EFFECTS OF WESTERN SPRUCE BUDWORM HERBIVORY ON FOREST SOILS AND LITTER DECOMPOSITION IN CENTRAL WASHINGTON

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by

Izak Roland Neziri

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Graduate Studies

We hereby approve the thesis of

Izak Roland Neziri

Candidate for the degree of Master of Science

APPROVED FOR THE GRADUATE FACULTY

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Dr. Clay Arango, Committee Chair

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Dr. Karl Lillquist

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Dr. Paul James

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Dean of Graduate Studies

ABSTRACT

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

by

Izak Roland Neziri

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TABLE OF CONTENTS

Chapter Page

I INTRODUCTION 1

First-Order Subheading 3

First-Order Subheading 4

First-Order Subheading 6

II METHODS 30

First-Order Subheading 30

First-Order Subheading 46

First-Order Subheading 55

III RESULTS 60

First-Order Subheading 60

First-Order Subheading 67

First-Order Subheading 75

IV DISCUSSION 78

First-Order Subheading 78

First-Order Subheading 81

First-Order Subheading 85

REFERENCES 91

LIST OF FIGURES

Figure Page

1 Title of figure—figures are captioned using sentence style

capitalization 1

2 Title of figure 4

3 Title of figure 60

4 Title of figure 63

5 Title of figure 65

6 Title of figure 66

7 Title of figure 72

8 Title of figure 73

9 Title of figure 75

11 Title of figure 75

LIST OF EQUATIONS

EQUATION Page

1 Title of figure—figures are captioned using sentence style

capitalization 1

2 Title of figure 4

3 Title of figure 60

4 Title of figure 63

10 Title of figure—some captions may wrap to more than one line, so use

this one as a model—multiple-line captions should be formatted in

inverted pyramid style (descending lines get shorter) 75

11 Title of figure 75

**I**

**INTRODUCTION**

Forests make up approximately one third of the Earth’s surface (Likens et al 1970). In these ecosystems, a large portion of the nutrients are generally retained within that system (CITATION). The general flow of nutrients within a forest include precipitation, which flows through the canopy as throughfall while leach nutrients from vegetation to soil, with those nutrients being taken up by plants and converted into biomass. When these systems are subject to disturbance, the cycling of forest nutrients is subject to changes and losses (CITATION). Common disturbances include fire, clear cut logging, insect outbreaks, and climate change. For example, in 1970, a famous experiment in the Hubbard Brook Experimental Forest found that experimental clear cutting dramatically increased annual runoff and loss of nitrate and other ions to downstream ecosystems (Likens et al, 1970). In these clear cut watersheds, Likens and others also found that the disturbed watershed had higher stream temperatures, as there was less shade, and saw an increase in turbidity, which lead to an increase in dissolved oxygen concentrations. If less canopy cover due to clear cutting can lead to increased temperature, increased decomposition rates, and increased nutrients, then it is possible that defoliating insects can also cause similar changes to forest ecosystem watersheds by removal of canopy cover. While all of these studies point to disturbance as a mechanism for nutrient loss for forests to downstream aquatic ecosystems, the effect of insect herbivores has not been studied in the western United States, which is expected to see an uptick in forest insect activity as global temperatures rise.

The process of defoliation is an important part of forest ecosystem health and function. When forests become too thick, defoliators such as herbivorous insects act as a negative feedback loop by killing trees and reducing the population to lower levels (Poirier 2017; Murdock et al. 2012). Defoliation also drives material cycling in forests through consumption of the canopy and excretion as frass which returns nutrients as organic matter to soils. Although defoliators are a natural part of forest material cycling, the activity of these insects is increasing due to fire suppression and climate change (Abatzolou and Williams 2016). As a result, the rate and severity of insect outbreaks has increased dramatically over the last century (Senf et al. 2016).

