., Hanssen, K. H., Jacobson, S., Kukkola, M., Luiro, J., Saarsalmi, A., ... & Tveite, B. (2011). Logging residue removal after thinning in Nordic boreal forests: long-term impact on tree growth. *Forest Ecology and Management*, *261*(11), 1919-1927.

Henderson, G. S. (1995). Soil organic matter: a link between forest management and productivity. *Carbon forms and functions in forest soils*, (carbonformsandf), 419-435.

Hicks, W. T. (2000). *Modeling nitrogen fixation in dead wood* (Doctoral dissertation).

Hicks, W. T., Harmon, M. E., & Myrold, D. D. (2003). Substrate controls on nitrogen fixation and respiration in woody debris from the Pacific Northwest, USA. *Forest ecology and management*, *176*(1), 25-35.

IEA – International Energy Agency (2005) Answers to ten frequently asked questions about bioenergy, carbon sinks and their role in global climate change. Compiled and edited by Robert Matthews and Kimberly Robertson. IEA Bioenergy, Task 38 "Greenhouse Gas Balances of Biomass and Bioenergy Systems". Second edition. 8 p

Jang, W., Keyes, C. R., & Page-Dumroese, D. (2015). Impact of biomass harvesting on forest soil productivity in the northern Rocky Mountains.

Janowiak, M. K., & Webster, C. R. (2010). Promoting ecological sustainability in woody biomass harvesting. *Journal of forestry*, *108*(1), 16-23.

Klein, D., Höllerl, S., Blaschke, M., & Schulz, C. (2013). The contribution of managed and unmanaged forests to climate change mitigation—a model approach at stand level for the main tree species in Bavaria. *Forests*, *4*(1), 43-69.

Klockow, P. A. (2012). *Impacts of biomass harvesting on biomass, carbon, and nutrient stocks in Populus tremuloides forests of northern Minnesota, USA* (Doctoral dissertation, UNIVERSITY OF MINNESOTA).

Lavelle, P., Blanchart, E., Martin, A., Martin, S., & Spain, A. (1993). A hierarchical model for decomposition in terrestrial ecosystems: application to soils of the humid tropics. *Biotropica*, 130-150.

Lewandowski, T. E., Forrester, J. A., Mladenoff, D. J., D'Amato, A. W., & Palik, B. J. (2016). Response of the soil microbial community and soil nutrient bioavailability to biomass harvesting and reserve tree retention in northern Minnesota aspen-dominated forests. *Applied Soil Ecology*, *99*, 110-117.

Lloyd, J., & Taylor, J. A. (1994). On the temperature dependence of soil respiration. *Functional ecology*, 315-323.

Lundmark, T., Bergh, J., Hofer, P., Lundström, A., Nordin, A., Poudel, B. C., ... & Werner, F. (2014). Potential roles of Swedish forestry in the context of climate change mitigation. *Forests*, *5*(4), 557-578.

Ma, S., Concilio, A., Oakley, B., North, M., & Chen, J. (2010). Spatial variability in microclimate in a mixed-conifer forest before and after thinning and burning treatments. *Forest Ecology and Management*, *259*(5), 904-915.

Mann, L., & Tolbert, V. (2000). Soil sustainability in renewable biomass plantings. *AMBIO: A Journal of the Human Environment*, *29*(8), 492-498.

Mariani, L., Chang, S. X., & Kabzems, R. (2006). Effects of tree harvesting, forest floor removal, and compaction on soil microbial biomass, microbial respiration, and N availability in a boreal aspen forest in British Columbia. *Soil Biology and Biochemistry*, *38*(7), 1734-1744.

Maser, C., & Trappe, J. M. (1984). The seen and unseen world of the fallen tree.

McGill, W. B., & Cole, C. V. (1981). Comparative aspects of cycling of organic C, N, S and P through soil organic matter. *Geoderma*, *26*(4), 267-286.

Mead, D. J. (2005). Opportunities for improving plantation productivity. How much? How quickly? How realistic?. *Biomass and Bioenergy*, *28*(2), 249-266.

Mika, P. G., & Vanderploeg, J. (1991). Six-year fertilizer response of managed second-growth Douglas fir stands in the intermountain northwest.

Mitchell, R. G., Waring, R. H., & Pitman, G. B. (1983). Thinning lodgepole pine increases tree vigor and resistance to mountain pine beetle. *Forest Science*, *29*(1), 204-211.

Murphy, K. L., Klopatek, J. M., & Klopatek, C. C. (1998). The effects of litter quality and climate on decomposition along an elevational gradient. *Ecological Applications*, *8*(4), 1061-1071.

O'Connell, A. M., Grove, T. S., Mendham, D. S., & Rance, S. J. (2004). Impact of harvest residue management on soil nitrogen dynamics in Eucalyptus globulus plantations in south western Australia. *Soil Biology and Biochemistry*, *36*(1), 39-48.

Padilla, F. M., & Pugnaire, F. I. (2007). Rooting depth and soil moisture control Mediterranean woody seedling survival during drought. *Functional Ecology*, *21*(3), 489-495.

Page-Dumroese, D. S., Jurgensen, M., & Terry, T. (2010). Maintaining soil productivity during forest or biomass-to-energy thinning harvests in the western United States. *Western Journal of Applied Forestry*, *25*(1), 5-11.

Page-Dumroese, D., Harvey, A., Jurgensen, M., & Graham, R. (1991). Organic matter function in the western-montane forest soil system. *AE Harvey, and LF Neuenschwander (Compilers)*,

ProceedingsÐManagement and Productivity of Western-Montane Forest Soils. USDA For. Serv. Gen.Tech. Rep. INT-280. Ogden, UT, 95-100.

Perala, D. A., & Laidly, P. R. (1989). Growth of nitrogen-fertilized and thinned quaking aspen (Populus tremuloides Michx.).

Perlack, R. D., & Stokes, B. J. (2011). US Department of Energy.(2011). US Billion-Ton Update: Biomass Supply for a Bioenergy and Bioproducts Industry. *ORNL/TM-2011/224. Oak Ridge National Laboratory, Oak Ridge, TN*.

Peterson, E. B., & Peterson, N. M. (1995). Aspen managers' handbook for British Columbia. *FRDA report*.

Piatek, K. B., & Allen, H. L. (1999). Nitrogen mineralization in a pine plantation fifteen years after harvesting and site preparation. *Soil Science Society of America Journal*, *63*(4), 990-998.

Piticar, M. A. (2016). The effects of thinning on a spruce stand, over a period of 40 years of monitoring. *Advances in Agriculture & Botanics*, *8*(2).

Powers, R. F., Alban, D. H., Miller, R. E., Tiarks, A. E., Wells, C. G., Avers, P. E., ... & Loftus Jr, N. S. (1990). Sustaining site productivity in North American forests: problems and prospects. *Sustained productivity of forest soils/edited by SP Gessel...[et. al.]*.

Powers, R. F., Scott, D. A., Sanchez, F. G., Voldseth, R. A., Page-Dumroese, D., Elioff, J. D., & Stone, D. M. (2005). The North American long-term soil productivity experiment: findings from the first decade of research. *Forest Ecology and Management*, *220*(1), 31-50.

Prescott, C. E. (2002). The influence of the forest canopy on nutrient cycling. *Tree physiology*, *22*(15 16), 1193-1200.

Prescott, C. E., Kabzems, R., & Zabek, L. M. (1999). Effects of fertilization on decomposition rate of Populus tremuloides foliar litter in a boreal forest.*Canadian Journal of Forest Research*, *29*(3), 393-397.

Purahong, W., Kapturska, D., Pecyna, M. J., Schulz, E., Schloter, M., Buscot, F., ... & Krüger, D. (2014). Influence of different forest system management practices on leaf litter decomposition rates, nutrient dynamics and the activity of ligninolytic enzymes: a case study from Central European forests. *PloS one*, *9*(4), e93700.

Ranger, J., & Turpault, M. P. (1999). Input–output nutrient budgets as a diagnostic tool for sustainable forest management. *Forest Ecology and Management*, *122*(1), 139-154.

Safford, L. O., & Czapowskyj, M. M. (1986). Fertilizer stimulates growth and mortality in a young Populus–Betula stand: 10-year results. *Canadian Journal of Forest Research*, *16*(4), 807-813.

Sedell, J., Sharpe, M., Apple, D. D., Copenhagen, M., & Furniss, M. (2000). Water and the forest service.

Shafii, B., Moore, J. A., & Newberry, J. D. (1990). Individual-tree diameter growth models for quantifying within-stand response to nitrogen fertilization. *Canadian Journal of Forest Research*, *20*(8), 1149-1155.

Smolander, A., Levula, T., & Kitunen, V. (2008). Response of litter decomposition and soil C and N transformations in a Norway spruce thinning stand to removal of logging residue. *Forest Ecology and Management*, *256*(5), 1080-1086.

Stupak, I., Nordfjell, T., & Gundersen, P. (2008). Comparing biomass and nutrient removals of stems and fresh and predried whole trees in thinnings in two Norway spruce experiments. *Canadian Journal of Forest Research*, *38*(10), 2660-2673.

Swift, M. J., Heal, O. W., & Anderson, J. M. (1979). *Decomposition in terrestrial ecosystems* (Vol. 5). Univ of California Press.

Tappeiner, J. C., Maguire, D. A., & Harrington, T. B. (2007). *Silviculture and ecology of western US forests*. Oregon State University Press.

Tate, R. L. (1992). Nitrogen in Terrestrial Ecosystems. Questions of Productivity, Vegetational Changes, and Ecosystem Stability. *Soil Science*, *154*(6), 508.

Thies, W. G., & Russell, K. W. (1983). Controlling root rots in coniferous forests of northwestern North America. In *Proceedings of the 6th Int. Conf. on Root and Butt Rots of Forest Trees. Australia* (pp. 379-386).

Thiffault, E., Hannam, K. D., Paré, D., Titus, B. D., Hazlett, P. W., Maynard, D. G., & Brais, S. (2011). Effects of forest biomass harvesting on soil productivity in boreal and temperate forests—a review. *Environmental Reviews*, *19*(NA), 278-309.

Union, E. (2009). Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union, 5, 2009.

Van Cleve, K., & MooRE, T. A. (1978). Cumulative effects of nitrogen, phosphorus, and potassium fertilizer additions on soil respiration, pH, and organic matter content. *Soil Science Society of America Journal*, *42*(1), 121-124.

Wall, A. (2008). Effect of removal of logging residue on nutrient leaching and nutrient pools in the soil after clearcutting in a Norway spruce stand. *Forest Ecology and Management*, *256*(6), 1372 1383.

Wardle, D. A., Nilsson, M. C., Zackrisson, O., & Gallet, C. (2003). Determinants of litter mixing effects in a Swedish boreal forest. *Soil biology and biochemistry*, *35*(6), 827-835.

Wickland, K. P., & Neff, J. C. (2008). Decomposition of soil organic matter from boreal black spruce forest: environmental and chemical controls. *Biogeochemistry*, *87*(1), 29-47.

Zeide, B. (2001). Thinning and growth: a full turnaround. *Journal of Forestry*, *99*(1), 20-25.

Chapter 2. Litter Decomposition under Logging Residues in a Thinned Inland Northwest Forest

2.1 Introduction

Globally, bioenergy is an alternative to fossil fuels that avoids increasing atmospheric carbon (C) (Mann and Tolbert, 2000; Demirbas, 2009). Therefore, the demand for woody residues for bioenergy production is increasing as the interest in woody biomass energy production increases. Although bioenergy may help reduce atmospheric C concentrations, there are increasing concerns that increased biomass removal will reduce forest productivity and alter soil properties (Stupak et al., 2008; Page-Dumroese et al., 2010; Jang et al., 2015). After thinning or harvest operations, logging residues left on the soil surface can affect the physical conditions of the forest floor (e.g., temperature and moisture). However, soil temperature and moisture can be altered whether the residues are retained or not (Henderson, 1995; Powers et al., 2005). More specifically, logging residues create shade on the forest floor which can lower temperature and prevent evaporative losses (Powers et al., 2005). However, removal of the overstory will also alter soil temperature and moisture.

Because climate (temperature and moisture) is one of the critical controls over OM decomposition (Swift et al., 1979; Wickland et al., 2008), changes in the physical condition of the forest floor resulting from biomass retention may increase organic matter (OM) decomposition. Decomposition of plant litter is the mechanism that returns OM and nutrients to forest soils (Aber and Melillo, 1980). Rising soil temperature, with an appropriate range of soil moisture, increases OM decomposition rate; however, without enough soil moisture, increased temperatures will not increase OM decomposition (Murphy et al., 1998; Chapin et al., 2011). Temperature and moisture play an important role in regulating the metabolism of microbial decomposers (Swift et al., 1979; Lavelle et al., 1993; Lloyd and Taylor, 1994; Chapin et al., 2011). Abundant soil moisture is crucial for microbial activity and forest productivity, especially on sites suffering from a summer drought, which can impede OM

decomposition (Padilla and Pugnaire, 2007). In other words, residue removals can lead to reductions in soil moisture and the activity of microbial decomposers (Hicks, 2000; Hicks et al., 2003; Chapin et al., 2011). Consequently, reducing OM decomposition rate (Finér et al. 2016) and resulting in decreased site sustainability (Ranger and Turpault, 1999). However, the limits for biomass retention in various ecosystems are not known.

One negative effect of woody residue removals is less N input. Leaving residues on the soil surface can be a good N source in forest ecosystems. Nitrogen is an essential element for tree growth and microbial decomposers; however, this N is commonly deficient in northwestern forests, limiting forest productivity (Binkley, 1991). In this regard, logging residue removal can decrease N input into the forest floor or the soil causing reductions in litter decomposition rates because microbial decomposers require N for their growth and enzyme production (Sinsabaugh, 2005). In addition, microbial decomposers have their own elemental C: N ratio (stoichiometry C: N ratio) which refers that how much N is required to build microbial biomass (Smolander et al., 2008). When decomposers do not satisfy their stoichiometry C: N ratio, microbial decomposition will be delayed or not completed since decomposers should find N from other sources. Helmesaari et al. (2011) insisted that N fertilization can compensate for the decreased N which resulted from residue removal. However, there is still lack of research about the impact N fertilization on decomposition rate with various biomass retention levels.

