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## **EVALUATING THE LONG-TERM EFFECTS OF LOGGING RESIDUE REMOVALS IN GREAT LAKES ASPEN FORESTS**

Michael I. Premer

*Michigan Technological University, miprem@mtu.edu*

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EVALUATING THE LONG-TERM EFFECTS OF  
LOGGING RESIDUE REMOVALS  
IN GREAT LAKES ASPEN FORESTS

By

Michael I. Premer

A DISSERTATION

Submitted in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

In Forest Science

MICHIGAN TECHNOLOGICAL UNIVERSITY

2015

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This dissertation has been approved in partial fulfillment of the requirements for the Degree of DOCTOR OF PHILOSOPHY in Forest Science.

School of Forest Resources and Environmental Science

Dissertation Co-Advisor: *Dr. Robert E. Froese*

Dissertation Co-Advisor: *Dr. Linda M. Nagel*

Committee Member: *Dr. Evan S. Kane*

Committee Member: *Dr. Gary A. Campbell*

School Dean: *Dr. Terry Sharik*

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## Preface

At the time of this writing, no chapter or material presented here has been accepted or published under any other copyright. Authorship and collaboration for each of the chapters as articles is listed:

Chapter 2, *Management strategies for northern temperate forest carbon stocks and implications for site productivity* - all analysis, figures, and writing performed by Michael Premer. Some direction and ideas were contributed by Robert Froese and Evan Kane.

Chapter 3, *Assessment of aspen forest productivity and stand dynamics under intensive harvesting regimes* - all analysis, figures, and writing performed by Michael Premer. Direction, editing, and revisions contributed by Robert Froese and Linda Nagel.

Chapter 4, *Logging residue removals leads to increased structural heterogeneity of forest understory vegetation communities*- all analysis, figures, and writing performed by Michael Premer. Direction, editing, and revisions contributed by Robert Froese, Linda Nagel, and Christopher Webster.

Chapter 5, *Incidental effects of contemporary harvest systems on stand regeneration* – all analysis, figures, and writing performed by Michael Premer. Direction, editing, and revisions contributed by Robert Froese and Linda Nagel.

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## Abstract

Commercial aspen (*Populus* spp.) forests of the Great Lakes region are primarily managed for timber products such as pulp fiber and panel board, but logging residues (topwood and non-merchantable bolewood) are potentially important for utilization in the bioenergy market. In some regions, pulp and paper mills already utilize residues as fuel in combustion for heat and electricity, and progressive energy policies will likely cause an increase in biomass feedstock demand. The effects of removing residues, which have a comparatively high concentration of macronutrients, is poorly understood when evaluating long-term site productivity, future timber yields, plant diversity, stand dynamics, and consequently, appropriate silviculture.

These practices were evaluated throughout the western Upper Peninsula of Michigan using forest inventory data from harvested stands over the last 45 years. Assessment provided a framework of ecosystem carbon patterns across stand rotation, and comparisons reflected moderate losses in residue-bound carbon pools when captured. However, residue loads exceeded those recommended by established residue retention guidelines even in stands where residues were recovered for bioenergy. Soil nutrient pools exhibited declines under residue removal treatments on coarse textured soils. Aboveground biomass and timber yields did not vary according to residue treatment, and development of the aspen cohort was similar across all soil types, regardless of nutrient levels, an unexpected result. Stand dynamics varied by soil productivity class in the late stages of stand rotation, and imply that encouragement of non-crop trees for ecological purposes does not affect yields of the aspen resource. Predictions based upon historical growth and yield models showed mixed levels of accuracy when compared to observations. Analyses of vegetation communities and multivariate ordination methods revealed temporal patterns in species richness and increased heterogeneity under residue removal treatments. A spatially explicit sampling design was utilized to examine advances in harvesting technology on variance in stand level regeneration, soil nutrients, and residue estimates. Operator and equipment induced patterns in harvesting activity were found to increase residue loads with a concomitant decline in stem density and height growth across a large portion of the regenerating stand.

Collectively, these findings suggest that residue removal results in a minor decline in aboveground C stocks, and coarse textured soils may be susceptible to reductions in forest soil nutrient pools. Further, variations in growing conditions at the forest floor due to high residue loads may adversely affect understory vegetation communities. Finally, harvesting equipment, cut-block layout, and residue distribution can influence stand regeneration patterns, and warrants consideration in harvest and silvicultural planning.



# Chapter 1

## Introduction

The work presented here utilizes a quantitative systems approach to assess aspen dominated ecosystem response to intensive harvesting regimes. Globally, forest resources are pressured by increased demands to provide sustainable long-term timber utilization, optimal carbon storage, and bioenergy feedstocks, while balancing ecosystem function and site productivity. Forests have long been recognized for their timber commodity resources, however only in the past few decades has forest ecosystem function been thoroughly assessed through advancing studies of ecophysiology and nutrient cycling dynamics. In light of these findings, foresters are developing silvicultural prescriptions which aim to mimic natural disturbance regimes to promote heterogeneous stand composition and structural components (Frelich, 2008; Puettman et al., 2009). Assessing the balance between increased utilization and preserving ecosystem structure and resilience is the principle component of successful contemporary forest management regimes.

The potential effects of increased atmospheric greenhouse gasses on ecosystem responses has been extensively covered and assessed in numerous meta-analyses and case studies (IPCC, 1997; Anderson-Teixeria and DeLucia, 2010; WRI, 2010). These reports propose a variety of management strategies for terrestrial carbon sequestration and energy feedstock substitution. In the United States, initial actions to address atmospheric CO<sub>2</sub> were implemented through cap and trade programs, with expanding interests in alternative energy feed stocks including wind, geothermal, solar, and biomass (Cabral, 2008). The variety of alternative fuel sources has indirectly created a regional approach to energy policies and production. In the Great Lakes region, utility providers have been required to supply a minimum of 10% of renewable energy sources in retail supply portfolios, and it is likely that there will be more aggressive alternative energy policies in the future (ACORE, 2012; PA 295, 2008).

Forest harvest residues are a readily available source of bioenergy feedstock as timber harvests are continuously occurring on private, state, and federal forest lands in the northern Great Lakes region. Residues include; (1) small diameter trees; (2) topwood – (fine limbs and branches < 10 cm small end diameter), not adequate for pulpwood production and (3) non-merchantable cull trees (Perala, 1990; MDNR, 2010). Residues have traditionally been retained on the forest site after felling, either scattered throughout the forest matrix and on skid trails or at the harvest landing. In some regions, pulp and paper mills already utilize these byproducts as fuel in combustion for heat and electricity. It has been well documented that harvest residues contain a high concentration of macronutrients when compared to the merchantable bolewood associated with a sawtimber or pulpwood harvest (Bartos and Johnson, 1978; Alban, 1985; Ruark and Bockheim, 1987). Therefore, large scale removals are of primary concern when assessing long-term site productivity and ecological sustainability of these practices.

For the needs of this study, it is imperative to establish a distinction between logging residues, which are a byproduct of forest harvesting, and what is commonly referred to in the literature as coarse woody debris (CWD) or down dead wood (DDW). This distinction is necessary as CWD/DDW that is present prior to harvest occurs through stand development processes and is not utilized as biomass due to; decreases in mass resulting from decay and decomposition processes, inefficient methods of collection, and low economic incentive for recovery given high water content. Residues and DDW differ in their methods of input, temporal distribution and spatial arrangement in the forest stand, decomposition rates, and volume (Harmon et al., 1986). Following harvest operations, incidental breakage during felling and whole-tree skidding results in scattered fine woody debris (FWD) throughout the stand, and is uncollectable by machinery given their dispersed pattern.

DDW will be referred to as the woody material originating during forest stand development and disturbances. These processes include: self-thinning, competition and successional mortality, diseased and dead trees, and wind throw/snap. These inputs result in a random spatial distribution at ranging levels of decomposition and are generally characteristic of a forest stand in demographic transition or relative old-growth conditions (Harmon et al., 1986; Bergeron, 2000; Frelich, 2008). The importance of DDW in ecological function has been illustrated through roles as long-term nutrient sinks, seedbed substrates, regulators in sediment transports, and structural influence on microenvironments and wildlife habitat (Harmon et al., 1986; Currie et al., 2003; Sun et al., 2004; Janowiak and Webster, 2010).

Harvest residues are deposited into the forest matrix at times of forest harvesting, the spatial arrangement and volume being determined by the silvicultural treatment, harvest type (clearcut/selection/geometric thinning), and operating equipment. Due to the inherent variability of soil types, forest cover and composition, nutrient cycling regimes, and management practices, any potential increase in residue extraction warrants further evaluation of site productivity and stand dynamics.

Ongoing research evaluating forest response to residue removal in the Scandinavian boreal forest offers limited long-term results. Progressive energy policies in Finland and Norway require electrical providers in the region to double the utilization of biofuels by year 2020, the majority of which is expected to be derived from forest industry biomass (Helmisaari, 2011). Major findings in these research programs suggest that residue removals result in a decrease in basal area increment (Engell, 1997; Helmisaari, 2011) and slight reductions in mineral soil carbon and macronutrients (Tamminen et. al, 2012). However, drawing broad conclusions from these findings warrants caution, as the boreal forest type and associated nutrient inputs, rates of productivity, and management regimes are inherently different than those in the North American temperate forests. Whole-tree harvesting and residue removal effects in the Great Lakes temperate forests have been investigated in the past (Lenz et al., 1984; Hendrickson et al., 1989). However, these studies have also been limited in temporal scope (Hendrickson et al., 1989) or

focused on forest site response to conversion from northern hardwood types to artificial stocking of *Pinus resinosa* and *Larix decidua* x *leptolepis* (Mroz et al., 1989).

The North American Long Term Soil Productivity (LTSP) was established to assess intensive forest management (increased compaction levels and organic matter removal) on long-term trends in soil nutrient pools and forest productivity. However, at this time the LTSP is limited to less than 20 years (approximately half of a commercial aspen rotation) and the treatments are highly controlled and deliberately more severe than operational impacts. While this program is invaluable to advancing our knowledge of management influences on forest productivity, any observed trends are unlikely to be indicative of large scale commercial management practices. Therefore, expanded investigation of long-term ecosystem response to varying levels of harvest intensity under realistic and practical management scenarios is warranted.

Aspen forests have historically been an expansive cover type across the Great Lakes region. Intensive harvesting and subsequent wildfires at the end of the 19th century initiated favorable germination conditions reflective of the species disturbance regimes and increased its geographical extent (Whitney, 1987). Today, aspen has the largest distribution of any species in North America, with approximately 5 million hectares across Michigan, Minnesota, and Wisconsin (FIA, 2013). In the absence of management or disturbance, aspen stand breakup sets the successional pathway for understory reinitiation and movement toward mid-late successional communities, including, *Acer-Tsuga-Dropteris* and *Acer-Tsuga-Mainthemum* in the northern Lake states (Coffman, 1984; Oliver and Larson, 1996).

Commercial aspen stands are harvested on a 40-year rotation, set by the culmination of mean annual increment. Silvicultural methods use even-aged clearcutting with coppice regeneration, utilizing the prolific vegetative reproductive mechanism that allows aspen to maintain site dominance (Bates et al., 1989; Perala, 1990). The stocking density of regenerating aspen suckers is proportionally dependent upon the pre-disturbance stand basal area (Greene and Johnson, 1999; Prévost and Pothier, 2003) and studies have reported as many as 25-100k stems ha<sup>-1</sup> (Zasada, 2001). Actively managed aspen forests in the region are primarily utilized for pulp and panel board products. However, harvest residues have been utilized at a minor scale in the past for heat and electricity co-generation in local pulp fiber mills. The abundance of aspen forests in the region under commercial management and readily available source of harvest residues offer potential feed stock to the growing bioenergy market.

Timber harvesting methods have continuously evolved in the Great Lakes region. Historically, timber was harvested using hand-felling, de-limbing, and bucking in the woods, while skidding was done by horse drawn wagon to a central landing, with non-merchantable timber left at the site of felling. Advances in harvesting technology initiated the use of chainsaw felling and topping with tractors to whole-tree skid merchantable stringer-length logs to the slasher for bucking. Innovation in machine capabilities has given way to high production timber harvesting equipment. Although

various methods are employed elsewhere in North American harvest operations, contemporary methods in the Great Lakes region utilize whole-tree skid or cut-to-length ground systems. These methods differ in their operations, stand and silvicultural applications, and in the subsequent distribution of residues.

This study leverages a historical database of stand-level residue management prescriptions on forest land originally held by the Mead Corporation and currently owned and managed by Plum Creek Timber Company. This approach offers an exclusive appraisal of residue recovery on long-term site patterns and productivity. The four parts of this work reflect stand productivity according to various scales and processes, and address intensive management effects on aspen forests through intricately related components of a dynamic ecosystem. These chapters include (i) evaluation of forest carbon storage and soil nutrient capital; (ii) a comparison of timber yields and stand dynamics under residue management prescriptions with a critical review of traditional growth and yield studies; (iii) assessments of vegetation communities using biodiversity and community heterogeneity as metrics of ecosystem resilience, and; (iv) a quantitative analysis of harvesting systems and impacts on stand regeneration.

It was expected that residue removal would result in a decline in ecosystem C storage and site nutrient capital, due to the disproportionate levels of macronutrient content. This was further expected to result in a reduction of forest productivity and aboveground yield estimates. I predicted that the removal of residues and associated nutrients would be reflected by a decrease in vegetation biodiversity and homogenization of community structure. The influence of harvest machinery on stand regeneration patterns was fortuitously discovered while collecting stand measurements, yet was expected to have a distinctive pattern on spatial heterogeneity of stand regeneration.

## Chapter 2 <sup>1</sup>

# Management strategies for northern temperate forest carbon stocks and implications for site productivity

## 2.1 Introduction

The continual increase in atmospheric CO<sub>2</sub> levels from fossil fuel combustion and land cover and land use change has prompted research investigation to identify alternative sources for energy production while maximizing terrestrial carbon levels. As forested ecosystems account a substantial amount of global C storage and cycling through vegetation and soil components (Chapin et al., 2011), these systems have been the primary focus of management efforts to achieve reductions in greenhouse gas fluxes and as a source of biofuels (Schlamadinger and Marland, 1996; Reijnderes and Huilbregts, 2003; Lal, 2004; Canadell and Raupach, 2008). However, increased pressures on forest resources could pose threats to long-term site productivity and ecosystem resilience.

The feasibility of forest derived biomass as a sustainable fuel source has been proposed under a range of scenarios. It is generally acknowledged that due to contemporary processing technology, there is less energy produced per unit of woody biomass than when compared with fossil fuel sources (Manomet, 2010; Mitchell et al., 2012). These studies further advance the concept of a ‘carbon debt’ to signify the temporal lag between the utilization of forest biomass and the recovery time required to replenish the aboveground stock. The time to repay the carbon debt varies appropriately with the scale being considered (stand/landscape) and whether the biomass is considered a direct substitute for fossil fuels. The source of forest biomass feedstock has been approached with contention. Several studies have inferred that forest bioenergy feed stocks will be derived through conversion of natural forest land to monoculture plantations based on short rotation systems (Schulze et al., 2012), or forest land currently managed for commodity supply products (Janowiak and Webster, 2010). However, these are unlikely scenarios due to current market demands for higher valued timber products such as pulp fiber and saw logs when compared to biomass prices (MDNR, 2015).

Efforts to enhance utilization of forest biomass as alternative fuel feedstock are paralleled, and sometimes contested, by efforts to optimize aboveground biomass for carbon sequestration through afforestation of cropland (Righelato and Spracklen, 2007) or maximizing standing tree carbon stocks (Ryan et al., 2010; Woodall et al., 2011). These approaches are perhaps a preferable scenario in short term planning (< 30 years), however largely overlook the specificity of site and biomass productivity (Marland and Schlamadinger, 1997). Further, increasing aboveground biomass stocks on forest land

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<sup>1</sup> The material contained in this chapter is in preparation to be submitted to *Canadian Journal of Forest Research*

may have indirect consequences including an increased susceptibility to disease and forest health issues and increased fuel loads.

Forest harvest residues (< 10 cm. diameter, comprised of fine shoots, buds, and leaves) are a byproduct during operational harvesting and have been largely unutilized in the past due to product specification standards in pulp fiber supply. Harvest residues are readily available in forested landscapes under active management and represent a minor proportion of ecosystem C when compared with live tree C. Residues may offer a readily available source to bioenergy production and a viable compromise between the “to sink or burn?” competing doctrines of forest biomass feedstocks or terrestrial carbon sequestration reservoirs (Kirschbaum, 2003).

Although the removal of forest residues as a bioenergy feedstock has the potential to be C neutral (Schlamadinger and Marland, 1996; Mitchell et al. 2012), these portions of aboveground biomass contain a disproportionate amount of nutrients critical to plant growth when compared with the traditionally merchantable portion of the stem (Alban, 1985; Barnes et al., 1998). Therefore, residue recovery may result in reductions soil nutrient pools, causing a decline in aboveground timber yields and threatening long-term site productivity. Silkworth and Grigal (1982) proposed that Ca removals from whole-tree harvesting in aspen forests would result in catastrophic losses exceeding atmospheric and weathering inputs far into the future, with pools unable to recover to pre-harvest conditions. Further, soil compaction indirectly caused by residue removal through additional passes from machinery could increase soil bulk density, thereby decreasing macropore space and reducing root respiration and water infiltration rates, ultimately altering soil nutrient content and ion exchange (Powers 1999; Fisher and Binkley, 2000).

In boreal forest plantation systems, removal of harvest residues has been noted to reduce aboveground productivity on subsequent rotations (Walmsley et al., 2009; Helmisaari, 2011) and declines in soil nutrient pools (Tamminen et. al, 2012). In the Great Lakes region, short-term studies have reported no effect of residue removals on aboveground biomass accumulation (Alban and Perala, 1989), minor reductions in exchangeable nutrient pools (Voldseth et al., 2011), and mixed results in soil carbon (Nave et al., 2010). Recently published meta analyses have identified the need for evaluation of long-term impacts of residue removals, as short-term responses are not consistently indicative of long-term patterns (Gollany et al., 2015).

To address these gaps in knowledge and limited temporal scope of past investigations, this study leverages a detailed historical harvesting database originally under ownership by the Mead Corporation, and now held by Plum Creek Timber Company. In the last 45 years, harvest residues have been utilized at a minor scale for heat and electricity co-generation at a local pulp fiber mill in Escanaba, Michigan, and allows for a unique perspective to compare long-term effects of stands with and without residue removals. Any trends detected from this study may serve as a baseline when assessing large-scale removals of harvest residues.

The objectives of this chapter are to: (i) examine temporal patterns in aboveground carbon in aspen forests by stand compartment under residue removal treatments and across varying levels of soil productivity and; (ii) determine the effects of residue removals on total and exchangeable soil macronutrients across the commercial aspen rotation. Forest stands located on soils with highest estimated productivity levels are likely to exhibit the greatest temporal increase in aboveground C components, however that residue removals would result in a decrease of these pools, and that these losses would be most severe on stands located on lower soil productivity classes. Further residue removals are expected to result in a decrease of forest soil nutrients, and coarse textured soils will likely exhibit the most severe losses.

## 2.2 Methods

### 2.2.1 Experimental design and site descriptions

To test these hypotheses, I utilized a database provided by Plum Creek Timber Company to identify aspen forest stands of interest to include in the sampling frame. To be included for consideration, stands were required to be; (i) classified as aspen forest cover type; (ii) containing a minimum stand area of 10 ha ; (iii) contain detailed records of harvest residue treatment (Whole-tree harvest – WTH, or Stem-only harvest – SOH), and; (iv) year of harvest. Once these criteria were met, I randomly selected stands from five-year age classes across the commercial rotation (40 years) for a maximum possible of 8 stands per treatment type.

I attempted to replicate this sampling design across a variety of productivity classes to represent the range of soil types that aspen forests occupy in the region under industrial ownership in the Lake States. Using this approach, I was able to establish a comparison of residue treatments across the entirety of the aspen rotation with a chronosequence design along several soil types and associated productivity levels. Site Index (SI<sub>50</sub>) was estimated for all soil types from NRCS records combined with models developed by Carmean (1979) to serve as a proxy for soil productivity levels, and ranged from 17.7-23.2m. The Onaway soil type (*Inceptic Hapludalf*) allowed for the most robust sampling intensity with almost a complete replication of the chronosequence (SOH, n = 14; WTH, n = 15). I included 7 (SOH, n = 4; WTH, n = 3), stands along the Charlevoix soil type (*Argic Endoquod*), followed by the Ensley Angelica (*Aeric Endoaquept*) and Rubicon (*Entic Haplorthod*), which each offered a total of 5 stands for sampling (SOH, n = 3; WTH, n = 2).

Stands included in the sampling frame are located in Baraga, Delta, Dickinson, and Menominee Counties of the Upper Peninsula of Michigan, and are owned and actively managed by Plum Creek Timber Company. These stands are managed with coppice regeneration methods and clearcut on a 40-year commercial rotation. Mean air temperatures in the study area range from -13° C to 24° C in January and July, respectively, with an average of 724 mm precipitation, most of which is received during

the growing season. Stands are located on soils that originated from glacial retreat during the Wisconsin age, and are comprised of moraines and outwash plains (NRCS, 2013). Stands are primarily comprised of *Populus tremuloides* (Michx.), with minor occurrences of *Populus grandidentata* (Michx.) and *Populus balsamifera* (L.) on the highlands and lowlands, respectively. Other species, including *Abies balsamea* (L.), *Picea glauca* (Moench.), *Quercus rubra* (L.), *Acer saccharum* (Marshall), and *Acer rubrum* (L.) are commonly found mixed in the mid and understory, however contribute to only a minor portion of stand level basal area (< 15 %). Common understory species include *Pteridium aquilinum* (L.), *Corylus cornuta* (Marshall), *Maianthemum canadense* (Desf.), *Prunus pennsylvanica* (L.), *Eurybia macrophylla* (L.), *Trientalis borealis* (Raf.), *Anemone quinquefolia* (L.), *Rubus idaeus* (L.) and *Dryopteris carthusiana* (Vill.).

