EFFECTS OF THE WESTERN SPRUCE BUDWORM ON NITROGEN CYCLING IN CENTRAL WASHINGTON

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A Thesis

Presented to

The Graduate Faculty

Central Washington University

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In Partial Fulfillment

of the Requirements for the Degree

Master of Science

Biological Sciences

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by

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July 2020

CENTRAL WASHINGTON UNIVERSITY

Graduate Studies

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Dean of Graduate Studies

ABSTRACT

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January 2020

ACKNOWLEDGMENTS

I would like to thank Dr. Clay Arango for taking me on as a graduate student under his budget and for helping me being my project related to the NSF EAGER Grant and his lab assistants for helping me with my research. I would also like to thank my committee members Dr. Karl Lillquist, and Dr. Paul James for helping me with my proposal and for offering advice on ways that I can improve my research. I also could not have done this without the people who helped me collect and process samples during the course of this project so a special thank you to Julia Bramstedt, Michael Dallas, Natalie Levesque, Antonia Stewart, and Sarah Trikha for all of your time and effort in helping me complete my thesis project.

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**I**

**INTRODUCTION**

The process of defoliation is an important part of forest ecosystem health and function. Defoliators such as herbivorous insects act as a negative feedback loop for forests when they become too thick by killing trees and reducing the population to lower levels (CITATION). Defoliation is an important part of material cycling in forests by returning nutrients in organic matter to soils through consumption of the canopy and excretion as frass. Although defoliators are a natural part of forest material cycling, trends towards WHAT are being seen. This can be attributed to the rate at which insect outbreaks are occurring as well as outbreak severity—which has increased dramatically over the last century (Senf et al. 2016).

Since the 1930s, intense fire suppression throughout the West has led to thicker forests with increased canopy cover (Keane et al, 2002). High frequency, low intensity wildfires that formerly maintained an open forest stand occur less often, leading to increased incidence of forest insect pests. Historic fire regimes used to maintain insect pests via two avenues. First, frequent low intensity fires increased distance between trees making it challenging for insects to disperse. This decreased the rate at which (defoliators) damaged the forest. Secondly, fires killed pests directly. A multi-decadal history of fire suppression, coupled with summer drought stress due to climate change, has generated conditions that encourage sustained insect outbreaks and disease in the forest (Keane et al, 2002). As climate change progresses, theses insect outbreaks are expected to intensify (CITATION).

A major defoliator of the coniferous forests of Central Washington, as well as western North America in general (Senf et al. 2016), is the western spruce budworm (WSB) (*Choristoneura freemani)*—a native lepidopteran that ranges from Southern British Columbia to Arizona and New Mexico (Fellin and Dewey, 1982). These insects emerge during budburst around mid-May to feed on the new growth of short needle conifers, specifically Douglas fir (*Pseudotsuga menziesii*) and grand fir (*Abies grandis*). They are known to feed on a handful of other species as well (Fellin and Dewey, 1982), until late June or early July. They then pupate and emerge as adults, taking flight around mid to late July for oviposition. Larvae then emerge the following year in mid-May to repeat their life cycle. In a more natural fire regime that maintained an open forest structure, WSB outbreaks would occur about once every decade (CITATION find Flowers 2014? Paper..i think she cites 30 years). In recent years, thicker forests from fire suppression and increased drought stress from climate change have created conditions that encourage more frequent and widespread WSB outbreaks (Willis et al, 2008; Lovett et al, 2006). Furthermore, the cold weather that would have normally killed off pests in the past is occurring less often. This allows these pests to stay out longer, causing more damage to plants more often than they otherwise would (Griffin and Turner, 2012). This shift in forest structure and herbivore behavior has the potential to change forest ecosystem nutrient dynamics with implications for forest-stream connectivity. It has also been suggested that pest outbreaks can lead to increased fires due to the dead and dying trees they leave behind (Schlesinger et al, 2015), but new research has shown that this may not be the case, and in fact may have the opposite effect. These insects are defoliators as opposed to wood burrowers and therefore potentially have different effects on ecosystem dynamics.