It is critical to establish regional criteria for managing forest residues as doing so will help to achieve sustainable forestry during the production of bioenergy. A detailed understanding of the mechanisms that control decomposition after harvesting will help determine limits for biomass removals; it will also help to figure out the effects of biomass removal on OM decomposition and therefore sustainable forestry. Therefore, the goal of this study was to determine if thinning and residue retention levels in a conifer forest in northern Idaho contribute to changes in litterbag decomposition. To meet this goal,

decomposition rates and nutrient pools should be similar to unharvested control rates within a short period after harvest activities.

I had two objectives: 1) determine the effects of thinning and biomass retention level on litterbag decomposition rates, moisture content, respiration, and nutrients' dynamics (C, N, Ca, Mg, and K) and 2) evaluate the capacity of N fertilization to compensate for biomass removal impacts on litter decomposition rate and nutrient concentrations in litter (C, N, Ca, Mg, and K). I approached these objectives using a 172-day short-term litterbag decomposition experiment in the two-different stands (mixed conifer and the ponderosa pine). Rates of litter decomposition are important because they provide a measure of how harvesting, fertilization, temperature and moisture affect microbial activity and the return of OM to the mineral soil (Aber and Melillo, 1980; Prescott, 2005). Understanding which factors are most influential will help managers develop best management practices that maintain nutrient cycling processes. Consequently, the developed model or management practices can be applied across many sites in northwestern regions to understand how decomposition responds to residue removal.

My hypotheses were:

I. Removal of biomass may reduce litter moisture content and respiration and consequently, cause a decrease in mass loss and decomposition rate.

II. Subsequent effect of slow decomposition rate in biomass removal treatments have lower mineralization of C and N and lower release of other nutrients (Ca, Mg, and K).

III. Fertilization can compensate for less input of nitrogen due to the removal of biomass in no biomass retention treatments and therefore, it maintains decomposition rate compared to retained biomass treatments.

2.2 Materials and methods

2.2.1 Site description

The study sites are located on the University of Idaho (UI) Experimental Forest, West Hatter Creek Unit, near Princeton, Idaho. The soil parent material under both sites is a granite bedrock. The soils are primarily of the Santa series (coarse-silty, mixed, superactive, frigid Vitrandic Fragixeralfs) which is formed in deep loess with a small amount of volcanic ash in the upper part (Web Soil Survey). These soils are moderately well-drained. Data from a weather station on site indicated the mean annual temperature averages 8.7℃, the warmest month is August (19.4℃, mean temperature), and the coldest month is December (-1.4℃, mean termperature). The mean annual precipitation averages 688mm most of which falls as snow from November to May. Both sites are located in naturally regenerated forests. One site is a ponderosa pine forest (Pinus ponderosa Lawson & C. Lawson) (46.85°N, 116.85°W) and the other is a mixed conifer (46.85°N, 116.84°W) forest consisting of Douglas-fir (Pseudotsuga menziesii (Mirb.) Franco), grand fir (Abies grandis (Douglas ex D. Don) Lindl.), and ponderosa pine. The two stands are 600 meters apart. The elevation of two sites ranged from 830 to 890 meters above sea level. The ponderosa pine stand has a higher elevation than mixed conifer stand. Additional site characteristics are shown in Table 2.1. During harvesting in the late 1980s, the both stands were harvested using a seed tree harvest method leaving approximately 5 trees/ha. In 2013, both stands were thinned and the intensity was based on Powell (1999) who recommended leaving a relative density of 40% by removing the most undesirable trees (approximately, 200 trees/ha with an average diameter of 12 cm). Litter used in the litterbags was collected from an adjacent mature stand and the initial litter nutrient contents are shown in Table 2.1.

Table 2.1 Study site characteristic including forest tree composition, topography, and initial value of nutrients in litter.

Tree Species	Ponderosa Pine Stand	Mixed Conifer Stand
Ponderosa pine	88%	22%
Douglas fir	9 %	16%
Lodgepole pine	2 %	23%
Grand fir	0.2%	34%
Western Larch	0.2%	5%
Cherry	0.2%	N/A
Site characteristics		
Elevation	880 (m)	838 (m)
Slope	6.5˚	5.7˚
Aspect	South-facing (169˚)	Southwestern-facing (204˚)
Soil series	Santa	Santa
Initial litter characteristics from six correction bags		
C (g)	8648.74 ± 171 (n = 6)	
N (g)	116.67 ± 4.35 (n = 6)	
C/N ratio	74.21 ± 2.87 (n = 6)	
Ca (mg/kg)	3290.8 ± 238.46 (n = 5)	
K (mg/kg)	746.78 ± 72.81 (n = 5)	
Mg (mg/kg)	314.6 ± 91.05 (n = 5)	

2.2.2 Experimental design

Study plots were established in 2013 in two thinned stands to determine the affect of removing woody biomass and adding soil amendments on tree growth (Sherman et al., 2017). I deployed the litterbags on residue treatments plots in 2016. I also used the plots with N fertilization. In short, in each stand study plots were established with 3 different levels of biomass retention and one unthinned control. The retention treatments were 0x (all biomass removed), 1x (all the harvesting biomass retained), and 2x (double the harvesting biomass retained). The additional amount of logging residue in the 2x plots was brought from the 0x plots (Figure 2.1).

Each biomass retention treatment can be divided by two fertilization treatments. Within each retention level and the unthinned plots, there was a control (C = no fertilizer) and a fertilized (F) plot. The N fertilizer was applied at the rate of 224 kg N/ha as urea (Ramirez et al., 2010).

In the plots, temperature and moisture probes were installed to collect temperature and moisture data at 10 cm depths in each of the 4-different biomass retention (3 biomass retention levels and 1 unthinned control) plots. There was one temperature/moisture probe collecting data in the control plot and in each of the 3 different biomass retention plots. Data collection was programmed for 2 hour intervals.

Unthinned control		**No biomass retention (0x)**	
No fertilizer	Fertilizer	No fertilizer	Fertilizer
Left biomass on the site (1x)		**Double-biomass retention (2x)**	
No fertilizer	Fertilizer	No fertilizer	Fertilizer

Figure 2.1. Experimental design for biomass and fertilization treatments in each mixed conifer and ponderosa pine stand at the University of Idaho Experimental Forest, Hatter Creek Unit, near Princeton, ID

2.2.3 Preparation and field Installation of litterbags

In March 2016, forest floor (inclusive of the Oa, Oe, and Oi horizons) was collected from an adjacent, mature mixed conifer stand at the UI Experimental Forest and returned to the USDA Forest Service Moscow Forestry Sciences Lab for processing. I did not use the litter that was produced in each stand in order to reduce the variation of litter quality. The forest floor was air-dried at room temperature and sticks, twigs, green needles, and cones were picked out and discarded. The remaining brown needles were homogenized and moisture content at time 0 was determined on a subsample at 65°C to a constant weight. Commercially available fiberglass screen (1 mm mesh) was used to construct the litterbags. A rotary cutter and straight-edge were used to cut the screen into 36 cm x 20 cm rectangles which were folded in half and an impulse sealer (TISH-300, TEW Electric Heating Equipment Co., Ltd, Taipei, Taiwan, R.O.C.) was used to seal the edges at 2.5 cm. Empty litter bag weights were recorded and each litter bag was filled with 20 grams of forest floor and sealed at 2.5 cm, creating an area 15.2 cm x 15.2 cm for the litter.

On April 7, 2016, approximately 3 years after the thinning treatments were applied (2013, spring), all litter bags were placed in the field. Litter bags were tagged and secured to the soil with two 15.2 cm landscape staple, one in each 2.5 cm margin beyond the seam (Figure 2.2). Within each thinning and fertilizer treatment, fifteen litter bags (5 sampling dates and 3 replicates) were randomly arranged in a grid of 3 rows and 5 columns with at least 20 cm between bags (Figure 2.3). In the 1x and 2x biomass retention treatments, the biomass was removed prior to litterbag placement and was replaced after their installation. The number of deployed litter bags in one stand was 120 litter bags. The total number of litter bags installed in both stands were 240 (2 stands * 4 biomass retention treatments (3 biomass retention levels and 1 unthinned control) * 2 fertilizer amendments * 5 sample dates * 3 replicates). In addition to the deployed bags, 7 litter bags were constructed as correction bags. These were transported to the field and installed as the other bags but were then collected and brought back to the lab to assess what amount of material might be lost to handling and transportation. The correction bags were used for initial values of nutrients (C, N, Ca, Mg, and K) and C/N ratio (Table 2.1) and were also used for correcting initial weight of litter bags. I measured the amount of mass loss during the transportation of litter bags from laboratory until litter bag deployment and averaged the mass loss from transportation; the averaged lost mass from the litterbags was subtracted from the initial weight of litter in the litter bags (W0) which was required to calculate remaining mass (%), mass loss (%), or decomposition rate (constant *k*).

Figure 2.2 A litterbag (Aluminum tag attached the other side of the litterbag)

Figure2.3 Deployment of the litterbags in the field

2.2.4 Litterbag removal and analysis

The litterbag collections occurred every 5 weeks from May to September in 2016. The first sample collection (May 11th) was 34 days after deployment and the subset of three randomly selected replicate bags were collected. Later collections occurred 71, 103, 137, and 172 days after the installation date (April, 7th). Every collected litter bag was placed into a labeled zip-type bag, kept in the cooler and transported directly to the Moscow Forestry Sciences Laboratory and cleaned of matter which was not part of the initial mixture including live plants which might grow through the mesh, seeds, bugs, and eggs of insects. After cleaning, samples were weighed for wet weight (g) and measured for a CO_2 level ($\mu mol \cdot mol^{-1}$) by using an EGM4 (PP Systems, Amesbury, MA) to determine respiration (Paudel et al., 2015). In turn, samples were dried at 65° C to a constant weight and weighed for the determination of moisture content and weight loss. The moisture value was calculated by subtraction dry weight from wet weight. From dry weight, remaining mass was displayed as a percentage of the final dry weight of the initial dry weight (before incubation).

2.2.5 Chemical analyses

Dried samples were coarse ground with a Wiley Mill using a 5 mm-mesh sieve (Thomas Scientific, Swedesboro, NJ) and split with a rifle-type splitter (Humboldt Manufacturing, Elgin, IL). One sub-sample was used as a representative subsample for further analyses. Samples were fine ground with an 8000D Mixer /Mill (Spex SamplePrep, Metuchen, NJ). Total C and N were determined on a Leco TruSpec CN analyzer (St. Joseph, MI) and used to determine the C/N for each sample. The TruSpec employs dry combustion of the samples at high temperature and uses an infrared (IR) detector to measure C and a thermal conductivity (TC) cell to determine N from the combustion gases (Nelson and Sommers 1996). Organic matter concentration was determined by loss-on-ignition (Nelson and Sommers 1996). Organic cations were extracted after dry ashing in a muffle furnace at 475°C for 5 hours and extracted with 2M nitric acid for estimating exchangeable cations (calcium (Ca), magnesium (Mg), and potassium (K); Jones Jr. et, al., 1990; Nelson and Sommers, 1996; Sumner and Miller, 1996; Thomas, 1996). Analysis of the cation extract was conducted on an Atomic Absorption Spectrometer (PinAAcle 500, Perkin Elmer, Shelton, CT).

2.2.6 Calculation and statistical analyses

Decomposition rate (*k*) for each residue retention levels (treatments) was calculated with the negative exponential decomposition function (1) (Olson, 1963):

$$k = -\ln (DW_t/DW_0)/t \quad \ldots\ldots\ldots (1)$$

where DW_0= initial dry mass of litter prior to the incubation, DW_t = dry mass remaining at time t, k = decomposition rate, and t = time (year).

Analysis of variance (ANOVA) was used to analyze all data incorporating the Tukey's honest significant difference (HSD) test implemented in R software to determine the differences ($P < 0.1$)

between different treatments at each sampling date (R Development Core Team, 2015). The exponential curve fitting was used for the remaining mass of litterbags.

2.3 Results

2.3.1 Residue retention effects on litterbag mass loss

During the whole incubation period (172 days), litterbags lost 15% to 25% of the initial mass (Figure 2.4). The incubation time was the most important factor controlling decomposition (Table 2.2). Litterbag mass loss in all biomass treatments was detected at the first sample date. Litterbag mass differed by biomass treatments during the incubation period (Figure 2.4, Table 2.2); in 103 days of incubation and 172 days of incubation, the mass loss in 0x was significantly higher than the mass loss in unthinned control. When all incubation days were combined, the unthinned control had slower decomposition rate than other biomass treatments although, among the three biomass retention levels, only 0x showed significantly higher decomposition rate compared to control (Figure 2.5). In general, the managed plots (three biomass retention levels) had higher decomposition rates compared to unthinned control. Removing all residual biomass (0x) resulted in the greatest litterbag mass loss as compared to the unharvested control or other biomass retention levels (1x or 2x).

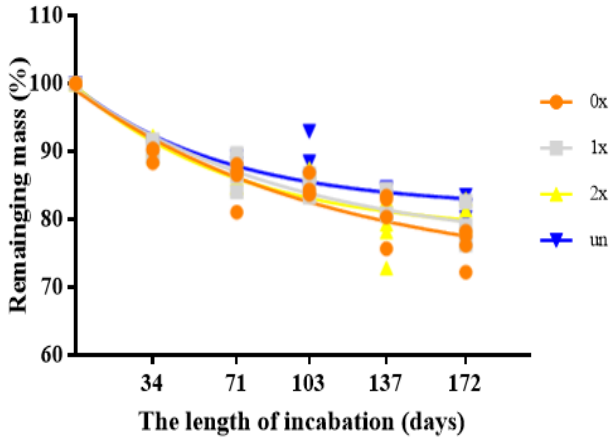

Figure 2.4 Remaining mass in each residue retention treatment during the incubation periods. (unthinned control (un)-Blue, 0x-Orange, 1x-Grey, and 2x-Yellow).