### 2.2.2 Field measurements and calculations

Field measurements were conducted during the growing season from 2011-2013 across a total of 46 aspen stands using a fixed area nested plot design. I used a two-stage sampling design with overstory vegetation measurements taken from late-May to early-July and forest floor and soil samples collected from late-July to early-September to avoid bias in estimates due to seasonal uptake of nutrients early in the growing season. A total of 10 fixed area plots were randomly established in each of the forest stands using ESRI ArcMap 10.1 random point generator, and located in the field using a handheld Global Positioning System receiver.

At each site, soil pedons were dug to confirm characteristics of soils of interest. At each of the 10 sampling plots (0.04 ha), overstory trees ( $\geq 10$  cm at 1.37 m) were recorded by species to the nearest 0.1 cm. A total of four nested regeneration surveys (0.001 ha) were established in each cardinal direction from the overstory plot center, where understory trees ( $\leq 9.9$  cm at 1.37 m) were recorded by species and diameter at dbh. Standing dead trees ( $> 1.37$  m in height) were recorded by dbh and height within the overstory plot and identified by species where applicable. I used species specific taper equations to estimate top diameter of each snag included in the plot (Weiskittel and Li, 2012), and used Smalian's formula to determine snag volume. To estimate snag biomass, I used standing dead tree biomass reduction factors according to equations by Domke et al. (2011).

At the center of each overstory plot, I used four Line Intersect Distance Sampling (LIDS) transects, extending in cardinal directions with a limiting distance of  $k = 800$  to include harvest residues and down dead wood (DDW) (Affleck, 2008). Each particle counted in the LIDS tally was recorded by decay class and species (where applicable) and biomass estimates were adjusted by corresponding reduction factors (Woodall and Monleon, 2008). All stumps ( $> 1.37$  m in height) were recorded within a 0.01 ha subplot centered by overstory plot center. Stumps were measured for height, with two perpendicular measurements taken at the top of the stump and averaged for top diameter. I used Raile's (1982) equations to estimate stump volume, then calculated stump biomass according to methods proposed by Woodall et al. (2011).



Forest floor samples were collected within a 0.000025 ha frame placed immediately adjacent to the north and south borders of each overstory plot, for a total of 20 forest floor samples per stand. Herbaceous and shrubby plants, along with fine woody debris (< 2.54 cm) and detritus were removed from the forest floor to expose the sapric layer of the organic soil layer. Once the organic soil was exposed, I used an AMS slide hammer to extract organic and mineral soil to a depth of 30 cm following procedures by Jurgensen et al. (1977), for a total of 20 organic and mineral soil samples per stand.

### *2.2.3 Laboratory Processing*

Soils collected in the field were kept on ice until returned to the forest soils laboratory at Michigan Tech. All leaf litter samples and soils were dried at 65° C for a total of 48 hours or longer to ensure a consistent dry weight. Soils were sieved using a 2 mm screen to separate all roots, rocks, and buried wood. Roots were weighed separately. Sieved soils were split multiple times and a subsample was ball milled to a consistent texture. Total soil carbon and nitrogen was determined using a Fisons NA 1500 Elemental Analyzer. I chose to test for total soil N rather than exchangeable as total estimates provide better insight of potential nutrient supply given than exchangeable pools, given rapid cycling and immobilization rates, and the relatively small proportion of soil N that is available for plant uptake (Fisher and Binkley, 2000; Brady and Weil, 2008). I used a 1M reagent of NH<sub>4</sub>CL with a 10:1 dilution factor to determine exchangeable Ca, K, and Mg which was processed with an Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES).

### *2.2.4 Statistical Analysis*

I conducted all statistical analysis in the R software programming environment (R Core Team, 2015). To compare temporal patterns of aboveground components and soil variables under the residue treatments and across the aspen rotation of the varying soil types, I used mixed effects analysis of covariance and linear and non-linear mixed effects models. This approach was taken due to the hierarchical study design, and implemented with the *nlme* package (Pinheiro et al., 2014) with restricted maximum likelihood (Kenward and Roger, 1997). Differences in treatment and soil type were considered statistically significant at  $\alpha = 0.05$ . I used the B.C. Ministry of Forests Catalogue of Curves (Sit and Poulin-Costello, 1994) to identify candidate models for non-linear temporal patterns in stand level carbon by individual compartment, and fit the equations with the *nls* function and *growthmodels* package (Perez, 2013) to estimate model coefficients.

To adjust for heteroskedasticity and adhere to assumptions of normality, I performed log transformations on estimates of stumps, snag, and residue carbon. When models failed to detect a difference in variables by soil or treatment type, observations were pooled to estimate temporal patterns. I assessed residue removal treatment on each soil type

individually with ANCOVA due to the hierarchical study design and inherent differences in texture and therefore nutrient holding capacity between soil types.

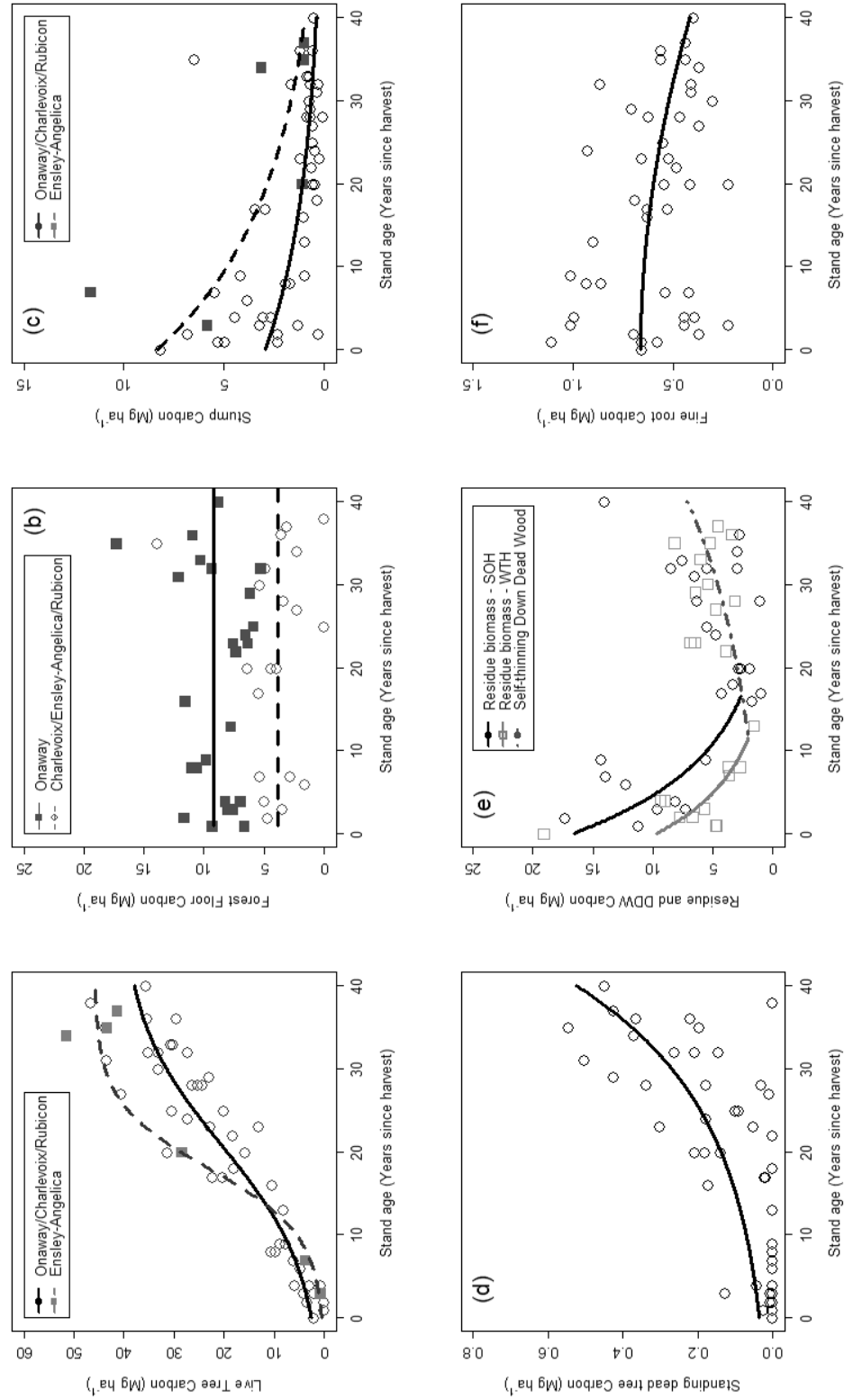
## 2.3 Results

### 2.3.1 Temporal patterns in aboveground forest carbon pools

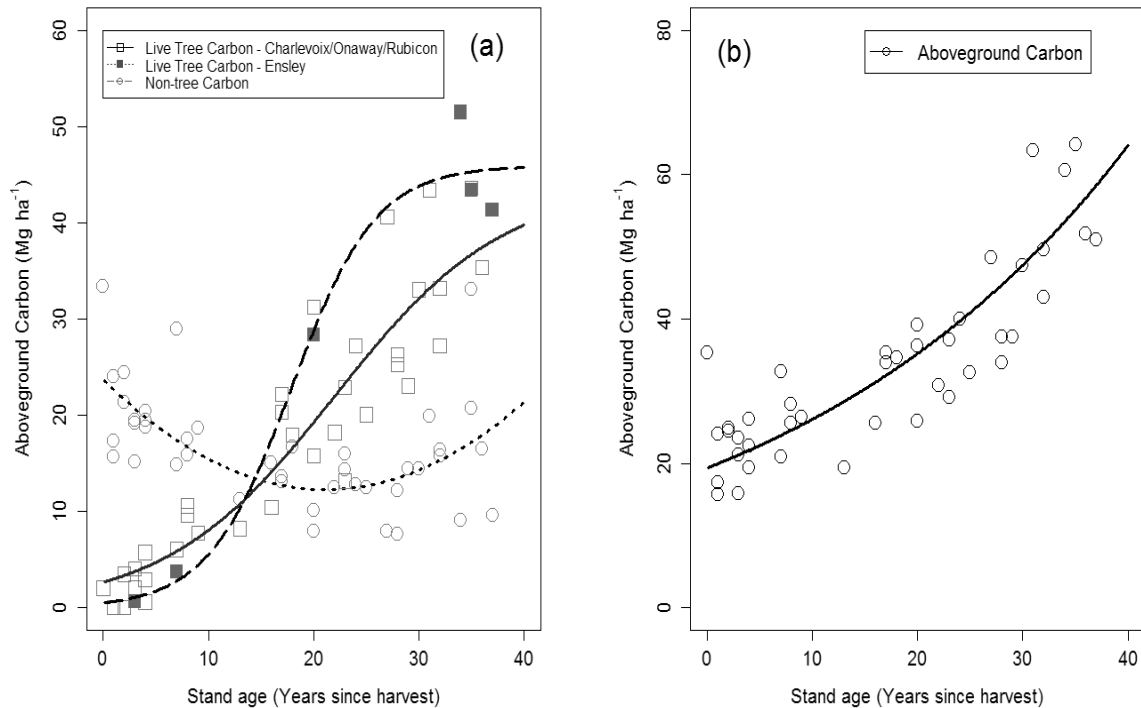
Temporal patterns were detected in most aboveground carbon stand components; further, there were several differences according to soil productivity class and residue treatment. A residue treatment effect was not detected on live tree carbon on any of the included soils at stand initiation ( $p = 0.126$ ), yet there were strong differences attributable to an age and soil type interaction ( $p = 0.012$ ). Stands located on the Ensley-Angelica soil complex exhibited highest levels of live tree carbon through the second half of the chronosequence, with a mean value of  $45.74 \text{ Mg ha}^{-1}$ , although there was no difference in the remaining soil types at  $41.03 \text{ Mg ha}^{-1}$ . Temporal patterns of live tree C on all soil types were best fit with a Chapman Richards function (Figure 2.1a.). Forest floor detritus was higher in the Onaway soil type at  $9.19 \text{ Mg ha}^{-1}$  than all other soil types at  $4.65 \text{ Mg ha}^{-1}$  ( $p = 0.035$ ); however tests failed to detect a temporal effect.

Temporal trends in stump carbon were fit with a logarithmic function, and showed a strong decrease with time since harvest ( $p < 0.001$ ), with the Ensley-Angelica exhibiting the highest mean values predicted at  $8.13 \text{ Mg ha}^{-1}$  compared with all other soil types at  $2.94 \text{ Mg ha}^{-1}$  ( $p = 0.027$ ) immediately post-harvest. These values correspond with the observed trends in live tree carbon in Figure 2.1a, and suggest that the Ensley-Angelica soil type maintains the highest live tree carbon compared with included textural classes. Trends in standing dead tree carbon increased with stand age ( $p < 0.001$ ) and exhibited no difference by either residue treatment or soil type ( $p = 0.225$ ) (Figure 2.1d).

Estimates of mean harvest residues were the only stand component where a treatment effect was observed. Tests failed to detect a difference in soil type ( $p = 0.335$ ), so observations were pooled across soils according to treatment type. At the stand level, estimates suggested a decrease from  $16.58 \text{ Mg ha}^{-1}$  in the SOH treatment to  $9.67 \text{ Mg ha}^{-1}$  ( $p = 0.004$ ), or roughly a 42 % decline (Figure 2.1 e) immediately post-harvest. Results suggest that decomposition of harvest residues on the SOH stands are more rapid than those on the WTH treatments ( $p = 0.008$ ), and the temporal residence of residues among both treatment type appears to be limited to approximately 15 years. After this point, there is no difference in residue pool estimates between residue treatment types ( $p = 0.256$ ) (Figure 2.1e). There was no effect of residue treatment ( $p = 0.59$ ) on fine root carbon, yet results suggest a moderate effect of stand age ( $p = 0.05$ ). Results indicated a decline in fine root carbon through stand rotation, and the pattern was best described with a quadratic function according to model residuals and model comparisons using the whole-model comparison anova function in R.



**Figure 2.1.** Results of Carbon storage by stand compartment across stand rotation. Top panels exhibit (a) Live tree; (b) Forest floor, and; (c) Stump Carbon, lower panels illustrate temporal trends in (d) Standing dead trees; (e) Residue and DDW biomass, and; (f) Fine root Carbon



**Figure 2.2.** Stand level estimates of Non-timber Carbon across pools and Live-tree Carbon (a), and; Total aboveground Carbon through stand rotation (b)

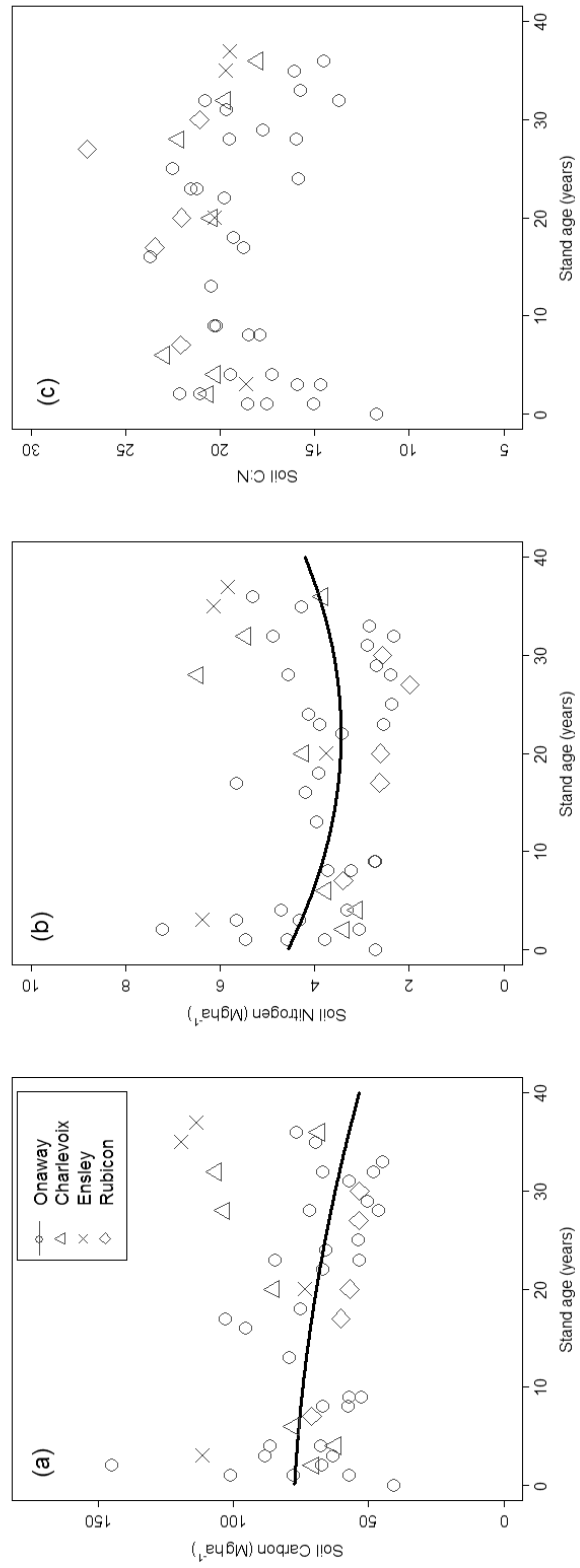
Combining all non-tree carbon pools (residues/DDW, stumps, standing dead trees, forest floor, and fine roots) resulted in post-harvest aboveground carbon levels at approximately 23.65 Mg ha<sup>-1</sup>, which decreases until roughly 22 years, when levels begin to increase. Temporal patterns of aboveground non-timber carbon were fit with a quadratic function (Figure 2.2a). When these values were added to live-tree carbon, observations followed an exponential pattern across stand rotation, and resulted in a predicted total of approximately 64.01 Mg ha<sup>-1</sup> by year 40 (Figure 2.2b).

### 2.3.2 Forest Soils

Total soil carbon and nitrogen varied widely across all of the included soil types. Estimated levels of soil C immediately post-harvest ranged from 69.61 to 98.77 Mg ha<sup>-1</sup> from the Charlevoix to Ensley, respectively. There were no statistical differences in total soil carbon between the soil types ( $p > 0.193$ ) (Figure 2.3a), nor was there a difference in treatment type in any of the included soils ( $p > 0.417$ ). Although estimates of the of soil carbon exhibited high variance across taxonomic classes, there was a significant decline in forest soil C carbon on the Onaway soil type ( $p = 0.022$ ). Estimates of total soil nitrogen ranged from 3.31 Mg ha<sup>-1</sup> in the Charlevoix soil types to 5.66 Mg ha<sup>-1</sup> in the Ensley-Angelica complex. Similar to trends in soil carbon, estimates of soil nitrogen

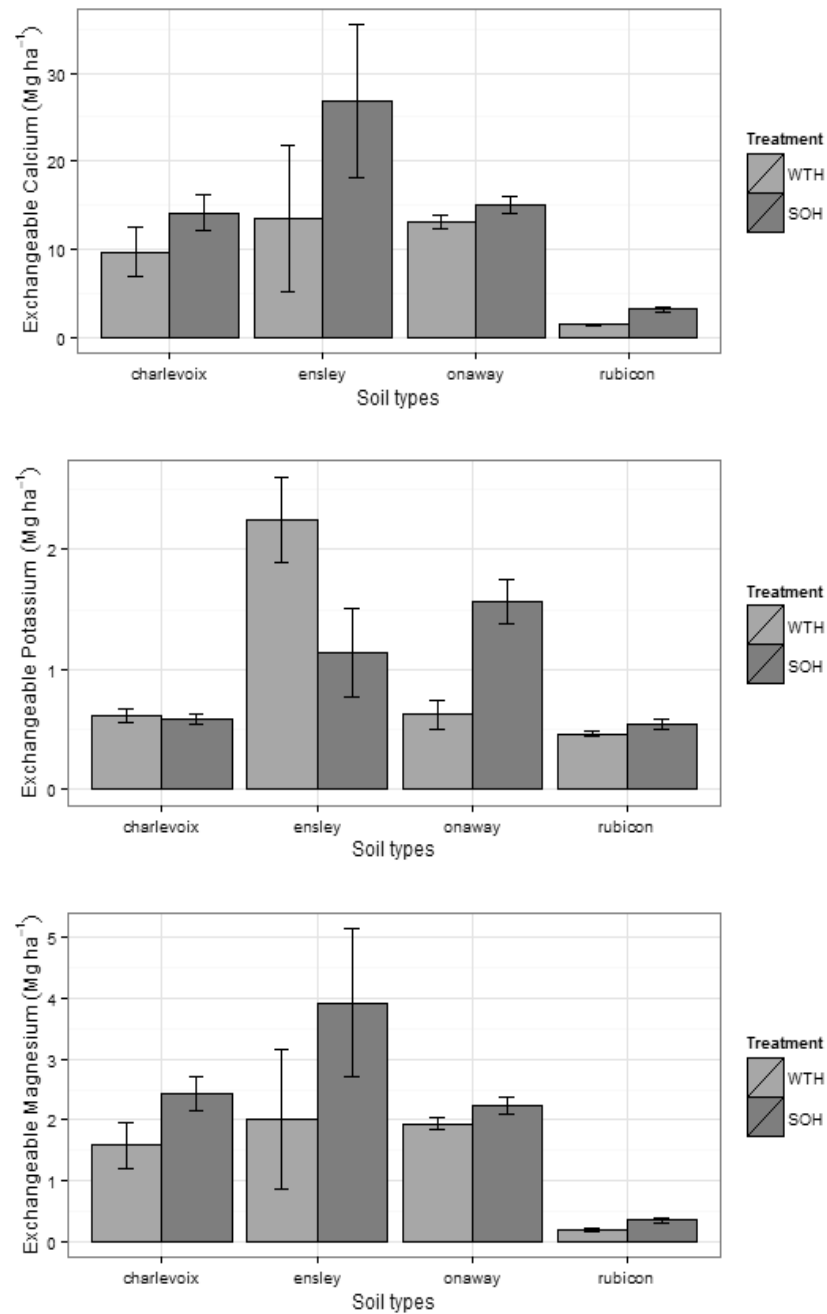
showed high variance within soil groups and therefore tests revealed weak evidence for differences attributable to soil group ( $p > 0.074$ ). Further, differences in N were not significant according to residue treatment in any of the tested soil types ( $p = 0.145$ ). Again, temporal effects on soil N was limited to the Onaway soil type, which showed highest levels immediately post-harvest and declined with stand age ( $p = 0.011$ ) (Figure 2.3b). Estimates of soil C:N across all soil types exhibited high variance (Figure 2.3c). The Ensley-Angelica exhibited the lowest values of C:N at 18.96, while the Rubicon exhibited the highest values at 21.965, however these were not statistically different ( $p = 0.101$ ).