This study examined some of the possible ecological effects of sustained WSB herbivory—including the rate of decomposition of mixed conifer needles to see whether or not that rate is increasing in areas highly impacted by WSB meaning that more nutrients would be added to the system. Under non WSB conditions, leaf litter would fall to the forest floor and be broken down by microbes over time, gradually releasing nutrients into the soil. Areas highly impacted by WSB have the potential to lead to increased nutrient availability in soils due to the large amounts of frass that these defoliators excrete that then falls to the forest floor. Once rainfall occurs, the leaching of frass frees up those nutrients, making them available for the forest system to use. From there, if nitrogen amounts are decreasing (net mineralization) then it can be inferred that nitrogen is taking the form of ammonium (NH4+) and is be taken up by plants and bacterial immobilization. If nitrogen levels are increasing (net nitrification) then it can be inferred that it is taking the form of nitrate (NO3-) that can then be exported to streams (Lewis and Likens, 2006). Defoliation by WSB also has the potential to increase microbial activity via the changing of an ecosystem’s chemistry through allowing more light and rainfall to reach the forest floor, in turn leading to a quicker break down in litter (Chapman et al, 2013). Pests, mixed with the current drought in the region are likely to alter the areas nutrient cycles on the forest floor as well as in soils (Schlesinger et al, 2015).

Any time an ecosystem experiences a major disturbance, there is an overall change in balance, leading to implications for both wildlife and for human concerns. It has been shown that in fish, removing even one key species in the food web can greatly alter an ecosystem's health (Taylor et al, 2006). If the WSB are altering the nitrogen and phosphorous cycles in soils, it is important to know how the process happens. Looking at total phosphorus, net nitrification/net mineralization, canopy damage, and decomposition rates will help to offer explanations as to the nature of the cycle change. This can show where there might be potential problems and may help lead to suggestions as to how we might be able to manage this pest outbreak. As outbreaks occur, there is a shift in biomass. Through knowing the degree of shift, we can then look at overall litter quality to provide more explanations of the effects of these pests (Genung et al, 2013).

Little research has been done on the western spruce budworm. Griffin and Turner (2012) did an extensive field study on *Dendroctonus pseudotsugae* (Douglas fir beetle) and *Dendroctonus ponderosae* (Mountain pine beetle) and found that herbivorous insect outbreaks cause noticeable changes to soil nitrogen cycling (2012). There is also no evidence that the new growth of conifers is occurring earlier or that it is lasting for longer in our region as it is in many flowering plants on the East Coast (Miller-Rushing and Primack, 2008).

To summarize, this study is important to local soil ecosystem dynamics. By looking at the rate of decomposition, it is possible to see if the rate of conifer leaf breakdown is influenced by herbivory and microbial activity Leading to the addition of supplemental nutrients to the soil. We can measure whether those soil nutrients are being taken up by plants or are accumulating with potential to enter the stream due to runoff to monitor changes in stream chemistry and the community food web. From that information we can look at whether those changes are significant and whether we should be concerned with the WSB outbreaks.

This project was part of an overarching research grant and is intended to help provide more data on WSB activity and their effect on PNW ecosystems. The main question of the grant being addressed was; are the WSB affecting aquatic food webs in local streams. To help answer that question, I looked at two smaller questions that led back to that main focus. The first question that was investigated was; are WSB changing that rate of decomposition of conifer litter on the forest floor in the grant’s study site. My project will be testing against the null hypothesis that there is no change to see whether WSB are affecting the rate of decomposition. A second question will also be looked at to support the data gathered on the rate of decomposition. The second question is; are the WSB changing net nitrification in the soils of the areas being investigated. This will also be tested against the null hypothesis of no change.

**II**

**METHODS**

Study Area

This study took place in the eastern Cascades in Washington State. Summers (May-September) are relatively dry, with seasonal drought and temperatures ranging from 15°C-25°C, and winters (October-April) are wet with temperatures ranging from -5°C-11°C. The average precipitation for the area is 720 mm (Northwest River Forecast Center, NOAA, https://www. ncdc.noaa.gov,accessed 7 September 2018) with most falling between month-month. Because of the distinct seasonal patterns in precipitation and temperature, eastern Cascades forests are characterized by a mix of Douglas fir (*Pseudotsuga menziesii)*, grand fir (*Abies grandis)*, ponderosa pine (*Pinus ponderosa*), western larch (*Larix occidentalis*) and at higher elevations, lodgepole pine (*Pinus contorta*).