For centuries, frequent, low intensity, naturally caused fires and fires ignited for landscape management by indigenous people have shaped the structure of coniferous forests across the United States (Klenner et al 2008), creating for example ponderosa pine forests with many grasses and shrubs growing between widely spaced trees. 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, decreasing the rate at which defoliators damaged the forest. Secondly, fires killed pests directly. Since the 1930s, intense fire suppression throughout the American West has led to thicker forests with increased canopy cover (Keane et al, 2002), and insect pests have increased with the thickening tree stands. Moreover, decreases in winter severity due to climate change has reduced cold winters that can kill insect pests (Murdock et al. 2012). Therefore, a multi-decadal history of fire suppression, coupled with summer drought stress and warmer winters due to climate change, has generated conditions that encourage sustained insect outbreaks and disease in the forest (Keane et al, 2002), and these insect outbreaks are expected to intensify as climate change progresses (Flower et al 2014).

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*), which have benefitted from fire suppression, but also spruce needles (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 (Flowers 2014). 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). This shift in forest structure and herbivore behavior has the potential to change internal forest ecosystem nutrient dynamics with implications for nutrient loss to nearby stream ecosystems that would alter forest-stream ecological connectivity.

Sustained WSB herbivory could alter internal forest nutrient cycling and/or forest-stream ecological connectivity through a variety of mechanisms. Under conditions without active defoliation, leaf litter would fall to the forest floor and be broken down by microbes over time, gradually releasing nutrients into the soil. However actively defoliating WSB, in combination with the increasing drought stress via climate change, are likely to alter the nutrient cycle in forest soils (Schlesinger et al, 2015). For example, the large amount of frass that these defoliators excrete falls to the forest floor and has the potential to increase nutrient availability in soils after rainfall leaches nutrients to soils, making them available for the forest nutrient cycle (CITATION FOR THE LEACHING MECHANISM?). In particular, nitrogen availability may increase (CITATION), and via various transformations into ammonium or nitrate, the increased nitrogen can meet a variety of fates. For example, ammonium (NH4+) can be taken up by plants or immobilized by bacteria or fungi, remaining in the ecosystem as organic N. Organic N in frass can be mineralized as ammonium via decomposition, where it can be converted to nitrate via nitrification whereby it is then subject to leaching losses to downstream ecosystems (Lewis and Likens, 2006). Furthermore, defoliation by WSB has the potential to allow more light and rainfall to reach the forest floor, increasing microbial activity via and leading to a quicker break down in litter via decomposition (Chapman et al, 2013).

Any time an ecosystem experiences a major disturbance, there is an overall change in ecosystem dynamics, leading to implications for both wildlife and for human concerns (CITATION?). However, little research has been done on the western spruce budworm so it is uncertain how the predicted increases in their outbreaks might alter ecosystem dynamics in the Pacific Northwest and other western coniferous forests. (THIS LINKS TO THE LAST PARAGRAPH IN THE INTRO)

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).

To better understand how WSB defoliation affects internal forest nutrient pathways, I studied a WSB outbreak in the east slope of the Cascades in Central Washington with an aim of understanding how WSB herbivory affected throughfall nutrient composition, leaf litter decomposition rate, soil chemistry, and soil nitrogen transformations. I hypothesized that WSB activity would accelerate the movement of nutrients from the canopy to soils which would: 1) increase throughfall nutrient concentration, 2) increase litter decomposition rate, 3) increase soil nutrient concentrations, and 4) increase net nitrification in soils. I also hypothesized that canopy opening via defoliation would alter soil temperature and moisture patterns, with implication for decomposition.

**II**

**METHODS**

Study Area

This study took place in the eastern Cascades in central Washington state. In the rain shadow of the Cascades, summers (May-September) are relatively dry, with seasonal drought and temperatures ranging from 15°C-25°C whereas 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 as snow between November-February. Because of the distinct seasonal patterns in precipitation and temperature, eastern Cascades forests are characterized by drought tolerant trees such as 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 concentrations. I established 4 study sites each within low and high budworm herbivory level stands (n=8 study sites; Figure 1), and at each study site I established three replicate plots approximately 15 m from each other from upstream to downstream (n=24 total sample plots). The low budworm sites were located in the Teanaway Community Forest in Washington state, approximately 40 miles northeast of Central Washington University (Figure 1) 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 (Figure 1). 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 difficulty of access. 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 roughly 20 km of each other.