Forest soil nutrients (Ca, K, and Mg) varied according to soil productivity class and residue treatment type (Figure 2.4), however there was no effect of stand age on nutrient loads on any of the included soil types ( $p > 0.648$ ). The Rubicon soil type showed significant declines from  $3.15 \text{ Mg ha}^{-1}$  to  $1.44 \text{ Mg ha}^{-1}$ , between the SOH and WTH stands, respectively ( $p = 0.021$ ), resulting in approximately a 55% loss. Trends in other soil macronutrients were highly variable across textural classes and residue treatment types, which masked any residue treatment effect.



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**Figure 2.3** Stand level mean estimates of forest soil total Carbon (a) Nitrogen (b), and C:N ratios across stand rotation and by soil type. Regression lines indicate temporal trends on the Onaway soil type. Other soil types exhibited no temporal patterns.



**Figure 2.4** Soil exchangeable cation by soil and treatment type. Top panel illustrates Exchangeable Calcium, and middle and lower panels exhibit Exchangeable Potassium and Magnesium, respectively. Error bars are the standard error across all stands within the soil and treatment category. Asterisks (\*) indicate significant differences at  $\alpha = 0.05$ .

## 2.4 Discussion

### 2.4.1 Aboveground forest carbon pools

The temporal trends reported here are typical of stand development patterns across a range of forested ecosystems. The Ensley-Angelica soil type contained the highest levels of aboveground live tree and stump C through stand rotation. When these findings are coupled with findings from Chapter 3, results suggest that the promotion of non-crop species can increase stand-level live tree C stocks has no detrimental impact on the development of the aspen cohort or timber yields, contrary to findings by Roth (2012). Therefore, live tree retention and protection of advance regeneration during harvesting activities should be considered if maximizing live tree C pools are a management priority.

The utilization of logging residues has been speculated to lead to a complete removal of coarse and fine woody debris pools (Janowiak and Webster, 2010; Berger et al., 2013). However, these and other studies commonly overlook harvesting equipment utilized during operations, as well as the minimal economic incentive for logging contractors to collect harvest residues given current market demand for biofuels. In commercial forest settings, comparing residue loads on stands under SOH to WTH methods have been noted to be highly variable. Rittenhouse et al. (2012) and Klockow et al. (2013), determined that residue removals resulted in declines of approximately 30-35% of fine woody debris pools in northern Wisconsin and Minnesota, respectively, which are slightly lower than the estimated 42% reduction presented here. It should be noted, however, that despite a lack of statistical evidence, fine woody debris pools in WTH systems reported by Rittenhouse et al., (2012) in Marinette County, WI, were indeed higher than those occurring on stands with residue retention. Similarly, observations in this study exhibited highest stand level estimates of residue-bound C in the WTH stands immediately post-harvest (Figure 2.1e.), and reflects high variability in residue loads. The reduction of roughly 36% of residues across this and the aforementioned studies are likely due to several factors, including; (i) high levels of naturally occurring DDW through self-thinning in stand development; (ii) incidental breakage during timber harvesting operations, and; (iii) inefficient collection methods of harvesting equipment. Therefore, substantial levels of residues remain on site after WTH operations.

As harvest residues have gained renewed interest as large scale biofuel feedstock, it is imperative to assess trade-offs between their potential as an alternative fuel source and role in forest C sequestration pools by examining the temporal dynamics of decomposition rates and fate of residue-bound C. Previous work in Great Lakes aspen forests reported rapid declines in slash C following harvest, with roughly a 50% reduction by 5-9 years post-harvest (Alban and Perala, 1989; Alban and Perala, 1992; Alban and Pastor 1993). Observations reported here support and extend these trends, with residue levels quickly decreasing and reaching minimum values at ages 12-15 (Figure 2.1e), therefore the ecological importance of residues may be minor (Johnson and Todd, 1998) .



At the stand level, it appears that residues and total non-timber C pools decline in their relative contribution to aboveground C as the stand ages up to 12-15 years post-harvest. At this point, live-tree biomass begins to be the dominant contributor to ecosystem C (Figure 2.2a). This trend continues for approximately 8-10 years, until stand-level growth of live-trees begins to decrease due to increasing individual stem mortality and incremental growth of residual stems (Assman, 1970), and results in increased levels of DDW.

#### 2.4.2 Forest soils

Forest soil C estimates were highly variable however levels corresponded with soil texture and taxonomic class. This high variability masked temporal effects, if any, across soil types with the exception of the Onaway, which exhibited highest soil C values immediately post-harvest and a slight decline across stand rotation. Meta analyses of harvesting effects on soil C have reported reductions in C levels immediately following harvests (Johnson and Curtis, 2001), however Nave et al. (2010) report that changes are minimal and largely dependent upon soil taxonomic class. Previous work conducted in Great Lakes aspen forests have shown mixed results in total soil C immediately after timber harvesting, however; trends of increased soil C reported by Ruark and Bockheim (1988) and Alban and Perala (1992) corroborate with levels detected in this study. Johnson and Curtis (2001) suggest that gains in soil C can be largely attributable to the addition of slash to the mineral soil following harvesting. While this proposition is feasible, it is reasonable to expect to find increases in soil C that would parallel estimates in residue loads. Therefore, harvest residues that have not been fragmented by harvest machine passes and directly incorporated into the forest floor could simply decompose in place and release residue-bound C back into the atmosphere. Therefore, the fate of C stored in harvest residues deserves additional investigation.

The role of fine and coarse root turnover is largely overlooked in carbon accounting studies and their contribution to forest soils post-harvest; however these pools contain a notable proportion of stand level C (Hendrickson et al., 1989; Steele et al., 1997). Live root biomass of aspen trees have been estimated to be approximately 20% of aboveground totals (Ruark and Bockheim, 1988; Jenkins et al., 2003) at the time of harvest. Using live tree biomass estimates from this study, approximately 8.21 – 9.15 C Mg ha<sup>-1</sup> of coarse roots are present directly post-harvest within the forest soil. Despite heavy regeneration density within the first two years post-harvest, aspen sucker leaf area mass and photosynthesis is far inadequate to support the root systems of parent trees (Zasada et al., 2001; Des Rochers and Lieffers, 2009), and root biomass has been suggested to decrease up to approximately age 13. Therefore, given the shallow rooting characteristics of *Populus tremuloides*, these patterns could in part explain the lack of treatment difference in this study (Hendrickson et al., 1989), and further supports the need to be cognizant of this often neglected contributor to soil C pools.

Unlike patterns exhibited in soil C, Nitrogen estimates shift on the Onaway soil type through stand rotation (Figure 2.3). High levels of total soil N early in stand

development likely reflect increased decomposition of detritus and aspen root tissues and declines in tree uptake. The frequent occurrence and high density of *Rubus* spp. and *Prunus pensylvanica* in the post-harvest understory community have been noted to accumulate a disproportionate amount of N in perennial tissues (Mou et al., 1993), and act as a buffer against nutrients lost through leaching losses following forest disturbance (Outcalt and White, 1981). The annual turnover of *Rubus* spp. and other herbaceous species coupled with the suppression of *Prunus pensylvanica* and shade intolerant shrubs provides N to the regenerating aspen suckers which quickly reoccupy the site. As stand density levels of the aspen cohort reach maximum levels, N is quickly relocated from the mineral soils to the perennial tissues of the aspen overstory, and results in a decline in total soil N at approximately 20 years post-harvest (Ruark and Bockheim, 1988). After this point, the decline in stem density through self-thinning reduces vegetative competition for soil N with additional inputs from annual litter fall, which incrementally increases N levels through the second half of stand rotation.

It is important to note that despite the vast differences in soil macronutrient pools across soil types, trends in aboveground biomass were similar regardless of soil texture and productivity class, with the exception of the Ensley/Angelica, which exhibited highest levels due to the occurrence of shade tolerant conifers. Forest soil macronutrient pools are dependent upon a balance of inputs from precipitation, weathering, and vegetative decomposition and outputs from harvesting and leaching losses. Early studies assessing nutrient pools under whole-tree harvesting systems of aspen forests cautioned against excessive topwood removals that could deplete cation reserves and accelerate acidification of soils. Boyle et al. (1973) hypothesized that nutrient pools would not be limiting to aboveground yields until approximately nine-30 year rotations (270 years), caused by inadequate amounts of Ca. Similar predictions were made by Silkworth and Grigal (1982), who were unable to determine the recovery time of Ca pools under whole-tree harvesting methods.

Results from this study partially support these predictions, as the Rubicon soil type exhibited reductions of approximately 54% in soil exchangeable Ca, and coarse textured soils have been suggested to be most susceptible to nutrient depletions through topwood removals (Thiffault et al., 2011). These patterns support the hypotheses that coarse textured soils would exhibit the greatest reduction in nutrient pools from residue removal due to low cation exchange capacity and high macropore space, yet these reductions did not appear to impact forest productivity. It is difficult to compare the impact of residue removal in other soil types due to the high variance encountered in estimates, especially in the Ensley-Angelica, which contains higher mean estimates of C and all measured nutrients than other included soil types. It is noteworthy that when comparing aspen growth across the wide range of nutrient levels, ranging from the lowest reported values found on Rubicon soil type to the highest on the Ensley-Angelica; there were no detectable differences in aboveground yields. While results contend that residue removals may not affect aspen growth even on those nutrient poor coarse textured soils, these reductions may have unforeseeable effects on soil nutrient capital and long-term site productivity.

## 2.5 Conclusions

The utilization of forest derived woody biomass is dependent upon energy policies, timber markets, harvesting methods, and variations in forest productivity which require a site-specific approach when analyzing potential outcomes of wide scale bioenergy production. Efforts to maximize C sequestration in living trees appear to have no effect on the productivity of forest yields and supply to commodity markets; therefore future efforts can aim to increase live tree C across a broad range of soil types and inherent productivity under aspen cover. Bradford and Kastendick (2010) propose that extending aspen stand rotations beyond the current ages of 40 years can increase terrestrial C stores at a higher rate than other forest types. Results from this work suggest that harvest residues may be a viable option for biofuel feedstock. Given the limited temporal residence and relatively minor decreases in residue loads under current WTH practices, it is fair to expect minimal time for C debt repayment under the conceptual bioenergy production model proposed by Mitchell et al. (2012). However, equipment collection efficiency and transportation distance of residues remain critical components to achieving the broader goal of carbon neutrality given their incremental contribution to CO<sub>2</sub> emissions.

Although there is a potential resource for woody biomass procurement and utilization in the Lake States through residue utilization, it should be kept in mind that large-scale removals could affect long-term productivity of these forest stands through the removal of nutrient rich residues, but are largely cover type and site dependent. Results reported here suggest that residue removals can impact macronutrient levels, therefore detrimental losses to soil nutrient pools might be ameliorated by alternating rotations with topwood removal. It is vital for foresters and land managers to approach the effect of residue removals as site specific, as in some locations fluctuating water tables can be sources of soil nutrient replenishment (Trettin et al., 2011). Overall, the effect of residue utilization on site productivity is largely dependent upon silvicultural program and is site specific.

## Chapter 3 <sup>2</sup>

### Assessment of aspen forest productivity and stand dynamics under intensive harvesting regimes

#### 3.1 Introduction

Predictive models of growth and yield production are critical to forest management especially when stand development is influenced by silvicultural activities that alter stand conditions and development trajectory. Efforts to optimize stand production to meet long-term objectives and provide products to commodity markets are commonly attained through forest site preparation and competition control (Sajdak, 1982; Wagner et al., 2006), selection of the level of removal at harvest (Rittenhouse et al., 2012), and control of stand densities and thinning regimes (Wyckoff and Lauer, 2014). Influences of genetic characteristics in planting stock and retained individuals coupled with varying levels of residual density and changes in site quality have variable effects on forest stand development and production (Smith et al., 1996; Frelich, 2002; Aubry et al., 2009).

In even aged systems, stand development and size density relationships are often quantitatively represented with the self-thinning rule and stand density index metrics (Reineke, 1933; Yoda et al., 1963; Pretzsch, 2010). These are characterized by decreasing stand density levels with concurrent increases in individual stem volume and crown growth through time (Smith and Hann, 1986). The temporal lag between culminations at the stand level prior to individual stem growth has been the primary economic consideration of density management regimes (Langsaeter, 1941; Long, 1985). These patterns are correlated to rates of canopy closure and full site foliar occupancy which is maintained in equilibrium by crown growth of surviving trees along with recruitment of individuals from the under and midstory (Long and Smith, 1984; Smith and Long, 2001). While these processes are continuous, viewing them in the context of discrete stages aids in interpreting ecological dynamics and can greatly assist in management activities. Attempts to represent patterns within this conceptual framework in a managed forest setting were initially taken by Borman and Likens (1979) who characterized stand development by the accumulation of aboveground biomass in deciduous hardwoods of New England. These concepts were extended to examine changes in physiological community structure (Long and Smith, 1984; Oliver and Larson, 1996) with additional work emphasizing site partitioning based on disturbance regimes (Frelich, 2002). It is generally recognized that site quality affects the rate of these processes (Assman, 1970); however, the effect of management practices can exacerbate or counteract these patterns and have important implications for further silvicultural intervention.

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<sup>2</sup> The material contained in this chapter is in preparation to be submitted to *Forestry*

The recent demands on global forests to provide traditional and emerging products, such as bioenergy feedstock, are paralleled by increasing expectations to provide ecosystem goods and services and concerns over forest landscape fragmentation. These factors have increased the use of intensive management regimes coupled with third party certification standards (Forest Stewardship Council – FSC; Sustainable Forestry Initiative – SFI) in industrial forests worldwide to meet these growing demands, with research programs initiated to investigate long-term forest productivity (Vance, 2010). Further increases in forest resource demand will likely result in increased prioritization for intensification on industrial forest lands rather than designated conservation and preservation areas, and warrants an examination of global forest conditions. Particularly, empirical growth and yield models may not be able to reflect changes in management intensity, and warrant re-evaluation and continued research.

The aspen forest cover type is circumboreal, with common genera and species across the North American and Eurasian continents (Larsen, 1980). Aspen is found in temperate, boreal, and mountainous regional forest types in North America (SAF cover types 16-217-251). In the Great Lakes region, aspen forests cover approximately 5.2 million ha, most commonly under industrial ownership, and are characteristic of an early seral community (Barnes et al., 1998; Domke, 2010; FIA, 2013). Silvicultural regimes have been driven by the regional pulp fiber market and rely on coppice regeneration techniques on a 40-yr biological rotation, utilizing the prolific vegetative reproduction strategies of aspen across the expansive range of soil types that the species is found to occupy (Perala, 1990). Any further increase in management intensity is likely to happen first on industrially managed aspen forests given their abundance, cost effective regeneration characteristics, and the relatively greater importance attached to financial considerations in this ownership class.

Forest mensuration research of this regional forest type was initiated by Kittredge and Gevorkiants (1929), who assessed fire-origin aspen stands on the Chippewa National Forest, Minnesota to quantify the vast aspen resource at the turn of the 20<sup>th</sup> century. This initial research quantified aspen productivity across a wide range of soil types through establishing Site Index (SI<sub>50</sub>) relationships, growth and yield volume predictions on a variety of merchantability standards, and provided stand density recommendations. With some minor modifications, these models have been the accepted paradigm for aspen forest management in the Great Lakes region. Lundgren and Dolid (1970) refined original SI<sub>50</sub> equations to model trends in stand mean height growth, while Schlaegel (1971) expanded the original models to include stand density (m<sup>2</sup> ha<sup>-1</sup>) as an additional variable to predict stand yield. Ek and Brodie (1974) updated these models to estimate future stand volume according to contemporary product standards, and Perala et al. (1996) established a multi-product yield model for *Populus* spp. across North America and Scandinavia based on the original work of Kittredge and Gevorkiants (1929). The use of these models in contemporary management warrants assessment given noted sub-regional differences in aspen species (Berrang et al., 2003; Rock et al., 2007), continually increasing minimum temperatures since model establishment (Allen et al., 2010), and

potential influences of repeated rotations using coppice regeneration silviculture (Frey, 2003).

Due to conventional and current product standards, harvest residues (topwood  $\leq 10$  cm diameter) on managed aspen forests in the region have generally been retained in the forest matrix post-harvest. However, some forest stands under industrial ownership have removed harvest residues in addition to conventional pulp fiber at a minor scale in the past 40 years for heat and electricity co-generation at local production mills. The removal of harvest residues, which contain a disproportionate concentration of stand level macronutrients (Alban, 1985) could result in a decrease of site nutrient pools and threaten long-term productivity. The degrees to which residues have been utilized largely depend on transportation distance and utility markets (K. Weyers and G. Wyckoff - Plum Creek Timber Company, personal correspondence). These levels of management intensity across the wide range of forest soil types and associated productivity levels that aspen occupies allows for a unique perspective to evaluate and quantify stand development patterns and long-term production. Any observed patterns may serve as a baseline when assessing this forest type for continued and potentially enhanced management scenarios given evolving demands on forest resources coupled with unknown future climatic and growing conditions.

The objectives of this study were to examine aspen forest growth patterns using a stand development framework and traditional growth and yield models after 40 years of intensive management across site productivity levels. Stand development patterns were expected to parallel the site productivity estimates; soil types with higher estimated  $SI_{50}$  would exhibit accelerated recruitment of competitors and higher density levels of non-aspen crop trees compared with less productive sites. It was further anticipated that stands with residues removed would exhibit a decrease in growth and yield trajectories, and that this pattern would become more pronounced on sites with lower estimated productivity. Finally, predictions based on historical growth and yield models were anticipated to be inaccurate beyond the established threshold of indifference of the equivalence test.

## **3.2 Methods**

### *3.2.1 Study area*

Aspen forest stands used in this study are located in Baraga, Delta, Dickinson and Menominee counties in the Upper Peninsula of Michigan, and are owned and managed by Plum Creek Timber Company. These stands have remained under the aspen cover type since extensive logging of the region of the mid 1800's to early 1900's, as a result of how aspen responds to disturbance and the production oriented management history of the land base. The study area has a cool continental climate with mean air temperatures reaching  $-13^{\circ}\text{C}$  in January to  $24^{\circ}\text{C}$  in July with an average of 724 mm of annual precipitation, most of which is received during the growing season. Physiographic

characteristics of the study region include ground moraines and outwash plains, originating from glacial retreat in the Wisconsin age (NRCS, 2013).

Aspen stands in the study area are primarily composed of *Populus tremuloides* (Michx.), with mixed occurrences of *Populus balsamifera* (L.) and *Populus grandidentata* (Michx.), in the fine and coarse textured soils, respectively. Silvicultural practices in these forest types rely on coppice regeneration methods to re-establish stands following harvest, with occasional single/patch tree retention of advance regeneration of sub and non-merchantable species. These commonly include *Abies balsamea* (L.), *Picea glauca* (Moench.), *Quercus rubra* (L.), *Acer saccharum* (Marshall), and *Acer rubrum* (L.). These trees are deliberately retained to adhere to third-party certification (SFI) guidelines for biological legacy retention and wildlife considerations, or indirectly due to harvesting operator behavior and mill demand. Harvest residues (branches and tops < 10 cm) and cull wood have been utilized on a small scale in some of these forest stands in the last 45 years for heat and electricity co-generation at a local pulp fiber production mill.

### 3.2.2 Study design

Forest stands were systematically identified from an inventory database provided by Plum Creek and that met specific criteria: classified as aspen forest cover type; records indicating harvest residue treatment (Stem Only Harvest [SOH]/Whole-Tree Harvest [WTH]) and year of harvest operation, and; a minimum of 10 ha in area. Stands were further classified by major soil type, with the most common types identified as the focus of this research. Stands were then randomly selected from each soil group and binned into five year age classes to form an observational chronosequence across commercial stand rotation and according to soil type and residue treatment (Table 3.1).

### 3.2.3 Site measurements and calculations

Measurements were conducted on a total of 480 sample plots in 48 aspen stands using a fixed area nested plot design. A total of 10 plots were randomly established in each stand of interest using the random point generator in ESRI Arc Map 10.1 (ESRI, 2012), restricted to a minimum distance of 25 m from stand edges and between sampling points. Each plot consisted of a 0.04 ha fixed area plot for overstory measurements, with four nested 0.001 ha regeneration plots established at a distance of 5.2 m from plot center in each cardinal direction. Plots were located in the field using a Garmin E76 Trex Global Positioning System; with soil pedons confirmed using diagnostic criteria of soils of interest (NRCS, 2012). At each of the ten 0.04 ha overstory plots, all live trees ( $\geq 10$  cm diameter at breast height [dbh]) were identified and measured for species, dbh (cm), and crown class (Smith et al., 1997). A subsample of each aspen crown class was randomly selected within each overstory plot for additional measurement of total height and length of crown. Saplings ( $\leq 9.9$  cm dbh) were tallied by species, height and dbh using a digital caliper, while all regeneration < 1.3 m in height was tallied by species and measured by height within the four nested regeneration plots. Height was recorded to the nearest 0.3 m on regeneration saplings and seedlings with a height pole.

### 3.2.4 Statistical Analysis

All analysis and graphing was conducted in the R software environment (R Core Team, 2012). I used the *nlme* package (Pinheiro et al., 2014) to examine the effect of soil type and stand age (years) on stand development patterns. Estimates were conducted at the plot level and analyzed using mixed effects restricted maximum likelihood regression and analysis of covariance (ANCOVA) models to account for hierarchical study design. Reineke's Stand Density Index (SDI) was calculated and used as an additional metric to characterize site occupancy and development stage across rotation by soil and treatment types, as suggested by Jack and Long (2006).

