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Development of soil lead and tree-ring methods for characterizing atmospheric deposition gradients and subsequent tree growth responses

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**Development of soil lead and tree-ring methods for characterizing atmospheric deposition
gradients and subsequent tree growth responses**

Bradley P. Breslow

Thesis submitted to the
Eberly College of Arts and Sciences
at West Virginia University
in partial fulfillment of the requirements
for the degree of

Master of Arts
in
Geography

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2011

Keywords: Acid deposition, forest ecology, deposition modeling, dendrochronology

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ABSTRACT

Development of soil lead and tree-ring methods for characterizing atmospheric deposition gradients and subsequent tree growth responses

Bradley P. Breslow

Atmospheric deposition of pollutants and nutrients can have a range of effects on aquatic and terrestrial ecosystems. Despite its ecological importance, it remains a challenge to model patterns of deposition across complex terrain and also to assess how forest ecosystems are reacting to chronic inputs of atmospheric deposition. I measured the lead (Pb) content of forest soils adjacent to atmospheric deposition monitoring stations throughout the central Appalachian Mountains to examine how well patterns of soil Pb relate to regional patterns of inorganic nitrogen (N) deposition. While I found no significant relationship between organic soil Pb content and N deposition, I did find a strong positive relationship between mineral soil Pb content and N deposition, ($R^2=0.68$, $p=0.08$). These results indicate that mineral horizon soil Pb is a potentially useful index of pollutant deposition in this region. Furthermore, the measurement of mineral soil Pb provides a practical, surrogate measure of depositional patterns for ecologists interested in observing the effects of atmospheric deposition on ecosystem function. I also examined the effects of acid deposition on tree growth by sampling three tree species (*Picea rubens*, *Liriodendron tulipifera*, and *Betula alleghaniensis*) along spatial gradients of deposition within the eastern United States. By studying tree-rings along the gradient, I was able to examine the inherently long-term and cumulative effects of deposition on tree growth and how this response changed through time. While a limited sample size and stand dynamics overwhelmed the ability to detect any growth response within the *Liriodendron* or *Betula* study sites, I did discover a strong negative relationship between the deposition gradient and *Picea* growth in the central Appalachian Mountains ($R^2=0.89$, $p=0.05$). Interestingly, the strength of the relationship within our *Picea* dataset changed through time, in parallel with known temporal patterns of deposition in the region. I found that the use of tree-ring analyses in carefully selected plots along environmental gradients provides the unique opportunity to not only assess growth responses to spatially-varying patterns of environmental change, but to also characterize how forests are reacting to slow cumulative changes through time. The promising results seen in this study should be expanded upon in the future.

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Introduction

Acid deposition represents a complex environmental stress that is variable across space and time and affects the health of forest ecosystems. Despite this complexity, research and management efforts over the last 40 years have resulted in decreases in acid deposition and better understanding of many of the subsequent ecosystem effects. However, it still remains a challenge to model patterns of atmospheric deposition across complex terrain and consistently measure growth responses of forest ecosystems to increased levels of deposition.

I proposed my thesis as one project, with the goal of empirically modeling deposition across known gradients and then measuring the growth responses of multiple tree species to spatially varying levels of nitrogen (N) deposition. However, as the research progressed, it became clear that I ultimately had two projects with related, but separate, objectives and conclusions. *Chapter 1* evaluates the use of soil lead (Pb) as an index of pollutant deposition in topographically complex regions. *Chapter 2* outlines the development of a methodological approach using tree rings along environmental gradients to investigate how forest ecosystems may be reacting to slow, cumulative environmental changes, such as chronic inputs of atmospheric deposition. Each project is organized as a journal manuscript draft, with appropriate sections required by the journals *Environmental Pollution* (*Chapter 1*) and *Methods in Ecology and Evolution* (*Chapter 2*).

While separate, these two manuscripts are connected by the theme of developing methodological approaches to investigate the complex environmental stress of acid deposition. The methods discussed here offer the potential to be expanded upon and more fully developed in the future.

Chapter 1.

Soil lead as an index of cumulative loading of atmospheric nitrogen deposition: an evaluation in the central Appalachian Mountains, USA.

Abstract

Understanding the effects of atmospheric deposition on ecosystem functioning has been a focus of scientific research for several decades, but the ability to accurately measure atmospheric deposition across large spatial scales remains difficult. We measured the lead (Pb) content of forest soils adjacent to atmospheric deposition monitoring stations throughout the central Appalachian Mountains to examine how well patterns of soil Pb relate to regional patterns of inorganic nitrogen (N) deposition. While we found no significant relationship between organic soil Pb content and N deposition, we found a positive relationship between mineral soil Pb content and N deposition, ($R^2=0.68$, $p=0.08$). Our results indicate that mineral horizon soil Pb is a useful index of pollutant deposition in this region and provides a practical, surrogate measure of depositional patterns for ecologists interested in observing the effects of atmospheric deposition on ecosystem function.

1.1 Introduction

Atmospheric deposition is an important source of nutrients and pollutants for many terrestrial ecosystems. Since nitrogen (N) is a common limiting nutrient in terrestrial ecosystems worldwide (LeBauer and Treseder 2008, Vitousek and Howarth 1991), atmospheric N deposition can have many important ecological effects, including positive impacts on forest growth (Bedison and McNeil 2009, Magnani et al. 2007, Thomas et al. 2010) and negative impacts, such as surface water eutrophication (Driscoll 2003, Galloway 2003). However, despite this

importance, the ability to measure deposition across broad spatial scales remains difficult, especially in mountainous terrain.

Numerous interpolation models have been developed that use point-based deposition measurements to predict patterns of deposition at various scales, from the landscape to the regional level (Grimm and Lynch 2004, Ito et al. 2002, Ollinger et al. 1993, Weathers et al. 2006). While these and other studies illustrate that spatial gradients remain largely constant through time, they also indicate that complex terrain and a sparse network of monitoring stations in mountainous regions create considerable uncertainty in deposition estimates interpolated from lower-elevation monitoring stations.

To reduce this uncertainty, the concentration of lead (Pb) in the forest floor has been proposed as an integrative index of deposition in mountainous terrain because it can preserve long-term deposition patterns (Johnson et al. 1982, Weathers et al. 2006, 2000, 1995). The major input of Pb into terrestrial ecosystems is through atmospheric deposition (Nriagu 1989) with mines, smelters, and fossil fuel combustion being the prime contributors (Johnson et al. 1982). Forest ecosystems tend to strongly retain Pb because inputs (i.e. atmospheric deposition) largely exceed outputs (i.e. Pb outflow in streams; Johnson et al. 1995, Wang et al. 1995), and because Pb has a strong affinity for soil organic matter (Smith 1990). Soil Pb can reflect patterns of atmospheric deposition (Weathers et al. 2006, 2000, 1995, Johnson et al. 1982) because it is relatively biologically immobile (Friedland 1984), it binds strongly with organic matter in the forest floor (Johnson et al. 1995, Smith 1990, Wang et al. 1995), and it has a long residence time (Friedland 1984, Smith et al. 1981, Wang et al. 1995). A strength of using soil Pb as an index of pollutant deposition is that continuous measurements are not necessary; a one-time measurement

can theoretically provide a time-integrated index of the cumulative deposition loading (Weathers et al. 2006).

While it has been established that atmospheric Pb is retained in the soil of forest ecosystems, it has more recently been identified to move through the soil profile faster than once predicted (Johnson et al. 1995, Johnson et al. 2010). Previously, it was generally believed that Pb would remain in the surface layer of forests for a period of centuries (Smith et al. 1981, Turner et al. 1985). More recent analyses, however, have shown faster than expected transport to the lower layers of the soil horizon (Johnson 1995, Johnson et al. 2010, Kaste et al. 2003, Watmough et al. 2004). Here, we explore how well organic and mineral horizon soil lead concentrations relate to measurements of atmospheric inorganic N deposition throughout the Central Appalachian Mountains.