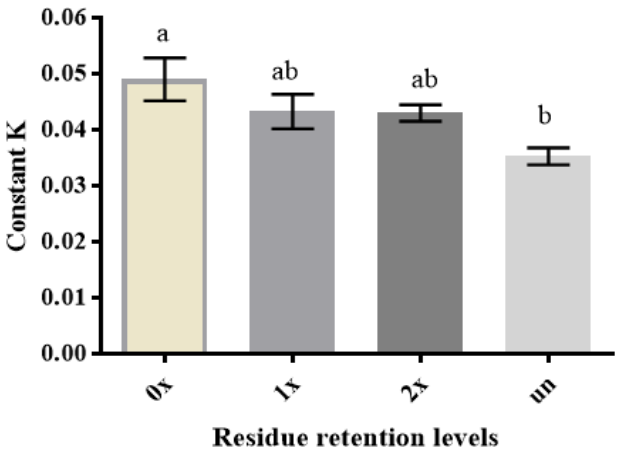

Figure 2.5 The decay constant *k* in each residue retention treatment (±standard error of the mean, n=4). Different letters indicate significant differences at p<0.1.

Table 2.2 The F- value and p-value from statistical analysis from R. The significant results are marked by bold text and * indicates significant level (*<0.1, **≤0.05, *<0.01)**

Treatments	Remaining mass of litter		Constant *k*		Moisture content of litter		Respiration		Carbon content		Nitrogen content		Carbon Concenration		Nitrogen Concentration	
	F	P	F	P	F	P	F	P	F	P	F	P	F	P	F	P
Stand (S)	0.184	0.669	1.412	0.262	9.555	**0.0028 (***)**	1.435	0.235	0.005	0.946	0.185	0.668	0.125	0.724	0.202	0.654
Biomass (B)	7.159	**0.0005 (***)**	4.258	**0.0351 (**)**	4.968	**0.0050 (***)**	3.23	**0.032 (**)**	2.201	**0.096 (*)**	3.098	**0.0322 (**)**	0.285	0.836	0.729	0.541
Fertilizer (F)	0.355	0.555	0	1	0.432	0.515	0.385	0.538	0.004	0.949	1.031	0.313	0.009	0.923	0.76	0.388
Time (T)	68.94	**<0.0001 (***)**	N/A	N/A	70.79	**<0.0001 (***)**	19.11	**<0.0001 (***)**	29.125	**<0.0001 (***)**	1.576	0.190	3.278	**0.0096 (***)**	**9.028**	**<0.0001 (***)**
B*F	1.613	0.201	3.547	0.076	0.443	0.724	1.03	0.390	0.392	0.759	1.943	0.130	0.09	0.965	0.521	0.67
B*T	1.474	0.175	N/A	N/A	1.566	0.142	1.54	0.150	1.120	0.372	2.094	**0.031 (*)**	1.274	0.271	1.2	0.317
F*T	0.74	0.571	N/A	N/A	0.347	0.845	0.185	0.945	0.866	0.493	0.837	0.510	0.738	0.572	0.765	0.555
B*F*T	1.422	0.196	N/A	N/A	0.766	0.681	1.055	0.421	0.902	0.553	0.899	0.556	0.837	0.613	0.487	0.91

Treatments	C/N ratio		Calcium Content		Magnesium Content		Potassium Content		Calcium Concentration		Magnesium Concentration		Potassium Concentration	
	F	P	F	P	F	P	F	P	F	P	F	P	F	P
Stand (S)	0.064	0.801	0.032	0.858	4.461	**0.0383 (**)**	0.004	0.951	0.094	0.76	5.2	**0.025 (**)**	0.004	0.947
Biomass (B)	0.420	0.740	4.313	**0.010 (**)**	1.307	0.286	3.100	**0.0374 (**)**	2.053	0.122	0.254	0.858	8.892	**0.047 (**)**
Fertilizer (F)	0.489	0.487	2.648	0.112	0.049	0.825	0.024	0.878	2.673	0.11	0.027	0.870	0.014	0.905
Time (T)	6.346	**0.0002 (***)**	17.819	**<0.0001 (***)**	15.04	**<0.0001 (***)**	0.664	0.621	20.25	**<0.0001 (***)**	2.973	**0.030 (**)**	0.841	0.507
B*F	0.394	0.758	0.625	0.603	0.078	0.972	1.242	0.307	0.47	0.705	0.095	0.962	1.283	0.293
B*T	1.337	0.237	1.363	0.224	0.579	0.846	0.713	0.730	1.437	0.19	0.679	0.760	0.644	0.791
F*T	0.775	0.548	0.751	0.563	0.817	0.522	1.104	0.368	0.774	0.549	0.851	0.501	1.014	0.411
B*F*T	0.511	0.895	0.689	0.752	1.200	0.317	0.964	0.498	0.58	0.845	0.904	0.551	0.86	0.592

2.3.2 Changes in litterbag moisture content and respiration

Moisture content and respiration rate of litterbags were affected by time and biomass treatments, but fertilization did not affect either moisture content or respiration (Table 2.2). Litterbag moisture content was significantly lower in the ponderosa pine than the mixed conifer stand (Figure 2.6), but there were no differences in litter respiration between the two stand locations ($p=0.235$). When all biomass retention treatments were combined, moisture content in the litterbags had a significant continuous decrease from May (34 days) to August (137 days), and a slight increase in September (Figure 2.7). Biomass retention level also significantly affected moisture content in litter compared to unthinned control but no there were differences among 0x, 1x, and 2x (Figure 2.8). The moisture content in the litter was the highest in unthinned control (16%) and next came 1x (13%), 0x (11%) and 2x (10%). Compared to unthinned control, 2x and 0x had a significantly lower moisture content in the litter but 1x was not statistically different from unthinned control. For moisture content in the litter, there were significant differences between stands; when all treatments were combined, the averaged moisture content of mixed conifer stand had approximately 27% more moisture than that of ponderosa pine stand.

The impacts of time, biomass treatment and fertilization treatments on litter respiration followed a similar pattern as moisture content. Litterbag respiration levels decreased during the summer as moisture content was reduced and then tended to increase in fall (after 137 days of incubation (September); Figure 2.9). This September increase corresponded with 31mm of rainfall compared to 25mm in July and 6mm in August (U.S Climate Data Website). Similar to moisture content, biomass retention levels significantly affected respiration in litter compared to unthinned control; however there were no significant differences among 0x, 1x, and 2x (Figure 2.10). There is a strong ($R^2 = 0.7383$; $p < 0.0001$) correlation between litter moisture content and litter respiration; when litterbag moisture content was low, litter respiration was also low, vice versa (Figure 2.11).

Respiration in the litterbags follows a similar trend as moisture content (Figure 2.10). Litterbag respiration was highest in the unthinned control and was significantly higher than 0x and 2x ($p = 0.032$). This respiration trend in Figure 2.10 was very similar to litterbag moisture content (Figure 2.8). However, unlike moisture content, respiration rate was not affected by stand location.

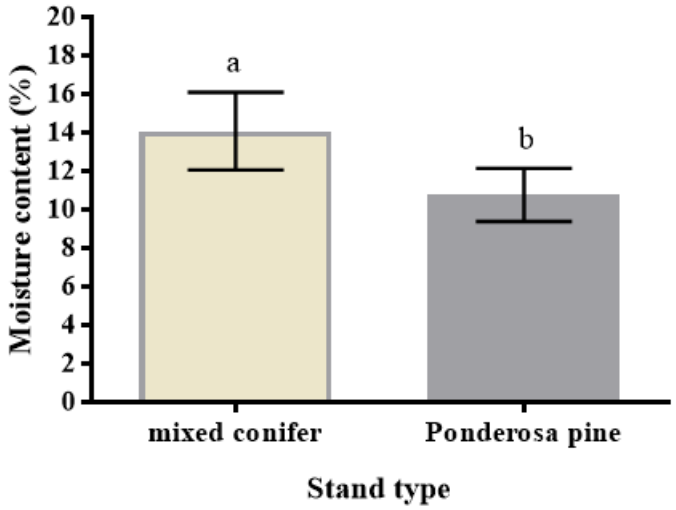

Figure 2.6 The impact of stand difference on moisture content in litterbags (±standard error of the mean, n=40). Different letters indicate significant differences among the stand at p<0.1.

The majority of the forest surface temperature data was lost because the field data loggers did not record. The available soil temperature data is shown in Figure 2.12. In this experiment, residue retention levels as well as thinning did not significantly affect the soil temperature at 10 cm depth. However, there were interesting relationships between soil temperature and mass loss in each residue retention treatments (Figure 2.13) indicating that as soil temperature increased so did mass loss. Among the residue retention levels, mass losses in 0x, 2x, and unthinned control were highly related to soil temperature ($R^2 > 0.6$).

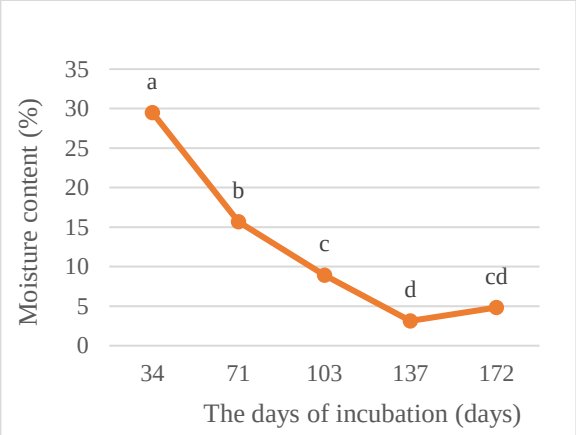

Figure 2.7 Litterbag moisture content as affected by incubation time when all residue retention treatments are combined (n=16). Different letters indicate significant differences among treatments (P<0.1).

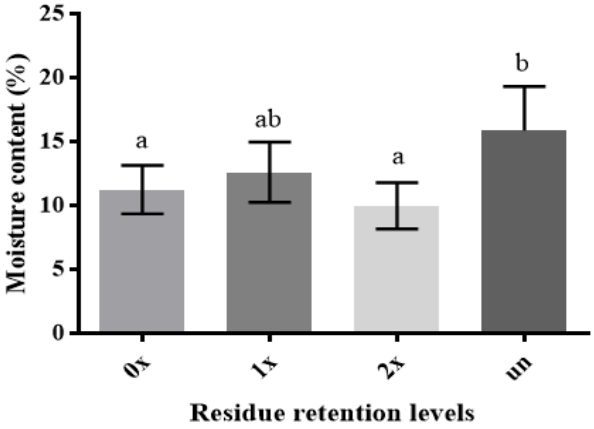

Figure 2.8 Average litterbag moisture (±standard error of the mean, n=20) as affected by residue retention level . All incubation dates are combined. Different letters indicate significant differences among treatment at p<0.1.

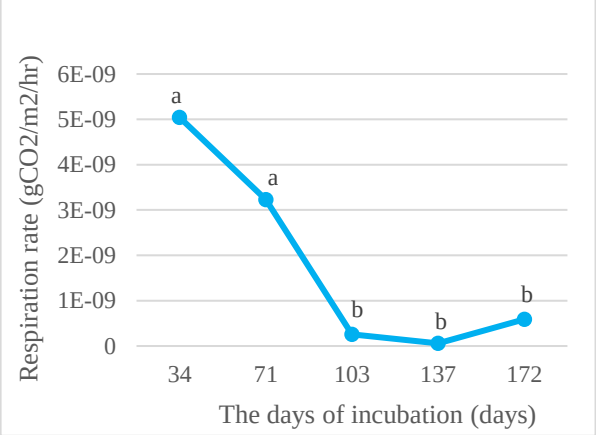

Figure 2.9 Litterbag respiration rate as affected by incubation time when all residue retention treatments are combined (n=16). Different letters indicate significant differences among treatments at p<0.1.

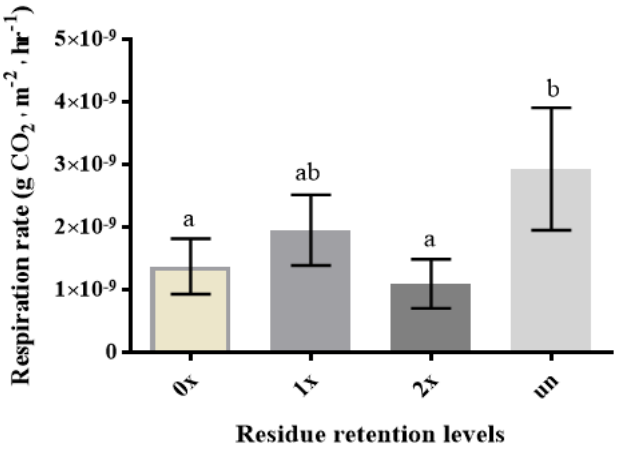

Figure 2.10 Average litterbag respiration (±standard error of the mean, n=20) as affected by residue retention level. All incubation dates are combined. Different letters indicate significant differences among treatment at p<0.1.

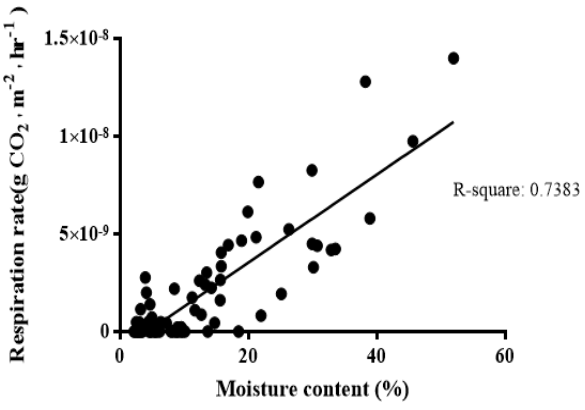

Figure 2.11 Relationship of microbial respiration and litterbag moisture content for all dates and residue retention levels.