I used a nested study design with repeated sampling through time to investigate how budworm herbivory influenced throughfall composition, litter decomposition, and soil nutrient dynamics. Within each budworm herbivory level (low versus high), I established study sites along 4 different streams (n=8). At each stream I established three replicate plots approximately 15 m from each other from upstream to downstream. At each replicate plot, I measured frassfall and litterfall, soil chemistry, soil organic matter and moisture content, and soil temperature 8 times between early September 2015 and early November 2016. Throughfall water chemistry was collected when accumulated precipitation allowed (> 100 mL). At each sample event, I collected decomposition bags to calculate one decomposition rate for each plot over the course of **the study.

Figure 1: Site locations with activity level shown in relation to major city.

The low budworm sites for this study were located in the Teanaway Community Forest in Washington State, approximately 40 miles northeast of Central Washington University on public land (Figure X). These study sites were located near the following creeks: Stand Up Creek (903 m a.s.l.) where sites were on a slope with light tree cover, Jungle Creek (824 m a.s.l.) where sites were often disturbed by free range cattle, Jack Creek (963 m a.s.l.) where sites were under moderately heavy tree cover, and Moonbeam Creek (973 m a.s.l. where sites were also under moderately heavy tree cover. The high budworm sites were located in the Swauk drainage in the Okanogan-Wenatchee National Forest in Washington State approximately 45 miles north of Central Washington University and east of the low budworm sites, also on public land (Figure X). These study sites were located near the following creeks: Cougar Creek (984 m a.s.l.) where sites were on a slope, Hurley Creek (978 m a.s.l.) where sites were located further away from the stream in comparison to other sites due to the stream being less accessible in a confined valley, Hovey Creek (1050 m a.s.l.) where sites were under moderately heavy tree cover, and Blue Creek (1055 m a.s.l.) where sites were also further away from the stream due to where the stream was in comparison to tree cover. Although each individual site varied based on microclimatic factors, sites were exposed to similar temperature and precipitation patterns based on similar elevation and being within X km of each other.

Throughfall

At each plot (n=24) a throughfall collector was installed under the canopy of a randomly selected tree close to each decomposition site. Each throughfall collector consisted of a funnel (20 mm diameter) that drained through tygon tubing into a 4-L acid-washed collection jug. To protect the tubing, I fed it through a PVC pipe, pounded into the ground with hole in the side so the tubing could enter the collection jug. The PVC pipe was stabilized by wiring it to a piece of rebar pounded into the ground. To prevent material from entering the collection jug, the opening was sealed with parafilm to keep the tubing in place, and polywool at the base of filter prevented litter from entering the jug from the funnel.

Upon rainfall, water entered the funnel and traveled through the tubing into the jug until I retrieved it within 48 h of the rain stopping. Upon collection, the total sample volume was recorded as the sample was transferred to an acid washed HDPE bottle and returned to the lab for filtration using a 1.0 μm glass fiber filter. Samples were frozen until later water chemistry analysis. In order to differentiate nutrients in bulk rainfall compared to throughfall that had percolated through the canopy, a total of four rainfall collectors were set up in areas with no canopy cover, two in the low budworm study sites and two in the high budworm study sites.

Throughfall and rainfall collectors were taken down November 8, 2015 just before lack of accessibility to sites due to snowpack and to prevent damage to the apparatus, and they were redeployed April 23, 2016 just after snowmelt to begin sampling again. All collectors were taken down on November 5, 2016.

Frass and Litter Measurements

To meaure organic matter matter movement from the canopy to the forest floor, I collect frass and litterfall at each site. One funnels (0.25 m2 diameter) made of tarp and garden hose connected to a one-liter Nalgene bottle was set up under one tree at each plot. These were sampled regularly during budworm feeding and less frequently after feeding. The samples were dried, sorted by frass versus litter, and weighed in the laboratory. Weights were then converted to a daily litter or frassfall rate by mg frass/m2d or mg litter/m2d. Frass collectors were taken down in November 5, 2015 due to lack of site accessibility and to prevent damage during winter snow accumulation, and they were reinstalled in April 23, 2016. Unfortunately, due to frequent rains in the spring months of 2016, samples decomposed before they could be collected and measured, so no data are available for the second half of the study.