I used Sit and Costello's (1994) Catalogue of Curves to identify candidate models for fitting observations. The *growthmodels* package (Perez, 2013) was used to estimate non-linear regression models to observed patterns in stand level basal area (BA) ( $\text{m}^2$ ), quadratic mean diameter-(QMD)(cm), height (m), stand density (stems  $\text{ha}^{-1}$ ), and SDI by aspen and non-aspen tree species along soil types through the commercial rotation.

A conceptual stand development framework (Borman and Likens, 1979; Long and Smith, 1984; Oliver and Larson, 1996; Frelich, 2002) was employed to interpret temporal and structural patterns of the aspen cohort and competing tree species. Importance values (mean of relative BA and relative density) were calculated for each species to assess site partitioning by silvics. I assessed the effects of harvest residue removals on rates of forest productivity as reflected in QMD, BA, height, and SDI mean estimates across the commercial rotation and across all soil types of both aspen and all combined tree species. Analysis was conducted using restricted maximum likelihood Analysis of Variance (ANOVA) ( $\alpha=0.05$ ) with mixed effects models using the *nlme* package (Pinheiro et al., 2014).

Finally, the accuracy of regional aspen growth and yield equations was evaluated in comparison to observations collected in the field using traditional published equations. Height predictions published by Lundgren and Dolid (1970) were estimated with  $\text{SI}_{50}$  values found from NRCS records (2013) and Carmean (1979) (Table 3.1), while Schlaegel's (1971) prediction models of QMD and mean height were estimated using stand age as the sole independent variable. Perala et al.'s (1996) estimates of QMD were calculated with stand age,  $\text{SI}_{50}$  values and the maximum set parameter of mean July temperature ( $17^\circ\text{C}$ ). Stand level BA estimates were calculated using equations from Ek and Brodie (1974), using stand age, mean tree height, and stand level aspen density as predictors.

All estimates obtained through historical published equations were compared to observations across the measured commercial rotation (40 years). Predictions obtained through these models were compared using the two-one sided test (TOST) of equivalence, which calculates confidence intervals around the observations and compares them to an established region of indifference of the model. This approach shifts the



structure of the hypothesis test to assume a null of difference between the predictions and observations, requiring a test statistic to prove otherwise (Robinson and Froese, 2004; Robinson et al., 2005). The confidence intervals of the observations must fall within the region of indifference for the model to be considered acceptable. Regions of indifference were specified at a 10% error allowance of the intercept and slope, following procedures by Pokharel and Froese (2008), and Pond (2012). Tests of equivalence were conducted using the *equivalence* package (Robinson, 2014) in R. A bootstrapping sampling method provided in the *equivalence* package was used to estimate variance, using 1000 samples.

### 3.3 Results

The geographic extent of the Onaway soil type (*Inceptic Hapludalf*) allowed for the most intensive sampling ( $n = 29$ ), offered the most robust set of stand ages for residue treatment comparison along the commercial rotation. The Charlevoix soil (*Argic Endoaquod*), allowed for treatment comparisons at both ends of the rotation ( $n = 8$ ), followed by the Rubicon (*Entic Haplorthod*) ( $n=6$ ) and Ensley (*Aeric Endoaquept*) ( $n=5$ ) (Table 3.1). These soils allowed for comparisons among a range of productivity levels under the aspen forest cover type.  $SI_{50}$  levels of each soil type was estimated from NRCS records combined with  $SI_{50}$  comparison models developed by Carmean (1979) for Upper Michigan, and ranged from 17.7-23.3 m across soil types.

**Table 3.1** Study site characteristics by soil and treatment type

|                                     | Onaway             | Charlevoix      | Rubicon          | Ensley           |
|-------------------------------------|--------------------|-----------------|------------------|------------------|
| Soil texture                        | Fine sandy loam    | Coarse loam     | Mixed sand       | Coarse loam      |
| Soil order                          | Inceptic Hapludalf | Argic Endoaquod | Entic Haplorthod | Aeric Endoaquept |
| No. of stands sampled               | 29                 | 8               | 6                | 5                |
| Mean age (years)                    | 17.39              | 20.75           | 21.13            | 20.12            |
| Sampling units and age range - WTH  | 11 (1-36 years)    | 3 (2-38 years)  | 3 (7-30 years)   | 2 (3-36 years)   |
| Sampling units and age range - SOH  | 14 (1-40 years)    | 5 (6-36 yeras)  | 3 (17-26 years)  | 3 (7-34 years)   |
| Site index $_{50}$ (m) <sup>1</sup> | 23.2               | 23.2            | 18.3             | 17.7             |

<sup>1</sup>Natural resource conservation service (2013); Carmean (1979)

Soil type failed to explain any of the variance in development of the aspen cohort ( $p > 0.409$ ); however there was a significant temporal effect on stand patterns ( $p < 0.001$ ) (Table 3.2).

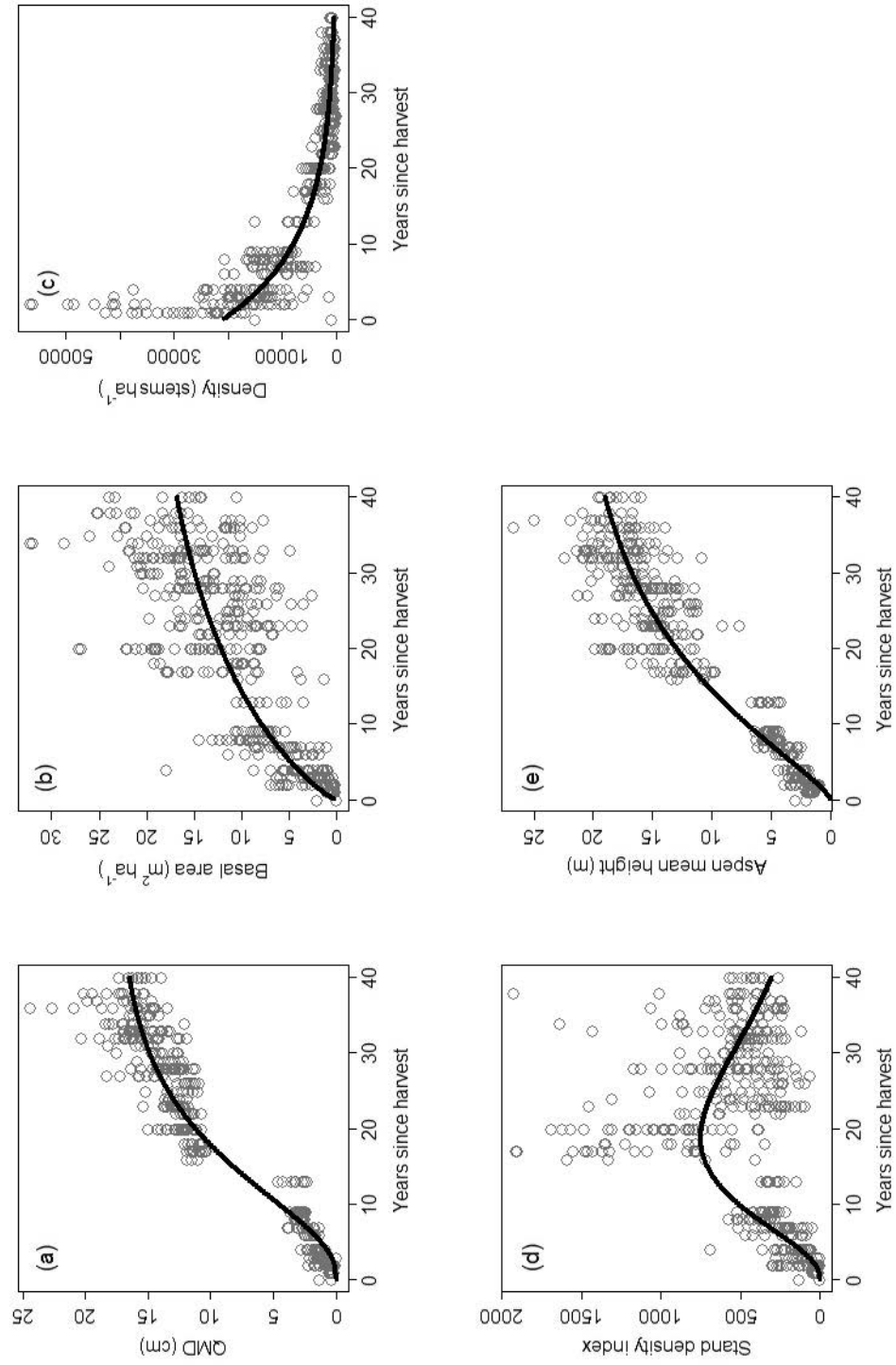
Aspen growth and yield patterns of QMD, BA, and stand mean height were characterized with a Chapman Richards function (Figure 3.1a,b,e respectively, and Table 3.3). Self-thinning rates of stand level density were expressed as a logarithmic function (Figure 3.1c) across stand rotation. Patterns of SDI were fit with an exponential power function (Figure 3.1d and Table 3.3), suggesting that the aspen cohort reaches a maximum SDI value at approximately year 18.

**Table 3.2** Mixed effects analysis of covariance effects of soil type and stand age on stand level variables

|   | Soil type |        |        |        | Stand Age |        |        |       | Soil type : stand age interaction |       |        |  |
|---|-----------|--------|--------|--------|-----------|--------|--------|-------|-----------------------------------|-------|--------|--|
|   | df        | F      | p      | df     | F         | p      | df     | F     | df                                | F     | p      |  |
| <i>Populus spp.</i>                           |           |        |        |        |           |        |        |       |                                   |       |        |  |
| quadratic mean diameter (cm)                  | 3, 44     | 0.721  | 0.545  | 1, 425 | 725.559   | <0.001 | 3, 425 | 0.039 | 3, 425                            | 0.039 | 0.990  |  |
| stocking (stems ha <sup>-1</sup> )            | 3, 44     | 0.896  | 0.451  | 1, 425 | 100.645   | <0.001 | 3, 425 | 1.521 | 3, 425                            | 1.521 | 0.209  |  |
| height (m)                                    | 3, 44     | 0.564  | 0.642  | 1, 425 | 513.629   | <0.001 | 3, 425 | 0.057 | 3, 425                            | 0.057 | 0.982  |  |
| stand density index                           | 3, 44     | 0.845  | 0.477  | 1, 425 | 14.682    | <0.001 | 3, 425 | 0.444 | 3, 425                            | 0.444 | 0.722  |  |
| basal area (m <sup>2</sup> ha <sup>-1</sup> ) | 3, 44     | 0.4339 | 0.730  | 1, 425 | 166.031   | <0.001 | 3, 425 | 0.753 | 3, 425                            | 0.753 | 0.521  |  |
| <i>All tree species</i>                       |           |        |        |        |           |        |        |       |                                   |       |        |  |
| quadratic mean diameter (cm)                  | 3, 44     | 0.168  | 0.918  | 1, 424 | 235.559   | <0.001 | 3, 424 | 1.023 | 3, 424                            | 1.023 | 0.382  |  |
| stocking (stems ha <sup>-1</sup> )            | 3, 44     | 0.524  | 0.668  | 1, 425 | 90.310    | <0.001 | 3, 425 | 5.343 | 3, 425                            | 5.343 | 0.001  |  |
| stand density index                           | 3, 44     | 7.373  | <0.001 | 1, 425 | 19.756    | <0.001 | 3, 425 | 5.546 | 3, 425                            | 5.546 | <0.001 |  |
| basal area (m <sup>2</sup> ha <sup>-1</sup> ) | 3, 44     | 6.261  | 0.001  | 1, 425 | 256.989   | <0.001 | 3, 425 | 2.944 | 3, 425                            | 2.944 | 0.033  |  |

**Table 3.3** Refitted models using the aspen cohort and all species in dominant aspen stands

| <i>Populus</i> development models  |                            |   |           |          |          |         |
|--|----------------------------|---|-----------|----------|----------|---------|
| Stand variable   | Description                | Equation form   | RMSE      | $b_0$    | $b_1$    | $b_2$   |
| quadratic mean diameter (cm)   | Chapman Richards function  | $QMD = b_0 [1 - \exp^{-b_1 \text{ stand age}}]^{b_2}$                 | 1.451     | 17.537   | 0.0953   | 2.78    |
| stocking (stems <sup>-1</sup> ha <sup>-1</sup> )                                     | Logarithmic function       | exp stocking = $b_0 - (b_1 \text{ stand age})$                        | 1.780     | 9.974    | 0.098    | -       |
| height (m)   | Chapman Richards function  | Height = $b_0 [1 - \exp^{-b_1 \text{ stand age}}]^{b_2}$              | 2.079     | 22.03    | 0.05667  | 1.3624  |
| aboveground biomass (kg <sup>-1</sup> ha <sup>-1</sup> )                             | Chapman Richards function  | Aboveground biomass = $b_0 [1 - \exp^{-b_1 \text{ stand age}}]^{b_2}$ | 16210.413 | 238277   | 0.01201  | 1.1719  |
| stand density index  | Exponential power function | SDI = $b_0 (\text{stand age})^B C \text{ stand age}^C$                | 278.723   | 7.2097   | 2.4      | 0.88    |
| basal area (m <sup>-2</sup> ha <sup>-1</sup> )                                       | Chapman Richards function  | Basal area = $b_0 [1 - \exp^{-b_1 \text{ stand age}}]^{b_2}$          | 4.077     | 20.801   | 0.03693  | 0.82797 |
| Stand development models   |                            |   |           |          |          |         |
| quadratic mean diameter (cm)   | Linear model               | $QMD = b_0 + b_1 \text{ stand age}$                                   | 2.583     | 0.826    | 0.316    | -       |
| stocking (stems <sup>-1</sup> ha <sup>-1</sup> )                                     | Logarithmic function       | exp stocking = $b_0 - (b_1 \text{ stand age})$                        | 1.994     | 10.04    | 0.06105  | -       |
| aboveground biomass (kg <sup>-1</sup> ha <sup>-1</sup> ) - Charlevoix/Onaway/Rubicon | Logistic function          | Aboveground biomass = $b_0 / (1 + e^{-b_1 b_2 \text{ stand age}})$    | 14730.144 | 84527.93 | 2.657    | 0.1274  |
| aboveground biomass (kg <sup>-1</sup> ha <sup>-1</sup> ) - Ensley                    | Logistic function          | Aboveground biomass = $b_0 / (1 + e^{-b_1 b_2 \text{ stand age}})$    | 20390.231 | 98310.28 | 4.531704 | 0.2487  |
| stand density index (Rubicon)  | Exponential function       | SDI = $b_0 b_1 \text{ stand age}$                                     | 965.162   | 303.02   | 1.07     | -       |
| stand density index (Charlevoix, Ensley, Onaway)                                     | Power function             | SDI = $b_0 \text{ stand age}^{b_1}$                                   | 742.371   | 304.62   | 0.3217   | -       |
| basal area (m <sup>-2</sup> ha <sup>-1</sup> ) - Charlevoix/Onaway/Rubicon           | Logistic function          | Basal area = $b_0 / (1 + e^{-b_1 b_2 \text{ stand age}})$             | 4.389     | 17.787   | 1.76     | 0.157   |
| basal area (m <sup>-2</sup> ha <sup>-1</sup> ) - Ensley                              | Logistic function          | Basal area = $b_0 / (1 + e^{-b_1 b_2 \text{ stand age}})$             | 5.261     | 23.998   | 2.8754   | 0.1957  |



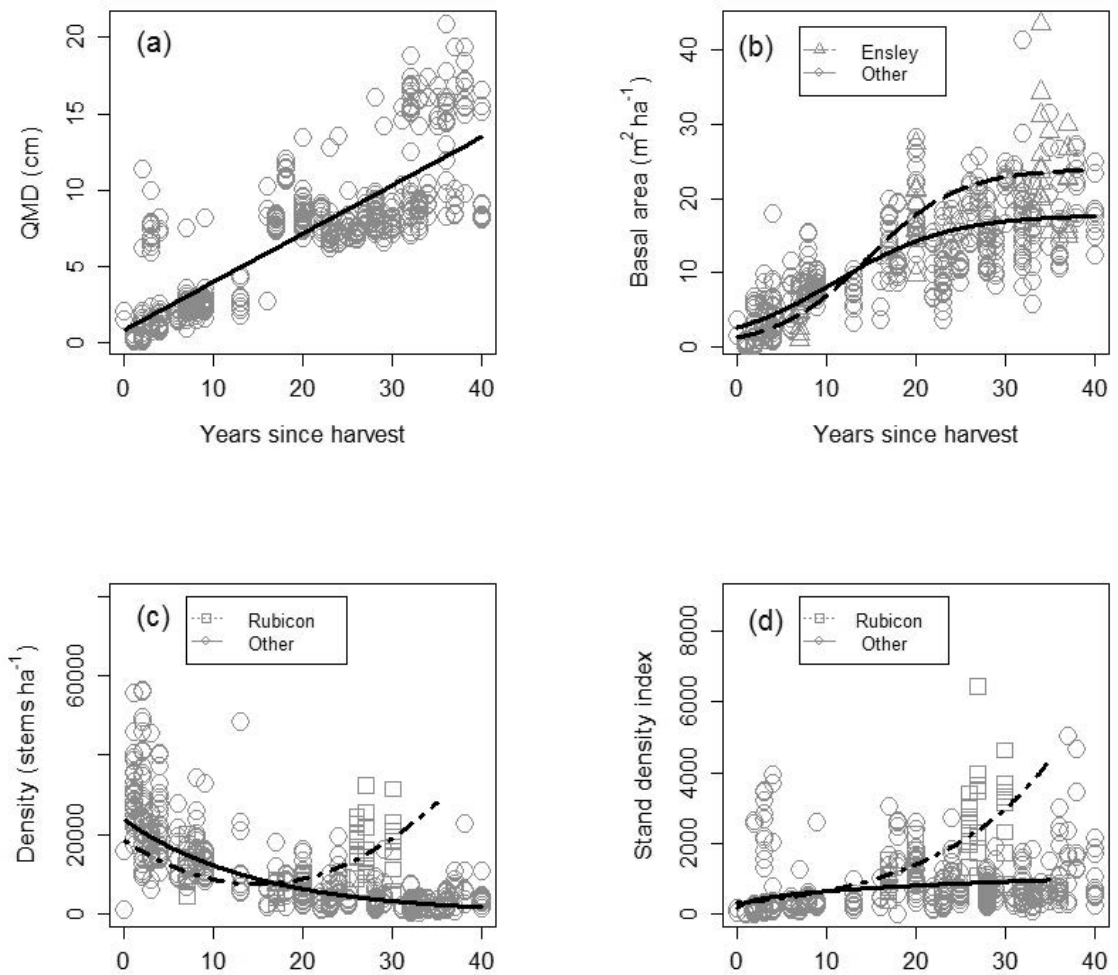
**Figure 3.1** Aspen cohort growth patterns across stand rotation (a) QMD (cm); (b) BA ( $\text{m}^2 \text{ha}^{-1}$ ); (c) Density ( $\text{stems ha}^{-1}$ ); (d) SDI, and; (e) Stand mean height (m)

When all tree species were combined for the analysis, tests revealed divergent patterns in stand level BA ( $p = 0.03$ ), density levels ( $p < 0.001$ ), and SDI ( $p < 0.001$ ) according to soil type (Figure 3.2b-d, and Table 3.4). The alteration in development patterns occurs at approximately 18-20 years, which corresponds to maximum SDI and canopy closure by the aspen cohort (Figure 3.1d). Increases in stand BA were characterized with a logistic function, however the Ensley exhibited higher values later in the rotation than all other soil types (Figure 3.2b), due to an increased proportion of large diameter *Abies balsamea* in the older stands (Table 3.6).

**Table 3.4** Harvest residue removals by soil type on stand level variables

|   | Onaway |       |       | Charlevoix |       |       | Ensley |       |       | Rubicon |       |       |
|---|--------|-------|-------|------------|-------|-------|--------|-------|-------|---------|-------|-------|
|   | df     | F     | p     | df         | F     | p     | df     | F     | p     | df      | F     | p     |
| <i>Populus spp.</i>                           |        |       |       |            |       |       |        |       |       |         |       |       |
| quadratic mean diameter (cm)                  | 1, 27  | 0.187 | 0.669 | 1, 6       | 1.164 | 0.322 | 1, 3   | 0.030 | 0.867 | 1, 4    | 0.032 | 0.867 |
| stocking (stems ha <sup>-1</sup> )            | 1, 27  | 0.008 | 0.929 | 1, 6       | 2.242 | 0.185 | 1, 3   | 0.348 | 0.597 | 1, 4    | 0.032 | 0.867 |
| height (m)                                    | 1, 27  | 0.205 | 0.655 | 1, 6       | 1.378 | 0.285 | 1, 3   | 0.056 | 0.829 | 1, 4    | 0.007 | 0.938 |
| stand density index                           | 1, 27  | 0.839 | 0.368 | 1, 6       | 1.559 | 0.258 | 1, 3   | 1.317 | 0.334 | 1, 4    | 2.953 | 0.161 |
| basal area (m <sup>2</sup> ha <sup>-1</sup> ) | 1, 27  | 0.220 | 0.643 | 1, 6       | 0.981 | 0.360 | 1, 3   | 0.568 | 0.506 | 1, 4    | 0.001 | 0.988 |
| <i>All tree species</i>                       |        |       |       |            |       |       |        |       |       |         |       |       |
| quadratic mean diameter (cm)                  | 1, 27  | 0.541 | 0.468 | 1, 6       | 0.263 | 0.627 | 1, 3   | 0.005 | 0.944 | 1, 4    | 0.708 | 0.447 |
| stocking (stems ha <sup>-1</sup> )            | 1, 27  | 0.001 | 0.980 | 1, 6       | 4.376 | 0.081 | 1, 3   | 0.320 | 0.611 | 1, 4    | 0.866 | 0.405 |
| stand density index                           | 1, 27  | 1.547 | 0.225 | 1, 6       | 0.110 | 0.751 | 1, 3   | 0.152 | 0.723 | 1, 4    | 0.124 | 0.743 |
| basal area (m <sup>2</sup> ha <sup>-1</sup> ) | 1, 27  | 0.230 | 0.635 | 1, 6       | 1.167 | 0.322 | 1, 3   | 0.181 | 0.699 | 1, 4    | 0.078 | 0.794 |

Stand level density patterns were not statistically different among the Charlevoix, Ensley/Angelica, or Onaway soil types, and were fit with a logarithmic function similar to the aspen cohort (Figure 3.2c; Table 3.3). However, stands along the Rubicon soil exhibited increasing density values after canopy closure. These patterns were characterized by a quadratic function, and can be attributed to relatively high density levels of *Acer rubrum*, *Acer saccharum*, and *Fraxinus americana* compared with other soil types (Table 3.6). Similarly, Rubicon SDI values diverged from the remaining soil types with an exponential increase after aspen canopy closure (Figure 3.3d, Table 3.3). Tests did not detect a difference in QMD according to soil type, and observations were fit with a linear model based on model comparisons. The removal of harvest residues had no significant effect on stand development patterns in the aspen cohort or all combined species (Table 3.4).