Figure 1: Site locations with budworm activity level derived from United States Forest Service aerial detection surveys (https://www.fs.fed.us/foresthealth/applied-sciences/mapping-reporting/gis-spatial-analysis/detection-surveys.shtml) flown in 2015.

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 late September 2016, roughly every 6 weeks with a break from sampling when snow pack precluded site access. At each sample event, I collected decomposition bags to calculate one decomposition rate for each plot over the course of the study. Throughfall water chemistry was collected on an event basis when accumulated precipitation allowed (> 100 mL). I measured net nitrification at each site twice; the first measurement aggregated net soil nitrogen dynamics between summer 2015 and fall 2015, and the second aggregated net soil nitrogen dynamics between fall 2015 and spring 2016. Thus, my study design included measurements taken before, during, and after, one complete WSB life cycle.

Throughfall

A throughfall collector was installed under the canopy of a randomly selected tree near each sample plot (n=24). Each throughfall collector consisted of a funnel (20 cm diameter) that drained through tygon tubing into a 4-L acid-washed collection jug. The tubing was protected by feeding it through a PVC pipe pounded into the ground with a hole in the side so the tubing could leave the PVC and 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, polywool was placed at the base of the funnel, and the opening of the jug was sealed with parafilm which also kept the tubing in place.

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 (described below). 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 a low budworm study site and two in a high budworm study site.

After deploying collectors on 25 Jun 15 and collecting 4 samples in 2015, throughfall and rainfall collectors were taken down November 8 2015 to prevent damage due to snowpack, and they were redeployed April 23, 2016 just after snowmelt to begin sampling again. All collectors were taken down on September 19, 2016 after collecting 6 samples in 2016.

Frass and Litter Measurements

To measure organic matter movement from the canopy to the forest floor, I collected frass and litterfall at each site. One funnel (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 sample plot (n=24). These were sampled regularly during budworm feeding and less frequently after feeding. The samples were dried, sorted by frass versus litter, weighed in the laboratory, and converted to a daily litter or frassfall rate (mg frass/m2d or mg litter/m2d). Frass collectors were taken down in November 5, 2015 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 plots. Ten bags at each 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 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 g of air-dried needles or leaves (Benfield, 1996), recorded the needle mass, and 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 13-month deployment. Mesh bags with needles or leaves were subsequently placed into plastic mesh 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 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 determine mass loss per bag during deployment and extraction, I deployed twenty bags, ten deciduous and ten coniferous, and retrieved them immediately. Mass loss per bag was averaged and applied to all bags extracted throughout the study, with separate calculations 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 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). Upon retrieval decomposition bags were hung on a clothesline in paper bags (Genung et al. 2013) and air dried in the lab to constant mass (Schweitzer, 2005). 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:

**Equation 1:** The rate of decomposition where k is the slope.

Soil Collection and Processing

Upon each collection of decomposition bags, I also used a thermocouple to measure temperature at three soil depths: 2 cm, 10 cm, and 20 cm, corresponding 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 when I collected litter bags. Soil cores were stored on ice for return to the laboratory whereupon the core from each plot was homogenized in a Ziploc bag. Soils were immediately analyzed for moisture content and percent organic matter and then frozen for later analysis of ammonium, nitrate, and inorganic P using methods detailed below.

*Moisture Content and Percent Organic Matter:*

Homogenized soil was sieved at 2 mm and a subsample was placed into an ashed aluminum pan and weighed immediately for initial 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 initial field mass and dry mass was used to calculate percent moisture.

**Equation 2:** The determination of moisture content in soil samples.

Then dried soil samples were ashed 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 ashed mass, and the difference between dry mass and ashed mass (ash free dry mass) was used to calculate percent organic matter.