Litter decomposition

At each replicate plot I deployed twenty 20x20cm mesh litter bags (García-Palacios et al., 2016) with a top sieve size of 2 mm (Genung et al, 2013) and a bottom sieve size of 0.5 mm (Schweitzer et al, 2005) to reduce content loss while still allowing small detritivores to enter the bags. I deployed a total of 480 bags across all sites. Ten bags at each site contained an air-dried, mixed conifer needle sample of Douglas fir, grand fir, and ponderosa pine in a 2:1:1 ratio to represent the most abundant species in the study area. The other ten bags at each replicate plot contained sugar maple (*Acer saccharum*) leaves which are non-native to the area but are commonly used in decomposition studies for comparison across biomes (Webster and Benfield 1969; Graça et al, 2005).

Within each litter bag, I placed ~3-5 grams of air-dried needles or leaves (Benfield, 1996) after recording the needle mass, and I added an aluminum tag with a unique ID. Bags were assembled by stapling the two sieve sizes together and by reinforcing them with super glue at the corners. The bags stayed intact throughout the 14-month deployment. Mesh bags with needles or leaves were subsequently placed into red peanut bags (mesh size ~ 3.1 mm) to further protect them during deployment and to simplify sample collection, and each individual bag was placed into a Ziploc for transport to the field.

On September 8, 2015, the mesh bags were deployed and strung together on an approximately 6 m nylon parachute cord held in place by 0.6 m pieces of rebar driven into the ground on either side. The rebar anchors and parachute cord prevented bags from being moved by the wind, displaced by hillslope runoff, or moved by animals. A coin flip determined which bags (conifers or deciduous maple) were placed upstream and downstream at each site. To quantify handling loss, I deployed twenty bags, ten deciduous and ten coniferous, and retrieved them immediately to determine mass loss per bag during deployment and extraction. Mass loss per bag was averaged and applied to all bags extracted throughout the study. This was done separately for conifer and deciduous leaves.

Bags were collected 7 times beginning October 11, 2015 and ending November 6, 2016 in approximately 1-2-month intervals with a 5-month break during winter snowpack (December 2015 to April 2016) when sites were inaccessible. During each retrieval from the field, one conifer bag and one maple bag was randomly collected from each plot for a total of 48 bags per sampling time and returned to the lab in a Ziploc bag to prevent additional leaf mass loss. On the final collection day, all remaining bags were collected from the sites (n=4 per leaf type at each plot). Decomposition bags were air dried in the lab to constant mass (Schweitzer, 2005) in paper bags (Genung et al. 2013) hung on a clothesline. After air drying, each bag was sorted to remove any noticeable debris that had become incorporated in the sample (Chapman et al. 2013). Because of natural loss of conifer needles from the canopy, it was difficult to determine what was originally in the bag and what had fallen into it, so the mass of conifer needles accumulated in the maple decomposition bags was sorted and used as a correction factor for the mass of conifer needles that entered the conifer bags. Decomposition rate was calculated as:

*Soil Analyses*

Upon each collection of decomposition bags, I also used a thermocouple to measure temperature at three soil depths: 2 cm, 10 cm, 20 cm. These corresponded approximately to the O horizon, the top of the A horizon, and within the A horizon respectively. A soil core of ~10 cm depth was also collected from each replicate plot at each stream site each time I collected litter bags. Soil cores were stored on ice for return to the laboratory whereupon each core was homogenized in a Ziploc bag. Soils were immediately analyzed for moisture content and percent organic matter, and soils were frozen for later analysis of ammonia, nitrate, and inorganic P using methods detailed below.

*Moisture Content and Percent Organic Matter:*

Soil was sieved at 2 mm and a subsample was placed into an ashed aluminum pan and weighed immediately for field mass. Pans were then placed in a drying oven at 60ºC until constant mass, cooled to room temperature, and weighed to obtain dry mass (DM). The difference between field mass and dry mass was used to calculate percent moisture.

Then dried soil samples were placed in a muffle furnace at 500ºC for 48 h to combust all organic matter. After ashing, samples were cooled to room temperature, rehydrated with Milli-Q water to rehydrate clays and colloids containing water molecules, and then placed again into a drying oven until constant mass. Pans were cooled to room temperature and reweighed to obtain ash-free dry mass, with the difference between dry mass and ash-free dry mass used to calculate percent organic matter.