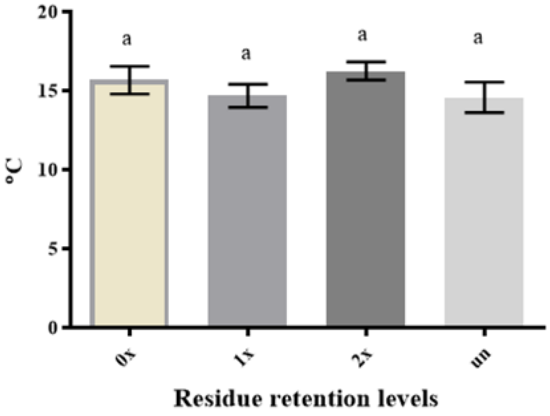

Figure 2.12 Soil temperature (±standard error of the mean) within residue retention treatments when all dates are combined (n=14, n=14, n=12, n=10).

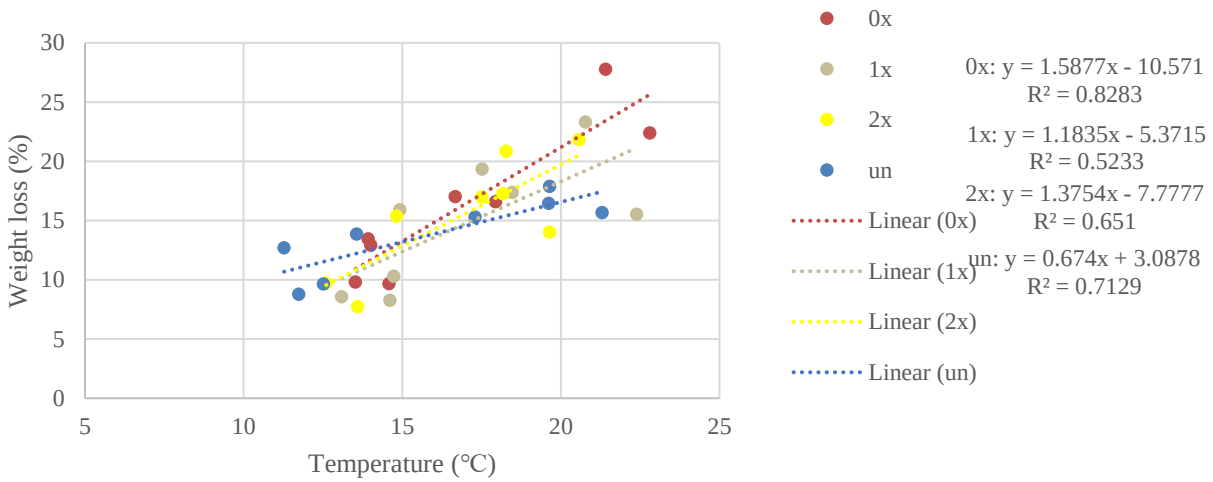

Figure 2.13 The relationship between weight loss and soil temperature. Soil temperature data from both mixed conifer and ponderosa pine stands were combined and averaged.

2.3.3 Residue retention effects on litterbag nutrient

In general, the nutrients in the litterbags significantly changed during the incubation period except for N content and K concentration. Specifically, both the concentration and content of carbon in the litterbags decreased significantly after the litter bag was deployed. Compared to the initial C content, C content decreased significantly over time (Figure 2.15 (a)). Carbon concentration showed the significant decreases after 71days of incubation compared to initial C concentration (Figure 2.15 (b)). Nitrogen content did not change ($p=0.1902$; Table 2.2; Figure 2.15 (c)). Unlike N content in the litterbags, N concentration in the litterbags significantly increased over time (Figure 2.15 (d)). Based on C and N changes, the C/N ratio also declined during the incubation periods (Figure 2.15 (e)). The C/N ratio was significantly lower in the last two sample dates as compared to the first 3 dates.

When all biomass retention treatments were combined, Ca and K concentrations exhibited similar patterns during the incubation periods. Calcium concentrations in litter increased until 103 days of incubation and decreased thereafter (Figure 2.16 (a); (e)). Calcium concentrations increased in the first 103 days of incubation and significantly decreased thereafter and as compared with initial Ca (Figure

2.16 (a)). Magnesium concentrations decreased significantly over time compared to the initial concentration and were lowest on day 137 (Figure 2.16 (c)).

The biomass retention treatments significantly influenced C content, N content, and K concentration but did not significantly affect the concentration of C, N, Ca and Mg and the C/N ratio (Table 2.2). The unthinned control litterbags had significantly higher C content compared to 0x; however, there were fewer significant differences among the three biomass retention levels (Figure 2.17). The nitrogen content in litterbags also showed significant differences between unthinned control and 0x having significantly lower N content (Figure 2.18). Unlike C content, N content had significant differences among the various biomass retention levels and in the decreasing orders are unthinned control> 1x> 2x> 0x. More specifically, 0x had the lowest N content and was significantly lower than in the 1x treatment where biomass from thinning was retained. On the contrary, the concentration in each C and N and the C/N ratio did not change among the biomass treatments. Based on C and N content (Figure 2.17; 2.18), 0x increased the mineralization of C and N in the litterbag into the soil compared to that of unthinned control.

Although the impact of biomass retention levels on K concentration in the 0x and 1x did not differ from unthinned control, K concentration in 2x was significantly higher than 0x (Figure 2.19). On the contrary, Ca and Mg in the biomass retention treatments did not show any significant changes compared to unthinned control. The impacts of biomass retention on Ca concentration were nonsignificant compared to unthinned control. Similarly, the concentrations of Mg were not affected by biomass treatments when compared to unthinned control.

In addition, an interaction between biomass treatments and time in N content in litterbags was found (Table 2.2). For 71 days, litterbag N content in all residue retention treatments consistently decreased. However, after 71 days, the trend of N content changed.

Among the nutrients, only the concentration of Mg was significantly different between the two stand locations (Table 2.2) where it was significantly higher in the ponderosa pine stand as compared to mixed conifer stand (Figure 2.14).

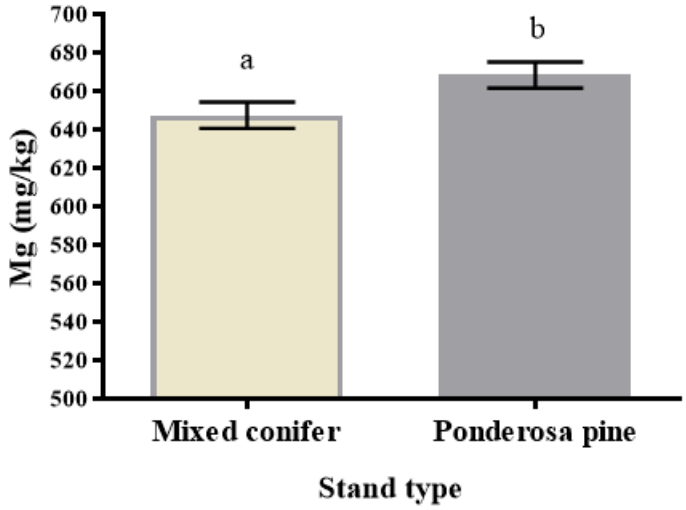

Figure 2.14 The impact of stand difference on Mg concentration (±standard error of the mean, n= 40). Different letters indicate significant differences among the stand at p<0.1.

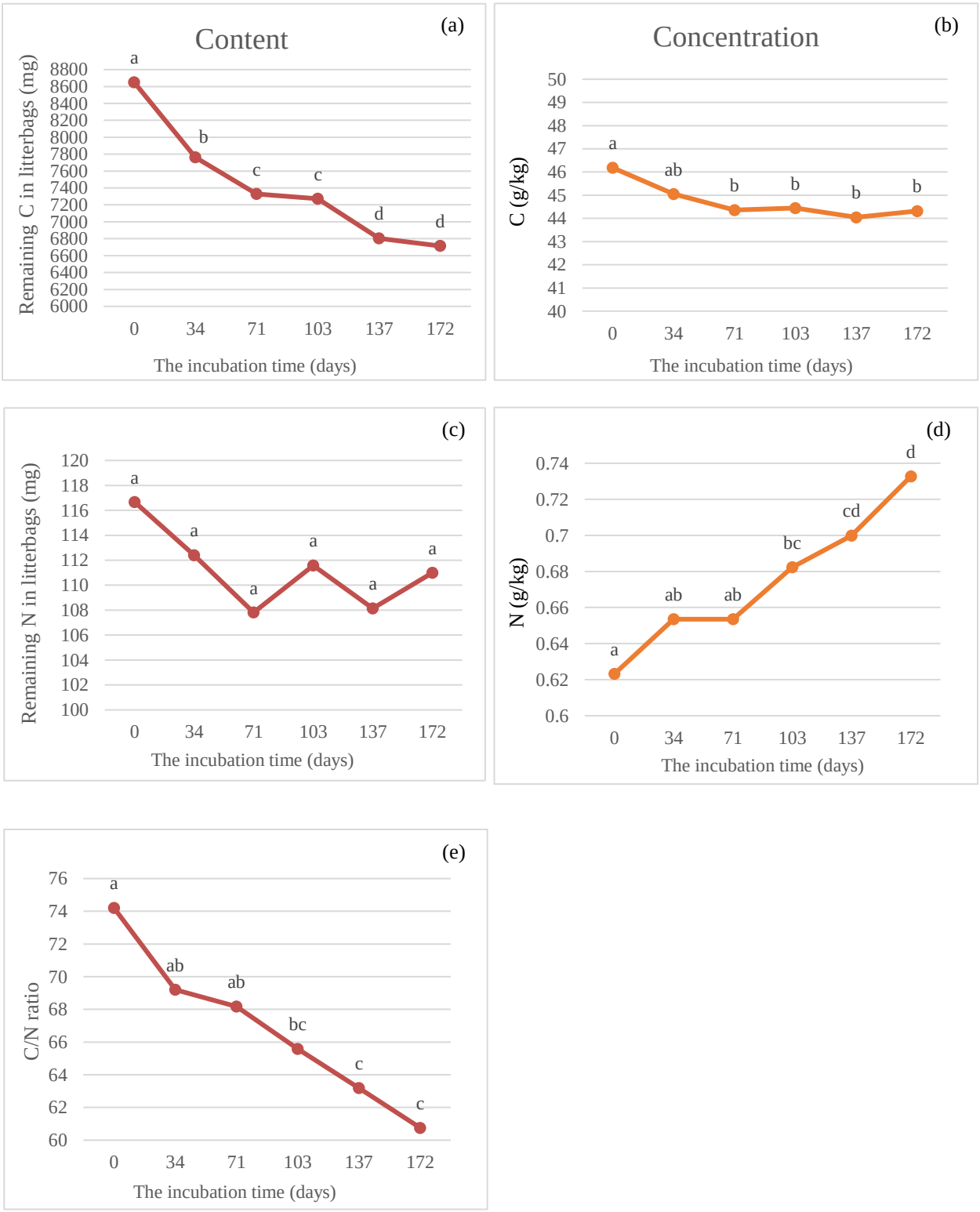

Figure 2.15 The influence of incubation time on (a) C content, (b) C concentration, (c) N content, (d) N concentration, and (e) C/N ratio. Different letters indicate significant differences among the incubation periods at p<0.1.

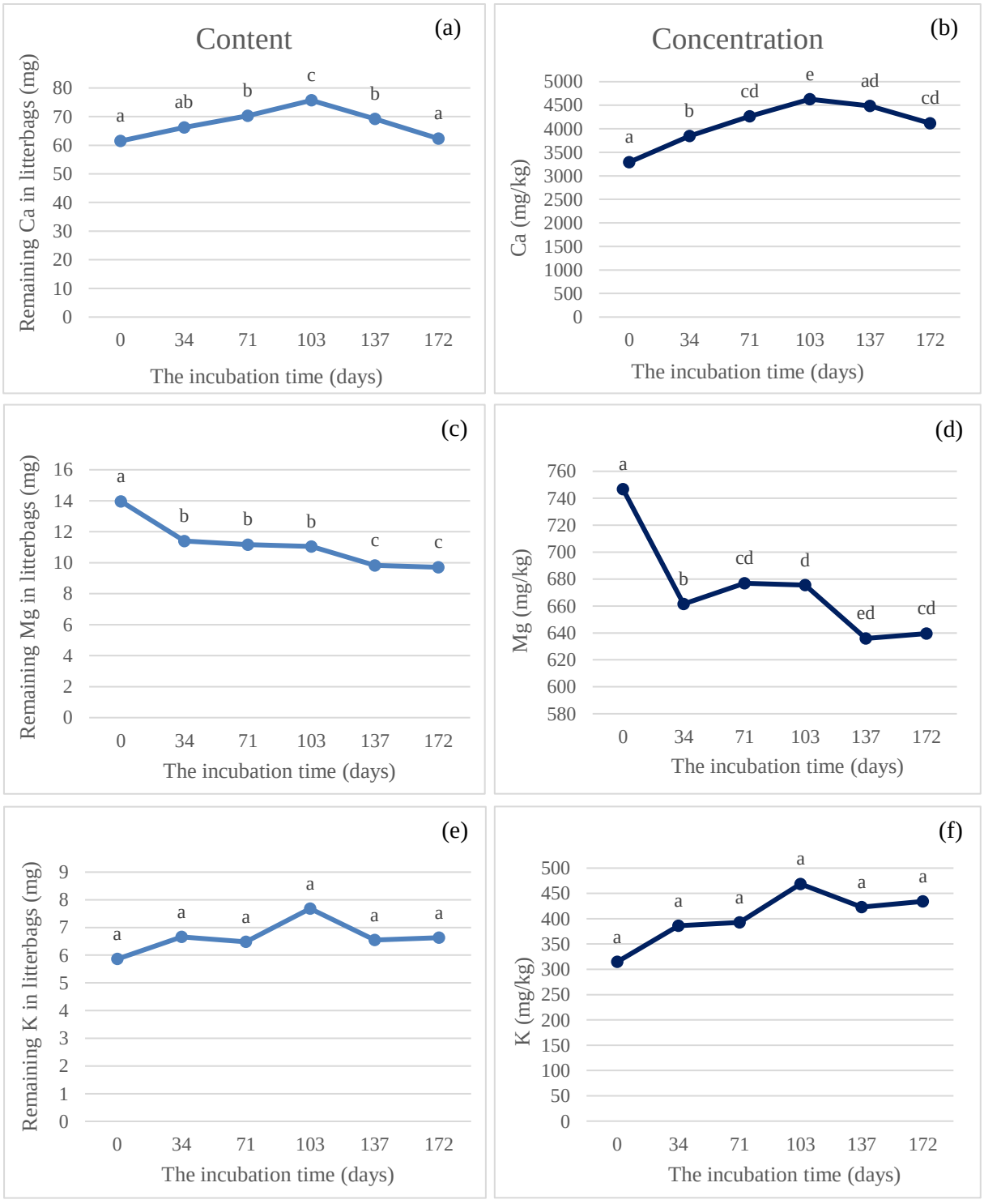

Figure 2.16 The influence of incubation time on (a) Ca content, (b) Ca concentration, (c) Mg content, (d) Mg concentration, (e) K content, and (f) K concentration. Different letters indicate significant differences among the incubation periods at p<0.1.