**Equation 3:** The determination of how much organic matter each soil sample contained.

*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:

**Equation 4:** Where N is the combination of ammonium and nitrate. Net increase in NH4+ indicates net mineralization, net increase in NO3- indicates net nitrification, and net decrease in either ion indicates immobilization of that ion (Griffin and Turner, 2012).

A 2M KCl extraction method (Keeney and Nelson 1987) was used to extract inorganic nitrogen from each soil sample. Five grams of air-dried soil were added to 37.5 mL 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.

The Bray P1 method was used to extract phosphorus from each soil sample (Bray and Kurtz 1945). One gram of air dried soil was added to 10 mL of the Bray P1 extractant solution (30 mLs 1 N NH4F to 50 mL 0.5 HCl) and shaken at 100 rpm for 15 minutes on a shaking 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.

Chemical Analyses for Throughfall and Soil

Samples were analyzed for NO3-+NO2- (hereafter referred to as NO3-) using the cadmium reduction method (U.S. Environmental Protection Agency (EPA) 1993) and NH4+ using the phenate method (Solórzano, 1969) on a Seal AQ1 Discrete Analyzer (Seal AQ1, Seal Analytical; Mequon, Wisconsin, USA). 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). DOC samples were measured using infrared methods (American Public Health Association (APHA) 1995) after acidifying the samples to a pH of less than 2 to remove inorganic carbon on a total organic carbon analyzer (TOC-L Total Organic Carbon Analyzer, Shimadzu, Kyoto, Japan)

Statistical Analysis

All data was analyzed in R version 3.6.1 (R Core Team 2019). Throughfall composition was analyzed using (R Core Team 2019) to see how patterns varied by budworm herbivory level and time. Frass and litterfall was compared by budworm level and time using a generalized least squares (GLS) model, and leaf decomposition was analyzed using a linear model (LM) with leaf type and location as interacting factors. I used 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, soil nutrients, and net nitrification/mineralization through time. LM models were also used to regress decomposition rates of both deciduous and conifers leaf litter on total N in throughfall and total water falling at each plot. To optimize models, I compared alternate model structures with an interaction between impact factors and sample event and models with a nested design (Zuur et al. 2009). Additional models were constructed with weighted variances to improve model residuals. Models were compared using the anova command in R and the model with the lowest AIC score was selected. To evaluate the assumptions of the model, I plotted the residuals using a Q-Q Normal Plot and normalized when applicable. For LME models that yielded significant results, estimated marginal means (EMMs) analysis was used as a post hoc test to determine which sample events differed significantly. All statistical tests were evaluated against  = 0.05.

**III**

**RESULTS**

Throughfall Chemistry



**Figure 2:** 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 study (Figure 2A). 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 between high and low budworm sites. Throughfall nitrate differed by sample event (LME, p<0.001) but not budworm activity level throughout the course of the study (Figure 2B). 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 and 21 Jul 16, which were generally during or after peak budworm herbivory. There was a general trend of increasing concentration of throughfall ammonium and nitrate during the time of WSB budworm activity between 8 May 16 and 13 Jul 16 (Figure 2).



**Figure 3:** 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 study with highest concentrations on two dates (8 Nov 15 and 21 Jul 16). However, SRP concentration did not differ between high and low budworm sites (LME, p=0.43) (Figure 3 A). 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) (Figure 3B). The biggest pulses of SRP and DOC from the canopy appeared on the same dates (8 Nov 15 and 21 Jul 16), which also coincided with the two rainfall events with the most water collected.

Frass and Litter Deposition

A picture containing map, boat, photo, sitting

Description automatically generated

**Figure 4:** The deposition of frass and litter in high impact and low impact sites with vertical dashed line indicating end approximate end of budworm feeding.

In high impact sites, frass content was greater than in low impact sites during peak herbivory times (GLS p=0.04). Once the budworm feeding season ended, frass input for high impact and low impact sites was virtually the same. Low budworm site collectors contained more leaf litter than high impact sights during the cooler months where the highest concentration of litter fell during the Oct sampling dates.