*Net changes in the soil inorganic N pool*

To measure changes in the soil inorganic N pool, at each site I also deployed a resin bag made of bleached nylons (to prevent color leaching that may affect results) filled with 20 g of ion exchange resin (IONAC NM-60 mixed bed exchange resin, strong acid/strong base; sulfonated alkyl quaternary ammonium polystyrene; J.T. Baker #JT4631-1) 10 cm deep after initial soil samples were taken. Bags were deployed in September 2015 and removed in November 2015, and fresh bags were deployed in November 2015 and removed in April 2016. Net changes in the inorganic N pool were calculated as (Griffin and Turner, 2012) Net nitrification was indicated by … and net mineralization was indicated by … etc.

*Nitrogen Analyses*

A 2M KCl extraction method was used to extract inorganic nitrogen from each soil sample. Five grams of air-dried soil were added to 37.5 mLs of 2M KCl and shaken at 100 rpm for 2 hours on a shaker table and then centrifuged at 10,000 g. The sample was then filtered with a syringe through a 1.0 µm glass fiber filter and stored in the freezer until analysis. Samples were analyzed for NO3-+NO2- (hereafter referred to as NO3-) using the cadmium reduction method (CITATION) and NH4+ using the phenate method (CITATION) on a Seal AQ1 Discrete Analyzer (Seal AQ1, Seal Analytical; Mequon, Wisconsin, USA).

*Phosphorous Analysis*

The Bray P1 method was used to extract phosphorus from each soil sample (Hamilton, 1997; Patton and Kryskalla, 2003). One gram of air dried soil was added to 10 mLs of the Bray P1 extractant solution (30 mLs 1 N NH4F to 50 mL 0.5 HCl) and shaken on a shaking table at 100 rpm for 15 minutes then centrifuged at 10,000 g. The sample was then filtered with a syringe through a 1.0 µm glass fiber filter and stored in the freezer until analysis. Samples were analyzed for inorganic phosphorous using the ascorbic acid method (Murphy and Riley, 1962) on a Seal AQ1 Discrete Analyzer (Seal AQ1, Seal Analytical; Mequon, Wisconsin, USA).

**Statistical Analysis**

All data was analyzed in RStudio version 3.6.2. Throughfall was analyzed using XXX (package). Frass and litterfall was compared using a generalized least squares (GLS) model (package). Decomposition was analyzed using a linear model (LM) with leaf type and location as factors as well as looking at the interaction between high impact and low impacted sites. A two-sample t-test to compare the two treatments; coniferous litter vs deciduous. I used GLS models and linear mixed effects (LME) models (Senf et al. 2016) to see how budworm herbivory level (low versus high) influenced percent soil moisture, percent organic matter, temperature, NO3-, NH4+, SRP, N:P ratio, total inorganic N, and net nitrification/mineralization through time and by site. Data was normalized when residuals did not meet the assumptions of the test. When selecting models, I compared ones with both an interaction between impact factors and sample event and ones with a nested design. I plotted the residuals using a Q-Q Normal Plot and normalized when applicable. Additional models were constructed with weighted variances to help reduce residual patterns. Models were compared using the anova command in R and the model with the lowest AIC score was selected.

For GLS and LME models that yielded significant results, estimated marginal means (EMMs) analysis (package) was used as a post hoc test on data to determine which sample events differed significantly. All statistical tests had were run with  = 0.05.

**III**

**RESULTS**

Throughfall Chemistry

Figure \_ Estimated marginal means (EMM) of (A) throughfall ammonium concentrations and (B) throughfall nitrate (NO3-) concentrations in low and high budworm stands by sample date. Significant interactions are noted with an asterisk.