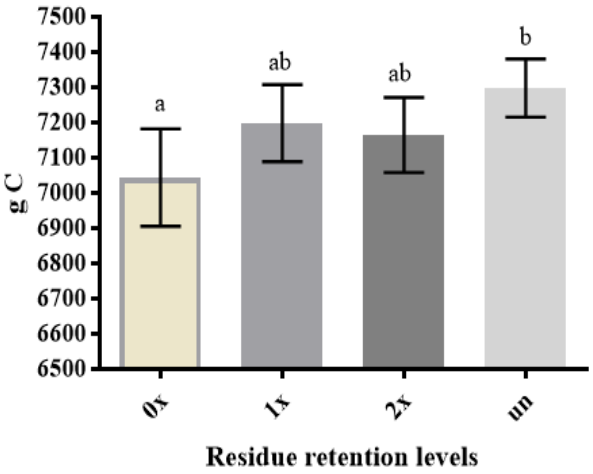

Figure 2.17 Average C content (±standard error of the mean, n=20) in litterbags as affected by residue retention treatments when all incubation dates are combined. Different letters indicate significant differences among treatment at p<0.1.

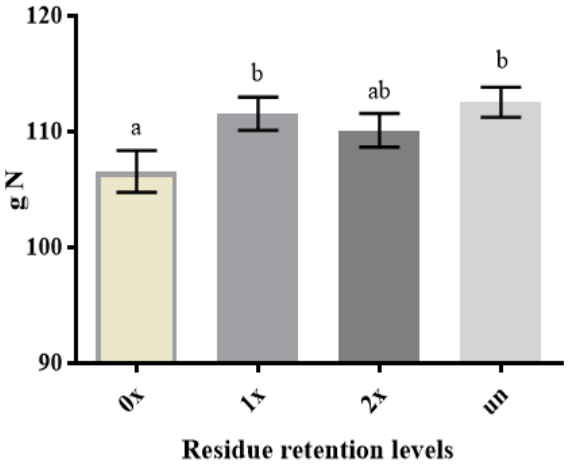

Figure 2.18 Average N content (±standard error of the mean, n=20) in litterbags as affected by residue retention treatments when all incubation dates are combined. Different letters indicate significant differences among treatment at p<0.1.

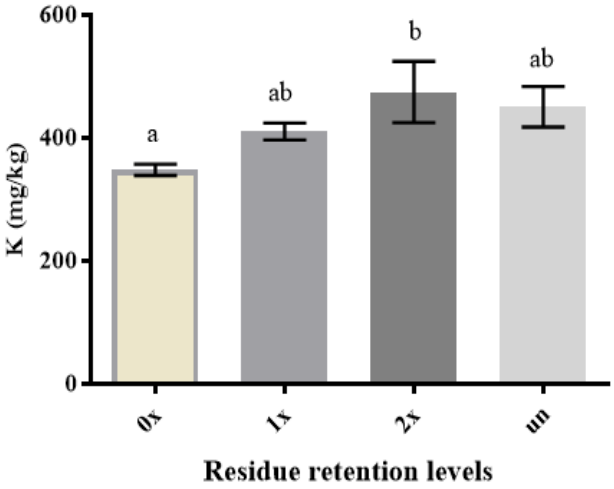

Figure 2.19 The impact of residue retention level on the concentration of potassium (K) when all incubation periods combined (±standard error of the mean, n=20). Different letters indicate significant differences among the biomass treatments at p<0.1.

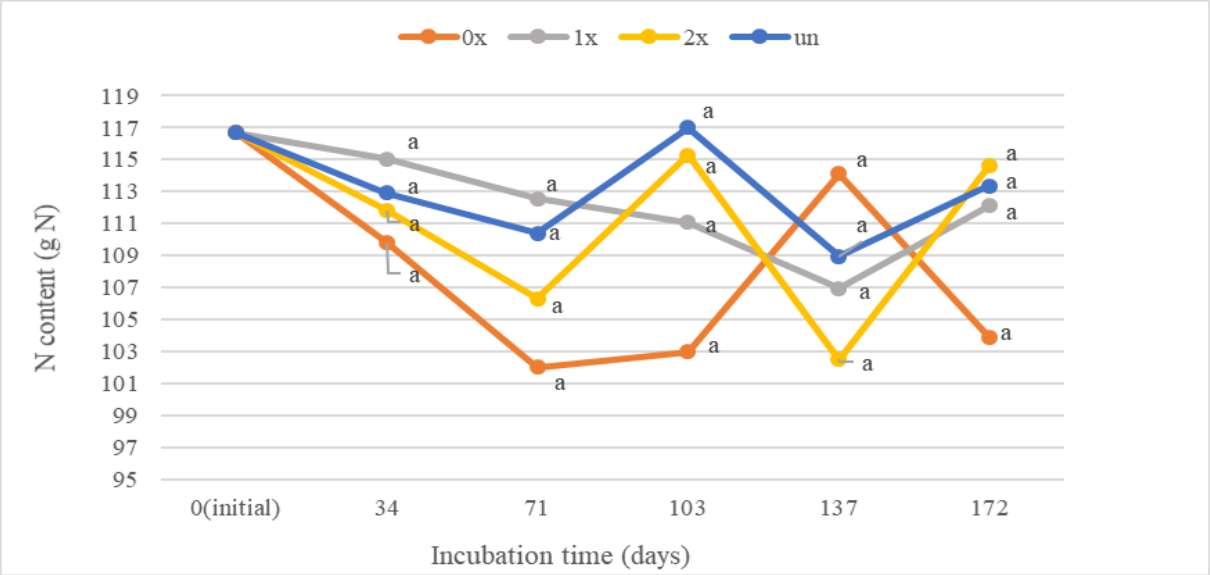

Figure 2.20 Changes in N content of litterbags in each residue retention treatment in each same date of incubation. Different letters indicate statistically significant difference ($p<0.1$) within each incubation date.

2.3.4 Fertilization and thinning impacts on litterbag decomposition rate and the properties on the forest floor and in mineral soil

I expected that N fertilization in 0x would maintain the decomposition rate compared to the other treatments with retained residue because removal residue might decrease the input of N. In addition, I expected increased decomposition rate in fertilized treatments compared to non-fertilized treatments. However, compared to non-fertilized treatments, N fertilized treatment did not affect N content, N concentration, mass loss, or decomposition rate in the litterbags (Table 2.2). More specifically, N fertilization did not show any significant effect on litter decomposition rate among the treatments. In addition, compared to other treatments, 0x did not reduce the N input into either the forest floor or mineral soil (Table 2.3).

While N fertilization did not alter C, N, OM content, and pH on the forest floor and in mineral soil, generally, the thinning increased C, N, and OM content in the forest floor, even in the 0x treatment with no residual biomass (Table 2.3). Soil pH in the forest floor was unaffected by thinning or residue treatment. The forest floor C/N ratio was lower post-harvest when compared to pre-harvest. By the

end of the study, the unthinned stand also had an increase in these forest floor properties. Similarly, mineral soil (to a depth of 20 cm) had increased C, N, and OM. The C/N ratio in the mineral soil increased significantly in all treatments. Although pH was significantly lower in the 2x treatment, it was likely not biologically significant. The C/N ratio of mineral soil in 2x was significantly higher than pre-harvest as well as the unthinned control ($p=0.094$). As expected, fertilization significantly increased soil C, N, OM, and significantly lowered the C/N ratio and pH in both the mineral soil and forest floor when compared to pre-harvest levels (Table 2.4).

Table 2.3 Pre-harvest and 3 year post-treatment forest floor and mineral soil (0-20 cm depth) C, N, C/N ratio, OM, and soil pH (± SE) by residue retention treatments. Different letters indicate statistical significance between pre-harvest and post-harvest sampling (p> 0.1) Pre-harvest forest floor (n=28) and mineral soil (n=64) and post-harvest forest floor (n=12) and mineral soil (n=24) were collected from each of the residue retention plots.

		pre-treatment Soil properties	Post-harvest soil properties			
			0x	1x	2x	unthinned
Forest floor	C (Mg/ha)	33.5 ± 53.0 a	41.8 ± 24.8 ab	107.9 ± 142.7 b	89.9 ± 58.9 ab	63.2 ± 56.8 ab
	N (kg/ha)	662.2 ± 710.7 a	1051.8 ± 660.1 ab	2115.5 ± 2781.5 b	2397.0 ± 1374.4 b	1687.6 ± 1469.7 ab
	C/N ratio	56.5 ± 13.1 a	43.8 ± 11.8 ab	48.4 ± 17.5 ab	39.6 ± 9.7 b	41.5 ± 14.9 b
	OM (Mg/ha)	60.7 ± 54.4 a	83.2 ± 42.7 ab	128.2 ± 103.3 ab	159.2 ± 102.1 b	124.1 ± 103.7 ab
	pH	5.07 ± 0.3 a	5.32 ± 0.3 a	5.22 ± 0.5 a	5.42 ± 0.4 a	5.19 ± 0.4 a
Mineral soil	C (Mg/ha)	22.0 ± 11.1 a	43.1 ± 7.7 b	44.8 ± 9.9 b	44.3 ± 7.4 b	44.0 ± 9.8 b
	N (kg/ha)	1251.8 ± 506.5 a	1538.5 ± 461.8 ab	1630.9 ± 561.9 b	1497.6 ± 432.1 ab	1646.2 ± 471.1 b
	C/N ratio	17 ± 3.1 a	29 ± 4.6 bc	29 ± 5.6 bc	**31 ± 5.3 c**	**27 ± 3.3 b**
	OM (Mg/ha)	42.9 ± 19.2 a	38.8 ± 17.7 a	44.2 ± 22.9 a	40.6 ± 18.0 a	44.4 ± 19.8 a
	pH	5.6 ± 0.2 a	5.5 ± 0.3 ab	5.6 ± 0.3 ab	5.4 ± 0.2 b	5.5 ± 0.2 ab

Table 2.4 Pre-harvest and 3 year post-treatment forest floor and mineral soil (0-20 cm depth) and C, N, C/N ratio, OM, and soil pH (±SE) by fertilization treatments. Different letters indicate statistical significance between pre-harvest and postharvest (p> 0.1). Pre-harvest forest floor (n=28) and mineral soil (n=64) and pos-t harvest forest floor (n=24) and mineral soil (n=48) were collected from each of the fertilization plots.

		Pre-harvest characteristics	Post-harvest characteristics	
			Non-fertilized (C)	Fertilized (F)
Forest floor	C (Mg/ha)	33.5 ± 33 a	65.5 ± 50 ab	85.9 ± 109 b
	N (kg/ha)	662.2 ± 711 a	1419.2 ± 1028 ab	2206.8 ± 2236 b
	C/N ratio	56 ± 13 a	47 ± 14 b	38 ±12 b
	OM (Mg/ha)	60.7 ± 54 a	124.1 ± 88 b	124.8 ± 100 b
	pH	5.07 ± 0.3 a	5.14 ± 0.33 ab	5.43 ± 0.38 b
Mineral soil	C (Mg/ha)	22.0 ± 11 a	44.0 ± 9 b	44.1 ± 8 b
	N (kg/ha)	1251.8 ± 506 a	1533.6 ± 449 b	1623.0 ± 511 b
	C/N ratio	16 ± 3 a	29 ± 4 b	28 ± 5 b
	OM (Mg/ha)	42.9 ± 19 a	44.0 ± 21 a	40.0 ± 18 a
	pH	5.6 ± 0.21 a	5.5 ± 0.25 b	5.5 ± 0.28 b

2.4 Discussions

2.4.1 Residue retention, thinning, and litter decomposition

The growing season in northern Idaho forest ranging from May to October, generally hot and dry; the mean temperature during the growing season is 14.9 ℃ and the mean annual precipitation averages 688mm most of which falls as snow during the non-growing season from November to May (U.S Climate Data Website; May in 2017). Therefore, I expected the moisture content and respiration in litterbags would decrease as time elapsed during the growing season. In addition, removal of logging residue would decrease litter decomposition because microbial activity in the litter can be reduced due to the lack of moisture content (Hicks, 2000; Hicks et al., 2003; Chapin et al., 2011).