**Figure 3.2** Stand development patterns across rotation for all tree species: (a) QMD (cm); (b) BA ( $\text{m}^2 \text{ha}^{-1}$ ); (c) Density ( $\text{stems ha}^{-1}$ ), and; (d) SDI

Tests of equivalence failed to validate published equations for aspen stand development in the study area within the specified range of acceptability (Table 3.5). Tests of the equation for stand mean height revealed mixed levels of acceptability when compared with compiled data, with Lundgren and Dolid's (1970) exponential function [2] model with a  $SI_{50}$  parameter being the only statistically acceptable model across rotation (Table 3.5). QMD predictions established by Schlaegel's (1971) linear model showed a significant departure from observations at the lower range of estimates in the intercept TOST confidence interval ( $C-B_0$   $C+B_0$ ). However, estimates derived from this model were statistically similar to observations at the upper range of values indicated through the confidence intervals ( $C-B_1$   $C+B_1$ ) (Table 3.7). Stand level BA models were limited to

those originally proposed by Ek and Brodie (1974). Using multiple predictors, including stand density and mean height, resulted in a bias overestimate when compared to observations recorded in the field (Table 3.5).

**Table 3.5** Historical growth model equivalence test bootstrapping test results. Values in bold indicate that the calculated confidence interval is within the established region of indifference ( $\mu_o - \mu_p = 0$ )

| Stand variable                      | Predict model                 | C-B <sub>0</sub> | C+B <sub>0</sub> | I-B <sub>0</sub> | I+B <sub>0</sub> | C-B <sub>1</sub> | C+B <sub>1</sub> | I-B <sub>1</sub> | I+B <sub>1</sub> |
|-------------------------------------|-------------------------------|------------------|------------------|------------------|------------------|------------------|------------------|------------------|------------------|
| Height (m)                          | Lundgren and Dolid (1970) [1] | 10.263           | 11.474           | 8.390            | 10.255           | 0.984            | 1.141            | 0.9              | 1.1              |
| Height (m)                          | Lundgren and Dolid (1970) [2] | <b>10.227</b>    | <b>11.469</b>    | 9.447            | 11.547           | <b>0.932</b>     | <b>1.069</b>     | 0.9              | 1.1              |
| Height (m)                          | Schlaegel (1971)              | 10.878           | 12.528           | 12.720           | 15.547           | 0.589            | 0.751            | 0.9              | 1.1              |
| QMD (cm)                            | Schlaegel (1971)              | 8.623            | 9.423            | 9.026            | 11.032           | <b>0.974</b>     | <b>1.088</b>     | 0.9              | 1.1              |
| QMD (cm)                            | Perala et al. (1996)          | 8.736            | 9.685            | 5.974            | 7.302            | 1.251            | 1.453            | 0.9              | 1.1              |
| Basal area (ft acre <sup>-1</sup> ) | Ek and Brodie (1974)          | 53.564           | 61.734           | 32.972           | 54.953           | 1.306            | 1.642            | 0.9              | 1.1              |

**Table 3.6** Species importance values by strata and soil type

|                               | Charlevoix |            | Ensley    |            | Onaway    |            | Rubicon   |            |
|-------------------------------|------------|------------|-----------|------------|-----------|------------|-----------|------------|
|                               | overstory  | understory | overstory | understory | overstory | understory | overstory | understory |
| <i>Abies balsamea</i>         | 7.0        | 10.6       | 10.4      | 5.9        | 5.7       | 3.5        | 5.4       | 4.4        |
| <i>Acer rubrum</i>            | 4.8        | 14.2       | 4.9       | 9.8        | 5.3       | 1.9        | 5.1       | 40.2       |
| <i>Acer saccharum</i>         | 3.1        | 1.5        | 6.8       | 0.1        | 6.8       | 0.9        | 3.1       | 3.4        |
| <i>Betula allgenhaniensis</i> |            | 0.2        |           |            |           |            | 4.1       |            |
| <i>Betula papyrifera</i>      |            |            | 3.0       | 0.8        |           |            | 4.0       | 1.1        |
| <i>Fraxinus americana</i>     | 3.0        | 0.8        | 6.1       | 8.1        | 3.2       | 4.3        | 3.0       |            |
| <i>Larix laricina</i>         |            | 0.1        |           |            |           |            |           |            |
| <i>Ostrya virginiana</i>      |            | 0.2        | 5.9       |            | 3.1       | 0.1        |           |            |
| <i>Picea glauca</i>           | 6.3        | 1.4        |           | 0.5        | 4.4       | 0.6        | 7.3       | 0.7        |
| <i>Picea mariana</i>          |            | 0.3        |           |            | 1.8       |            |           |            |
| <i>Pinus banksiana</i>        |            |            |           |            |           |            | 3.9       |            |
| <i>Pinus resinosa</i>         |            |            |           |            |           |            | 3.6       |            |
| <i>Pinus strobus</i>          |            |            |           |            |           | 0.1        | 6.1       | 0.3        |
| <i>Populus balsamifera</i>    | 13.9       | 6.3        | 18.6      | 7.1        | 8.0       | 9.8        | 1.6       | 0.6        |
| <i>Populus grandidentata</i>  | 7.9        |            |           |            | 5.2       | 3.7        | 10.7      | 17.3       |
| <i>Populus tremuloides</i>    | 49.6       | 58.6       | 35.9      | 67.0       | 45.9      | 70.8       | 31.4      | 24.0       |
| <i>Prunus serotina</i>        | 4.6        | 5.8        | 2.2       | 0.7        | 3.5       | 4.3        | 2.4       | 2.3        |
| <i>Quercus rubra</i>          |            |            |           |            |           |            | 4.1       | 5.7        |
| <i>Thuja occidentalis</i>     |            |            | 6.1       |            | 2.8       |            |           |            |
| <i>Tilia americana</i>        |            |            |           |            | 4.3       |            |           |            |
| <i>Tsuga canadensis</i>       |            |            |           |            |           |            | 4.1       |            |
| <i>Ulmus americana</i>        |            | 0.1        |           |            |           |            |           |            |

### 3.4 Discussion

Attempts to characterize aspen stand development were taken by Ruark and Bockheim (1987), who utilized a chronosequence approach on the Rubicon soil type to quantify patterns according to the biomass oriented stand development model of Borman and Likens (1979). While this work thoroughly assessed spatial and temporal biomass allocation patterns, findings are likely confounded by differences in stand origin and disturbance type and intensity which can severely alter species composition, stand structure, and long-term trajectory (Frelich, 2002).

The prolific vegetative reproduction of aspen allows for rapid reoccupation of a site immediately following harvest, exhibiting the highest levels of tree density in the first three years of stand development in the stand initiation stage Figure 3.2c (Oliver and Larson, 1996). Krasny and Johnson (1992) reported highest aspen density rates one year following harvest, with approximately 50% mortality by year three. Although stand density and self-thinning rates are correlated to harvesting effects, edaphic conditions, (Bella, 1986; Stone and Eliof, 1997; Arikian et al., 1998; Voldseth et al., 2011) and fluctuations in carbohydrate reserves, (Mundell et al., 2008), initial differences in density subside by age three due to increased self-thinning rates in stands with higher initial tree density (Bella, 1986). High levels of regeneration density coupled with rapid growth rates of aspen results in a relatively limited open growth stand initiation period with accelerated progression toward full stand foliar occupancy (Perala, 1990; Long and Smith, 1984). In this study, there was no observed difference in the development of the aspen cohort across the commercial rotation according to soil texture classes and estimated  $SI_{50}$  values. These findings suggest that aspen growth is not limited by site resources on any of the observed soil types, however the high variance in observations of both aspen height and basal area estimates warrant caution when making broad conclusions.

In the current study, the declining rate of self-thinning occurs simultaneously with the decline in growth rate of stand level QMD, BA, and mean height at approximately age 18-20. These events further coincide with maximum SDI values of the aspen cohort (Figure 3.1d), and is indicative of the stem exclusion stage as characterized by Oliver (1981) and Oliver and Larson (1996). As the aspen cohort declines in stand level growth following maximum SDI, gaps created through mortality recruit other tree species from the under- and mid-story strata. After this point in the rotation, the decline in aspen density initiates redistribution of the canopy layer among competitors and regeneration. These dynamics signify the shift to the understory reinitiation (Oliver and Larson, 1996) or demographic transition (Frelich, 2002) phase. The sampling design makes it difficult to interpret patterns past the stem exclusion stage (ages 18-20), therefore site partitioning according to canopy strata and individual species shade tolerance may provide insight to observed trends in observations (Table 3.6).

The increased BA values on the Ensley-Angelica soil later in stand rotation is due to the relatively higher occurrences of *Abies balsamea*, *Acer saccharum*, *Fraxinus americana*,



*Thuja occidentalis* (L.), and *Acer rubrum* in the overstory strata, as illustrated by the relative importance values (Table 3.6), and the lack of significant differences in QMD, tree density, and SDI (Figure 3.2a, c, d, Table 3.2). Given the high shade tolerance and slow growth rates associated with these species (with the exception of *Fraxinus americana*), it is likely these stems were advance regeneration at the last time of harvest (Burns et al., 1990). It is worth noting that an increase of approximately 26% in BA along the Ensley/Angelica soils by the end of the rotation by including competing species (Figure 3.2b) had no discernible detrimental effect on development of the aspen cohort. These trends suggest that promotion of non-aspen trees can enhance stand structural and compositional diversity without impacting long-term production, as suggested by Long and Shaw (2010).

The Rubicon soil type exhibited higher SDI and density levels from ages 18-30 than all other soil types (Figure 3.2d, Table 3.2), which is due to higher occurrences of *Acer rubrum* saplings and seedlings (Table 3.6). Observed SDI values along the Rubicon soil type are slightly higher than maximum possible SDI values estimated through specific gravity by Woodall et al. (2005). This is likely attributed to the limitation of stems >12.5 cm dbh in the testing procedures of the aforementioned study, whereas estimates include stand density regardless of commercial threshold. These patterns suggest that forest stands along the Rubicon soil type are reaching the understory reinitiation/demographic transition stage of development at an accelerated rate. Meng et al. (2002) modeled temporal species composition in boreal mixed aspen forests of Alberta, and found that while aspen dominance decreased after age 30 across sites, this trend was more pronounced on coarse textured soils of lower estimated productivity. These findings corroborate with those by Shields et al., (1981) and Perala (1990), which noted that the breakup of the aspen cohort occurs earliest on nutrient poor sites and in the southern limits of its range. And while these trends are inconsistent with reports by Larson et al. (2008), who found that competition in *Pseudotsuga menziesii* (Mirb.) stands of the Pacific Northwest was positively related to inherent site productivity, recent work by Coates et al. (2013) suggests that these patterns are strongly species dependent.

Observed trends in the Rubicon soil could provide insight when evaluating future stand dynamics given increases in minimum temperatures, accompanied trends in soil evapotranspiration, and potential shifts in species ranges associated with climate change models (Iverson and Prasad, 2001; Allen et al., 2010). While climate change models predict a considerable loss in the southern range of aspen in eastern North America and the Great Lakes region, it is likely that this transition will be occur gradually and perhaps over several rotations. Rather than a loss of aspen forests throughout the eastern region, it is perhaps reasonable to expect shortened biological rotations of this early-seral cohort and accelerated successional trajectories with a higher occurrence of hardwood competitors.

Renewed interest in logging residue utilization as a renewable energy source has warranted caution in some forest types, as harvest residues contain a disproportionate amount of stand level nutrients that are critical to long-term tree growth and stand productivity (Alban 1985; Barnes et al., 1998; Fisher and Binkley 2002; Pare et al.,

2002). Effects of residue removal on forest productivity have been the subject of significant study in the European and Scandinavian boreal forests, and have been found to vary according to forest type and silvicultural regime. Several research studies have suggested a significant decrease in tree and stand BA increment following residue removal in thinning trials of *Picea abies* and *Pinus sylvestris* stands in Norway (Nord-Larsen, 2002), Sweden (Engell and Leijon, 1997), and Finland (Helmisaari et al., 2011). Similarly, Walmsley et al. (2009) concluded that residue removal caused a significant decrease in individual tree diameter and stand level BA by 10.3% and 15.3%, respectively, in second rotation *Picea sitchensis* stands of Wales, UK. While invaluable, observed trends in these studies may be region and forest type specific, and are perhaps not broadly applicable. There are substantial differences in climate, soil properties and associated productivity rates, and silvicultural programs (i.e. stump removal/ disking-trenching), between conifer plantation forests of northern Europe and the temperate forests of the North American continent. These factors caution against making general conclusions about residue removals in global forest types.

Initial efforts to investigate the impact of intensive management and residue removal effects on forest productivity in North American temperate forests were established in the 1970-1980's. Many of these studies were limited in temporal scope (Silkworth and Grigal, 1982; Johnson et al., 1985; Henrickson et al., 1989) or focused on stands undergoing conversion to an alternative forest type (Mroz et al., 1985). Early findings from aspen forests under intensive organic matter removals and compaction levels in the North American Long-Term Productivity Study (LTSP) had varying results by residue removal and soil type at five years of age.

Stone (1998) reported a significant increase in stand density following WTH in aspen forests on calcareous clay and silt loam soil types, while this increase was not significant on the coarse sand soil type. Further, results suggested an increase in individual tree diameter and height on the clay and silt loam forests following whole tree harvesting; however stems on the sandy soil showed a significant reduction in these metrics following all levels of organic matter removal. By year ten, these previously observed differences had decreased to non-significant levels (Voldseth et al., 2011). Although a notable merit of the LTSP is the highly controlled and replicated experimental design, results presented here provide a pragmatic and operational context to the removal of harvest residues in industrial forest settings. Results failed to detect an effect of residue removal on stand level growth patterns on a variety of soil types across commercial rotation (Table 3.4). These trends could support hypotheses proposed by Boyle et al (1973) that nutrient deficiencies might not be affecting tree growth until after nine, 30-year crop rotations (270 years).

The equivalence test results suggest that traditional growth and yield equations have mixed levels of acceptability in application to forests in Upper Michigan. This is unsurprising, given that original models were based on a limited geographic extent of fire-origin stands of northern Minnesota in the early 1900's. Given the wide distribution of aspen, it is probable that there may be moderate climatic, geographical, and clonal

influence and introduced variation in growth patterns and trajectory (Berrang et al., 1991; King et al., 1999; Silim et al., 2009). Rogers et al. (2014) found high variety in stand dynamic patterns of aspen across the western region and attempted to describe these within distinct functional frameworks. Further, results from this work show that only one of the six models tested passed the equivalence tests; therefore, there is an apparent need for updated and localized equations within the Great Lakes region. Growth rates observed and modeled in section 1.4.2 may serve as potential updates and improvements in the sub region and under a similar silvicultural regime, however further testing and validation on these models is required.

### **3.5 Conclusions and Management Recommendations**

By using a chronosequence sampling approach to leverage detailed stand history records, we were able to quantify aspen forest stand development across a commercial chronosequence along major soil types of the Great Lakes region according to harvest intensity. There was no difference in aspen growth and yield according to soil type or harvest residue treatment. Forest stands on *Aeric Endoaquept* soils exhibited higher BA levels later in the rotation than all other soil types due to increased levels of *Abies balsamea* and *Thuja occidentalis* density and area, and had no discernible effect of productivity of the aspen cohort. Stand density and SDI values on the *Entic Haplorthod* were significantly higher than those on other soil types later in the rotation. These patterns suggests faster breakup of the aspen cohort and accelerated stand succession with higher occurrences of hardwood competitors on coarse textured soils. Taken together, these findings suggest that aspen forests managed for pulp and biofuel production maintain stand structural and compositional diversity under the current silvicultural regimes. Given the results from equivalence tests, growth and yield equations for aspen in the Great Lakes region should be applied cautiously and conservatively, as they may require re-fitting.

## Chapter 4<sup>3</sup>

# Logging residue removals leads to increased structural heterogeneity of forest understory vegetation communities

### 4.1 Introduction

Conservation of plant community diversity has been gradually integrated into contemporary forest management regimes throughout North America. Forests with high levels of species richness and diversity are shown to have enhanced ecological resilience (Peterson et al., 1998), mitigated losses of plant limiting nutrients (Likens et al., 1970; Mou et al., 1993) and positive feedbacks with soil fertility (Dybzinski, 2008). However, there has been considerably less emphasis on quantifying the effects of intensified timber harvest programs on changes to community composition and structure as part of long-term silvicultural planning. Thorough assessment of community dynamics enhances our understanding of floristic response to environmental conditions and disturbance (Lavorel and Garnier, 2002), in addition to the relative abundance metrics provided by species richness and diversity.

Forest vegetation communities and patterns are dependent upon light, nutrient, and water resources, the availability of which is dependent upon latitude, soil texture, and land management activities (Chapin et al., 2011). Variations in these resources contribute to niche partitioning and competition by individual plant species which further develop into distinct vegetation communities and successional trajectories. Community assemblages are further dependent upon forest successional stage and associated disturbance regime (Tilman, 1985; Johnson and Miyanishi, 2007). It has been proposed that the over- and understory strata communities are linked by both reciprocating influences and responses to environmental gradients and disturbance. The overstory stratum directly modifies the understory community through competition for soil resources, alterations in light quality, and detrital inputs (Gilliam and Roberts, 2003; Messier et al., 1998), while herbaceous plants have been shown to influence success and spatial patterning of tree regeneration (Donoso and Nyland, 2006). Gilliam and Roberts (2003) further proposed, after assessing stratum response to environmental gradients in eastern temperate forests that both strata respond in parallel to edaphic conditions and disturbance.

As demands on forest resources increase to supply traditional products and evolving notions of ecosystem services, forests currently under industrial ownership will likely be the first utilized for intensified management (Vance et al., 2010). The growing interest in forest biomass feedstock supply to bioenergy producers has gained particular interest in various regions of North America. A readily available source for bioenergy production are harvest residues, which are the byproduct of a commercial timber harvest and

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<sup>3</sup> The material contained in this chapter is in preparation to be submitted to *Forest Ecology and Management*

therefore comprised largely of fine woody debris (FWD, <10 cm diameter). Little is known, however, about the potential impacts of residue removal on soil nutrient resources, vegetation communities, and long-term productivity, given the disproportionate amount of nutrients concentrated in topwood (Alban, 1985; Thiffault et al., 2011). Therefore, caution must be exercised when assessing intensified management effects to the vegetation community, as shifts in diversity, composition, or structure of either stratum may disrupt nutrient cycling patterns and threaten ecosystem stability and long-term productivity (Saleska et al., 2002).

The aspen (*Populus* spp.) forest cover type is the most widely distributed in North America, and is characteristic of both mixed boreal and northern temperate forests (Barnes et al., 1998). In the Great Lakes region, nearly 5 million hectares of forest land are classified as aspen cover type (SAF forest type 16; FIA 2014), which is characteristic of an early seral community across a broad range of physiographic features (Zasada et al., 2001). Forest management regimes rely on coppice regeneration silviculture which favors aspen re-establishment at high densities (Perala, 1990). In the past, harvest residues (topwood < 10 cm. and non-merchantable bolewood) have been utilized at a minor scale for heat and electricity at local pulp fiber production mills under vertically integrated management programs in the Great Lakes region. The response of forest vegetation to varying levels of harvesting have been thoroughly assessed, however research programs investigating the effects of residue removals have primarily focused on aboveground timber production (Voldseth et al. 2011). As such, there has been less focus on changes in forest vegetation community diversity, composition, and heterogeneity. While increased levels of plant biodiversity are generally regarded as beneficial, these metrics are limited to relative abundance, and generally overlook ecological changes in community dynamics. Shifts in vegetation composition and structure have been suggested to alter successional trajectories and threaten long-term site productivity (Scheffer et al., 2001; Scheffer and Carpenter, 2003). Given the potential increase in biomass utilization with alternative energy policies, understanding the effects of residue removal is critical to assessing long-term site productivity and ecosystem stability.

The purpose of this study was to evaluate the effects of harvest residue removal on vegetation communities of commercially managed aspen forests as reflected in (i) species richness, diversity, and evenness and; (ii) community composition, and structural heterogeneity of the over- and understory forest strata. The removal of residues was expected to result in decreases forest vegetation biodiversity metrics across stand rotation, and that that this pattern would stimulate a shift in community composition and both in the over- and understory strata.

## 4.2 Materials and Methods

### 4.2.1 Field sites

The study was conducted in commercial aspen stands in Delta, Dickinson, and Menominee counties in the Upper Peninsula of Michigan, under ownership of Plum Creek Timber Company. The study area has a cool continental climate with mean air temperatures reaching -13° C in January to 24° C in July with an average of 724 mm of annual precipitation, most of which is received during the growing season. All stands are located on an *Inceptic Hapludalf* of the Onaway series, which is common throughout the region and originated during the Wisconsin age (NRCS, 2013). Aspen forests under this ownership have been kept in the aspen cover type since the extensive logging of the region as part of the production-oriented history of the land base. Stands are composed of *Populus tremuloides* (Michx.), with mixed occurrences of *Populus balsamifera* (L.) and *Populus grandidentata* (Michx.). Harvest residues (< 10 cm.) have been utilized at a minor scale in some of these forest stands in the last 45 years for heat and electricity co-generation at a local pulp fiber production mill in Escanaba, MI.