1.2 Material and Methods

1.2.1 Study Area

The central Appalachians are downwind of the Ohio River Valley which is an industrial center with an abundance of coal-fired power plants. As a result, the region receives some of the highest rates of acid deposition in the nation (National Atmospheric Deposition Program [NADP], 2010). Generally, the spatial pattern of atmospheric deposition in the region decreases from north to south (Figure 1.1). Therefore, we sampled forest soils adjacent to deposition monitoring stations throughout the region spanning this known gradient of N deposition

1.2.2 Nitrogen Deposition Data

We obtained total wet inorganic N deposition data for all of our NADP monitoring stations from the NADP National Trends Network (NTN) (Table 1.1). For ‘Station C’, a site within the Fernow Experimental Forest, we obtained data from the USDA Forest Service

Northern Research Station Timber and Watershed Lab- Parsons, West Virginia. Because we were interested in examining the relationship between soil Pb and chronic deposition, we calculated an index of long-term deposition loading by averaging annual total inorganic N in wet deposition at each site. For each site, we used all available years of data since site establishment and excluded any years that did not meet NADP/NTN completeness criteria (data available online). All included stations had at least 10 years of available deposition data. The deposition monitoring stations used in this analysis all occur at lower elevations and because wet and dry deposition generally have concurrent spatial patterns at lower elevations (Weathers et al. 2006), we did not incorporate any component of dry deposition into our index of long-term deposition loading.

1.2.3. Field Methods

Similar to methods used successfully by others (e.g. Weathers et al. 2006), we established circular (5-m radius) sample plots in forested areas adjacent to each of our National Atmospheric Deposition Program (NADP) sites and at ‘Station C’ within the Fernow Experimental Forest. Each plot was divided into four quadrants from which we collected two cores (5 cm diameter) of the O soil horizon (O_e & O_a) and two cores of the top 5 cm of mineral soil (2.2 cm diameter) for a minimum of eight subsamples from each horizon. To reduce the influence of confounding factors, all sample plots were: 1) located ≥ 100 m from the nearest primary road; 2) positioned ≥ 20 m uphill from any trails; 3) free of any evidence of recent fire activity; and 4) contained in a forest of intact canopy trees (> 12 m tall or > 25 cm diameter trees). At each site, we combined all subsamples into a single sample for each site for analyses.

1.2.4 Lab Methods

Once returned to the lab, we dried all samples in paper bags (≥ 48 hours at 65°C) and removed all rocks and debris from each of the samples prior to recording their mass and grinding them to pass a #40 mesh using a Wiley Mill. After samples were ground, we added 5 mL of concentrated nitric acid (HNO_3) and three glass beads to 0.2 g subsamples in 75 ml digest tubes and allowed them to sit for >24 hours. We then gently boiled the samples at 125°C using a block digester and added 2 mL of 30% hydrogen peroxide (H_2O_2) every hour, until the digestion was complete (i.e. all organics were oxidized and the solution was clear). Digested samples were diluted to a total volume of 75 mL by the addition of deionized water and then filtered through No. 42 ashless Whatman filter paper. For purposes of quality control, we treated laboratory blanks (nitric acid only) and standard reference material (National Institute of Standards and Technology [NIST], 2781, domestic sludge) in the same manner as the soil samples being analyzed. The concentration of lead in the acid-digested samples was measured using a Varian 220FS atomic absorption spectrophotometer (Varian, Inc., Mulgrave, Vic., Australia) equipped with a graphite furnace. To account for any matrix effects we used a 3 point standard addition method for each sample. We converted raw sample Pb concentrations (mg/kg) into total Pb content per area (mg/m^2) using the bulk density of the sample soil material.

1.3 Results and Discussion

Mean annual inorganic N deposition at the monitoring stations ranged from 3.78 to $7.86 \text{ kg ha}^{-1} \text{ yr}^{-1}$. Total Pb content in the organic horizon ranged from 135.62 to 637.59 mg/m^2 and total Pb content in the mineral horizon ranged from 214.92 to 1206.63 mg/m^2 . Total soil Pb content (organic + mineral) ranged from 350.54 to 1547.85 mg/m^2 . While we found no significant relationship between organic soil Pb and inorganic N deposition, we did find a

positive relationship between mineral soil Pb content per area and inorganic N deposition, ($R^2=0.68$, $p=0.08$) (Figure 1.2).

We suggest that the lack of a significant relationship between organic soil Pb and N deposition, but presence of a significant relationship between mineral soil Pb and N deposition can be attributed to several factors. First, and most importantly, we suggest that movements of Pb through the soil profile obscured relationships among organic horizon Pb and deposition loadings. The major input of lead (Pb) into terrestrial ecosystems is through atmospheric deposition, with the majority of atmospheric Pb deposition being a result of additives in gasoline between ca. 1940 until ca. 1985 (Nriagu 1989). Concern over Pb pollution in the United States resulted in the restriction of leaded gasoline in 1977. With the prohibition of leaded gasoline, atmospheric deposition of Pb decreased dramatically, with current rates only making up a few percent of deposition rates from 25 years ago (Johnson et al. 2010). Johnson et al. (2010) documented an obvious decrease in forest floor Pb in the mid-Atlantic region, which is attributable to decreasing emissions coupled with the fact that soil Pb is moving faster through the profile than it is being added (Friedland et al. 1992, Johnson et al. 1995, Johnson et al. 2010, Kaste et al. 2003, Watmough et al. 2004). Because an estimated 40% of atmospheric Pb comes from sources other than leaded gasoline (Johnson et al. 1982), Pb deposition still occurs, but the accumulation of Pb in the organic layer of forests has nevertheless reduced markedly in the last 30 years.

Second, there is also evidence that variations in forest floor thickness and litter turnover rates can affect rates of Pb migration through the soil profile (Johnson et al. 2010). These variables are largely controlled by patterns of temperature and litter quality; which, with cautious sampling, can be held relatively constant along regional gradients (Johnson et al. 2010).

However, because there were no large variations in organic horizon thickness or temperature among our sites, spatial patterns of litter turnover are unlikely to have strongly affected our results.

Finally, another important consideration is that our gradient of measured N deposition (3.78 to 7.86 kg ha⁻¹ yr⁻¹) is relatively small, which could have hampered our ability to detect a relationship among deposition and Pb in the organic horizon. For instance, using seven sites spread across a broader gradient in the northeastern US (4.20 to 11.00 kg ha⁻¹ yr⁻¹) Weathers et al. (2006) reported a positive linear relationship ($R^2 = 0.72$) between N deposition and organic soil Pb content. While a relatively small deposition gradient may have hampered our ability to detect a relationship between deposition and organic soil Pb, the presence of this relationship in the mineral horizon suggests that our results were more strongly affected by a migration of Pb from the organic horizon. In addition to the benefit that soil lead is relatively easy to measure across a landscape, mineral Pb measurements provide a way to contextualize long-term spatial patterns and cumulative ecosystem effects of deposition along shrinking gradients of deposition (NADP 1985-2010).

While our data suggest that atmospherically deposited Pb has transferred through the forest floor profile, we demonstrate that soil Pb can serve as a useful index of N deposition when it is sampled in the mineral horizon. We suggest future research should build on this promising result by incorporating more study sites spanning larger gradients of N deposition. It is important to note that Pb is only a surrogate measure of other pollutants (e.g. nitrogen), so establishing regional-specific calibrations are needed for using soil Pb to detect patterns of deposition in other areas. In addition, because we were able to demonstrate that Pb concentrations in the mineral horizon reflect patterns of deposition, future research could benefit

by examining how this relationship changes with soil depth and if/how it changes by soil horizon. By addressing these concerns, we believe a methodology for implementing a time-integrated index of chronic deposition could be more fully developed.

1.4 Conclusions

Our results support the use of mineral soil Pb as an index of deposition loading. However, aligned with evidence indicating that atmospherically deposited Pb is moving through the soil profile faster than once believed (Johnson et al. 1995, Johnson et al. 2010, Kaste et al. 2003, Watmough et al. 2004), our results suggest that organic and total soil Pb content are not viable indices of depositional patterns. We suggest dense networks of mineral soil Pb measurements can capture important spatial patterns of deposition and could be especially useful for understanding spatial patterns in ecosystem functioning within complex mountainous areas that are sparsely measured by national deposition monitoring networks.

Figures and Tables- Chapter 1

Figure 1.1. Study area and all stations sampled. Modeled deposition data from Grimm and Lynch (2004) show the mean annual total inorganic N deposition for years 1995 through 1999. Under represented stations in the region were excluded from the analysis ($N > 10$ years of available deposition data).

Table 1.1. Deposition stations sampled throughout the region.