Concentrations of throughfall ammonium differed in low and high budworm stands (LME, p=0.015) and by sample event (LME, p<0.001) throughout the course of the two-year study (Figure XA). There was a significant interaction (LME, p<0.001) whereby on four dates (11 Sep 15, 21 Jun 16, 13 Jul 16, and 21 Jul 16) throughfall NH4+ was higher in high budworm stands, but on 4 Jun 16, it was higher in low budworm stands. Generally speaking, in times where budworms were inactive (11 Oct 15, 29 Oct 15, 8 Nov 15, 9 Sep 16), there was no difference in throughfall NH4+concentration. Throughfall nitrate differed by sample event (LME, p<0.001) but not budworm activity level throughout the course of the two-year study (Figure XB). There was a significant interaction (LME, p<0.001) whereby the low budworm stands had higher concentration NO3- on 8 May 16, but the high budworm stands had higher concentration on 13 Jul 16, 21 Jul 16, which were generally during or after peak budworm herbivory.



Figure \_ Estimated marginal means (EMM) of (A) throughfall soluble reactive phosphorous (SRP) concentration and (B) dissolved organic carbon (DOC) concentration in low and high budworm stands by sample date. Significant differences among sample events are noted with letters.

Throughfall SRP concentration differed by sample event (LME, p<0.001) throughout the course of the two-year study with highest concentration on two dates (11 Sep 15 and 21 Jun 16). However, SRP concentration did not differ between high and low budworm sites (LME, p=0.43). Throughfall DOC concentration also differed by sample event (LME, p<0.001) with 8 Nov 15 having the highest concentration. Like SRP, DOC did not differ between high and low budworm sites (LME, p=0.26)

Decomposition Rates



Figure \_ Decomposition rates (-*k*) of deciduous and coniferous leaf litter in high and low budworm sites.

Decomposition of coniferous and deciduous leaf litter did not vary by leaf type (p=0.68) however decomposition was faster in low budworm sites for both leaf litter types (p=0.0024, LME). ![A close up of a map

Description automatically generated]()

Figure \_ Regression analysis of throughfall DIN and deciduous decomposition rate.

Rainfall did not affect the rate of decomposition for deciduous or coniferous leaf litter. DIN was positively associated with the deciduous decomposition rate (R2=.15, p=0.033) but not coniferous (p=0.13).

Soil Chemistry:



Figure \_ Estimated marginal means (EMM) of soil (A) ammonium (NH4+), (B) nitrate (NO3-), and (C) soluble reactive phosphorous (SRP) concentrations in low and high budworm stands by sample date. Significant sample events are noted with letters and significant interactions are noted with an asterisk.

Soil ammonium concentrations differed by sample date (LME, p<0.001) with higher concentrations on 8 Nov 15 and 8 May 16 compared to 13 Jun 16. These were times when budworms were generally not active, however there was no difference between high and low budworm site (p=0.33, LME). Although soil nitrate did not differ between high and low budworm sites (p=0.76, LME), it did differ by sample event (p<0.0001, LME) with a significant interaction between sample event and budworm (p=0.003, LME). In the interaction, high budworm sites had higher soil NO3- concentration than low budworm sites on 6 Nov 16 whereas as XXXX on 4 Aug 16. Usually soil NH4+ was 60times higher than soil NO3-. Soil SRP was significantly higher in high impact sites for every sample event (p=0.047, LME) but did not differ by sample event (p=0.91). Net Nitrification did not differ by budworm impact (p=0.53, LME).



Figure \_ Estimated marginal means (EMM) of (A) soil moisture and (B) soil organic matter in low and high budworm stands by sample date. Significantly different sample events are noted with letters.

Soil moisture varied among sample events (p<0.001, LME) and was greater during the sample events 11 Oct 15, 8 Nov 15, 4 Aug 16, and 19 Sep 16, but there was no difference between high and low budworm sites (p=0.86, LME) (Figure \_ A) Soil organic matter did not differ between high and low budworm sites (p=0.49, LME) or among sample dates (p=0.70, LME) (Figure \_ B). 

Figure \_ A regression analysis comparing the temperature at 2cms deep and air temperature in Celsius with a P<0.0001 and R2 of 0.78.

Soil temperature followed the expected pattern of increasing during spring and summer months and decreasing during winter and fall months (data not shown), and soil temperature was strongly correlated with air temperature (R2 = 0.78, p<0.0001, linear regression). Temperature increased and decreased more rapidly at shallow depths compared to deeper measurements, also following the expected pattern that as soil depth increases, the change in temperature changes at a slower rate (data not shown).