Decomposition Rates



**Figure 5:** Decomposition rates (-*k*) of deciduous and coniferous leaf litter in high and low budworm sites.

![A close up of a map

Description automatically generated]()The decomposition rate 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; Figure 5). The total mass of DIN deposited by throughfall was positively associated with the deciduous decomposition rate (R2=0.15, p=0.033; Figure 6) but not the coniferous decomposition rate (p=0.13), and the decomposition rate for both leaf types was unrelated to total rainfall sampled. Because I only measured DIN and precipitation while samplers were deployed, these values do not represent actual totals of DIN or precipitation.

**Figure 6:** Regression analysis of throughfall DIN and deciduous decomposition rate.

Soil Chemistry



**Figure 7:** 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). These times also coincided with the end of and the beginning of the growing season respectively. 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 low budworm sites had higher NO3-on 4 Aug 16. Usually soil NH4+ was 60 times higher than soil NO3-. Soil SRP was significantly higher in high impact sites (p=0.047, LME) but did not differ by sample event (p=0.91). Changes in the soil N pool indicated net nitrification (instead of net immobilization or net mineralization), but net nitrification did not differ by budworm impact (p=0.53, LME). Although the very high NO3- value on 6 Nov 16, suggested nitrification for that recorded NO3- spike, it did not differ between high and low budworm.



**Figure 8:** 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 8A). 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 8B). 

**Figure 9:** A regression analysis comparing soil temperature at 2 cm depth and air temperature (p<0.0001, 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). Budworm herbivory level did not influence soil temperature. As expected, temperature increased and decreased more rapidly at shallow compared to deeper depths, and soil temperature differences among dates were less variable in the deepest measurement at 10 cm (data not shown).

**IV**

**DISCUSSION**

In this study I investigated how WSB herbivory affected throughfall chemistry, leaf litter decomposition, and soil chemistry in the eastern Cascades of central Washington. Although budworm herbivory increased N loss from the canopy, especially for NH4+, WSB did not affect throughfall SRP and DOC. Instead, higher concentrations of SRP and DOC in throughfall were seen in two heavy rainfall events suggesting hydrologic control. Budworm herbivory increased organic matter loss from the canopy as frass in summer in comparison to low budworm sites which lost more organic matter as litter in fall. Unexpectedly, decomposition rates were faster in low budworm sites compared to high budworm sites for non-native deciduous litter and for native coniferous litter. Decomposition of deciduous litter was additionally positively influenced by total N deposited by throughfall. Seasonality was the main driver of differences in soil moisture, soil temperature, and soil ammonium whereas budworm herbivory and seasonality interacted in soil nitrate concentrations. Soil phosphorus concentrations were clearly higher in high compared to low budworm sites. Unexpectedly, budworms did not influence net nitrification rate.

Throughfall Nitrogen

I hypothesized that budworms would increase the amount of DIN in throughfall, and throughout this study, there was an interaction between WSB level and sample date for throughfall ammonium. On three of four sample dates, 21 Jun 16, 13 Jul 16, and 21 Jul 16, I observed higher concentration of ammonium coinciding with budworm feeding or immediately after feeding. During pest outbreaks in Scots pine forests, N fluxes from throughfall increase during defoliation events, and then decrease in the fall in the absence of defoliation, supporting my findings for WSB (Grüning et al, 2017). On a fourth date, 11 Sept 2015, I also observed higher ammonium concentrations in high budworm stands, but this date was well after budworm feeding in 2015. It is possible that ammonium generated by budworm feeding was stored in the canopy and washed out during the first major rain event in months that happened on 10 Sept 2015. A similar pattern was observed in subtropical wet forests in Puerto Rico where NH4 was X higher in the first rainfall after a season of defoliation (Heartsill-Scalley et al. 2007). Although these findings are consistent with defoliation accelerating ammonium loss from the canopy to forest soils, right as budworm feeding was beginning in 2016, a 4 Jun 16 throughfall showed the opposite pattern whereby low budworm sites had a higher ammonium concentration. In a regenerating southern Appalachian forest, regression analysis of NH4+ and NO3- canopy change rates showed that these nutrients are positively correlated and instead claim that they are being absorbed from precipitation suggesting uptake of ammonium, and could explain what I am seeing in the high budworm canopy sites (Potter et al, 1991).