### 4.2.2 Study design

An inventory database provided by Plum Creek Timber Company was used to systematically select forest stands with the following criteria: (i) classified as aspen cover type; (ii) contained records indicating harvest residue treatment (stem only harvest – SOH or; whole tree harvest - WTH), (iii) harvest year, and; (iv) minimum of 10 ha in size. I attempted to sample along a chronosequence of aspen stands across the commercial rotation (40 years) in each residue treatment (SOH / WTH) category on a common and widely distributed soil type. The Onaway soil type (Inceptic Hapludalf) was the sole soil type included in this chapter due to a robust availability of sites along the chronosequence and to avoid complications of rarefaction on other soil types. Stands were randomly selected from each treatment group (SOH or WTH) and binned into 5-yr age classes to represent a full chronosequence producing a nearly complete replicate, for a total of 29 stands (SOH, n=14; WTH, n = 15).

Vegetation sampling was conducted in early June through mid-July to effectively allocate sampling efforts and coincide with the seasonal flush of summer-green herbs. Soil sampling was conducted between mid-July and late August to coincide with decreased rates of plant uptake of exchangeable soil nutrients (Alban, 1985). Ten random sampling locations were established in each of the 29 aspen stands using the random point generator in ArcMap v 10.0 (ESRI, 2011), with a minimum distance between plots and from stand edges of 30 m.

#### *4.2.3 Data collection*

Each sampling location was identified in the field with a recreation-grade Garmin E76trex Global Positioning System (GPS). At each location, a 0.04 ha subplot was used to measure overstory trees ( $> 10$  cm) at 1.37 m (diameter at breast height- dbh) by species and to the nearest 0.1 cm. Four subplots, each 0.001 ha, were established in each cardinal direction from overstory plot center at a distance of 5.64 m. In each subplot, tree saplings  $\leq 9.9$  cm dbh were tallied by species and dbh measured with a digital caliper, with all stems  $< 1.37$  m in height tallied by species. I established two 1 m<sup>2</sup> quadrats directly outside of the overstory plot at north (0/360°) and south (180°), within each all herbaceous and shrub species were identified and tallied. Estimated percent cover of individuals, as well as FWD, coarse woody debris (CWD  $\geq 10$  cm), leaf litter, and bare soil cover were recorded to the nearest 0.1 cover class. I calculated plot level down dead wood (DDW – all size classes) values using a proportional probability to size method with the Line Intersect Distance Sampling (LIDS) protocol (Affleck, 2008) oriented on the overstory plot center.

I sampled forest floor and mineral soil by establishing two subplots adjacent to the east boundary of both vegetation sampling quadrats, and used 25 cm<sup>2</sup> subplots to remove vegetation and forest floor detritus. Once the organic layer of the soil was exposed, a soil core extractor (5 cm width by 15 cm depth) was inserted with a soil slide hammer, following procedures by Jurgensen et al., (1977). Soil sampling was conducted at two depths, (0-15 cm and 15-30 cm) which aimed to capture the mineral soil macronutrient content directly available to shallow lateral roots of aspen and associated vascular plants (Pregitzer and Friend, 1996), and further coincides with a restricting fragipan on the Onaway soil type between 33-53 cm in depth (NRCS, 2013). A total of four canopy closure measurements were taken with a spherical densiometer at the center of each regeneration subplot in each cardinal direction.

#### *4.2.4 Laboratory processing and soil analysis*

Soil samples collected in the field were stored on ice and transported to the Forest Soils and Forest Ecology laboratories at Michigan Technological University in Houghton, MI and remained frozen until processing. All soils were oven dried at 65° C to ensure a consistent dry weight. Soils were then sieved through a 2 mm mesh screen to separate all roots, rocks, and buried woody debris. Sieved soils were then milled to a consistent texture and processed for total carbon and nitrogen with a Fisons NA 1500 Elemental Analyzer. Exchangeable Ca, K, and Mg were determined using a NH<sub>4</sub>Cl dilution factor of 10:1 and processed in an Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES).

#### 4.2.5 Data Analyses

To assess the effect of residue removal on forest strata alpha diversity, species richness, Shannon's diversity index, and evenness values were calculated at the plot level for the overstory ( $n = 290$ ), and at the subplot level for the understory strata ( $n = 580$ ) for each of the measured stands. These values were then compared by treatment across stand rotation using linear mixed effects models to account for the hierarchical nature of the data. Analyses were conducted using the package *nlme* 3.1-119 (Pinheiro et al., 2015) with a specified  $\alpha = 0.05$  in the R software environment (R Core Team, 2012). Normality and homogeneity of variance assumptions were tested by examining residuals with  $q$ - $q$  plots.

Nonmetric multidimensional scaling (NMDS) ordination was used to assess variation in vegetation communities between the residue treatment types. Separate ordinations were conducted for both the overstory and understory communities. Plot values were averaged at the stand level for ordination procedures. The metaMDS function was used for ordination procedures in the R package *vegan* 2.0-10 (Oksanen et al., 2013), using the Bray-Curtis similarity measure and 100 iterations. NMDS is a preferred ordination method as it uses rank distances between species occurrences, thereby averting assumptions of linear relationships between variables (McCune and Grace, 2002). Species matrices were derived from importance values for all overstory tree species ( $IV = \text{relative BA} + \text{relative density} / 2$ ) and understory herbaceous and shrub species ( $IV = \text{relative cover} + \text{relative density} / 2$ ). Aspen was removed from the overstory species matrix to avoid circularity with variables included in the environmental matrix, such as relative aspen density and basal area. Species with less than 5% occurrences were excluded from analysis as suggested by McCune and Grace (2002).

Environmental matrices included in the ordination differed according to strata; however, both included stand age, elevation, cover of forest floor attributes, such as FWD, CWD (collectively DDW) leaf litter, mineral soil nutrients (total soil C, total soil N, C:N, and exchangeable Ca, K, and Mg), and overstory conditions (canopy closure, relative aspen density, relative basal area, and Reineke's Stand Density Index). LIDS estimates of residues and DDW were omitted from the ordination due to spatial dependence of the LIDS design which was centered on overstory plot center and did not accurately represent DDW conditions in the vicinity of the herbaceous subplots based on the sampling design. Therefore, only estimated cover of DDW (%) taken from the 1m<sup>2</sup> quadrats were used in the environmental matrix of the ordination.

Biodiversity metrics of the over- and understory strata were contrasted to investigate any influence or link between strata (i.e. overstory species richness, diversity index and evenness were included in understory environmental matrix, and conversely, understory metrics were included in the overstory environmental matrix). A total of 25 stands (SOH=13; WTH=12) were used in the NMDS (Table 4.1). The relative influence of environmental variables on community composition was assessed with the envfit function in package *vegan* 2.0-10 (Oksanen et al., 2013). Comparisons of vegetation composition



between treatment types were conducted using a nonparametric Multi-Response Permutation Procedure, with the MRPP function in *vegan*. This approach assumes a null hypothesis that each arrangement of objects, in this case soils and treatment type, is equally probable (Mielke and Berry, 2007). Bray-Curtis similarity distance measures were used, with 1000 permutations for the mrpp. Finally, I used a permutational multivariate analysis of variance (Anderson, 2006) to test for homogeneity of multivariate group dispersions based on treatment type with the betadisper and permutest functions in package *vegan* 2.0-10.

### 4.3 Results

Results from the mixed effects regression analysis suggest no significant effect of harvest residue treatment, stand age, or any interaction effect of these variables on overstory species richness, diversity, or evenness metrics over the 40 year aspen rotation (Table 4.2; Figure 4.1a-c). Several tree species including *Fraxinus nigra* (Marsh.), *Tilia americana* (L.), and *Salix spp.* were minor species in the SOH treatment stands, however they were not abundant enough to detect any difference in richness or diversity metrics.

The final solution of the overstory NMDS resulted in a two-dimensional structure, with a final stress of 16.9, and Axes 1 and 2 representing approximately 0.187 and 0.155 of the variance, respectively (Figure 4.2; Table 4.3). By plotting individual tree species in sample space, *Acer rubrum* (L.) was strongly negatively correlated to Axis 2, with both *Picea glauca* (Moench) and *Pinus strobus* (L.) exhibiting positive relationships with this axis. Results detected a weak correlation between fine woody debris cover and ordination Axis 2 (Table 4.3). *Picea glauca*, and to a lesser extent, *Thuja occidentalis* (L.) were more commonly found in the WTH stands, while *Populus spp* frequency was similar among both treatment types.

Post-hoc linear regression models show strong evidence for increasing *Acer rubrum* importance values through stand rotation ( $df = 23$ ,  $F = 11.876$ ,  $p = 0.002$ ). This temporal pattern is reversed in FWD cover, as values decrease through time ( $df = 23$ ,  $F = 11.971$ ,  $p = 0.002$ ). Tests did not detect a difference in species composition by residue treatment based on the MRPP results ( $A = -0.008$ ,  $p = 0.724$ ). Tests of homogeneity of multivariate dispersions failed to reveal a difference in multivariate space between the residue treatments ( $F = 0.359$ ,  $p = 0.564$ , 999 permutations), with distances to group centroids at 0.362 and 0.399 for the WTH and SOH groups, respectively.

**Table 4.1** Site characteristics of measured aspen stands in Delta, Dickinson, and Menominee Counties, in the Upper Peninsula of Michigan. Values indicate mean estimates and values in parentheses indicate stand deviations.

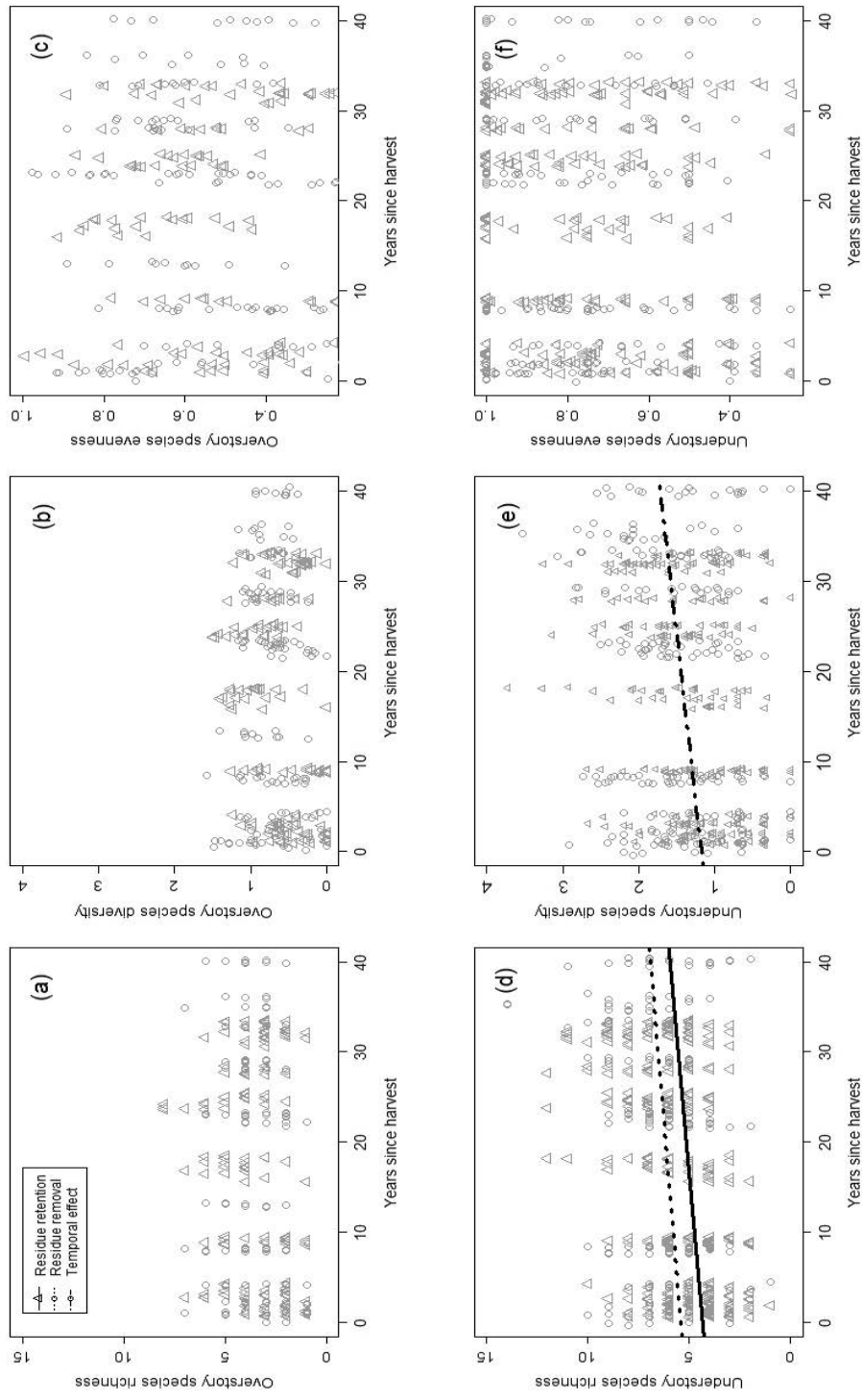
| Site attributes                                | Treatment               |                          |
|--|-------------------------|--------------------------|
|  | Stem only harvest (SOH) | Whole tree harvest (WTH) |
| Elevation (m)                                  | 291.15 (34.23)          | 283.19 (19.02)           |
| <b>Overstory strata</b>                        |                         |                          |
| Aspen stocking index (%)                       | 0.68 (0.25)             | 0.67 (0.26)              |
| Aspen basal area index (%)                     | 0.91 (0.12)             | 0.94 (0.05)              |
| Aspen stand density index                      | 445.11 (389.3)          | 317.81 (183.45)          |
| Canopy closure (%)                             | 63.12 (29.15)           | 65.20 (29.35)            |
| <b>LFH horizon</b>                             |                         |                          |
| Leaf litter (% cover)                          | 65.75 (23.96)           | 64.94 (14.77)            |
| Fine woody debris (% cover)                    | 16.71 (12.70)           | 12.94 (6.01)             |
| DDW (% cover)                                  | 12.74 (7.21)            | 12.66 (6.67)             |
| DDW biomass early decay (Mg ha <sup>-1</sup> ) | 11.73 (5.93)            | 9.07 (5.85)              |
| DDW biomass late decay (Mg ha <sup>-1</sup> )  | 4.49 (4.74)             | 3.39 (1.93)              |
| <b>Mineral soil</b>                            |                         |                          |
| Total soil carbon (Mg ha <sup>-1</sup> )       | 38.33 (14.45)           | 35.18 (6.92)             |
| Total soil nitrogen (Mg ha <sup>-1</sup> )     | 2.12 (0.80)             | 2.10 (1.04)              |
| C:N  | 18.53 (2.51)            | 18.70 (3.31)             |
| Exchangeable Ca (Mg ha <sup>-1</sup> )         | 5.17 (3.83)             | 4.69 (3.38)              |
| Exchangeable Mg (Mg ha <sup>-1</sup> )         | 0.74 (0.44)             | 0.71 (0.59)              |
| Exchangeable K (Mg ha <sup>-1</sup> )          | 4.90 (4.4)              | 2.20 (1.8)               |

Values indicate means with standard deviations in parentheses.

Aspen stocking and basal area indices indicate aspen proportional to all species

**Table 4.2** Mixed effects regression results of the effect of age and treatment type on vegetation diversity metrics of the over- and understory strata

| Stand age and residue treatment Analysis of Covariance (ANCOVA) results |        |         |                   |       |         |                                   |        |         |       |
|---|--------|---------|-------------------|-------|---------|-----------------------------------|--------|---------|-------|
| Stand age (years)   |        |         | Residue treatment |       |         | Stand age : Treatment interaction |        |         |       |
| df  | F      | p value | df                | F     | p value | df                                | F      | p value |       |
| <b>Overstory strata</b>   |        |         |                   |       |         |                                   |        |         |       |
| Species Richness  | 1, 259 | 0.779   | 0.378             | 1, 27 | 0.030   | 0.865                             | 1, 259 | 0.186   | 0.667 |
| Shannon's Divesity Index  | 1, 259 | 1.210   | 0.272             | 1, 27 | 0.293   | 0.593                             | 1, 259 | 0.091   | 0.763 |
| Species Evenness  | 1, 259 | 2.068   | 0.152             | 1, 27 | 0.564   | 0.459                             | 1, 259 | 0.144   | 0.705 |
| <b>Understory strata</b>  |        |         |                   |       |         |                                   |        |         |       |
| Species Richness  | 1, 256 | 32.494  | < <b>0.001</b>    | 2, 26 | 5.832   | <b>0.008</b>                      | 2, 256 | 1.735   | 0.179 |
| Shannon's Divesity Index  | 1, 256 | 12.264  | < <b>0.001</b>    | 2, 26 | 3.198   | 0.057                             | 2, 256 | 0.431   | 0.650 |
| Species Evenness  | 1, 256 | 3.342   | 0.069             | 2, 26 | 1.571   | 0.227                             | 2, 256 | 0.695   | 0.500 |



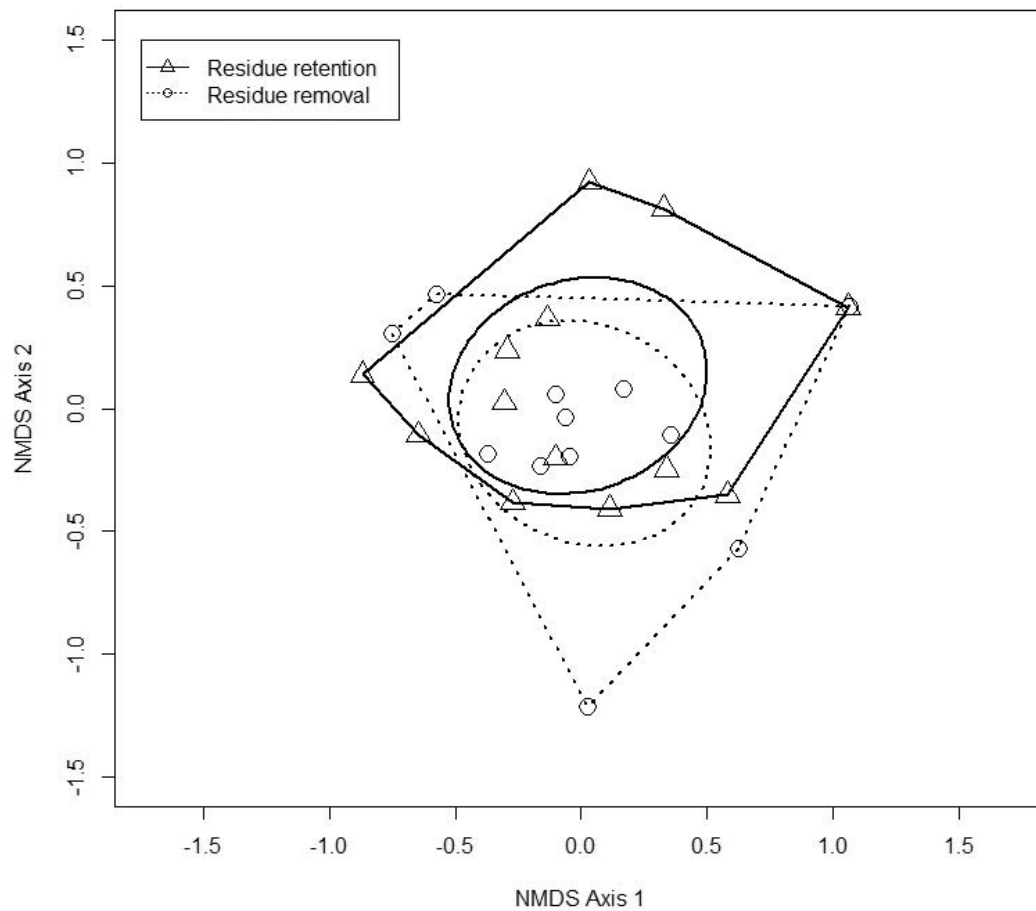
**Figure 4.1** Temporal and residue treatment effects on vegetation biodiversity metrics by strata. Panels a-c correspond to overstory (a) species richness, (b) Shannon's index, and (c), species evenness. Panels d-f represent understory species richness (d), Shannon's index (e), and species evenness (f). Only significant ( $p < 0.05$ ) regression surfaces are displayed. X values are uniformly jittered to avoid overplotting and aid in visual interpretation.

In contrast to the lack of observable patterns in the overstory strata, mixed effects regression models of understory strata diversity metrics detected a temporal effect in species richness ( $p < 0.001$ ). Understory species richness in the residue removal stands exhibited higher initial values post-harvest ( $p = 0.008$ ), as illustrated in the model intercept, while there was no difference found in temporal patterns between residue treatment type ( $p = 0.179$ ) (Figure 4.1d; Table 4.2). These differences may be attributed to a higher abundance of various graminoid species and the presence of vine species (*Parthenocissus quinquefolia* – L. Planch) in two of the sampled stands in the WTH treatment.