During the incubation period which ranged from April to September, the moisture content and respiration in litterbags decreased from May to August and slightly increased thereafter which is paralleled with the air temperature data. Also, the litter in litterbags continuously lost mass and reached a maximum mass loss of approximately 25% at the end of the incubation period. This is much less than the averaged mass loss of decomposition for a wide variety of litter material in northern forests (Berg, 2000). This indicates that at the end of my study mass loss was not complete. This is because my research included just one growing season which was not long enough to expect complete decomposition of litter.

My hypothesis about residue retention treatments was that the 0x would have lower litterbag moisture content and litterbag respiration than 1x, 2x, and unthinned control due to the lack of residue retention. Consequently, 0x would have a slower decomposition rate and less mass loss than other treatments since 0x would have lower moisture content and respiration in litterbags among the residue retention treatments. Although the effect of residue retention treatments showed significant differences in litterbag moisture retention (Figure 2.8) and respiration (Figure 2.10) compared to the unthinned control, there were no significant differences among the three biomass retention levels in thinned plots. However, the comparison to unthinned control implies the negative effect of all biomass

removal (0x) on moisture content and respiration in the litterbags. Normal biomass retention (1x) helped retain a similar litterbag moisture content and respiration as compared to the unthinned control, but when all biomass was removed (0x) there was significantly lower litterbag moisture content and respiration. The reason why the unthinned control had the highest moisture content compared to the thinned plots might be because uncut forests generally have greater canopy coverage than thinned stands resulting in decreased surface soil temperature and water loss through evaporation (Prescott, 2002). Therefore, unthinned stands in my study may retain more moisture and respiration (Figures 2.9; 2.10). My results indicate that when woody biomass is removed after thinning there can be a decrease in decomposition rate due to the lack of moisture content.

In my study, however, either higher moisture content or respiration in litterbags did not lead to faster decomposition rate or larger mass loss. To explain, although the unthinned control plots had significantly higher moisture content and respiration than 0x, decomposition rate and mass loss in unthinned control were significantly lower than in 0x. There are two possible explanations. One possible cause is that during the summer the moisture content in 0x may not have been low enough to significantly impede decomposition. The other possible explanation is that low moisture content might be less influential on decomposition (Hicks 2000; Hicks et al. 2003) than water-logged conditions (Smyth et al. 2016). The higher decomposition rate in 0x than unthinned control might be explained by a decomposition study which was reported by DeCatanzaro and Kimmins (1985). According to DeCatanzaro and Kimmins (1985), they examined the effect of moisture gradient on decomposition rate, but they found no differences in decomposition in northwestern forests; the reason why the decomposition rates in three different moisture gradients were similar in their study might be because 1) moisture content did not very important to litter decomposition in northwestern forest and/or 2) the other factors controlling decomposition such as substrate quality or microbial composition were more important than moisture gradients in their study. Likewise, in my research, other controlling factors could more strongly affect litter decomposition than moisture content did. I used the same litter for both ponderosa pine and mixed conifer stands; the effect of litter quality cannot affect the rate of

decomposition in this study. Therefore, the possible effect on litter decomposition rate might be thinning operation which can increase the temperature on forest surface. In addition to moisture content, temperature also plays a key role in regulating decomposition (Swift et al., 1979; Lloyd and Taylor, 1994; Wickland et al., 2008). Clear-cut harvesting increased litter decomposition because less canopy coverage increased forest surface and soil temperatures and subsequently increased litter decomposition (Prescott et al., 2000; Lee et al., 2002). In this regard, retaining no surface biomass after thinning likely resulted in more solar radiation reaching the forest floor. Increased solar radiation will increase the temperature on the forest floor at which the litterbags located. Consequently, if moisture is not limiting, rising temperatures can increase microbial decomposition (Chapin et al., 2011). During litterbag incubation, adequate litter moisture content and rising forest surface temperature in 0x possibly increased the decomposition rate and mass loss as compared to the unthinned control; with the 1x and 2x having intermediate levels of mass loss. In my study, I was not able to get forest surface temperature data because of broken field data loggers; however, based on the study by Thibodeau et al. (2000), I postulated that forest surface temperature would be warmer than soil temperature which is measured (Figure 2.12) and that the surface temperature might be more susceptible to residue retention levels. In addition, wood decomposition studies also support the temperature effect more than moisture when wood stake placed on the top of the forest floor (Yatskov et al. 2003; Mackensen et al. 2003; Jurgensen et al. 2006; Risch et al. 2013; Finér et al., 2016). The activity of decomposing fungi has been found to increase 1.4 - 4.8 fold for every 10º C rise in mean annual temperature (Q10; Mackensen et al., 2003; Yatskov et al., 2003; Hermann and Bauhus, 2013). I found that the k-rate increased approximately 28 % from the unthinned to the thinned plots. The reason for higher k-rate in the thinned plots might result from the higher temperature in the thinned plots. Figure 2.13 showed the relationship between soil temperature and weight loss in this study; the weight loss in most residue retention treatments except for 1x ($R^2 = 0.5233$) were highly related to soil temperature at 10 cm depth. Therefore, my decomposition results might imply that the thinning operation increased decomposition. These are similar to research on thinned white spruce forests in

Alaska (Piene and Van Cleve, 1978) and precommercial thinning in a balsam fir forest in Quebec (Thibodeau et al., 2000). In both studies, thinning increased decomposition rate compared to unthinned treatments.

Soil microorganisms are critical for nutrient cycling processes (Paul and Juma, 1981; Ross and Sparling, 1993). Forest management activities may reduce soil organic matter and change microbe levels (Ladd et al., 1985). I did not find decreases in soil OM after thinning which also explains the lack of changes in cations. However, in the long-term, removal of all thinning biomass may have a negative influence on nutrient pools and decomposition in northwestern forests. Although this study included just one growing season, no biomass retention in thinned plot decreased moisture content and litterbag respiration compared to normal biomass retention.

Another possible reason why decomposition occurred faster in 0x might be due to photodegradation of litter in the litterbags. Photodegradation refers to litter decomposition which is caused by light exposure or solar radiation. In many studies, light exposure or solar radiation increased mass loss and decomposition through photodegradation (Austin and Vivanco, 2006; Brandt et al., 2010; Liu et al., 2014). Moreover, in an arid and semi-arid area, the photodegradation is believed to be a direct contributor to litter decomposition. Thus, I believed that during the hot and dry summer in northern Idaho 0x had more solar radiation and light exposure when compared to other treatments that had the retained residue or canopy. Therefore, photodegradation probably had more significant effects on the litterbags in 0x plots and it might make the larger mass loss and the higher decomposition rate in 0x among the residue retention treatments.

2.4.2 Changes in litterbag C and N

I expected that 0x would have a higher amount of remaining C and N (content) in the litterbags compared to other treatments (1x, 2x, and unthinned control) because it would have a slower mass loss and decomposition rate. However, C loss in the litterbags was larger in 0x compared to other treatments (1x, 2x, and unthinned control) due to faster decomposition rate and more mass loss. Even though carbon loss was not statistically different among the three biomass retention levels (0x, 1x, and 2x), the remaining C content in 0x was significantly lower than unthinned control. The significant differences in remaining C content between 0x and unthinned control might result from litterbag mass loss. As each treatment lost the mass of litterbags, the corresponding C content was lost into the soil or was respired into the air from litterbags. These changes in mass loss and C content resulted in the similarity of C concentration among the residue retention treatments.

The nitrogen content in the litterbag was changed more in each biomass retention levels than C content. The remaining N of litterbags in 0x was significantly lower than in the 1x and unthinned treatments (Figure 2.18). The higher loss of the litterbag N might result from the faster decomposition rate and higher mass loss in 0x. My results of litterbag N content contradicted those of Smolander et al. (2008). They reported that whole-tree harvest resulted in lower N mineralization (more remaining N content in litterbags) than stem-only harvest. However, my results parallel a study by Purahong et al. (2014) who found that greater forest management harvest intensities resulted in the greater mass loss and N content loss in the litterbags when compared to unmanaged control and lower forest management intensity plots. Therefore, since the treatment with no residual woody biomass remaining on the soil surface (0x) was the more intensive treatment as compared to retaining a normal amount of biomass (1x), the 0x treatment had lower litterbag N content. Similar to C concentration, N concentration did not show any significant differences among the residue treatments. This might be because the treatment, which had the high litterbag mass loss, also lost the significant N content in

litterbags at the same time. Therefore, those changes in litterbag mass loss and N content created the similarity of N concentration among the residue retention treatments. As decomposition proceeded, C concentration decreased over time (Figure 2.15). This is because C content in litterbag decreased over time as litter in litterbags broken down (Figure 2.15). The overall correlation coefficient between remaining mass and C content is 0.9101 suggesting that higher mass loss may be due to loss of C. Therefore, the decreases in both mass and C content in litterbags are the causes of the decreases in C concentration. However, the concentration of N increased over time. During the incubation period, the increased N concentration in litterbags was often observed in many studies (Goya et al., 2008; Smolander et al., 2008); where increased N concentration resulted from the increased N content in litter by microbial translocation. However, since the N content in my study did not change over time, there was no microbial import from surrounding soil into litter in the litterbags. Therefore, the increased N concentration of my study might be because N content in litterbag stayed stable while the mass of litterbags decreased over time.

Figure 2.20, after 71days each treatment showed different trends; nitrogen content in litterbags increased in 0x, 2x, and unthinned control while the N content in 1x continuously decreased. I have inferred that the increased N content in litterbag might result from microbial translocation from surrounding soil into the litter. After 103 days until the end of the incubation period, 0x showed and opposite trend compared to other treatment; while litterbag N content in other treatments decreased, N content in 0x increased and vice versa. One possible reason for the opposite direction in 0x was that 0x lost much N content from litterbag compared to other treatments during the initial phase of the incubation period. Consequently, the decomposing litter in 0x might have been a shortage of N and microbial decomposers probably needed to import N from surrounding soil into the litter. The other treatments followed 0x with the same trend but they are a step behind. This fluctuation between N release and accumulation by microbial decomposers was often found in many studies (Bååth and Söderström, 1979; Berg and Staaf, 1981; Aber and Melillo, 1982; Boberg, 2009).

When all biomass treatments were combined litterbags C/N ratio was found to decrease over time. In this case, while C is respired as CO_2, litterbag N content remained stable during the incubation period (Figure 2.15). This decreasing C/N ratio during decomposition is often observed (Smolander et al., 2008; Klockow et al., 2014; Purahong et al., 2014). However, I did not detect any changes in the litterbag C/N ratio among the residue retention treatments. This is contradictory to results of Smolander et al. (2008) which found that whole-tree harvesting resulted in a higher C/N ratio than stem-only harvest. It is likely that my litterbags had no C/N ratio differences among the biomass retention levels because litterbags in each treatment lost C (Figure 2.17) and N (Figure 2.18), simultaneously.

My study is short-term, including just one growing season, which means that one growing season could not cover a yearly cycle in soil temperature and moisture. Also, my research data is only applicable to ponderosa pine and mixed conifer stands in northern Idaho. Long-term studies lasting 10 to 15 years note that removing residual biomass can negatively affect C and N mineralization and decrease N inputs to the soil (Olsson et al., 1996; Piatek and Lee Allen, 1999; O'Connell et al., 2004; Smolander et al., 2008). Therefore, for the future, forest manager should figure out more about this positive effect of biomass removal on N release through long-term study.

2.4.3 Cation changes in the litterbags

I hypothesized that 0x would have a higher remaining cation concentration in the litterbags (lower release of cations) compared to other treatments due to the subsequent effect of slow decomposition rate. Also, I expected that the concentration of Ca, Mg, and K in the litterbags would decrease as decomposition proceeds. The concentration of Ca increased for 103 days and decreased thereafter. This increase in Ca concentration for 103 days might result from the increased Ca content, while litterbag mass decreased. In other studies, the two-stage pattern of Ca dynamics, which initially increased and decreased thereafter, has often been observed (Berg and Staaf, 1987; Hasegawa and

Takeda, 1996; Bhatta, 2000; Palviainen et al., 2004; Osono and Takeda, 2004). Calcium content and concentration might be increased by microbial translocation; calcium can be imported by fungi into the litterbags from the soil; this is because Ca support the growth of certain fungi species and Ca is used for decomposition of lignin (Cromack et al., 1975; Connolly and Jellison, 1995). Therefore, the initial Ca content and concentration increases in my study might be explained by microbial translocation. The latter decrease in Ca concentration was probably due to the reduced Ca content. The concent of Ca can decrease during microbial decomposition of cell structure components. Since Ca is one of the components of cell structure and Ca is incorporated into a plant cell, accordingly, the release of Ca mostly depends on microbial decomposition. Therefore, Ca content might decrease as litter broken down by microbial decomposers (Salisbury and Ross, 1992; Rustad and Cronan, 1988; O'Connell and Grove, 1996; Ukonmaanaho and Starr, 2001; Palviainen et al., 2004; Goya et al., 2008). Thus, as decomposition proceeds and the structure of litter is broken down, litterbag Ca concentration might decline.

Unlike Ca concentration, the concentration of Mg did not show the two-phase pattern, but did continuously decrease over time. The possible reason for this Mg trend might be due to the existing status of Mg in a plant cell. To explain, similar to Ca, Mg is also associated with a structural component in plant cells; however, magnesium is more mobile than Ca (Lousier and Parkinson, 1978; Bhatta, 2000) because magnesium exists as a solution in the plant cells while Ca is structurally bound in plant cells. Thus, Mg can be released through both leaching and microbial decomposition while the release of Ca generally depends on microbial decomposition. Therefore, I have inferred that Mg concentration in my study may continuously decrease through both leaching and microbial decomposition as decomposition proceeds.