Similar to ammonium, budworms activity interacted with sample date to affect throughfall nitrate concentrations, so a generalized conclusion cannot be drawn. Interestingly, 13 Jul 16 and 21 Jul 16 have higher nitrate in high budworm stands that coincided with higher ammonium, which suggests canopy nitrification similar to coniferous throughfall in Adirondack Mountains of New York (Chen et al, 1983). More recent studies have challenged the idea of canopy nitrification being such a large factor. For example, leaching of partially consumed or damaged leaves in the canopy from mature trees, which become less hydrophobic as they age, allowing for more anions and cations to be released in water droplets during rain and wind events (Tukey 1966; Reynolds et al 2000; Hunter et al 2001). An experiment on the effect of nitrogen fertilizer on a mature spruce-hemlock forest in Maine also suggests that canopy nitrification was not responsible for nitrate increases in throughfall?, instead suggesting that it was due to dry deposition of nitrate washed off by rain events (Gaige, et al 2007). Although I did not collect the data required to discriminate between these alternate mechanisms of NO3- loss from the canopy, the data I did collect suggest that WSB herbivory is associated with NO3- loss from the canopy to forest floor.

Throughfall ammonium and nitrate generally increased concentration from 1.5x to 2x as herbivory intensified relative to the low herbivory stands (Figure 2), consistent with a sustained and increasing budworm effect during their active feeding times. Winter moths have been shown to increase canopy N during herbivory outbreaks in Oak forests in Germany and claimed that herbivores affect the canopy much more than they do soils (LeMellec et al, 2011). The generalized increase in throughfall inorganic N coinciding with the summer growing season could have implications for plant uptake during that time. An experiment using Gala apples showed that photosynthesis rates do increase with what?, leading to new growth after defoliating events, and although this study only measured carbohydrates, plants also need nitrogen to grow. If photosynthesis rates increase after defoliation, then nitrogen consumption must also increase (Zhou and Quebedeaux, 2003). Therefore the increased N in throughfall during the budworm feeding period may benefit the growth of understory plants, or it could contribute to continued leaf growth in the defoliated trees after WSB feeding ends with pupation.

Throughfall SRP and DOC

The observed pattern of throughfall SRP data does not support my hypothesis of increased SRP in high WSB sites, but SRP concentrations did differ by sample event. During 8 Nov 15 and 21 July 16, there were two large rainfall events, and here I saw higher concentrations of SRP. I would have expected that phosphorus levels would be higher in heavily impacted areas due to an increase in frass input and increased SRP from leaching of partially consumed leaves as seen in multiple herbivore ecosystem interactions (Hunter et al, 2001). However, not all studies have shown that herbivory leads to increased nutrient fluxes. In a study in western Oregon on Douglas fir trees, throughfall data suggests that precipitation plays a much bigger role in throughfall nutrients than defoliation did, consistent with my findings (Schowalter, 1999). Another study showed that the low level consumption of red maple trees and black locust trees in the southern Appalachians by canopy arthropods may not significantly alter throughfall concentrations for PO43-, but attributed imprecise method design for a potential reason for not being up to pick up small changes (Seastedt et al. 1983). Therefore, SRP fluxes from the canopy to soils appear to be more strongly influenced by hydrology than by herbivory.