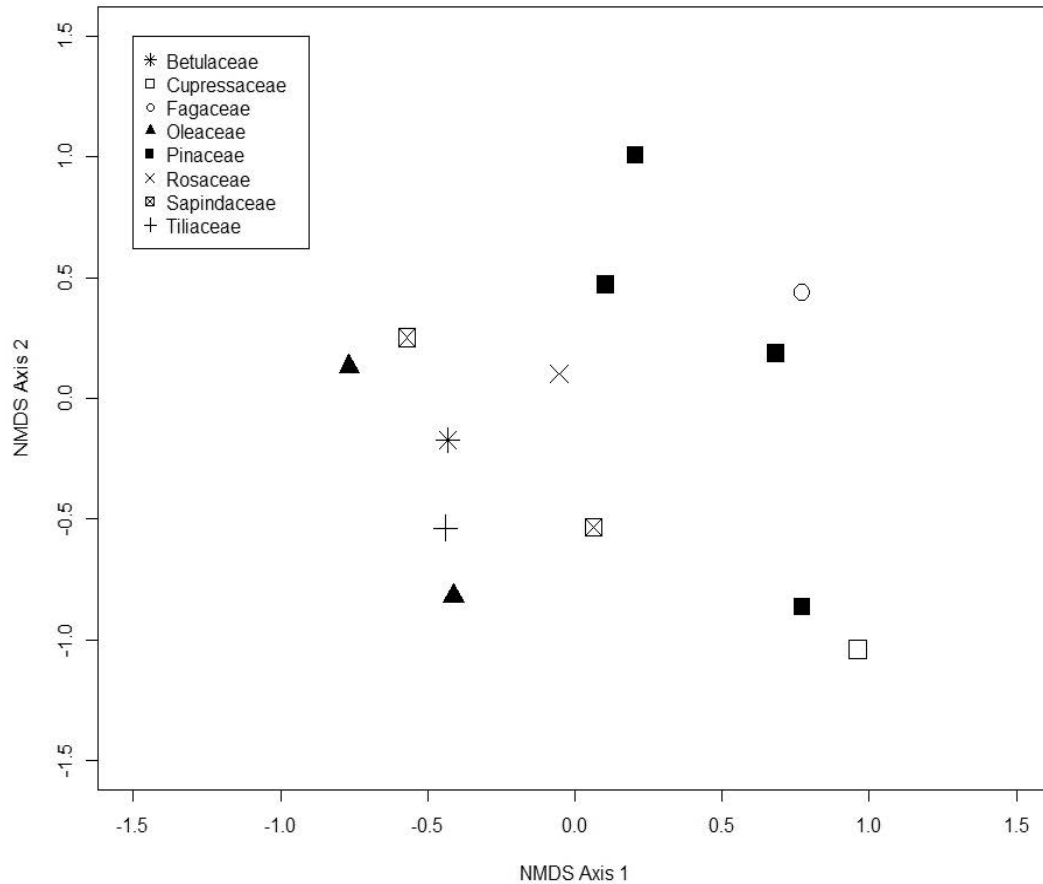
There were no observations of exotic species in stands of either treatment types. *Verbascum thapus* (L.) and *Cirsium arvense* (L. Scop.) were found in stands of both SOH and WTH treatments, while *Rosa multiflora* (Thunb.), *Tanacetum vulgare* (L.), and *Rhamnus cathartica* (L.) were found only in the SOH stands, and *Leucanthemum vulgare* (L.), and *Hypericum perforatum* (L.), were found exclusively in the WTH stands. Understory species diversity increased through stand rotation ( $p < 0.001$ ), however an effect of residue treatment type was not detected ( $p = 0.057$ ), and the temporal effect was represented with a single linear model (Figure 4.1e). Finally, there were no detectable temporal or treatment patterns in understory species evenness across stand rotation (Figure 4.1f; Table 4.2).

**Table 4.3** Non-metric multidimensional scaling results of the environmental variables correlation to the ordination axes 1 and 2 of the overstory strata

| Environmental variable       | Axis 1   |          | Axis 2   |          |
|------------------------------|----------|----------|----------|----------|
|                              | <i>r</i> | <i>p</i> | <i>r</i> | <i>p</i> |
| <b>Stand attributes</b>      |          |          |          |          |
| Stand age                    | 0.077    | 0.714    | -0.176   | 0.400    |
| Elevation                    | 0.142    | 0.498    | 0.127    | 0.545    |
| <b>Overstory conditions</b>  |          |          |          |          |
| Canopy closure               | -0.195   | 0.350    | -0.320   | 0.119    |
| Aspen stand density index    | -0.118   | 0.574    | -0.107   | 0.611    |
| Relative aspen stocking      | -0.064   | 0.761    | 0.288    | 0.163    |
| Relative aspen basal area    | 0.254    | 0.221    | -0.012   | 0.955    |
| <b>Understory strata</b>     |          |          |          |          |
| Understory species richness  | 0.104    | 0.621    | 0.017    | 0.936    |
| Understory species diversity | 0.125    | 0.552    | -0.053   | 0.801    |
| Understory species evenness  | 0.105    | 0.617    | 0.093    | 0.658    |
| <b>LFH Horizon</b>           |          |          |          |          |
| Fine woody debris cover      | 0.109    | 0.604    | 0.342    | 0.094    |
| Down dead wood cover         | -0.226   | 0.277    | -0.120   | 0.568    |
| Leaf litter cover            | -0.099   | 0.638    | -0.064   | 0.761    |
| <b>Mineral soil</b>          |          |          |          |          |
| Total soil carbon            | 0.043    | 0.838    | 0.123    | 0.558    |
| Total soil nitrogen          | -0.022   | 0.917    | 0.109    | 0.604    |
| C:N                          | 0.100    | 0.634    | -0.220   | 0.291    |
| Exchangeable soil Ca         | -0.038   | 0.857    | 0.064    | 0.761    |
| Exchangeable soil Mg         | -0.028   | 0.894    | 0.015    | 0.943    |
| Exchangeable soil K          | -0.102   | 0.628    | -0.208   | 0.318    |
| Proportional $r^2$           | 0.187    |          | 0.150    |          |
| Cumulative $r^2$             | 0.187    |          | 0.337    |          |



**Figure 4.2** Non-metric multidimensional scaling ordination sample plot scores for Axes 1 and 2 for the overstory community. Triangles and solid lines represent SOH plots while circles and dotted lines represent WTH plots. Ellipses represent the standard deviation of the group centroid, and the hull draws a polygon around all items in the sampling group.



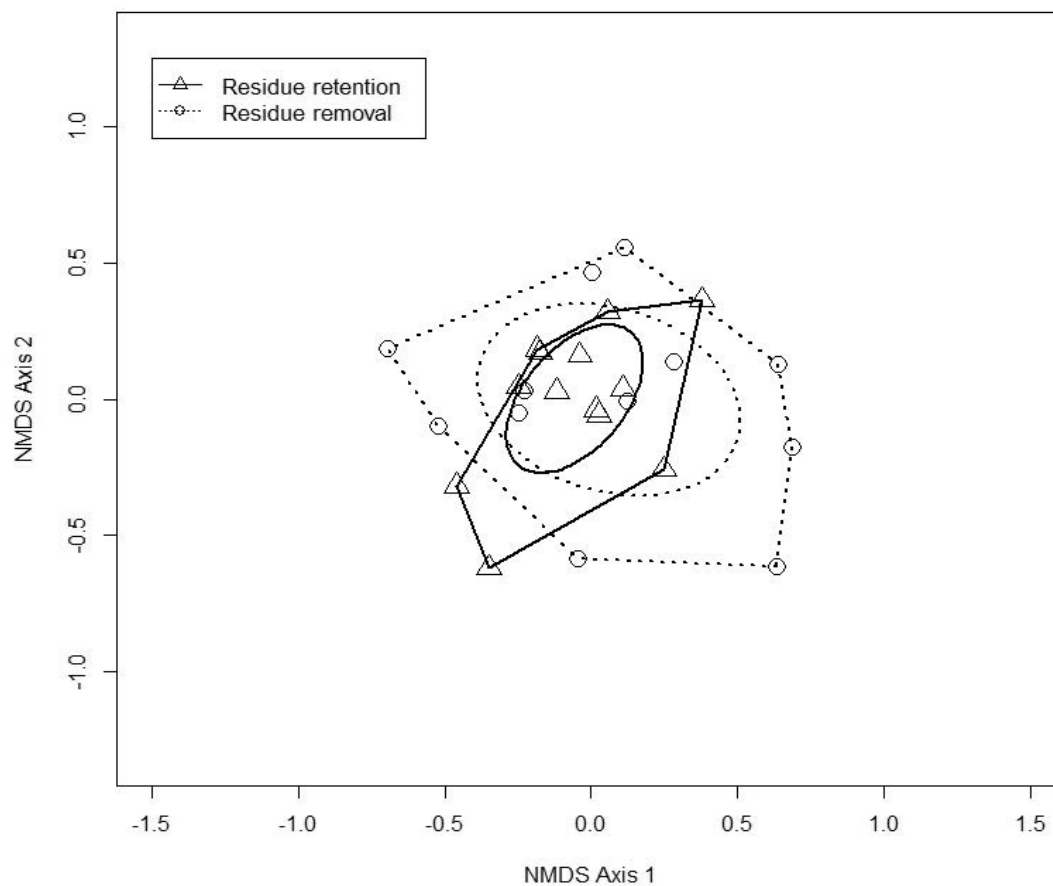
**Figure 4.3** Non-metric multidimensional scaling ordination of tree family types plotted in sample space. Species are summarized according to family.

The understory NMS resulted in a three dimensional solution (final stress = 14.6), with Axis 1-3 representing 0.120, 0.106, and 0.079 of the variance ( $r^2 = 0.305$ ) (Figure 4.4 and Table 4.4). While each individual species varied in their correlation to the three axes, cryptogams, exotics, and graminoids were most strongly correlated with Axis 1. Forbs, shrubs, and vines were most strongly correlated with Axis 2. Assessment of the environmental variables suggested a weak correlation between DDW and leaf litter cover with Axis 1, while Axis 2 was found to be strongly related to soil nitrogen, and to a lesser extent, stand elevation. Tests revealed weak correlations of both overstory species richness and evenness to Axis 3, suggesting dynamic interactions between forest strata (Table 4.4). Plotting species guilds in plot space revealed the association of forbs, shrubs, and graminoids with ordination axes (Figure 4.5, Table 4.6). Understory species composition between the residue treatment groups did not differ significantly according to MRPP results ( $A = 0.007$ ;  $p = 0.133$ ). Finally, results from the test of multivariate dispersions revealed a difference in distance to group centroid between treatment types ( $F = 6.125$ ,  $p = 0.019$ , 999 permutations). Distances to centroids for the individual treatment units were estimated at 0.377 and 0.303 for the WTH and SOH groups.

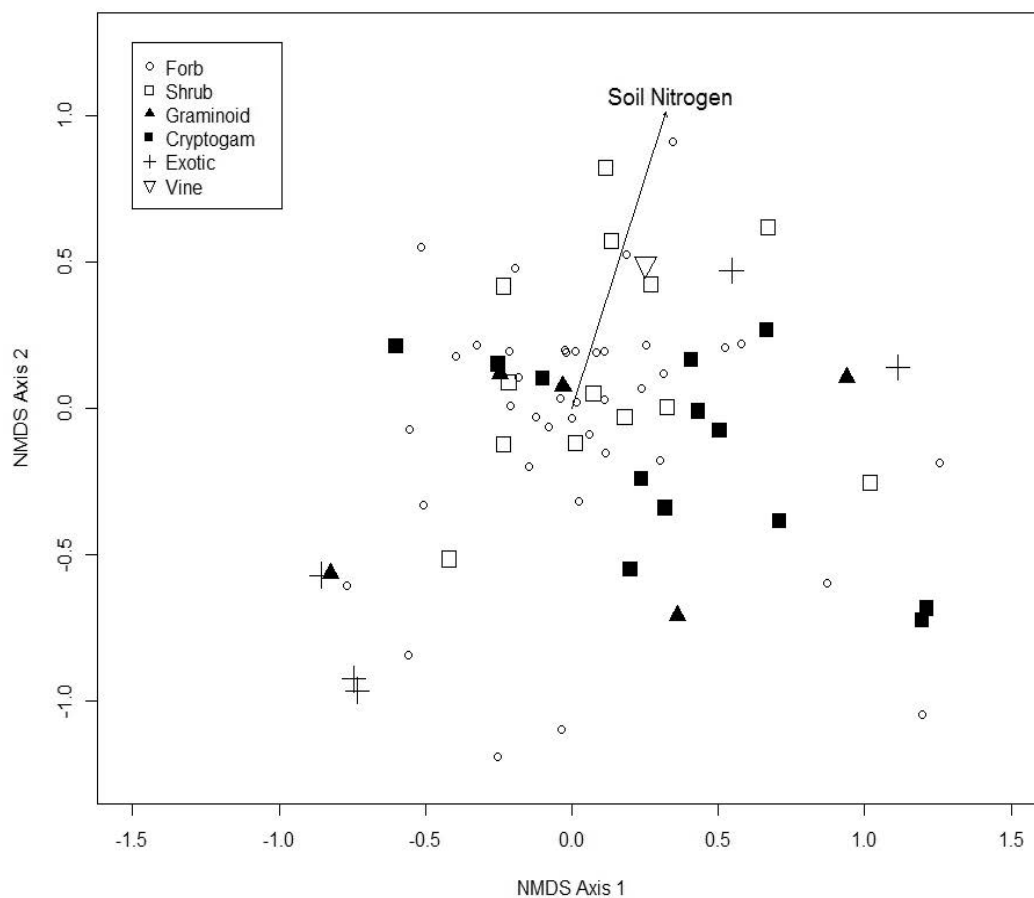


**Table 4.4** Non-metric multidimensional scaling results of the environmental variables correlation to the ordination axes 1-3 of the understory strata

| Environmental variable      | Axis 1 |       | Axis 2 |              | Axis 3 |       |
|-----------------------------|--------|-------|--------|--------------|--------|-------|
|                             | r      | p     | r      | p            | r      | p     |
| <b>Stand attributes</b>     |        |       |        |              |        |       |
| Stand age                   | -0.076 | 0.718 | 0.021  | 0.921        | 0.005  | 0.981 |
| Elevation                   | -0.135 | 0.520 | -0.358 | 0.079        | -0.147 | 0.483 |
| <b>Overstory conditions</b> |        |       |        |              |        |       |
| Canopy closure              | -0.060 | 0.776 | -0.140 | 0.504        | 0.103  | 0.624 |
| Aspen stand density index   | 0.071  | 0.736 | -0.203 | 0.330        | 0.246  | 0.236 |
| Relative aspen stocking     | 0.044  | 0.835 | -0.212 | 0.609        | 0.062  | 0.768 |
| Relative aspen basal area   | -0.117 | 0.397 | -0.227 | 0.275        | -0.182 | 0.389 |
| Overstory species richness  | 0.154  | 0.462 | 0.221  | 0.288        | 0.376  | 0.064 |
| Overstory species diversity | 0.130  | 0.536 | 0.318  | 0.121        | -0.130 | 0.536 |
| Overstory species evenness  | 0.090  | 0.669 | 0.282  | 0.172        | -0.357 | 0.080 |
| <b>LFH Horizon</b>          |        |       |        |              |        |       |
| Fine woody debris cover     | 0.279  | 0.177 | 0.241  | 0.246        | -0.214 | 0.304 |
| Down dead wood cover        | 0.348  | 0.088 | -0.009 | 0.966        | -0.128 | 0.542 |
| Leaf litter cover           | -0.359 | 0.078 | -0.227 | 0.275        | 0.170  | 0.417 |
| <b>Mineral soil</b>         |        |       |        |              |        |       |
| Total soil carbon           | 0.152  | 0.468 | 0.307  | 0.136        | 0.179  | 0.392 |
| Total soil nitrogen         | 0.124  | 0.555 | 0.444  | <b>0.022</b> | 0.046  | 0.827 |
| C:N                         | -0.083 | 0.693 | -0.220 | 0.291        | 0.005  | 0.981 |
| Exchangeable soil Ca        | 0.103  | 0.624 | 0.317  | 0.123        | 0.115  | 0.584 |
| Exchangeable soil Mg        | 0.071  | 0.736 | 0.256  | 0.217        | 0.077  | 0.714 |
| Exchangeable soil K         | -0.074 | 0.725 | 0.163  | 0.436        | 0.016  | 0.939 |
| Proportional r <sup>2</sup> | 0.120  |       | 0.106  |              | 0.079  |       |
| Cumulative r <sup>2</sup>   | 0.120  |       | 0.226  |              | 0.305  |       |



**Figure 4.4** Non-metric multidimensional scaling ordination sample plot scores for Axes 1 and 2 of the understory strata. Triangles and solid lines represent SOH plots while circles and dotted lines represent WTH plots. Ellipses represent the standard deviation of the group centroid, and the hull draws a polygon around all items in the sampling group.



**Figure 4.5** Non-metric multidimensional scaling ordination of plant guilds plotted in sample space. The environmental gradient associated with NMDS Axis 2 is plotted as a vector to illustrate correlation with individual guild.

## 4.4 Discussion

### 4.4.1 Overstory

Harvest residues and DDW can influence forest vegetation diversity and structure through additions to soil nutrient resources and providing physical variation and seedbed substrates in the forest floor environment (Lee and Sturges 2001; Wijesinghe et al., 2005; Thiffault et al., 2011). Overstory strata diversity appeared to be independent of both residue treatment and time since disturbance. This may be partly attributable to the presence of advance regeneration at the time of harvest and site partitioning according to shading regimes of observed species at the time of canopy closure (Palik and Pregitzer, 1995).

This study suggests a moderate correlation of FWD to the overstory tree community. Post-hoc results show a strong negative correlation of FWD and *Acer rubrum*, which may be explained by a lurking temporal effect as the species becomes more abundant through time while FWD loads decrease, rather than a direct physical influence, or may be an artifact of the chronosequence study design. The results of the MRPP and tests of multivariate dispersions suggest that there was no discernible difference in overstory species composition or heterogeneity according to residue treatment group. The lack of differences may reflect the limited temporal residence of residues (Alban and Perala, 1992), rapid early growth rates of tree species, and a variety of shading tolerance of competing species in secondary succession trajectories. While the retention of harvest residues to the forest floor has been suggested as a way to provide refugia from ungulate browse and provide favorable micro conditions to seedlings (Ripple and Larsen, 2001), no differences were detected in tree community diversity, composition, or structure when residues were captured. Collectively, these results suggest that residue treatment has no observed effect on overstory tree species diversity and community structure.

### 4.4.2 Understory

Understory species richness was significantly greater in stands where residues were removed across the commercial rotation, due to higher occurrences of graminoids, shrubs, and particularly, vine species associated with the successional trajectory of this soil type (Burger and Kotar, 2003). However, there was only weak support for a difference in species diversity between treatment types. Olsson and Staaf (1995) observed higher occurrences of graminoids in SOH thinning treatments of *Picea abies* and *Pinus sylvestris* stands in northern Sweden, which they attributed to presence of logging residues. Despite the lack of treatment difference in species diversity trends, results detected an increase in diversity and richness through time and across stand rotation, which further reflects the initial decline in these metrics following timber harvesting (Figure 4.1a,b). Other studies have shown an increase in vascular plant richness and diversity through time due to a decline in nutrient demand by the overstory

community (Grime, 2006), increasing heterogeneity of soil resources, and consequently, species niche partitioning (Grubb, 1977; Tilman, 1985; Bartels and Chen, 2010).

Alterations in community composition are noted to be reflective of harvest level of intensity (Frederickson et al., 1999; Peltzer et al., 2000; Campione et al., 2012; but see Bock and Van Rees, 2002). Despite the observed increase in species richness, results failed to detect a significant difference in community composition between residue treatment groups. However, results from the permutation test indicate a difference in community heterogeneity, and indicates higher variance in proportional representation of individual species within the community in WTH treatments. In disturbed forests, nutrient and environmental resource heterogeneity has been suggested to be an equally if not more influential component in community diversity and structure as resource quantity (Bartels and Chen, 2010). As results from Chapter 2 of this work show a 42% reduction in harvest residues under WTH methods, this moderate reduction in residues may result in increased soil scarification which in turn promotes occupancy of a variety of plant functional types.

The increased levels of structural heterogeneity in stands with residue removal may reflect an unintended and overlooked influence of residue management on plant communities. Additional site disturbance from either; (a) soil scarification from whole-tree skidding, or (b) multiple passes from forwarding equipment for topwood removal in residue removal treatments, may indirectly influence vegetation communities. The increased soil disturbance associated with whole-tree skidding harvest operations has been shown to enhance post-harvest heterogeneity of soil resources and the understory community (Mou et al., 1993) and increase species diversity (Roberts and Zhu, 2002) through scarification and removal of the forest floor and competitors (Harvey and Brais, 2002). Likewise, with the increased use of cut-to-length (CTL) processing systems, multiple equipment passes for residue collection can increase site disturbance levels.

It is important to note that harvesting equipment that is currently in widespread use by logging contractors is not specifically designed for extraction of residues. Therefore even when residues are captured during harvest operations, studies in Great Lakes aspen forests have only noted a reduction in residue volume and biomass estimates by approximately 30-35 % (Klockow et al., 2013; Rittenhouse et al., 2013). This can further be attributed to incidental breakage during felling and skidding activities in tree-length harvest operations, and may result in a more heterogeneous and even distribution of residues across the site (MFRC, 2007). In contrast, the retention of logging residues, especially when using CTL processors, can result in concentrated accumulations that correspond to harvest unit layout and specific equipment utilized (Chapter 5 of this work). Heavy slash and residue concentrations can decrease soil temperatures (Slesak, 2013), and increase surface air temperature through limiting air movement and convective heating (Zabowski et al., 2000). Further, the addition of slash to the forest floor can minimize soil disturbance and scarification associated with harvesting, thereby decreasing the frequency of scarified microsites and colonization by ruderal species. Therefore, residue retention may be reducing stand-level resource heterogeneity, which is

being reflected by lower diversity values and relatively homogenous representation of species when compared to stands with residues removed.

It was unsurprising to observe the strong correlation between total soil nitrogen with the understory community, as nitrogen availability is critical to nutrient cycling rates, species specific competition, and ultimately, vegetation community dynamics and stability (Tilman, 1985; Aerts, 1999; Barnes et al., 1998; Gilliam, 2006; Chapin et al., 2011). The high standard deviation estimates of total soil nitrogen (Table 4.1) further supports the importance of resource heterogeneity in vegetation community dynamics. The weak correlation of stand elevation with ordination variance is explained by the inclusion of sites across a minor geographic range included in this study, reflecting a random artifact of the chronosequence sampling design rather than a localized topographic effect.

These results reveal moderate correlations between overstory species richness and evenness to the understory community. The primary influences of overstory trees on herbaceous and shrub communities are primarily related to light quality and transmittance (Messier et al., 1998), substrate characteristics and associated nutrient cycling (Harmon et al., 1986), and stem flow/through fall chemistry (Barbier et al., 2008). Berger and Puettmann (2000) discovered positive relationships between understory species diversity values and proportional aspen basal area while these trends were reversed for increasing hardwood basal area in Northern Minnesota, however the proportion of hardwood to conifer species is further influenced by soil type and productivity class (Tilman, 1985). Therefore, developing silvicultural prescriptions aimed at enhancing biodiversity and community structure need to be site and strata specific, as disturbances and edaphic conditions may affect these communities at different temporal and spatial scales.