Station (Operator)	Location	Latitude	Longitude	Elevation	Est. Date
WV05 (NADP)	Gilmer County, WV	38.8794	-80.8478	234 meters	1999
WV04 (NADP)	Fayette County, WV	37.9796	-80.9525	753 meters	1983
VA28 (NADP)	Madison County, VA	38.5225	-78.4358	1074 meters	1981
VA13 (NADP)	Giles County, VA	37.3294	-80.5579	916 meters	1978
Station C (Fer. Exp. Forest)	Tucker County, WV	39.0579	-79.6919	842 meters	1982

*Other stations were sampled but excluded in the analysis because data did not meet requirements (N>10 years). For stations used in the analysis, all years of available data since site establishment were used to calculate mean deposition loading, excluding years where data did not NADP/NTN Completeness Criteria.

Figure 1.2. Relationship between total soil lead (Pb) content and mean N deposition in the organic horizon (a), mineral horizon (b), and total (organic +mineral) (c).

Chapter 2.

Observing slow ecological change with tree rings: a methodological evaluation using atmospheric deposition gradients in the eastern United States.

Abstract

It remains a challenge to assess how forest ecosystems are reacting to slow, cumulative environmental changes, such as chronic inputs of atmospheric deposition. As a result, there is still much debate on the effects of atmospheric deposition upon forest growth. Initially, most analyses concluded that increased acidic conditions lead to decreased tree growth, with a focus on a few select species. More recent analyses have suggested that increased N deposition is generally increasing forest growth. These mixed and context-specific results regarding the effects of atmospheric deposition are illustrative of a more general challenge: a limited ability to assess how forest ecosystems are reacting to gradual or accumulating environmental stresses. In this study, we explore the efficacy of tree-ring analyses along spatial gradients of N deposition to examine how atmospheric deposition affects three functionally-different, but common tree species (*Picea rubens*, *Liriodendron tulipifera*, and *Betula alleghaniensis*) in the eastern United States, and also to explore how this response has changed through time. While a limited sample size and stand dynamics overwhelmed our ability to detect any growth response within our *Liriodendron* or *Betula* study sites, we did discover a strong negative relationship between the deposition gradient and *Picea* growth in the central Appalachian Mountains. Interestingly, the strength of the relationship within our *Picea* dataset changed through time, in parallel with known temporal patterns of deposition in the region. We conclude that the use of tree-ring analyses in carefully selected plots along environmental gradients provides an unique opportunity to not only assess growth responses to spatially-varying patterns of environmental

change, but to also characterize how forests are reacting to slow cumulative changes through time.

2.1 Introduction

Despite its ecological importance, the effect of atmospheric deposition upon forest growth remains a subject of uncertainty (Magnani et al. 2007; De Vries 2009; Bedison and McNeil et al. 2009; Thomas et al. 2010). Most studies of increased pollutants and forest growth have relied upon experimental, high-dose treatments over short time periods. While certain fertilization studies have shown that N additions can provide growth enhancements (Hyvönen et al. 2008), others have demonstrated that N saturation can induce mortality, which in turn decreases the amount of carbon (C) stored in forest biomass (Magill et al. 2004; Wallace et al. 2007). These fertilization studies, while informative, are somewhat limited in their ability to examine long term vegetative responses to continual, low level inputs of nutrients, such as atmospheric deposition. Numerous studies have utilized gradients of N deposition to examine effects of deposition upon ecosystem function, but the extent to which N deposition affects forest C sinks is still a contentious topic (McNulty et al. 1991; Boggs et al. 2005 Magnani et al. 2007; DeVries et al. 2009; Solberg et al. 2009; Laubhann et al. 2009; Bedison and McNeil 2009; Thomas et al. 2010). Magnani et al. (2007) concluded that forest C sequestration was strongly correlated with increased N deposition, but the findings of this study were controversial and spawned a discussion on the plausibility of a large “N-induced C sink” (de Vries et al. 2008; Magnani et al. 2008). Thomas et al. (2010) helped resolve the debate by using long-term tree basal area data, collected by the United States Forest Service (USFS) Forest Inventory & Analysis (FIA) Program, to demonstrate that ambient rates of N deposition are generally increasing growth of some species, but also illustrated that many species are not affected or have

declining growth with increased N deposition. Because FIA data have a limited historical record, tree-rings provide a complementary approach to examine when changes in growth can first be detected and how this response has changed through time. Mixed and context-specific results of deposition induced tree growth responses are illustrative of a more general challenge: a limited ability to assess how forest ecosystems are reacting to gradual or accumulating environmental stresses over time. Here, we examine the utility of tree-ring records to examine temporally-changing growth responses to acidic deposition. Furthermore, based on our promising results, we suggest that this methodological approach can be applied to other examples of accumulating environmental stresses, such as gradients of ozone pollution, invasive species infestation, or new temperature/precipitation gradients induced by climate change.

A large body of work within the field of dendrochronology shows that tree-rings serve as a reliable and standardized proxy for overall forest health (Cook et al. 1987; Swetnam and Betancourt 1998; Gedalof and Berg 2010; Silva et al. 2010). In fact, tree-rings were widely used to document declining growth trends of *Picea rubens*, largely attributed to acid deposition, throughout the Appalachian Mountains and portions of New England during the 20th century (Johnson and Siccama 1983; Adams et al 1985; Hornbeck et al. 1986; Smith et al 1990; Hornbeck and Kochendeffe 1998). Dendrochronology has also been used to link anthropogenic impacts (i.e. air pollution) to declines in forest growth or physiological stress in trees (Freyer and Belacy 1983; LeBlanc 1993; LeBlanc 1998; Duchesne 2002; Savard et al. 2002, 2004; Boggs et al. 2005; Li et al. 2010).

Because tree-rings offer the ability to examine growth trends through time, many researchers have successfully combined tree rings with environmental gradients (e.g. topographic, vegetation, elevational, etc.) to address complex ecological questions. Some have

used topographic gradients to identify limiting factors most directly related to physiological changes and trends in radial growth (Donnegan and Rebertus 1999; Ruffner and Abram 2003; Knutson and Pyke 2008), while others have sampled along latitudinal or elevational gradients to predict potential growth responses to climate change, or to investigate the climatic sensitivity of species across their geographic range (Jacobi and Tainter 1988; Hofgaard et al. 1999; White et al. 2011).

In this study, we extend the proven ability of using tree rings along environmental gradients to examine anthropogenic ecological changes, specifically atmospheric deposition. Studying tree-rings along atmospheric deposition gradients offers the ability to examine the inherently long-term and cumulative effects of varying amounts of deposition on forest growth and it also offers the potential to contextualize when changes in growth may have begun to occur. The few existing studies utilizing tree-rings along deposition gradients have shown promising results (LeBlanc 1993; LeBlanc 1998; Duchesne 2002; Boggs et al. 2005), but most have focused on few species and do not capture recent trends during a period of decreasing emissions (National Atmospheric Deposition Program [NADP] 2011)

Here, we examine the effects of deposition on tree growth by sampling three tree species, *Picea rubens* (red spruce), *Liriodendron tulipifera* (tulip poplar), and *Betula alleghaniensis* (yellow birch), along spatial gradients of deposition within the eastern United States. Attempting to choose mature stands of intact forest with no evidence of recent disturbance, we created plots for each species spanning ambient gradients that maximized variability in deposition. We chose to look at three functionally different species to characterize how various forest types of the eastern US have been affected by chronic inputs of atmospheric deposition.

2.2. Methods

2.2.1 Study Area

Spatial variation in atmospheric deposition occurs at many scales and regional gradients of deposition in the eastern United States have been well documented for some time (Ollinger et al. 1993; Ito et al. 2002; Grimm and Lynch 2004). Based on these regional patterns we chose to focus on two well-documented gradients of N deposition within two sub-regions of the eastern United States. We sampled along a deposition gradient in the central Appalachian Mountains (CAM) (Grimm and Lynch 2004) and a deposition gradient in the Adirondack State Park (ADK) (Ito et al. 2002). Because the CAM and ADK are downwind of the Ohio River Valley, an industrial center with an abundance of coal-fired power plants, both sub-regions receive some of the highest rates of atmospheric deposition in the nation (NADP 2010). The spatial pattern of atmospheric deposition in the CAM generally decreases from north to south (Grimm and Lynch 2004), while the spatial pattern of atmospheric deposition in the ADK generally decreases from southwest to northeast (Ito et al. 2002) (Figure 2.1).

In the CAM, we selected eight study locations that contained either *Picea rubens* or *Liriodendron tulipifera*. Since the two study species in this region naturally occur at differing elevations within each study location, we sampled *Picea rubens* at high elevation (1125-1345 meters) mountaintop sites and *Liriodendron tulipifera* on lower elevation sites (730-915 meters). In the ADK, we selected four study locations that maximize variability in N deposition and contain populations of *Betula alleghaniensis* within an elevation range of 510-700 meters (Figure 2.1).