Like with throughfall SRP, there were also significant sample event differences for throughfall DOC but no differences for budworm impact. Again, I hypothesized higher DOC concentrations in high budworm impact sites as many studies have shown that insect herbivory leads to increased throughfall DOC and N (LeMellec et al, 2011; Kindlmann and Stadler 2004Stadler and Michalzik, 2001). Given the relatively consistent observation that herbivory increased DOC fluxes from the canopy, it is unclear why this pattern was not observed for WSB. (CHECK YOU CONCENTRATIONS COMPARED TO OTHER STUDIES. ARE YOU CONCENTRATIONS PARTICULARLY HIGH OR LOW IN COMPARISON? THAT MIGHT EXPLAIN SOMETHING). However, the samples dates with highest DOC coincided with the highest SRP concentrations, again suggesting hydrologic control over DOC delivery to forest floors in this study system.

Frassfall and Litterfall

Peak litter fall occurred for both high and low budworm areas during late fall, but litter fall was greater for low budworm sites compared to high budworm sites. In contrast, peak frass fall was greater in high budworm sites during the summer budworm feeding season. In the southern Appalachians native herbivores cause peak frass input between the months of June and August with temporal differences attributed to the elevation gradient and correspondingly different growing seasons (Hunter et al, 2003). Moreover, increased frass deposition during the growing season was positively correlated with throughfall nitrogen inputs (Hunter et al 2003), consistent with the ammonium and nitrate throughfall patterns I observed. While this study showed that spatio-temporal effects also played a large role in? when they did see positive correlations between frass inputs and nutrient availability, they were relatively strong correlations. Because frass has fewer complex organic molecules that need to be broken down, it is readily susceptible to leaching (Hunter et al, 2003) allowing for quick microbial immobilization of ammonium, and nitrate export to local watersheds. The consistent findings of higher throughfall ammonium and nitrate and more frass deposition during feeding strongly suggest that frass is a mechanism inorganic nitrogen delivery to forest soils.

Leaf Litter Decomposition

I had hypothesized that decomposition would occur at a faster rate in high herbivory areas, as herbivory would decrease forest canopy to allow more water to reach the forest floor, and herbivory would increase nitrogen simulating to stimulate bacterial growth (Lovett et al., 1995). My results showed the opposite effect whereby litter in high budworm sites decomposed at a slower rate than litter in low budworm sites. Interestingly cottonwood leaf litter decomposition response to galling aphids (*P. betae*) where galled leaf litter decomposed 34-40% slower than non-galled leaf litter because of decreased leaf litter quality (Schweitzer et al 2005) suggesting that leaf litter from defoliated trees might have reduced quality. Although I used leaf packs standardized across study sites, perhaps DOC leached from defoliated trees slowed decomposition. I cannot test that hypothesis because I only measured DOC concentration and not DOC quality. Whereas I had hypothesized that microclimate differences associated with herbivory (i.e., open canopy allowing more light and/or water to reach the forest floor), ultimately climate has a small effect on late stage decomposition; even with less canopy cover, greater amounts of light reaching the forest floor, and warm dry months, the rate of decay should not increase (Berg and Meentemeyer 2002). Fungi are less able to contribute to decomposition in N rich environments (Diepen et al 2017), suggesting that as more N enters the soil from throughfall, decomposition rates could decrease. My findings are consistent with literature in that decomposition rates are faster in low budworm impact areas compared to high due to high budworm sites having lower quality leaf litter, and having high concentrations of N overall, lessening fungal decomposition ability, and suggest that WSB could have the potential to alter ecosystem nutrient dynamics if their current outbreak cycles continue.

Soil Moisture and Organic Matter

I hypothesized that budworms would affect soil moisture, but I found no evidence for this. As defoliation by budworms occurred, the canopy could have had more openings for moisture to reach the forest floor during rain events, thus increasing soil moisture during wet periods and decreasing soil moistureduring warmer dry periods. While I did see an association between large rainfall events and higher soil moisture, for example on 11 Oct 15 and 4-Aug-16, in high impact sites and on 8-May-16 and 6-Nov-16 in low impact sites, there was no consistent difference between high and low impact sites. It is possible that even in high herbivory sites, most herbivory occurs in the new growth on the fringe of the trees, so herbivory might not open the canopy significantly until it becomes heavy enough to actually kill trees. Alternatively, in this mountainous environments, nearby sites might have had enough microclimate differences to exceed any variability caused by budworms.