In contrast to Ca and Mg, K concentrations did not change over time (Figure 2.16). It was surprising that both concentration and content of K stayed stable during incubation periods even though the concentration and content of Ca and Mg, which are less mobile than K in litter during decomposition

(Lousier and Parkinson, 1978; Blair, 1988; Hasegawa, 1996; Bhatta et al., 2000), significantly changed over time. These trends of both K concentration and K content are different from other studies that reported K content or K concentration rapidly decreased during decomposition (Palviainen et al., 2004; Goya et al., 2008; Purahong et al., 2014). The trend of K concentration in my study may be explained by the status of litter. The litter in the litterbag might have lost K before it was collected for my study. To explain, I made the litterbag by using the fresh litter which probably fell during the autumn 2015 and before I collected the litter in March 2016. During the winter until I collected litter, K may have already leached out of the litter because, in general, K is rapidly released from the litter.

I hypothesized that residue retention treatments would affect the concentration in the litterbags cation (Ca, Mg, and K). Since losses of both Ca and Mg are associated with decomposition. Goya et al. (2008) observed higher losses of Ca and Mg concentration on the site with higher decomposition rate or mass loss. However, I did not find any significant differences of Ca and Mg concentration among the biomass retention levels since there were no significant differences in decomposition rate or mass loss among three biomass retention levels. In contrast, the concentration and content of K were significantly affected by biomass treatments. The 2x treatment increased K concentrations in the litterbags (Figure 2.19). The increased K concentrations in 2x are likely caused by K leaching from the downed material into the litterbags. As I mentioned earlier, the mobility of K in plant residue can be relatively high and more easily released from logging residue when compared to Ca or Mg, which are bound to tissue structure (Lousier and Parkinson, 1978; Barber and Van Lear, 1984; Fahey et al., 1991; Laskowski et al., 1995; Palvianen et al., 2004). Since K is not bound to plant tissue structure, it exists as water-soluble salts that can be leached without microbial activity (Tukey, 1970; Salisbury and Ross, 1992). Thus, logging residues or canopy can increase K concentration in forest floor and therefore it can be a good source of K in the soil. My results for Ca, Mg, and K agree with reported data indicating that biomass retention does not affect Ca and Mg concentrations, but can increase K concentrations (Wall and Hytönen, 2011).

Therefore, removing all biomass after thinning can reduce the site productivity if there is a K limitation. Furthermore, the impact of biomass retention levels on the concentration of Ca and Mg were insignificant in short-term but were significant on K concentration. Therefore, the impact of biomass removal on litter cations is minimal. However, long-term changes from repeated biomass removals may alter site productivity.

2.4.4 Stand effects on decomposition

There were two significant differences between stands: litterbag moisture content (Figure 2.6) and Mg concentration (Figure 2.14). The mixed conifer stand had more moisture and may be related to higher levels of canopy cover which was collected on these sites as part of other research. Although not statistically significant, the mixed conifer stand had approximately 7% greater canopy cover compared to the ponderosa pine stand; leading to greater shading and reduced evaporation. In addition, canopy cover at each plot varied considerably and ranged from 4-90% in the mixed conifer stand and 9-80% in the ponderosa pine stand. The amount of overstory shading is likely responsible for the differences in litterbag moisture content.

Litterbags in the ponderosa pine stand had the higher Mg concentration and 0.2 % greater mass than those in the mixed conifer stand. Magnesium is associated with the structure of litter and as decomposition proceeded it decreased with the mass loss (Laskowski et al., 1995; Palvianen et al., 2004). Therefore, higher Mg concentrations in the ponderosa pine stand might result from higher remaining mass.

2.4.5 Nitrogen response to fertilization and biomass retention

My N fertilization hypothesis was 1) N fertilization will compensate for less input of N due to the biomass removal and therefore, 2) N fertilization will increase decomposition rate compared to the

non-fertilized treatments to maintain the decomposition rate as treatments with more residue retention. However, my results did not support these hypotheses. In the other studies, it was often observed that residue removals did not lead to significant decreases in soil N (Carter et al., 2002; Mendham et al., 2003; Belleau et al., 2006). According to Smolander et al. (2010), compared to normal biomass retention treatment, double amounts of retained biomass did not result in significant differences in parameters that they measured. For example, there were no significant differences in N mineralization in the soil between normal residue retention treatment and double amount residue treatment. In addition, C mineralization in double biomass retention treatments had the lowest C mineralization as compared to the normal residue retention. The possible explanation for the situation that 2x did not make any changes compared to 1x might be due to uneven distribution of thinning residue on the forest surface.

This might be related to the uneven distribution of thinning residues on the soil surface, which may also have been the case for my study plots. Likewise, differences in soil N content within the residue retention levels in my results also were insignificant (Table 2.3) contradicting the first part of my hypothesis (residue removals would decrease N input).

However, when compared to pre-harvest levels, thinning resulted in increased N in both the forest floor and mineral soil. This is likely due to the increased amount of biomass on the forest floor after thinning and the alteration of soil temperature and moisture which may lead to faster decomposition of retained residue. Forest floor pH in the fertilized plot was significantly higher as compared to pre-harvest but it did not differ from the non-fertilized plot. This significant increase may be due to higher levels of OM on the forest floor which elevated base cations in forest floor compared to pre-harvest (DeByle 1980; Megahan 1990; Jones and Jacobsen, 2005). Mineral soil pH decreased in fertilized plot compared to pre-harvest although it was not different from non-fertilized plot. Although I did not measure nitrification or ammonification on these study sites, N fertilization has been shown increase nitrification which may reduce soil pH (Chase et al. 1967).

The second part of my hypothesis about fertilization that N fertilization would maintain decomposition rates was also not supported by my results. One reason that N fertilization may not be a significant factor for litter mass loss on my sites is that they were fertilized 3 years before the litterbags were installed. It is likely that the fertilization effect on microbial populations was short-lived (Hobbie and Vitousek, 2000) or had no significant effect (Prescott 1995). This could be because our initial litter (initial N was 116.67g N in litterbag) had high enough N concentration for microbial decomposition (Sinsabaugh et al. 2002), but other environmental factors (e.g., temperature or moisture) may have effected litterbag decomposition to a greater extent (Knorr et al. 2005).

2.5 Conclusion

Understanding the impact of thinning on site processes is critical for developing best management practices for maintaining a long-term site and soil productivity. Although the scope of inference for my study is limited to the two sites where I installed litterbags, this provides needed baseline data on the impacts of thinning on litter decomposition rates. The findings of my study also have important implications for the effect of biomass retention on ecosystem function, especially nutrient cycling. Residue (biomass) removal had no negative effects on litter decomposition, cation concentration, C, N, and C/N ratio despite decreased litterbag moisture content and respiration. Instead, removal of biomass increased litterbag decomposition rate and input of C and N into the forest floor or the soil. In addition, the high (2x) residue retention treatment had increased litterbag K concentration, indicating that residues may be a source of K on nutrient-limited sites. However, N fertilization in this study did not have a compensating effect on litter decomposition by providing N into the soil or forest floor. This could be because fertilizer was applied 3 years before my litterbags were installed and N had already been used by microbes or leached into the mineral soil. Therefore, I conclude that removal of residue will not damage short-term ecosystem function in these northern Idaho soils and that the

application of N fertilization three years before was not important for maintaining the current nutrient cycle. Other above- and belowground long-term data are needed to assess the effects of residue retention levels on other site processes.

References

Aber, J. D., & Melillo, J. M. (1980). Litter decomposition: measuring relative contributions of organic matter and nitrogen to forest soils. *Canadian Journal of Botany*, *58*(4), 416-421.

Aber, J. D., & Melillo, J. M. (1982). Nitrogen immobilization in decaying hardwood leaf litter as a function of initial nitrogen and lignin content. *Canadian Journal of Botany*, *60*(11), 2263 2269.

Austin, A. T., & Ballaré, C. L. (2010). Dual role of lignin in plant litter decomposition in terrestrial ecosystems. *Proceedings of the National Academy of Sciences*, *107*(10), 4618-4622.

Bååth, E., & Söderström, B. (1979). Fungal biomass and fungal immobilization of plants nutrients in Swedish coniferous forest soils. *Revue d'Ecologie et de Biologie du Sol*, *16*(4), 477-489.

Barber, B. L., & Van Lear, D. H. (1984). Weight loss and nutrient dynamics in decomposing woody loblolly pine logging slash. *Soil Science Society of America Journal*, *48*(4), 906-910.

Belleau, A., Brais, S., & Paré, D. (2006). Soil nutrient dynamics after harvesting and slash treatments in boreal aspen stands. *Soil Science Society of America Journal*, *70*(4), 1189-1199.

Berg B., & Staaf, H. (1987). Release of nutrients from decomposing white birch leaves and Scots pine needle litter. *Pedobiologia*, *30*, 55-63.

Berg, B. (2000). Litter decomposition and organic matter turnover in northern forest soils. *Forest ecology and Management*, *133*(1), 13-22.

Berg, B., & Staaf, H. (1981). Leaching, accumulation and release of nitrogen in decomposing forest litter. *Ecol. Bull*, *33*, 163-178.

Berg, B., Staaf, H., & Wessen, B. (1987). Decomposition and nutrient release in needle litter from nitrogen-fertilized scots pine (pinus sylvestris) stands. *Scandinavian Journal of Forest Research*, *2*(1-4), 399-415.

Bhatta, B. K. (2000). Effect of soil nitrogen on decomposition and nutrient dynamics of Japanese black pine litter. *Appl For Sci Kansai*, *9*, 75-81.

Binkley, D. (1991). Connecting soils with forest productivity. *USDA For. Serv. Gen. Tech. Rep. INT GTR-280*, 66-70.

Blair, J. M. (1988). Nutrient release from decomposing foliar litter of three tree species with special reference to calcium, magnesium and potassium dynamics. *Plant and Soil*, *110*(1), 49-55.

Brandt, L. A., King, J. Y., Hobbie, S. E., Milchunas, D. G., & Sinsabaugh, R. L. (2010). The role of photodegradation in surface litter decomposition across a grassland ecosystem precipitation gradient. *Ecosystems*, *13*(5), 765-781.

Carter, M. C., Dean, T. J., Zhou, M., Messina, M. G., & Wang, Z. (2002). Short-term changes in soil C, N, and biota following harvesting and regeneration of loblolly pine (Pinus taeda L.). *Forest Ecology and Management*, *164*(1), 67-88.

Chapin III, F. S., Matson, P. A., & Vitousek, P. (2011). *Principles of terrestrial ecosystem ecology*. Springer Science & Business Media.

Chase, F. E., Corke, C. T., & Robinson, J. B. (1968). Nitrifying bacteria in soil. *The ecology of soil bacteria. University of Toronto Press, Toronto, Canada*, 593-611.

Connolly, J. H., & Jellison, J. (1995). Calcium translocation, calcium oxalate accumulation, and hyphal sheath morphology in the white-rot fungus Resinicium bicolor. *Canadian Journal of Botany*, *73*(6), 927-936.

Cromack, K., Todd, R. L., & Monk, C. D. (1975). Patterns of basidiomycete nutrient accumulation in conifer and deciduous forest litter. *Soil Biology and Biochemistry*, *7*(4-5), 265-268.

DeByle, N. V. (1980). Harvesting and site treatment influences on the nutrient status of lodgepole pine forests in western Wyoming. In *Proceedings of Symposium on Environmental Consequences of Timber Harvesting in Rocky Mountain Forests. USDA For. Ser. Gen. Tech. Rep. INT 90* (pp. 135-155).

DeCatanzaro, J. B., & Kimmins, J. P. (1985). Changes in the weight and nutrient composition of litter fall in three forest ecosystem types on coastal British Columbia. *Canadian Journal of Botany*, *63*(6), 1046-1056.

Demirbas, A. (2009). Progress and recent trends in biodiesel fuels. *Energy conversion and management*, *50*(1), 14-34.

Fahey, T. J., Stevens, P. A., Hornung, M., & Rowland, P. (1991). Decomposition and nutrient release from logging residue following conventional harvest of Sitka spruce in North Wales. *Forestry: An International Journal of Forest Research*, *64*(3), 289-301.

Finér, L., Jurgensen, M., Palviainen, M., Piirainen, S., & Page-Dumroese, D. (2016). Does clear-cut harvesting accelerate initial wood decomposition? A five-year study with standard wood material. *Forest Ecology and Management*, *372*, 10-18.

Goya, J. F., Frangi, J. L., Pérez, C., & Dalla Tea, F. (2008). Decomposition and nutrient release from leaf litter in Eucalyptus grandis plantations on three different soils in Entre Ríos, Argentina. *Bosque*, *29*(3).

Hasegawa, M. (1996). Carbon and nutrient dynamics in decomposing pine needle litter in relation to fungal and faunal abundances. *Pedobiologia*, *40*, 171-184.

Helmisaari, H. S., Hanssen, K. H., Jacobson, S., Kukkola, M., Luiro, J., Saarsalmi, A., ... & Tveite, B. (2011). Logging residue removal after thinning in Nordic boreal forests: long-term impact on tree growth. *Forest Ecology and Management*, *261*(11), 1919-1927.

Henderson, G. S. (1995). Soil organic matter: a link between forest management and productivity. *Carbon forms and functions in forest soils*, (carbonformsandf), 419-435.

Hermann, S., & Bauhus, J. (2013). Effects of moisture, temperature and decomposition stage on carbon loss from coarse woody debris (CWD) of important European species. *Scand J For Res*, *28*, 346-357.

Hicks, W. T. (2000). *Modeling nitrogen fixation in dead wood* (Doctoral dissertation).

Hicks, W. T., Harmon, M. E., & Myrold, D. D. (2003). Substrate controls on nitrogen fixation and respiration in woody debris from the Pacific Northwest, USA. *Forest ecology and management*, *176*(1), 25-35.

Hobbie, S. E., & Vitousek, P. M. (2000). Nutrient limitation of decomposition in Hawaiian forests. *Ecology*, *81*(7), 1867-1877.