2.2.2 Study Species

A recent study found that growth of *Picea rubens* was among the most adversely affected by N deposition (-0.1% growth change per kg N ha⁻¹ yr⁻¹) whereas *Liriodendron tulipifera* was

among the most positively affected (16% growth change per kg N ha⁻¹ yr⁻¹; Thomas et al. 2010). In that study, all species that were positively affected had arbuscular mycorrhizal associations, and all species that were negatively affected were evergreen conifers with ectomycorrhizal associations (Thomas et al. 2010). Based on the findings from that study and their hypothesis that mycorrhizal association could be a driving force of the relationship between N deposition and tree growth, we expect *Picea rubens* to be negatively affected by increasing deposition and *Liriodendron tulipifera* to be positively affected by increasing deposition. We chose *Betula alleghaniensis* as a “functional bridge” in the sense that it has the same ectomycorrhizal association as *Picea*, but is a deciduous hardwood species that displays physiological traits more characteristic of *Liriodendron*. Based on the current limited understanding linking mycorrhizal associations, functional types, and growth responses, we expected the response of *Betula alleghaniensis* to either have no response or to have a very weak positive response. Because N deposition may cause a preferential growth enhancement in less shade tolerant species (McNeil et al. 2007), it is important to note that that *Picea* is a late-successional, shade tolerant species while *Liriodendron* and *Betula* are less shade tolerant, earlier successional species.

2.2.3 Atmospheric Deposition Data

We obtained spatially-explicit estimates of atmospheric deposition for the ADK and CAM sub-regions of the eastern US from two, highly-resolved, validated spatial models (Ito et al. 2002; Grimm and Lynch 2004). While the Adirondack gradient has been used to examine numerous ecosystem responses to atmospheric deposition (e.g. Ito et al. 2002; McNeil et al. 2007; Bedison and McNeil 2009), the application of the CAM gradient is, to our knowledge, new to ecological science. To further validate the deposition model defining this gradient in the

CAM (Grimm and Lynch 2004), we sampled soil lead (Pb) at 6 of our 8 sites within the central Appalachian Mountains. Because soil lead can preserve long term patterns of pollutant deposition (Weathers et al. 2006), our observations of a strong correlation among mineral soil lead and modeled N deposition ($R^2 = 0.90$, $p = 0.03$) provides added confidence in the spatial model used to define deposition loadings to our plots in the CAM (Figure 2.2).

2.2.4 Tree-ring Data

During the summer of 2009 at our *Betula* sites and during the summer of 2010 at our *Liriodendron* and *Picea* plots, we established center points and selected the nearest 20 dominant canopy trees of the selected species for sampling. In order to minimize bias from uneven radial growth, we removed two increment cores from each tree parallel to the contours of the slope and recorded the diameter at breast height (DBH) of each tree sampled to the nearest half centimeter.

In the lab, we processed increment cores according to standard dendrochronological techniques (Stokes and Smiley 1968). We sanded mounted cores with progressively finer sand paper ending in 1500 grit or when individual cells were distinguishable under a stereomicroscope. We visually crossdated all samples before measuring annual ring widths using a Velmex Measuring System (0.01mm precision; Velmex Inc., 2009) combined with the program Measure J2X (Voor Tech Consulting, 2008). Crossdating accuracy was statistically validated using the computer program COFECHA (Holmes 1983; Grissino-Mayer 2001). COFECHA allows for the identification and correction of missing or false rings by comparing individual ring measurements across the entire series. Once each core was measured, crossdated, and statistically validated, we assigned a calendar year to each annual tree-ring.

Because our goal was to characterize growth trends, we converted crossdated tree ring series into annual basal area increment (BAI) using the DBH for each tree measured in the field.

As trees mature, ring width declines, and therefore growth trends are not readily detectable based on changes in ring widths alone (Phipps 1984; Phipps and White 1988; LeBlanc; McCarroll et al. 2004). Unlike patterns in raw ring width, age related trends in unstandardized BAI generally yield positive trends over time (Weiner 2004), therefore a negative trend in BAI can provide a strong indication of growth decline (Phipps and White 1988; LeBlanc 1990). Also, because atmospheric deposition can have cumulative, long term effects on tree growth, unstandardized BAI provides a better metric to preserve growth trends and examine the cumulative effects of varying levels of deposition on tree growth (Duchesne 2002).

Here, BAI is shown annually, where 1 year=1 full ring (late wood and early wood), assuming that growth was uniform along each ring. BAI or annual basal growth (mm^2) was calculated as the basal area (πr^2) of the current-year minus the basal area (πr^2) of the previous year (Equation 1). We combined the values of BAI from the two cores from each tree to obtain the mean annual BAI for each tree.

$$\text{Equation 1. } \mathbf{BAI} = \pi(\mathbf{r_n^2 - r_{n-1}^2}),$$

where r is the tree radius,

and n is the year of ring formation.

2.2.5 Statistical Analysis

To examine how each species responds to varying levels of N deposition at the stand level, we pooled the individual tree data to obtain plot-level values of mean BAI at each of our 12 plots. We then binned our data into fifty year segments (e.g. 1959-2009, 1909-1959) and five year segments (e.g. 2004-2009, 1999-2004, etc.) to not only examine recent growth responses, but to investigate how/if this response has changed through time. Because we were interested in

species-level responses to varying levels of N deposition, we evaluated each species' response to deposition separately. We used univariate linear regression to examine how varying levels of deposition have been affecting growth through time. We used JMP(v 8.0.1, SAS Institute, Cary, North Carolina, USA) for all statistical analyses.

2.3. Results

Mean age of sampled trees in the 12 plots ranges from 49 to 159 yrs (Table 2.1). At all plots 20 trees were sampled, but cores that could not be cross-dated or statistically validated with COFECHA were thrown out of the analysis (Table 2.1). Average unstandardized BAI trends ranged from 300-1880 mm²· yr⁻¹ for the *Betula* plots, 500-5000 mm²· yr⁻¹ for the *Liriodendron* plots, and 50-3000 mm²· yr⁻¹ for the *Picea* plots (Figures 2.3, 2.4, 2.5).

As marked by arrows in Figure 2.3, a large windstorm and canopy blow-down event struck plots 1 and 3 in 1950 and was followed by rapid growth. These marked responses to disturbance confounded any pattern observable along the deposition gradient within our *Betula* sites. Within our *Liriodendron* sites, plot 4, the lowest deposition plot, established much later than the other plots, and caused differences in growth attributable to stand history to overwhelm any detectable response along the deposition gradient (Figure 2.4).

Our data shows a sustained drop in growth among our *Picea* plots beginning around 1960, particularly within the high deposition plot where there was a steep drop in decadal-averaged growth between the 1950s and 1970s (Figure 2.5). We found a significant negative relationship between deposition and growth during the 50-year period of 1959-2009 ($R^2 = 0.89$, $p = 0.05$) (Figure 2.6). When examined at 5 year increments, the strength of the relationship between N deposition and BAI becomes very strong around 1970, is sustained until 2000, and

declines thereafter, generally following patterns in deposition over the same period of time (Figure 2.7).

2.4. Discussion

The Challenges of Studying of Earlier Successional Species

Within our *Betula* plots, large variations in disturbance history, tied with a small sample size confounded our ability to detect a growth response to the variation in chronic N deposition loading (Figure 2.3). At the time of sampling, we were unaware of the discrepancy between the four plots and later found historical evidence of the mapped blowdown event in 1950. Despite the fact that this large variation distorted our ability to examine a relationship between deposition and *Betula* growth in the ADK, we were able to elucidate the influential effect that stand dynamics can have on any forest growth analysis. Site selection is paramount when designing a regional-scale, gradient-based study, especially because so many factors affect tree growth. Boggs et al. (2005) found a non-linear relationship between *Betula alleghaniensis* basal area growth and N deposition in the Southern Appalachians. Because they saw the steepest growth declines within their high deposition plots, the authors inferred that their plots could be showing evidence of possible N saturation (Aber 1989). However, the authors examined the time period 1980-1998 and did not explore any temporal trend within the tree-ring record. While we did not see a similar negative trend within our data, an increased sample size and careful site selection could likely provide a better picture of this relationship and how it has changed through time.