**IV**

**DISCUSSION**

The intent of this study was to examine how WSB herbivory affected throughfall chemistry, decomposition, and soil chemistry in the Central Cascades. Although budworm herbivory seemed to positively influence X, Y, Z , seasonality seemed to be a stronger factor. Moreover, budworm herbivory and seasonality interacted in X Y.

**Throughfall Nitrogen**

suggesting an N limited system.

This could be due to fluctuation of nitrification and mineralization throughout the duration of this study. These results are similar to soil nitrogen, as we also see a lot of variability in NH4 as well. In addition to the similarities between throughfall and soil NH4, NO3 concentrations were also comparable. I saw the same peaks in the nitrate throughfall as I did in the soil, and confirmed this using the ion exchange resin beads, again seeing two large pulses of nitrate entering the system. Although we see spikes in the data shape, the concentrations are still relatively low in comparison to SRP.

**Throughfall SRP**

which was not what I hypothesized. I would have expected that phosphorus levels would be higher in heavily impacted areas due to an increase in frass input.

**Throughfall DOC**

which in turn will eventually be exported to streams, which has the potential to increase stream metabolism.

**Leaf Litter Decomposition**

I had hypothesized that in high herbivory areas, decomposition would occur at a faster rate, as a decrease in forest canopy would allow more water to reach the forest floor, simulating fungal and bacterial growth (Source). It is possible that with less cover, greater amounts of light could reach the forest floor during the warmer months, drying out the forest floor, and slowing the rate of decay (Source).

**Soil Nitrogen**

This could be due to fluctuation of nitrification and mineralization throughout the duration of this study, and it suggests N limitation. More available NH4 would suggest there is more potential for nitrification.

Soil tends to have less nitrogen during the winter (<https://link.springer.com/article/10.1007/BF02183092>) but the data did not follow that pattern as seen in the late fall and early spring sampling events. I do not suspect that plants were taking up extra nitrogen during that time as production rates tend to be lower in the cooler months (<https://link.springer.com/article/10.1007/s00442-005-0044-1>). Since throughfall samples showed low NH4 input during these sampling events, I can only attribute the increase in soil NH4 to nitrogen-fixing microorganisms due to N limitation.

**Soil SRP**

There is potential for SRP to be washed into the near by streams during rain events. Although SRP is important for productivity in stream ecosystems, an excess amount of SRP can lead to over productive systems, causing algae blooms, which will eventually lead to mass die off events and oxygen depletion. This also suggests that due to accumulating levels of soil SRP, it is not a limiting resource in the soil systems that I studied.

because this was not seen in the SRP samples from throughfall, it suggests that the WSB in highly impacted areas are adding more phosphorous than can be taken up by soil microbes

**Future Studies**

This study thoroughly investigated soil and throughfall nutrients, and their implications in both forest soil health and stream ecosystem health. Future studies could expand on the nutrients measured to include organic N and P, to help support the findings in this study that only looked at inorganic N and P.

In additional to looking at nutrients, a study to look at the invertebrate, fungal and microbial communities in the forest soil to help support missing aspects of this study, such as what happens to the inorganic nutrients. It would give us more insight as to whether they are being incorporated into those communities or being exported into stream systems, having different implications for the effects of WSB on forest ecosystems.

**REFERENCES**

Taylor, B. W. Flecker, A. S. Hall Jr., R. O. 2006. Loss of a Harvested Fish Species Disrupts Carbon Flow in a Diverse Tropical River. Science, 313, 833-836.

Miller-Rushing, A. J. Primack, R. B. 2008. Global Warming and Flowering Times in Thoreau’s Concord: Community Perspective. Ecology, 89(2), 332-341.

Zhao, T. Krokene, P. Hu, J. Christiansen, E. Bjorklund, N. Langstrom, B. Solheim, H. Borg-Karlson, A.K. (2011). Induced Terpene Accumulation in Norway Spruce Inhibits Bark Beetle Colonization in a Dose-Dependent Manner. Plos One, 6(10), 1-8.

Benfield, E. F. 1996. Leaf Breakdown in Stream Ecosystems. Methods in Stream Ecology, 579-589.

Willis, C. G. Ruhfel, B. Primack, R. B. Miller-Rushing, A. J. Davis, C. C. 2008. Phylogenetic patterns of species loss in Thoreau’s woods are driven by climate change. PNAS, 105(44), 17029-17033.