I hypothesized that more frass input would lead to higher soil organic matter content. Despite seeing higher frass deposition in high budworm sites, I did not observe any differences in organic matter between high and low budworm sites. Due to the possibly lower carbon quality of consumed leaf litter (CITATION FROM ABOVE), it is possible that frass had not decomposed enough to enter the soil pool as organic matter in my relatively short study. Alternatively, herbivoryes might not have a consistent effect on soil organic matter. For example, grasslands worldwide show that soil organic carbon can increase, decrease, or stay the same in the presence of herbivory by grazers, over a wide range of temperatures, precipitation levels, and elevation, but bulk density either increases or stays the same in response to herbivory (Piñeiro et al, 2010). Regardless of the mechanism, budworm herbivory does not appear to influence soil organic matter.

Soil Nutrients

I hypothesized that soil nutrients would increase in the presence of budworms, but this did not occur for ammonium despite higher throughfall ammonium concentrations. The lack of consistency between throughfall and soil ammonium concentrations suggests very high immobilization of ammonium via soil bacterial production or understory plant uptake. There was however a significant sample event effect indicating that ammonium concentrations differed through time. Highest concentrations of soil ammonium occurred on 8 May 2016 and 6 Nov 16, which are before and after the growing season, respectively. Higher soil ammonium during these times may represent reduce plant uptake from soil ammonium pools and/or net mineralization (THERE MUST BE A REFERENCE FOR THIS).

Soil nitrate concentrations interacted between sample event and budworm herbivory, but concentrations were relatively low throughout the study. Nitrate is taken up at similar rates during growing season (Nadelhoffer et al, 1984), which could explain the low concentrations for most of the sample dates. Similar to ammonium, there did not seem to be a concordance between throughfall and soil nitrate concentration. For example, on 8 May 2016 and XX June 2016 pulses of throughfall nitrate were not seen in the soil, suggesting rapid microbial immobilization or plant uptake into biomass. I did observe two pulses of nitrate in soils, one during the growing season (August 2016) and one right before winter after the growing season (November 2016). There was a large rainfall event just prior to both sampling events, and rainfall has been shown to leach nitrates into the soil solution (Wang et al, 2010) with potential for runoff to streams (Wang, 2020). Like soil ammonium concentration patterns, soil nitrate suggested rapid immobilization of N leached from the canopy, which implies a relatively tight link between canopy N losses via WSB followed by soil and/or plant retention of canopy-leached N.

SRP concentrations in soils were higher in high budworm sites, supporting my hypothesis. SRP was higher in high budworm sites at all sample dates, a trend that has also been seen in tropical forests experiencing herbivory (Metcalfe et al, 2013). Because SRP throughfall concentration did not differ by herbivory level in my study, it suggests that the WSB in highly impacted areas are adding more phosphorous to soils in particulate forms such as frass, molts, dead adults, or damaged leaf litter than can be taken up by soil microbes as seen in other systems (Metcalfe et al, 2013). A study with potted Douglas fir seedlings found that in soils containing high levels of basalt, WSB increased soil P concentrations (Kolb et all, 1999), suggesting that budworms can increase soil P in systems that are not limited by P. The central Cascade region is high in basalt, which coupled with apparently rapid immobilization of inorganic N, would suggest this is not a P limited system. In systems not limited by P, excess P has the potential to be leached into the nearby streams during rain or snow melt, and excessive P leaching can lead to eutrophic downstream systems. Finally, while evidence suggests a role for WSB to influence soil P concentration, because my study sites are not interspersed between the Swauk and Teanaway drainages due to where budworms were active, I cannot dismiss the possibility that Swauk soils generally have higher P than Teanaway soils in the absence of budworms.

Conclusion

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 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.

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