Within our *Liriodendron* dataset, we also found no relationship between BAI and N deposition. Similar to the response seen by Thomas et al. (2010), we hypothesized a positive growth response in the *Liriodendron* sites as deposition increased. Based on its mycorrhizal association and physiological characteristics, *Liriodendron* could theoretically benefit from extra

inputs of usable inorganic N. We consider the lack of response is likely because our low deposition plot (Litu 4) established in the 1950s, with more ingrowth occurring in the 1970s (Figure 2.4). Since the other three plots were similarly aged, the lack of a relationship could be an artifact of Litu 4 being so young. High initial growth, consistent with trends of stand initiation in young mid-successional stands, caused Litu 4 to have consistently inflated BAI for the whole study period in relation to the other plots. Similar to the confounding effects of disturbance history we found within our *Betula* dataset, variations in stand age altered our ability to capture a response within the *Liriodendron* sites. The *Betula* and *Liriodendron* data from our study are a valuable example of the challenges inherent in using tree rings along environmental gradients. In particular, they serve as important evidence documenting that researchers must carefully choose sites of similar stand age and disturbance histories, especially when studying earlier successional species whose rapid growth rates can easily obscure any current visual evidence (e.g. from stand structure) of past events.

A New Spatio-temporal View on Red Spruce Decline

In agreement with responses seen throughout the literature, we found that increased deposition led to declines in tree growth within our *Picea rubens* plots (Johnson and Siccama 1983; Adams et al 1985; Hornbeck et al. 1986; Smith et al 1990; Hornbeck and Kochendeffe 1998, Thomas et al. 2010). Additionally, by using a gradient of deposition coupled with tree-ring analysis, our results offer a unique view on *Picea rubens* growth decline during the 20th century. When designed carefully, this approach offers the ability to examine the effects of spatially-varying amounts of deposition on growth, and tree-rings illustrate with fine temporal resolution that this response changed through time. The variability of the relationship through

time, suggests four distinct stages of the relationship between deposition and tree growth (Figure 2.7).

Because industrial emissions have caused increases in atmospheric concentrations of NO_x , and SO_2 for over 150 years (Li et al. 2010), we emphasize that our plots were still receiving inputs of atmospheric deposition during, stage I, or “initial deposition” (Figure 2.7). We categorize this time period as “initial deposition” because ice core records from northern Greenland indicate steep increases in N and S concentrations after 1950, reflecting patterns in US emissions during the same time period (Fisher et al. 1998). Also, while acid deposition was first recognized in the 19th century (Smith 1872), it did not emerge as an ecological issue in the United States until the 1960s (Likens et al. 1972). The lack of a relationship between growth and deposition within the “initial deposition” stage could partially be attributed to the trend of increasing pollutant emissions arising after 1950, but the lack of a response is also tied to the fact that the trees constituting our plots were predominantly immature during this time period, so variations in growth are more likely driven by stand dynamics within each plot. Similar to the inherent challenges we found within our *Betula* and *Liriodendron* datasets, we once again illustrate the difficulty in comparing growth in young stands.

Stage II, or “increase in pollutant deposition” marks a period in the eastern United States when pollutant emissions continued to increase markedly (Fisher et al. 1998; Driscoll et al. 2001). Also, at this point during the study period our plots begin to approach stand maturity allowing for more straightforward comparison of growth responses to a spatially varying external disturbance (i.e. inputs of atmospheric deposition). During this period the strength of the relationship between deposition and tree growth begins to rise (Figure 2.7).

Stage III, “sustained period of chronic deposition”, supports the hypothesis that increased deposition lead to decreased *Picea rubens* growth in the CAM. Total emissions of SO₂ peaked at 28.8 million metric tons in 1973, while total emissions of NO_x reached 21.8 million metric tons in 1990 (USEPA 2000). During this prolonged period of sustained emissions we saw a strong negative relationship between tree growth and N deposition in alignment with widely cataloged evidence of decreases in *Picea rubens* growth throughout the eastern United States (Johnson and Siccama 1983; Adams et al 1985; Hornbeck et al. 1986; Smith et al 1990; Hornbeck and Kochendeffer 1998, Thomas et al. 2010). We designed our study along a gradient of N deposition, but patterns of S and N deposition have highly concurrent patterns (Nelleman and Thomsen 2001; Sullivan et al. 2006; NADP 2010), so while our modeled gradient represents N deposition, the near 40 year period of sustained negative correlation and recent weakening of this relationship represents patterns in both S and N emissions. During stage III, SO₂ emissions began to steadily drop, largely a result of amendments to the Clean Air Act of 1970 and 1990. Emissions of NO_x remained fairly constant until 1990, when the EPA initiated controls on N emissions from electric utility sources (Driscoll et al. 2001).

Adams et al. (1985) examined tree rings from 190 *Picea rubens* individuals across West Virginia and Virginia and documented evidence of accelerated growth decline from 1961 until the end of their measurements in 1981, with similar trends reported for other regions of the eastern United States during the same period (Johnson and Siccama 1983; Adams et al 1985; Hornbeck et al. 1986; Smith et al 1990; Hornbeck and Kochendeffer 1998). This strong negative relationship in our dataset similar to that seen by Adams et al. (1985) holds constant from the 1970s until a weakening of the relationship since 2000, possibly suggesting stage IV, “red spruce recovery”.

Hornbeck and Kochenderfer (1998) documented stabilizing trends in *Picea rubens* growth in West Virginia and found evidence of increasing growth throughout the region since the 1990s. The authors do not suggest the cause of increased growth, which could be attributable to many factors, but they affirm that *Picea rubens* forests in the region are healthy and can be highly productive. Stage IV, “red spruce recovery”, in our dataset suggests a weakening of the relationship between deposition and growth and also highlights a period of increased growth among all of our *Picea* plots. While these results are preliminary, the noted decline similar to that seen by Adams et al. (1985), and the positive trends in BAI since 2000 similar to the stabilization noted by Hornbeck and Kochenderfer (1998) offer preliminary evidence of *Picea rubens* recovery throughout our study area. While this result merits further exploration, we conclude that the combination of tree-ring analyses and spatial deposition gradients provides the unique opportunity to not only assess growth responses to spatially-varying amounts of deposition, but to also characterize this response through time.

2.5 Conclusions

We show that tree ring analysis along environmental gradients can provide a valuable opportunity to examine how forests may be reacting to slow environmental changes, such as sustained inputs of atmospheric deposition. Being able to contextualize how systems are reacting through time to a naturally occurring range of disturbances provides very useful information on how ecosystems reacted in the past, how they are reacting in the present, and offers the ability to predict how they might react in the future. These promising results signify the need for future research examining the effects of deposition on forest growth. Savard et al. (2009) illustrated that the $\delta^{15}\text{N}$ stable isotope in annual growth rings may provide a potential tracer of N deposition, and the $\delta^{13}\text{C}$ stable isotope can be used to examine “industrial induced

physiological stress on trees” (Li et al. 2010). We suggest that our method can become even more valuable with careful site selection, the inclusion of a larger number of sample plots along a gradient, and analysis of tree ring isotopes.

Figures and Tables- Chapter 2

Figure 2.1. Study areas, with field plots shown: Central Appalachian Mountains (a), Adirondack State Park (b). At each plot location, 20 trees of the selected species were chosen for sampling.

Figure 2.2. Relationship between estimated N deposition and mineral horizon soil Pb.

Table 1. Study Sites

Plot	Latitude °N	Longitude °W	Elevation (m)	Est. N deposition (kg·ha ⁻¹ ·yr ⁻¹)	Trees sampled	Trees used	Mean Age (yrs)	Mean annual minimum temperature (°C)	Mean annual precipitation (mm)
Piru 1	38.981	-79.697	3750	9.72	20	16	104	2.39	1079.74
Piru 2	38.613	-79.840	4357	8.19	20	20	80	1.62	1379.10
Piru 3	38.198	-80.296	4296	6.75	20	18	107	1.20	1534.48
Piru 4	37.378	-80.514	3720	3.83	20	18	125	3.57	1124.40
Litu 1	39.054	-79.687	2480	6.95	20	19	91	2.46	1404.29
Litu 2	38.712	-79.898	2514	6.38	20	19	78	2.29	1327.87
Litu 3	38.238	-80.448	2924	5.74	20	20	83	3.83	1419.60
Litu 4	37.430	-80.507	2891	3.76	20	20	49	3.66	1251.22
Beal 1	43.737	-74.701	698	6.66	20	14	159	-1.96	1258.86
Beal 2	43.741	-74.460	668	6.15	20	19	50	-2.18	1290.30
Beal 3	43.975	-74.478	550	5.66	20	13	125	-1.41	1071.26
Beal 4	44.249	-74.236	510	4.90	20	16	97	-1.84	1003.90

Figure 2.3. Annual average BAI for *Betula alleghaniensis* plots along an atmospheric deposition gradient. As marked by arrows, a large windstorm and canopy blow-down event struck plots 1 and 3 in 1950 and was followed by rapid growth. These marked responses to disturbance confounded any pattern observable along the deposition gradient.