Smith, R. M. Kaushal, S. S. 2015. Carbon cycle of an urban watershed: exports, sources, and metabolism. Biogeochemistry, DOI 10.1007/s10533-015-0151-y.

Genung, M. A. Bailey, J. K. Schweitzer, J. A. 2013. The Afterlife of Interspecific Indirect Genetic Effects: Genotype Interactions Alter Litter Quality with Consequences for Decomposition and Nutrient Dynamics. Plos, DOI 10.1371/journal.pone.0053718.

Lewis, G.P. Likens, G. E. 2006. Changes in stream chemistry associated with insect defoliation in a Pennsylvania hemlock-hardwoods forest. Forest Ecology and Management, 238, 199-211.

Lovett, G. M. Canham, C. D. Arthur, M. A, Weathers, K. C. Fitzhuge, R. D. 2006. Forest Ecosystem Response to Exotic Pests and Pathogens in Eastern North America. BioScience, 56(5), 395-405.Bott, T. L. 1996. Primary Productivity and Community Respiration. Methods in Stream Ecology, 533-556.

Liu, L. Wang, X. Lajeunesse, M. J. Miao, G. Piao, S. Wan, S. Wu, Y. Wang, Z. Yand, S. Li, P. Deng, M. 2015. A cross-biome synthesis of soil respiration and its determinants under simulated precipitation changes. DOI 10.1111/gcb.13156.

Webster, J. R. Ehrman, T. P. 1996. Solute Dynamics. Methods in Stream Ecology, 145-160.

Hamilton, S. 1997. Analysis of nitrate in Michigan waters.

Hamiltion, S. 1997. Combined persulfate digestion for analysis of total nitrogen and phosphorous in Michigan waters.

Graça, Manuel A.S., Bärlocher, Felix, Gessner, Mark O. 2005. Methods to Study Litter Decomposition: A Practical Guide.

Griffin, J. M. Turner, M. G. 2012. Changes to the N cycle following bark beetle outbreaks in two contrasting conifer forest types. Oecologia, 170, 551-565.

Goodale, C. L. Fredriksen, G. Weiss, M. S. McCalley, C. K. Sparks, J. P. Thomas, S. A. 2015. Soil Process drive seasonal variation in retention of 15N tracers in a deciduous forest catchment. Ecology, 96(10), 2653-2668.

Schlesinger, W. H. Dietze, M. C. Jackson, R. B. Phillips, R. P. Rhoades, C. C. Rustad, L. E. Vose, J. M. 2015. Forest Biogeochemistry in Response to Drought. DOI 10.1111/gcb.13105.

Chapman, S. K. Newman, G. S. Hart, S. C. Schweitzer, J. A. Koch, G. W. 2013. Leaf Litter Mixtures Alter Microbial Community Development: Mechanisms for Non-Additive Effects in Litter Decomposition. Plos, DOI 10.1371/journal.pone.0062671.

Schweitzer, J. A. Bailey, J. K. Hart, S. C. Whitham, T. G. 2005. Nonadditive Effects of Mixing Cottonwood Genotypes on Litter Decomposition and Nutrient Dynamics. Ecology, 86(10), 2834-2840.

Johnson, L. T. Tank, J. L. Dodds, W. K. 2009. The influence of land use on stream biofilm nutrient limitation across eight North American ecoregions. Can. J. Fish. Aquat. Sci., 66, 1081-1094.

Schweitzer, J. A. Bailey, J. K. Hart, S. C. Wimp, G. M. Chapman, S. K. Whitham, T. G. 2005. The interaction of plant genotype and herbivory decelerate leaf litter decomposition and alter nutrient dynamics. Oikos, 110(1), 133-145.

Kryskalla, J. R. Patton, C. J. 2003, Methods of Analysis by the U.S. Geological Survey National Water Quality Laboratory---Evaluation of Alkaline Persulfate Digestion as an Alternative to Kjeldahl Digestion for Determination of Total and Dissolved Nitrogen and Phosphorus in Water. USGS, 1-33.

Senf, C. Campbell, E. M. Pflugmacher, D. Wulder, M. A. Hostert, P. 2016.

A multi-scale analysis of western spruce budworm outbreak dynamics. Landscape Ecol, 32, 501-514.