Jang, W., Keyes, C. R., & Page-Dumroese, D. (2015). Impact of biomass harvesting on forest soil productivity in the northern Rocky Mountains.

Jones Jr, J. B., & Case, V. W. (1990). Sampling, handling and analyzing plant tissue samples. *Sampling, handling and analyzing plant tissue samples.*, (Ed. 3), 389-427.

Jones, C., & Jacobsen, J. (2005). Plant nutrition and soil fertility. *Nutrient management module*, *2*, 11.

Jurgensen, M., Reed, D., Page-Dumroese, D., Laks, P., Collins, A., Mroz, G., & Degórski, M. (2006). Wood strength loss as a measure of decomposition in northern forest mineral soil. *European Journal of Soil Biology*, *42*(1), 23-31.

Klockow, P. A., D'Amato, A. W., Bradford, J. B., & Fraver, S. (2014). Nutrient concentrations in coarse and fine woody debris of Populus tremuloides Michx.-dominated forests, northern Minnesota, USA. *Silva Fenn*, *48*, 962.

Knorr, M., Frey, S. D., & Curtis, P. S. (2005). Nitrogen additions and litter decomposition: A meta analysis. *Ecology*, *86*(12), 3252-3257.

Ladd, J. N., Jackson, R. B., Amato, M., & Butler, J. H. A. (1983). Decomposition of plant material in Australian soils. I. The effect of quantity added on decomposition and on residual microbial biomass. *Soil Research*, *21*(4), 563-570.

Laskowski, R., Berg, B., Johansson, M. B., & McClaugherty, C. (1995). Release pattern for potassium from decomposing forest needle and leaf litter. Long-term decomposition in a Scots pine forest. IX. *Canadian Journal of Botany*, *73*(12), 2019-2027.

Lavelle, P., Blanchart, E., Martin, A., Martin, S., & Spain, A. (1993). A hierarchical model for decomposition in terrestrial ecosystems: application to soils of the humid tropics. *Biotropica*, 130-150.

Lee, J., Morrison, I. K., Leblanc, J. D., Dumas, M. T., & Cameron, D. A. (2002). Carbon sequestration in trees and regrowth vegetation as affected by clearcut and partial cut harvesting in a second growth boreal mixedwood. *Forest Ecology and Management*, *169*(1), 83-101.

Liu, S., Hu, R., Cai, G., Lin, S., Zhao, J., & Li, Y. (2014). The role of UV-B radiation and precipitation on straw decomposition and topsoil C turnover. *Soil Biology and Biochemistry*, *77*, 197-202.

Lloyd, J., & Taylor, J. A. (1994). On the temperature dependence of soil respiration. *Functional ecology*, 315-323.

Lousier, J. D., & Parkinson, D. (1978). Chemical element dynamics in decomposing leaf litter. *Canadian Journal of Botany*, *56*(21), 2795-2812.

Mackensen, J., Bauhus, J., & Webber, E. (2003). Decomposition rates of coarse woody debris—a review with particular emphasis on Australian tree species. *Australian Journal of Botany*, *51*(1), 27-37.

Mann, L., & Tolbert, V. (2000). Soil sustainability in renewable biomass plantings. *AMBIO: A Journal of the Human Environment*, *29*(8), 492-498.

Megahan, W. F. (1990). Erosion and site productivity in western-montane forest ecosystems. *Proceedings, Management and Productivity of Western-Montana Forest Soils*, 146-150.

Mendham, D. S., O'connell, A. M., Grove, T. S., & Rance, S. J. (2003). Residue management effects on soil carbon and nutrient contents and growth of second rotation eucalypts. *Forest Ecology and Management*, *181*(3), 357-372.

Murphy, K. L., Klopatek, J. M., & Klopatek, C. C. (1998). The effects of litter quality and climate on decomposition along an elevational gradient. *Ecological Applications*, *8*(4), 1061-1071.

Nelson, D. W., & Sommers, L. E. (1996). Total carbon, organic carbon, and organic matter. *Methods of soil analysis part 3—chemical methods*, (methodsofsoilan3), 961-1010.

O'Connell, A. M., & Grove, T. S. (1996). Biomass production, nutrient uptake and nutrient cycling in the jarrah (Eucalyptus marginata) and karri (Eucalyptus diversicolor) forests of south-western Australia. *Nutrition of Eucalypts. Canberra, Australia. CSIRO*, 155-189.

O'Connell, A. M., Grove, T. S., Mendham, D. S., & Rance, S. J. (2004). Impact of harvest residue management on soil nitrogen dynamics in Eucalyptus globulus plantations in south western Australia. *Soil Biology and Biochemistry*, *36*(1), 39-48.

Olson, J. S. (1963). Energy storage and the balance of producers and decomposers in ecological systems. *Ecology*, *44*(2), 322-331.

Olsson, B. A., Staaf, H., Lundkvist, H., Bengtsson, J., & Kaj, R. (1996). Carbon and nitrogen in coniferous forest soils after clear-felling and harvests of different intensity. *Forest ecology and management*, *82*(1-3), 19-32.

Osono, T., & Takeda, H. (2004). Potassium, calcium, and magnesium dynamics during litter decomposition in a cool temperate forest. *Journal of Forest Research*, *9*(1), 23-31.

Padilla, F. M., & Pugnaire, F. I. (2007). Rooting depth and soil moisture control Mediterranean woody seedling survival during drought. *Functional Ecology*, *21*(3), 489-495.

Page-Dumroese, D. S., Jurgensen, M., & Terry, T. (2010). Maintaining soil productivity during forest or biomass-to-energy thinning harvests in the western United States. *Western Journal of Applied Forestry*, *25*(1), 5-11.

Palviainen, M., Finér, L., Kurka, A. M., Mannerkoski, H., Piirainen, S., & Starr, M. (2004). Release of potassium, calcium, iron and aluminium from Norway spruce, Scots pine and silver birch logging residues. *Plant and soil*, *259*(1-2), 123-136.

Parton, W., Silver, W. L., Burke, I. C., Grassens, L., Harmon, M. E., Currie, W. S., ... & Fasth, B. (2007). Global-scale similarities in nitrogen release patterns during long-term decomposition. *science*, *315*(5810), 361-364.

Paudel, E., Dossa, G. G., de Blécourt, M., Beckschäfer, P., Xu, J., & Harrison, R. D. (2015). Quantifying the factors affecting leaf litter decomposition across a tropical forest disturbance gradient. *Ecosphere*, *6*(12), 1-20.

Paul, E. A., & Juma, N. G. (1981). Mineralization and immobilization of soil nitrogen by microorganisms. *Ecological Bulletins (Sweden)*.

Piatek, K. B., & Allen, H. L. (1999). Nitrogen mineralization in a pine plantation fifteen years after harvesting and site preparation. *Soil Science Society of America Journal*, *63*(4), 990-998.

Piene, H., & Cleve, K. V. (1978). Weight loss of litter and cellulose bags in a thinned white spruce forest in interior Alaska. *Canadian Journal of Forest Research*, *8*(1), 42-46.

Powell, D. C. (1999). *Suggested stocking levels for forest stands in northeastern Oregon and southeastern Washington: an implementation guide for the Umatilla National Forest*. US Department of Agriculture, Pacific Northwest Region, Umatilla National Forest.

Powers, R. F., Scott, D. A., Sanchez, F. G., Voldseth, R. A., Page-Dumroese, D., Elioff, J. D., & Stone, D. M. (2005). The North American long-term soil productivity experiment: findings from the first decade of research. *Forest Ecology and Management*, *220*(1), 31-50.

Prescott, C. E. (1995). Does nitrogen availability control rates of litter decomposition in forests?. *Plant and Soil*, *168*(1), 83-88.

Prescott, C. E. (2002). The influence of the forest canopy on nutrient cycling. *Treephysiology*, *22*(1516), 1193-1200.

Prescott, C. E. (2005). Do rates of litter decomposition tell us anything we really need to know?. *Forest Ecology and Management*, *220*(1), 66-74.

Prescott, C. E., Blevins, L. L., & Staley, C. L. (2000). Effects of clear-cutting on decomposition rates of litter and forest floor in forests of British Columbia. *Canadian Journal of Forest Research*, *30*(11), 1751-1757.

Purahong, W., Kapturska, D., Pecyna, M. J., Schulz, E., Schloter, M., Buscot, F., ... & Krüger, D. (2014). Influence of different forest system management practices on leaf litter decomposition rates, nutrient dynamics and the activity of ligninolytic enzymes: a case study from Central European forests. *PloS one*, *9*(4), e93700.

R Development Core Team (2015). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Ramirez, K. S., Craine, J. M., & Fierer, N. (2010). Nitrogen fertilization inhibits soil microbial respiration regardless of the form of nitrogen applied. *Soil Biology and Biochemistry*, *42*(12), 2336-2338.

Ranger, J., & Turpault, M. P. (1999). Input–output nutrient budgets as a diagnostic tool for sustainable forest management. *Forest Ecology and Management*, *122*(1), 139-154.

Risch, A. C., Jurgensen, M. F., Page-Dumroese, D. S., & Schütz, M. (2013). Initial turnover rates of two standard wood substrates following land-use change in subalpine ecosystems in the Swiss Alps. *Canadian journal of forest research*, *43*(10), 901-910.

Ross, D. J., & Sparling, G. P. (1993). Comparison of methods to estimate microbial C and N in litter and soil under Pinus radiata on a coastal sand. *Soil Biology and Biochemistry*, *25*(11), 1591 1599.

Rustad, L. E., & Cronan, C. S. (1988). Element loss and retention during litter decay in a red spruce stand in Maine. *Canadian Journal of Forest Research*, *18*(7), 947-953.

Salisbury, F. B., & Ross, C. W. (1992). Plant physiology. 4th. *Wadsworth Pub. Co*.

Sherman, L., Page-Dumroese, D., & Coleman, M. D. (2017) Idaho Forest Growth Response to Post Thinning Energy Biomass Removal and Complementary Soil Amendments. *GCB Bioenergy*. DOI:10.1111/gcbb.12486

Sinclair, T. R. (1992). Mineral nutrition and plant growth response to climate change. *Journal of Experimental Botany*, *43*(8), 1141-1146.

Sinsabaugh, R. L. (2005). Fungal enzymes at the community scale. *Mycology Series*, *23*, 349.

Sinsabaugh, R. L., Carreiro, M. M., & Repert, D. A. (2002). Allocation of extracellular enzymatic activity in relation to litter composition, N deposition, and mass loss. *Biogeochemistry*, *60*(1), 1-24.

Smolander, A., Kitunen, V., Tamminen, P., & Kukkola, M. (2010). Removal of logging residue in Norway spruce thinning stands: long-term changes in organic layer properties. *Soil Biology and Biochemistry*, *42*(8), 1222-1228.

Smolander, A., Levula, T., & Kitunen, V. (2008). Response of litter decomposition and soil C and N transformations in a Norway spruce thinning stand to removal of logging residue. *Forest Ecology and Management*, *256*(5), 1080-1086.

Smyth, C. E., Titus, B., Trofymow, J. A., Moore, T. R., Preston, C. M., Prescott, C. E., & CIDET Working Group. (2016). Patterns of carbon, nitrogen and phosphorus dynamics in decomposing wood blocks in Canadian forests. *Plant and Soil*, *409*(1-2), 459-477.

Son, Y., Jun, Y. C., Lee, Y. Y., Kim, R. H., & Yang, S. Y. (2004). Soil carbon dioxide evolution, litter decomposition, and nitrogen availability four years after thinning in a Japanese larch plantation. *Communications in Soil Science and Plant Analysis*, *35*(7-8), 1111-1122.

Stupak, I., Nordfjell, T., & Gundersen, P. (2008). Comparing biomass and nutrient removals of stems and fresh and predried whole trees in thinnings in two Norway spruce experiments. *Canadian Journal of Forest Research*, *38*(10), 2660-2673.

Sumner, M. E., & Miller, W. P. (1996). Cation exchange capacity and exchange coefficients. *Methods of Soil Analysis Part 3—Chemical Methods*, (methodsofsoilan3), 1201-1229.

Swift, M. J., Heal, O. W., & Anderson, J. M. (1979). *Decomposition in terrestrial ecosystems* (Vol. 5). Univ of California Press.

Thibodeau, L., Raymond, P., Camiré, C., & Munson, A. D. (2000). Impact of precommercial thinning in balsam fir stands on soil nitrogen dynamics, microbial biomass, decomposition, and foliar nutrition. *Canadian Journal of Forest Research*, *30*(2), 229-238.

Thomas, G. W. (1996). Soil pH and soil acidity. *Methods of Soil Analysis Part 3—Chemical Methods*, (methodsofsoilan3), 475-490.

Tukey Jr, H. B. (1970). The leaching of substances from plants. *Annual review of plant physiology*, *21*(1), 305-324.

U.S Climate Data Website (https://www.usclimatedata.com/climate/moscow/idaho/united states/usid0170; May in 2017

Ukonmaanaho, L., & Starr, M. (2001). The importance of leaching from litter collected in litterfall traps. *Environmental monitoring and assessment*, *66*(2), 129-146.

Wall, A., & Hytönen, J. (2011). The long-term effects of logging residue removal on forest floor nutrient capital, foliar chemistry and growth of a Norway spruce stand. *Biomass and Bioenergy*, *35*(8), 3328-3334.

Web Soil Survey Website (https://websoilsurvey.sc.egov.usda.gov/App/WebSoilSurvey.aspx)

Wickland, K. P., & Neff, J. C. (2008). Decomposition of soil organic matter from boreal black spruce forest: environmental and chemical controls. *Biogeochemistry*, *87*(1), 29-47.

Yatskov, M., Harmon, M. E., & Krankina, O. N. (2003). A chronosequence of wood decomposition in the boreal forests of Russia. *Canadian Journal of Forest Research*, *33*(7), 1211-1226.