Figure 2.4. Annual average BAI for *Liriodendron tulipifera* plots along an atmospheric deposition gradient. Plot 4, the lowest deposition plot, established much later than the other plots. Large variation in stand history likely overwhelmed any detectable growth response along the deposition gradient.

Figure 2.5. Annual average BAI for *Picea rubens* plots along an atmospheric deposition gradient. Note the sustained drop in growth beginning around 1960, particularly within the high deposition plot between the 1950s and 1970s.

Figure 2.6. Relationship between mean annual N deposition and mean BAI since 1959 at Picea plots. We found a significant negative relationship between deposition and growth during this 50-year period.

Figure 2.7. Strength of the relationship (r) between N deposition and BAI in *Picea rubens* plots over the last 85 years broken into 4 stages: I) initial deposition; II) increase in deposition; III) sustained period of chronic deposition; and IV) red spruce recovery (a). Note a strong negative relationship from 1970 to 2000, following patterns and increases in deposition over the same period of time. Linear relationship between deposition and BAI during the initial deposition (b) and sustained deposition (c) stages.

Conclusions and Future Directions

The promising results seen in this study should be expanded upon in the future. In *Chapter 1*, we were able to demonstrate that mineral horizon soil Pb is a useful index of pollutant deposition in this region and provides a practical, surrogate measure of depositional patterns. Furthermore, we suggest dense networks of mineral soil Pb measurements can capture important spatial patterns of deposition and could be especially useful for understanding spatial patterns in ecosystem functioning within complex mountainous areas that are sparsely measured by national deposition monitoring networks. Currently, this project is being expanded by sampling soil Pb in locations unmeasured by NADP monitoring stations and by implementing a network of deposition collectors throughout the region to better understand patterns of deposition in the region

In *Chapter 2*, we were able to demonstrate that the use of tree-ring analyses in carefully selected plots along environmental gradients provides the unique opportunity to assess how forests are reacting to slow cumulative changes through time. We suggest that this method can become even more valuable with careful site selection, the inclusion of a larger number of sample plots along a gradient, and analysis of tree ring isotopes. We are currently pursuing all of these steps and are eager to investigate whether isotopes in the tree ring record indicate physiological stress in agreement with declining growth trends during the 20th century.

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Appendix 1: Total soil Pb and N deposition at all sample sites.

Site	Soil type	Total Pb content (mg/m ²)	N deposition (kg/ha/yr)
WV05	Organic	637.59	4.53
WV04	Organic	407.36	5.34
VA13	Organic	135.62	3.78
VA99*	Organic	345.55	n/a
VA28	Organic	465.64	4.71
MD08*	Organic	512.20	n/a
WS 4-2	Organic	341.22	7.86
WV18	Organic	341.22	6.31
W. Dep B*	Organic	1079.38	n/a
WV05	Mineral	544.05	4.53
WV04	Mineral	452.05	5.34
VA13	Mineral	214.92	3.78
VA99*	Mineral	464.84	n/a
VA28	Mineral	888.86	4.71
MD08*	Mineral	781.25	n/a
WS 4-2	Mineral	1206.63	7.86
WV18	Mineral	1206.63	6.31
W. Dep B*	Mineral	2425.25	n/a
WV05	O+M	1181.64	4.53
WV04	O+M	859.41	5.34
VA13	O+M	350.54	3.78
VA99*	O+M	810.39	n/a
VA28	O+M	1354.50	4.71
MD08*	O+M	1293.45	n/a
WS 4-2	O+M	1547.85	7.86
WV18	O+M	1547.85	6.31
W. Dep B*	O+M	3504.64	n/a

*Stations sampled, but excluded because not enough available N deposition data (N>10 years)

Appendix 2: Basal area increment (BAI) by plot – *Betula allegheniensis*

Year	Beal 4	Beal 3	Beal 2	Beal 1	Year	Beal 4	Beal 3	Beal 2	Beal 1
1850	487.9715			406.7623	1880	571.3974		353.5442	484.5715
1851	533.9408			426.0204	1881	557.9463		431.8492	440.902
1852	371.6958			336.1015	1882	471.4012		259.4989	411.9113
1853	474.4049			350.361	1883	630.3015		481.0272	412.8961
1854	344.0155			251.6403	1884	452.7823		483.1702	459.1495
1855	403.9635			225.1043	1885	551.9638		625.2843	479.7072
1856	400.011			207.907	1886	564.8903		650.6219	464.1633
1857	537.4144			241.137	1887	577.8022		745.6392	486.8492
1858	483.7535			221.8953	1888	652.4376		782.7911	436.8764
1859	402.8839			190.8393	1889	658.5776		930.987	474.3249
1860	427.9093		213.9126	238.6642	1890	740.2966		843.2436	513.185
1861	408.1244		275.8493	233.6273	1891	525.5767		696.8295	465.7058
1862	471.7189		377.7583	214.1452	1892	736.5625		880.2098	422.0502
1863	505.4935		315.0232	200.5295	1893	761.2766		701.2316	413.162
1864	537.2014		133.3422	217.5071	1894	722.2199		688.2274	464.0112
1865	491.8586		177.839	255.0607	1895	523.4561		574.9865	428.5744
1866	586.5046		178.5611	293.0867	1896	796.3667		717.8376	384.496
1867	601.9502		207.3324	354.8544	1897	593.193		436.5053	452.2176
1868	433.6607		210.8716	314.8931	1898	750.4801		664.4142	418.9113
1869	475.8691		320.7289	364.2888	1899	518.2415		524.1333	362.8855
1870	381.9306		274.3068	358.0415	1900	619.3892		433.9887	382.526
1871	595.0426		273.4702	276.5543	1901	664.1006		463.1439	361.549
1872	446.1149		213.9008	415.8244	1902	841.2574		374.878	348.1036
1873	745.4584		701.3796	512.9519	1903	946.9258		449.7503	418.5929
1874	671.1711		523.6781	466.397	1904	800.7643		473.3979	360.222
1875	770.0438		627.3671	550.5861	1905	683.7868		424.6464	422.833
1876	603.4313		512.9845	541.323	1906	767.2735		562.516	427.9119
1877	789.4576		488.436	494.3696	1907	677.4512		495.3535	351.804
1878	761.7242		413.7594	409.1185	1908	710.4052		656.3693	435.1987
1879	510.1555		293.5565	482.1269	1909	652.7735		598.085	345.4003

□

1910	735.573	560.7993	389.1854	1941	898.4095	1301.571	435.7596
1911	788.1879	504.6688	324.4184	1942	1033.108	1259.187	394.669
1912	703.0178	469.6344	336.8715	1943	1000.435	1332.782	456.9455
1913	681.9633	465.6169	425.1786	1944	1270.48	1267.092	404.4685
1914	705.7285	447.6289	338.6181	1945	1068.558	1162.849	477.6517
1915	975.7955	502.1374	333.6089	1946	993.7432	1225.133	478.5663
1916	1162.619	636.0435	359.2075	1947	1326.737	1399.431	557.8578
1917	1000.355	635.5065	401.4128	1948	923.3213	822.6239	558.273
1918	1042.775	593.2195	420.4586	1949	867.2874	1135.906	700.9338
1919	817.1192	567.7523	368.649	1950	829.0347	1394.469	832.3683
1920	710.9076	679.2583	317.0261	1951	1174.447	1571.914	1119.467
1921	633.4415	754.2192	352.2839	1952	1160.974	1557.546	1131.078
1922	811.4696	814.2657	314.2838	1953	656.3227	1264.731	1202.926
1923	777.4138	730.3147	361.2701	1954	801.2054	1354.838	1223.504
1924	1044.294	835.553	300.5087	1955	1064.987	1267.057	1285.518
1925	1102.285	903.4854	363.9593	1956	987.3913	1113.176	1244.2
1926	1121.765	902.6254	395.1055	1957	1076.486	1427.669	1087.698
1927	1179	1043.132	377.8615	1958	920.6345	1386.511	1021.214
1928	1077.478	1117.626	416.2567	1959	1131.197	1469.262	997.0701
1929	1073.101	1082.768	384.9365	1960	1358.634	1442.694	1025.82
1930	1102.581	1160.634	382.0377	1961	1204.894	1671.947	1148.784
1931	1059.108	1129.114	345.4007	1962	831.5193	1207.713	879.0523
1932	1037.799	1310.961	334.3497	1963	1165.054	1177.734	939.6099
1933	900.415	1102.769	374.1774	1964	1205.115	1708.803	944.7096
1934	689.8535	1006.543	420.8123	1965	1083.15	1314.606	879.9122
1935	804.9531	1063.783	342.5375	1966	691.535	963.7321	690.9445
1936	638.2159	831.6954	308.1266	1967	857.9445	999.1749	800.6462
1937	754.2302	920.4758	281.3932	1968	1056.028	1274.8	784.0495
1938	999.4142	1372.199	364.4704	1969	1050.807	1502.157	920.6131
1939	1061.425	1469.614	398.6902	1970	1130.04	1564.073	849.9722
1940	1018.441	1528.374	436.8017	1971	1024.137	1615.627	862.8606

□

1972	1243.324	1098.296	1820.071	992.2346	2000	1025.119	1190.16	1235.17	846.1401
1973	1255.251	1202.439	1551.391	1084.752	2001	1135.601	1183.329	1105.806	884.733
1974	1090.225	1100.351	953.1492	732.0999	2002	1032.627	1227.982	1002.582	909.6049
1975	1199.556	1102.785	1256.216	822.9923	2003	1026.915	779.7557	610.4751	829.8912
1976	968.9532	1138.268	1303.402	682.3528	2004	1673.621	1524.809	1138.987	1144.491
1977	1031.998	1361.947	1740.215	947.643	2005	1105.032	1196.913	1025.093	1096.481
1978	1144.588	1364.614	1956.784	959.8293	2006	948.3488	1257.185	950.7915	882.5083
1979	843.2404	1174.789	1432.893	817.8215	2007	1278.386	1347.803	1078.049	1094.063
1980	1219.957	1402.104	1817.957	858.6832	2008	1265.166	1552.359	1249.763	1121.914
1981	1239.303	1198.455	1778.85	870.7984					
1982	1542.491	1320.151	1766.122	1067.211					
1983	1199.026	893.6389	722.3945	723.1263					
1984	1261.394	1267.261	342.5219	701.1416					
1985	1262.193	1229.461	685.6601	895.0854					
1986	1364.843	1258.196	1501.623	1007.713					
1987	1175.913	1410.601	1484.317	899.2094					
1988	1212.969	1145.753	1677.554	757.7985					
1989	1421.21	1426.75	1953.734	781.7135					
1990	1402.74	1294.661	1900.664	886.0103					
1991	1270.11	1370.262	1901.074	891.7					
1992	1281.587	1206.608	1383.934	837.6057					
1993	1420.819	1327.563	1411.317	906.2404					
1994	1565.744	1511.532	1661.356	960.0193					
1995	1227.874	1295.387	1428.301	1043.623					
1996	1357.113	1462.744	1553.294	1144.117					
1997	1309.874	1308.51	1463.612	970.0242					
1998	1265.333	1578.411	1356.944	1044.047					
1999	1012.499	1368.274	1432.216	1068.126					

Appendix 2: Basal area increment (BAI) by plot – *Liriodendron tulipifera*

Year	Litu 1	Litu 2	Litu 3	Litu 4	Year	Litu 1	Litu 2	Litu 3	Litu 4
1916	948.5899	1737.296	403.7773	8155.932	1946	2166.481	1834.195	3211.391	4312.795
1917	842.4659	1289.477	462.7604	7119.145	1947	2181.797	1812.472	4034.355	5075.55
1918	769.6258	947.5823	289.8515	6123.15	1948	2048.196	1895.273	3620.747	5958.424
1919	1202.31	1110.285	420.1617	9497.059	1949	3029.796	2666.958	3983.566	7708.874
1920	1182.816	1042.246	670.9607	11340.75	1950	2414.847	2503.397	3287.499	6498.112
1921	1270.609	1192.978	1005.718	8325.424	1951	2258.01	2806.468	3336.376	6003.275
1922	1404.284	1414.73	1103.985	10793.81	1952	2501.275	2251.82	3802.206	2601.78
1923	1531.249	1356.707	1083.353	8120.802	1953	2230.19	2558.665	3928.819	2372.431
1924	1372.672	1500.533	1141.861	9268.596	1954	2290.374	1665.697	3593.643	2177.93
1925	1586.582	1070.744	460.7968	2393.136	1955	2527.896	1891.728	3697.47	2458.424
1926	1819.789	992.9696	1119.019	5963.84	1956	2273.379	2008.445	3685.185	2341.023
1927	1673.195	1251.452	1346.643	7265.985	1957	2787.371	2051.931	3502.195	2665.228
1928	1906.917	1982.588	1722.89	9632.601	1958	2484.871	2237.008	4046.089	3444.896
1929	1648.205	1585.512	1592.313	7288.725	1959	2475.308	1898.327	2933.215	2474.346
1930	1331.797	1115.653	787.3183	2795.838	1960	2269.132	2316.184	3446.893	2333.431
1931	1534.003	1457.264	1708.225	4701.065	1961	2304.592	2368.653	2162.431	3155.41
1932	1619.335	1689.204	2122.078	6713.248	1962	2012.265	1977.697	1696.544	3448.189
1933	1668.2	1879.166	2621.321	7381.428	1963	1871.898	1594.79	1646.515	2012.163
1934	1547.485	1511.382	2138.367	8858.484	1964	1923.914	1845.347	2691.829	2291.633
1935	2043.87	2553.717	2819.745	9256.121	1965	1752.153	1425.414	2479.773	2785.421
1936	682.6986	1356.534	1588.232	3150.419	1966	1336.137	657.8076	1678.872	1733.502
1937	1261.11	1754.588	2822.435	6240.969	1967	1841.521	1195.596	2209.107	2679.895
1938	1746.524	1917.92	2614.76	7843.787	1968	1760.738	771.0803	2353.233	4105.647
1939	1574.204	1775.051	2774.811	8300.893	1969	2240.513	1862.837	2803.107	3915.967
1940	1744.727	1494.477	3288.058	7253.98	1970	2217.32	2009.831	2943.63	3186.845
1941	1546.297	1586.148	2763.764	8158.672	1971	2645.359	2538.678	3361.949	4043.425
1942	2155.101	2742.722	4083.346	6764.54	1972	2698.418	2375.362	3131.201	4049.035
1943	2119.241	2278.057	3607.087	5184.193	1973	2565.386	3069.315	3023.05	4707.673
1944	2000.593	1553.67	2852.377	3385.794	1974	3011.463	3734.02	3202.125	4931.843
1945	2502.169	2071.975	3967.847	3837.861	1975	2728.741	3157.992	3247.061	4251.536

1976	2580.675	2898.254	2266.15	3948.252	2006	3166.257	3753.969	3286.601	5766.458
1977	1727.921	1960.34	743.611	3449.811	2007	3822.755	4266.319	2798.632	3661.015
1978	2595.138	3599.535	3323.698	3844.424	2008	3762.163	5001.733	3322.895	4557.615
1979	2590.191	2762.773	2531.021	3936.882	2009	2911.282	3683.946	2586.087	5345.569
1980	2360.294	2793.766	3511.335	4317.03					
1981	2390.222	2121.644	3119.612	4577.881					
1982	2577.628	2454.284	3025.528	4397.741					
1983	2689.14	3653.316	3448.714	4266.151					
1984	2390.804	2720.217	1807.532	3702.14					
1985	3449.352	3278.376	2504.263	4813.876					
1986	3870.486	3657.656	3239.318	4023.334					
1987	2637.832	3549.925	2670.488	4520.695					
1988	2023.62	2427.372	1715.966	3672.548					
1989	1748.846	3060.366	1788.696	4392.025					
1990	3661.236	5186.449	3554.426	4849.685					
1991	3010.741	3700.224	3487.879	4964.352					
1992	4474.324	5900.982	3752.589	5244.248					
1993	3577.175	2825.683	3541.31	4887.514					
1994	4251.064	3721.075	3270.486	4997.802					
1995	4270.76	4089.639	3520.884	6390.891					
1996	3055.151	3007.06	3024.608	4550.437					
1997	3824.022	3925.324	3133.183	3855.872					
1998	4123.021	4392.27	3692.612	4241.782					
1999	3370.692	2710.689	2734.794	3953.643					
2000	3249.994	2586.59	3527.444	5335.043					
2001	4523.121	3954.327	3169.014	5963.736					
2002	3634.507	3036.356	2169.036	3540.583					
2003	3758.295	4682.295	3550.64	6118.347					
2004	3816.41	4152.444	3699.981	5747.806					
2005	4115.804	4524.164	3596.782	5603.355					

Appendix 2: Basal area increment (BAI) by plot – *Picea rubens*

Year	Piru 1	Piru 2	Piru 3	Piru 4	Year	Piru 1	Piru 2	Piru 3	Piru 4
1858	237.5247		192.8892		1888	139.0968		160.2358	337.1082
1859	358.4757		231.441		1889	142.3386		215.2089	434.683
1860	302.8125		336.3735		1890	151.9482		189.6972	361.0435
1861	236.0585		332.6238		1891	172.6638		168.3018	382.1117
1862	210.3806		303.3131		1892	152.0672		183.7636	393.308
1863	160.3717		342.6356		1893	216.3549		175.6611	392.8636
1864	161.2331		256.7925		1894	247.7891		170.4529	327.0914
1865	208.7204		400.5382	517.7495	1895	312.5921		177.7871	317.1347
1866	228.2145		431.6647	446.5809	1896	281.2266		146.9943	386.6608
1867	231.8153		386.1049	501.0919	1897	593.1345		218.1834	444.381
1868	250.4348		469.5587	162.1557	1898	479.6167		173.3532	391.1989
1869	225.7435		445.0612	155.6149	1899	371.99		134.8846	453.9287
1870	248.596		416.2096	179.5807	1900	396.7148		164.9955	347.5423
1871	215.3232		429.8346	203.2709	1901	294.5447		147.7735	312.8945
1872	220.9016		418.3801	221.4409	1902	432.5325		149.5818	357.7073
1873	190.6384		383.437	412.4752	1903	421.9961		196.1954	302.5155
1874	162.3846		260.5337	316.6134	1904	348.3565		144.2692	273.0579
1875	140.2656		302.3682	286.2309	1905	384.5082		132.3229	266.1992
1876	194.8353		311.3547	260.1659	1906	400.2384		164.4873	358.5191
1877	173.2321		280.0751	235.3109	1907	442.6735		161.2988	310.1953
1878	169.338		369.9463	313.839	1908	503.5386		163.0709	339.3942
1879	143.206		307.7535	217.6275	1909	335.6364		158.8354	337.2904
1880	146.0687		291.3894	186.0016	1910	281.915		138.6021	327.9893
1881	138.8128		200.946	221.5609	1911	207.8332	484.7905	98.05197	320.8541
1882	113.2653		247.2139	218.7742	1912	290.0467	404.1169	127.9928	373.5276
1883	108.8845		278.9688	201.9115	1913	426.8084	400.4777	222.9187	372.4868
1884	134.754		287.7379	305.1129	1914	279.1701	665.0585	198.962	317.1231
1885	147.4752		273.3115	261.8761	1915	403.8409	757.9934	276.1403	301.0213
1886	190.8073		264.0107	305.8728	1916	469.6238	149.8716	441.8913	308.5904
1887	111.1893		218.0459	268.8011	1917	571.9751	216.5676	432.3143	283.7019

1918	665.9132	537.6475	447.2249	243.3446	1948	1365.962	732.7602	1882.417	1148.398
1919	655.7423	956.5003	520.435	321.6313	1949	1660.482	881.3585	1921.742	1175.329
1920	908.8404	1107.106	782.4148	380.6808	1950	1707.528	942.2412	2043.233	1363.03
1921	913.8353	910.0478	785.7821	334.8395	1951	1739.991	1005.333	1859.889	1328.774
1922	1078.121	885.7248	1205.52	352.7851	1952	1558.112	1025.07	1899.852	1325.955
1923	906.8611	708.5046	1071.142	348.5154	1953	1190.702	1031.025	1642.38	1218.452
1924	923.5486	641.4898	1280.457	427.4173	1954	1239.241	945.1423	1629.297	1233.739
1925	888.3994	443.0735	1207.039	517.1652	1955	1396.9	931.8461	1844.415	1209.287
1926	942.9572	547.176	1188.538	553.2124	1956	1276.148	967.3053	1818.667	1313.229
1927	975.8779	549.7659	1179.249	595.0249	1957	1609.781	1033.053	2364.324	1630.813
1928	1074.512	622.5472	1520.334	711.5763	1958	1524.069	987.0521	1902.825	1820.701
1929	1217.898	606.611	1823.413	786.7713	1959	1675.377	840.1961	2220.649	1895.274
1930	911.9433	475.3114	1111.823	652.6021	1960	1571.44	898.0474	1760.433	1818.955
1931	1042.002	652.2319	1496.738	691.3906	1961	1419.189	805.5594	1720.022	1746.45
1932	1166.029	802.0061	1609.433	664.7234	1962	1277.234	859.8898	1586.749	1601.972
1933	1173.865	662.4728	2033.619	725.5839	1963	1482.921	926.3526	1750.722	1806.39
1934	1261.086	688.1077	1829.002	740.1473	1964	1263.696	815.6204	1530.603	1615.425
1935	1271.475	849.0084	2156.632	879.0142	1965	1209.139	839.1968	1478.864	1613.542
1936	1286.22	883.1945	1800.54	882.904	1966	1058.568	568.8399	1280.743	1412.89
1937	1324.002	949.4875	1816.092	808.549	1967	944.8303	649.663	1180.762	1506.838
1938	1502.548	851.7794	2406.249	885.17	1968	1137.201	612.4869	1602.034	1678.582
1939	1558.738	916.3483	2251.621	903.3992	1969	859.2615	608.1181	1290.196	1671.661
1940	1534.245	873.041	2150.032	877.7208	1970	666.3762	791.4534	1467.781	1642.62
1941	1566.144	862.7663	2130.935	926.3176	1971	636.4679	880.7816	1347.082	1679.576
1942	1551.165	897.0595	2065.541	1035.606	1972	543.901	705.3672	1348.627	1460.778
1943	1316.686	753.2177	1801.802	925.0586	1973	721.6861	843.1663	1360.44	1607.91
1944	1153.659	807.8417	1476.143	942.6383	1974	511.6325	872.0204	1396.74	1796.806
1945	1413.796	842.7812	2017.535	843.634	1975	609.0441	945.7047	1523.346	1781.699
1946	1283.132	634.5557	1732.265	1023.649	1976	655.9519	1119.958	1631.82	2152.421
1947	1284.308	837.2224	1867.433	1055.599	1977	601.7561	807.8627	1129.111	2164.195
1948	1365.962	732.7602	1882.417	1148.398	1978	575.9844	798.1786	1140.027	2331.669

1979	541.6123	849.1362	966.966	1845.122
1980	714.2337	1028.295	1257.224	2385.929
1981	645.3999	950.1472	905.5981	2272.585
1982	698.1926	1112.777	1063.906	2263.675
1983	746.3167	1160.391	1260.302	2220.92
1984	570.152	949.2101	742.5664	1677.023
1985	776.0232	1070.78	1076.769	1852.829
1986	958.2654	1119.843	1374.053	2051.59
1987	996.3127	1083.655	1280.084	2072.086
1988	680.8718	783.1489	799.5407	2044.07
1989	629.349	813.0597	887.1854	2255.809
1990	836.0301	903.1935	1157.905	2113.576
1991	1111.379	881.9474	1418.422	1831.365
1992	1033.814	1024.505	1457.119	1870.554
1993	971.4563	1170.046	1623.484	2381.946
1994	882.2298	967.2679	1569.182	2234.086
1995	1002.908	1068.371	1741.324	2460.692
1996	680.4857	710.4321	1273.815	2237.712
1997	973.0635	861.1056	1552.63	2773.537
1998	702.0421	922.0876	1473.031	2789.572
1999	1087.981	900.764	1852.929	2613.009
2000	1312.884	873.4082	1987.245	2641.767
2001	1423.505	929.3874	1898.327	2909.44
2002	1676.387	989.5618	2284.802	2873.049
2003	1417.109	1027.015	1908.842	2600.893
2004	1426.878	1029.22	2102.69	3113.072
2005	1625.152	1253.745	2564.368	3031.298
2006	1136.056	1066.333	1940.132	2751.43
2007	1737.673	1123.039	2330.883	2741.617
2008	1522.854	968.4374	1996.587	2414.47
2009	1630.251	859.0519	1984.23	2010.514