Results further highlight moderate correlations of the understory community to DDW and leaf litter cover. Large diameter particles of DDW can provide microsites that are favorable to plant establishment and growing conditions (Harmon et al. 1986; Scheller and Mladenoff, 2002). Lee and Sturges (2001) reported that species assemblages on logs > 20 cm in diameter were different than those occurring on the adjacent forest floor, however these initial differences decreased through time and corresponding progression through decomposition. It is imperative to emphasize here that in the managed Great Lakes aspen forest type, inputs of large diameter DDW are limited to those occurring through natural mortality of legacy trees or self-thinning of the aspen cohort, and that residues associated with harvest are consistently < 10 cm. Therefore, the removal of residues should not be expected to negatively impact ecological benefits associated with DDW presence.

The role of leaf litter cover can influence forest vegetation communities through substrate quality, decomposition rate, and subsequent nutrient cycling (Fischer and Binkley, 2000). A meta analyses conducted by Xiong and Nilsson (1999) concluded that plant leaf litter can inhibit successful germination through physical and allelopathic impacts, and ultimately effect community composition and dynamics. Given moderate overstory

species richness levels throughout the stand rotation (Figure 4.1a), leaf litter of various species may further be contributing to understory heterogeneity.

The long-standing history of production oriented management on the land base used in this study has created multiple age classes and harvest units connected by heavily trafficked timber harvest roads. These roads may serve as vectors of propagule dispersion and increase stand susceptibility to invasion by exotic or noxious competitors (Birdsall et al., 2012; Moore et al., 2013), thereby altering community composition and structure. Observations of exotic species highlight the need to integrate assessment of vegetation composition and structure in stand prescriptions and forest management goals in addition to goals of conserving biodiversity. These additional metrics can extend silvicultural options to managers to meet long-term stand objectives.

Stands with residue removal included in this study had lower mean values of FWD and measured soil nutrients (Chapter 2). While the variability is relatively high as indicated by the standard deviations of treatment means in Table 4.1, these values suggest that residue removal could be triggering a decrease in soil nutrient pools. Nutrients associated with harvest residues contribute directly to soil resource availability (Alban, 1985), therefore the distribution of residues can affect resource heterogeneity, and can be equally important in vegetation dynamics and species diversity (Bartels and Chen, 2010). Boyle and Ek (1973) suggested declines in stand level Ca over nine WTH rotations; however, Silkworth and Grigal (1982) predicted deficiency of Mg and N within two rotations of residue removals. Results from this study suggest that nutrient losses are soil type specific, and that vegetation communities along the Onaway soil type may be more influenced by K levels than other cations. Ongoing results from the North American Long Term Soil Productivity study indicate a significant decrease in available K ten years after residue removal, and a minor decrease in C, N, Mg, and Na on a Haplic Glossudalf soil, which is similar in diagnostics to the Onaway and reported findings here (Voldseth et al., 2011).

## **4.5 Conclusions**

The removal of harvest residues has varying effects on forest components, and is likely cover type and site dependent. In this study, the removal of residues had no effect on the overstory strata; however, WTH treatments increased understory species richness and community heterogeneity. These patterns can be attributed to increased disturbance from WTH logging operations, which in turn alters the spatial distribution of logging residues and may result in increased soil scarification. Harvest unit layout, operating equipment, and residue management should be strongly considered when assessing goals to control or enhance biodiversity and community composition of forest strata, as residues can directly affect growing conditions and resource availability. Further, management goals of biodiversity and community structure should be strata specific, as results show that understory shrub and herbaceous species can exhibit significant changes in structure and richness that are independent of the overstory tree community. A reduction in soil nutrient resources was detected when residues were captured, especially K, however this

was not observed to be correlated with patterns in vegetation communities. In closing, these results suggest that aspen forest vegetation communities on productive soils may be insusceptible to degradation through the removal of residues; however communities along differing soil types require further investigation.



## Chapter 5 <sup>4</sup>

### Incidental effects of contemporary harvest systems on stand regeneration

#### 5.1 Introduction

Advances in forest harvesting and processing technology have increased extraction and utilization efficiency of timber resources while seeking to minimize production costs. However, harvesting operations can have notable effects on forest soil health and stand growing conditions (Alban, 1991; Marshall, 2000) and may result in alterations in vegetation structure and soil conditions, and fluctuations in associated nutrient cycling rates (Hendrickson et al., 1989; Likens and Bormann, 2001). Post-harvest regeneration and rapid early growth is fundamental to silvicultural programs to promote early stand development and minimize costly intervention. As such, state devised Best Management Practices (BMPs) and third party certification programs have become prominent in large scale forest management programs to ensure long-term site productivity. These guidelines commonly include recommendations for equipment operations and maintenance, harvest unit layout, and protection of soil resources and water quality to minimize detrimental impacts on stand conditions and optimize regeneration. Cumulatively, these guidelines may inadvertently result in management constraints increase production costs and potentially reduce timber yields (Van Deusen et al., 2012).

Aspen forests (SAF Cover type 16) are an expansive forest cover type in the Great Lakes region, and have increased in extent since widespread hand-felling and subsequent slash fires in the early 20<sup>th</sup> century. This forest type covers approximately 5 million hectares across the region today, a large portion of which is held under industrial ownership (FIA, 2013). These forests have long been managed for their supply to the bolt and pulp fiber markets, and traditional methods of hand-felling were replaced by the widespread mechanization of forest harvesting since the 1960-70's. Harvest disturbance studies have reported adverse effects to forest soil conditions and regeneration challenges posed by ground based feller-bunching and skidding equipment in tree-length removal operations. Multiple passes of harvesting equipment has resulted in soil compaction (Shetron et al., 1988) and scarification (Bates et al., 1992), and may damage shallow root systems of the aspen cohort (Shepperd, 1993). Heavy equipment traffic used in tree-length systems can result in increased soil disturbance and lead to a loss of stocking density and height growth in aspen regeneration (Bates et al., 1992; Zenner et al., 2007; Puettmann et al., 2008).

Over the last two decades, harvesting technology has continued to progress, and reflects changes in silvicultural prescriptions, commodity products, and increased efforts to alleviate site impact. The advent of cut-to-length (CTL) processing equipment has

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<sup>4</sup> The material contained in this chapter is in preparation to be submitted to *Forest Science*

enabled managers and loggers to minimize damage to residual trees during thinning-operations (Puttock et al., 2005), protect advance regeneration during harvesting (Waters et al., 2004), and decrease transport distance to landings (Han et al., 2009). Further, this change in harvest equipment has indirectly resulted in a change in residue management. Traditional feller-buncher and skidder operations transport whole trees to a central landing, where a slasher or de-limber bucks and sorts logs to appropriate product lengths to a minimum merchantable diameter (commonly 7.5-10 cm. in Lake States). With these systems, harvest residues are piled at the landing, dispersed back into the forest matrix (mainly on skid trails), or chipped and used for heat and electricity co-generation in local production mills. In contrast, during CTL operations, the logging slash is either: (i) retained at the site of felling and processing within the forest stand at the time of felling and only the merchantable logs are transported to the roadside, or; (ii) piled adjacent to processed logs and removed with an additional pass by a forwarder. When residues are not utilized in CTL systems, they are commonly left on the forest floor in front of the path of equipment in efforts to mitigate soil compaction (Han et al., 2009), and damage to fine roots posed by equipment

Renewed interest in utilizing harvest residues as a bioenergy feedstock has resulted in new research programs and revisiting some of the original work conducted in whole-tree harvesting studies of the 1970-80's. Some studies have shown the removal of harvest residues and associated nutrients exceeds natural inputs, result in growth deficiencies and potentially threatens long-term site productivity (Silkworth and Grigal, 1982; Helmisaari et al., 2011; Thiffault et al., 2011). In response to these findings, states in the Great Lakes region have developed highly generalized biomass management guidelines and recommend retention of roughly 10-30% of harvested tree topwood (MFRC 2007; Herrick et al., 2009; MDNRE, 2010). Considerably less attention has been paid, however, to harvest operations and subsequent spatial orientation of harvest residues in stand regeneration processes. Heavy slash loads following WTH methods in aspen clearcuts have been shown to reduce sucker regeneration, however the impacts are fairly limited in areal extent (Bella, 1986; Bates et al., 1989). Therefore, while the retention of logging residues may be reducing soil and forest floor disturbance, higher slash loads in CTL operations (Benjamin et al., 2013; Rittenhouse et al., 2013) may have a potentially overlooked influence on regeneration patterns.

The goal of this study was to assess the presence, spatial variation, and influence of residues in regenerating aspen stands harvested with CTL equipment. The first objective was to quantify residues and sucker regeneration to determine if variability in these estimates can be attributed to spatially explicit patterns of timber harvesting operations. My second objective was to examine soil bulk density, total soil C and N, and forest floor C, directly on and adjacent to CTL equipment tracks. This approach investigates the effectiveness of CTL harvesting in mitigating forest soil disturbance through the addition of residues to the forest floor during harvesting operations. This also aims to isolate any effect on regeneration patterns (such as soil compaction or residues loads). Finally, an additional objective was to assess early stand development to form long-term predictions of persistence of residues and trends in crop-tree height growth. It was hypothesized that

harvest operations result in high levels of residue concentration confined to distinct rows that correspond to equipment trafficking. Residue retention in equipment rows was expected to mitigate soil compaction, and increase soil C and N relative to adjacent areas off of the equipment trail, while these patterns would be reversed for regeneration patterns and forest floor biomass (i.e. fewer and smaller saplings on CTL trail).

## 5.2 Methodology

### 5.2.1 Study area

Forest stands included in the study are located in Baraga, Delta, Dickinson, and Menominee counties, in the Upper Peninsula of Michigan and are owned and managed by Plum Creek Timber Company. The study area receives approximately 724 mm annual precipitation, most in the growing season, with mean air temperatures ranging from -13° C to 24° C in January and July, respectively (NCDC, 2015). Forest stands are primarily composed of *Populus tremuloides* (Michx.), with some minor mixed occurrences of *Populus grandidentata* (M.) and *Populus balsamifera* (L.). Other minor tree species found across the study sites included *Acer rubrum* (L.), *Acer saccharum* (Marshall), *Abies balsamea* (L. - Mill.), and *Prunus serotina* (Ehrh.). Common herbaceous and shrub species included *Pteridium aquilinum* (L. - Kuhn), *Solidago* spp. (L.), *Carex* spp. (L.), *Rubus* spp. (L.), *Hamamelis Virginia* (L.), *Corylus cornuta* (Marshall), and *Equisetum* (L.). Stands are managed with coppice silvicultural methods and clearcut to initiate aspen suckering and reclamation of the site following harvest, and reach biological rotation age at approximately 35-40 years. These forests have been under aspen cover since the expansive cut-over period of the early 20<sup>th</sup> century, due to intolerant shading characteristics and exposed mineral seedbed requirements of the early seral *Populus* genus and the management history of the land base.

### 5.2.2 Study design

This chapter is part of a larger study investigating the long-term effect of residue removal on aspen forest productivity across the 40 yr commercial rotation. Stands were systematically selected from an inventory database supplied by Plum Creek Timber Company. To be considered for sampling, forest stands had to adhere to specific criteria: (i) aspen forest cover type (SAF type 16); (ii) records of residue treatment and harvest year, and; (iii) met a minimum of 10 ha in area. Stands meeting these specifications were subjected to field reconnaissance. Several stands identified as treated with residue retention were visually noted to have regularly spaced (~10-15 m) strips that exhibited reductions in height and stocking. Upon further investigation, I discovered that these areas were very high in residue concentration in comparison to areas with high regeneration density. I compiled a list of stands obtained through the Plum Creek Timber database as having records of residue retention and cross-referenced these with PCL staff forester's records of equipment type. A total of seven stands were identified ( $n = 7$ ) where I could confirm CTL was the sole harvesting equipment utilized. I used publicly

available imagery in ESRI ArcMap v 10.1 to determine mean distance spacing between equipment trails, and confirmed these estimates in the field to formulate a sampling protocol to characterize the observed spatial patterns. Using the sampling approach similar to those employed by Han (2006) and Puettmann et al., (2008) to investigate WTH and skidding impacts on aspen stand regeneration and soil patterns of northern Minnesota, I designed the sampling plots to run across and perpendicular to the CTL equipment trail (Figure 5.1). A total of 10 plots within each stand of interest were established using the random point generator in ESRI ArcMap v 10.1 with a minimum criterion of 20 m from stand boundaries and  $> 20$  m between sampling points. CTL trails that were closest to random points were designated for sampling.

### *5.2.3 Field Methods*

Random points were located in the field using a recreational grade Garmin 76 ETrex Global Positioning System, from which the CTL trail with the minimum distance from the established point were identified visually. Identification of CTL trails was based upon machine rutting  $> 0.1$  m at the 3.1 m width of CTL machinery, and abundance and high concentration of harvest residues. Given the prominence of these features in the stand (Figure 5.1), there was no difficulty locating them in the field. At each designated trail, I established three subplots; one directly in the middle of the CTL trail, and two set at 1 and 5 m off of the outside of the CTL track (Figure 5.2). At each subplot, a 10 m<sup>2</sup> regeneration plot was used to record all regenerating suckers and saplings by species and height to the nearest 0.3 m with a height pole. Stems  $\geq 1.34$  m in height were measured for dbh (mm) with a digital caliper.

I used four Line Intersect Distance Sampling (LIDS) transects centered on the same plot as the regeneration survey, extending in cardinal directions with an adjusted limiting distance ( $k = 200$ ) to increase probability of inclusion of harvest residues ( $< 10$  cm. in diameter) in the tally (Affleck, 2008). Classification of decay classes according to Woodall and Monleon (2008) were used for each particle included in LIDS measurements. In addition, I used a 25 cm<sup>2</sup> quadrat to remove shrubs, herbaceous plants, detritus, and fine wood debris ( $< 2.54$  cm) to expose the organic layer of the soil. Once exposed, an AMS slide hammer was used to extract organic and mineral soil to a depth of 15 cm following procedures by Jurgensen et al., (1977). An additional sample of forest floor and forest soil directly in the equipment track of the trail was included to test bulk density.

For relative comparisons of potential findings, I utilized a dataset from a larger sampling framework to provide a comparison of trends observed in stands harvested with CTL equipment and residue retention to WTH methods with feller-bunchers and skidding equipment. Observations from the rotation chronosequence study utilized 40-100 m<sup>2</sup> regeneration subplots nested within 10 overstory (400 m<sup>2</sup>) plots. Aspen suckers were measured for height and dbh if  $> 1.34$  m in height in regeneration subplots, while all trees with dbh  $> 10$  cm in overstory plots were recorded by dbh with a subset of two dominant and co-dominant tree heights measured in each plot. I used a LIDS limiting distance of  $k$

= 800 to increase probability of including both residues and natural inputs to DDW pools. I limited comparison of CTL and WTH methods to regeneration height, density, and residues, due to the spatial orientation of the CTL sampling design while WTH stands were sampled randomly.

#### *5.2.4 Laboratory processing and soil analysis*

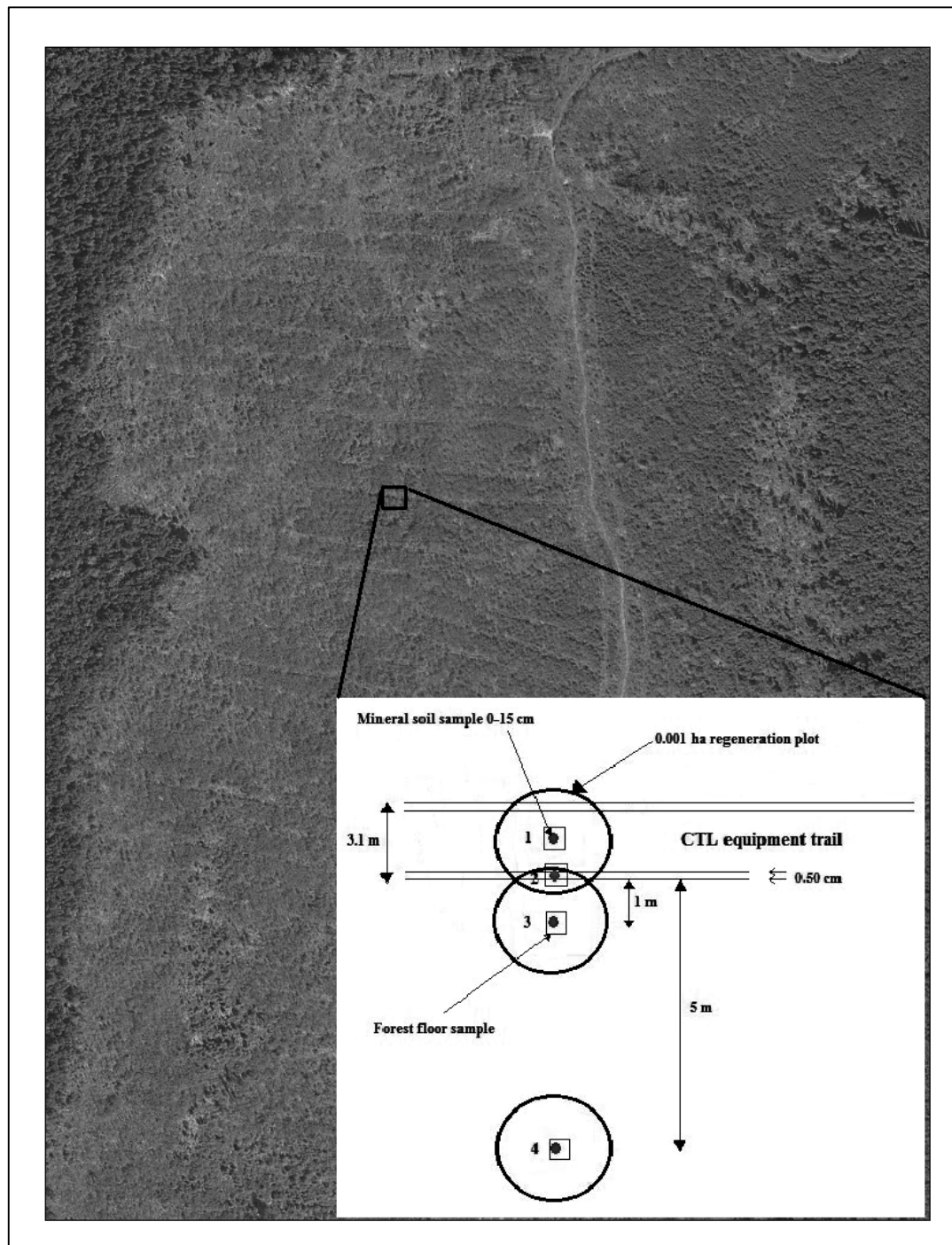
Soils collected in the field were frozen, stored, and transported to the Forest Soils laboratory at Michigan Technological University in Houghton, MI. All soils were oven dried at a temperature of 75° C for 48 hours to ensure consistent dry weight. Dry soils were sieved through a 2 mm mesh screen to separate all roots, rocks, and buried woody debris. Sieved soils were homogenized with a Wiley ball mill to a consistent texture and processed for total carbon and nitrogen with a Fisons NA 1500 Elemental Analyzer.

#### *5.2.5 Statistical Analysis*

Statistical analyses were performed in the R software environment (R Core Team, 2015). To assess temporal and spatial trends of residues and regenerating vegetation, I used the *nlme* package 3.1-119 (Pinheiro et al., 2015) with forest stand as a random effect, and treatments, stand age, and distance to skid trail (subplot) as fixed effects. LIDS transects were combined by subplot. Residue volume at each subplot was calculated according to Affleck (2008) and estimated residue biomass using density reduction factors according to observed decay class (Harmon et al., 2008). Stands were pooled and I used linear mixed effects models to test the effect of distance to trail, stand age, and an interaction effect on harvest residues and stand regeneration patterns with alpha specified at  $\alpha = 0.10$ . A log transformation of residue biomass and sapling density was performed for analysis and model residuals assessed with the *ggplot2* 1.0.0 (Wickham, 2009) package and *q-q* plots.

I assessed patterns in forest soil total carbon, total nitrogen, and bulk density ( $\text{g cm}^{-3}$ ), within CTL stands according to distance from equipment trail with mixed effects analysis of variance and post hoc orthogonal contrasts of subplot level observations. Subplots were pooled and a Tukey multiple comparisons test was used with the *mcp* function in R only if results from the *F*-test were significant. The relative contribution of soil and residue estimates to patterns in sucker height and density was tested using linear mixed effects models and ranking models using Akaike's Information Criterion (AIC) and Akaike weights (*w*).

Temporal trends in mean dominant height and residue biomass were summarized by subplot and compared to stand measurements obtained from stands treated with residue removals as part of a larger study design across the commercial rotation ( $n = 15$ ). In order to quantify any alteration in trajectory of measured stand variables, I used sequential  $F$  tests to compare each individual subplot to stands treated with WTH methods in the early stage of stand development (0-10 years). I extrapolated patterns in height growth in CTL stands across the entirety of the stand rotation with non-linear functions. Sit and Poulin-Costello's Catalogue of Curves (1994) aided in estimating non-linear trends in residue biomass and mean dominant height of WTH observations, and the *growthmodels* 1.2.0 package (Perez, 2013) was used to estimate model coefficients.



**Figure 5.1** Aerial photo of Stand 980010764 – Ensley Angelica soil type located in Delta County, Michigan. Imagery sourced from ESRI ArcMap 10.2. Inset depicts the sampling transect regime based on CTL trail center

**Table 5.1** Study locations, soil characteristics, dates of measurements, and estimated area of stand impact by equipment trails (AOI – Area Of Impact)

| Stand Number | Soil Type and Texture     | Latitude | Longitude | Stand age (years) | Stand area (ha) | Date installed | Estimated AOI % |
|--------------|---------------------------|----------|-----------|-------------------|-----------------|----------------|-----------------|
| 980009574    | Onaway fine sandy loam    | 45.9828  | -87.5446  | 1                 | 27.5            | Aug-13         | 27.6 (0.8)      |
| 980004569    | Onaway fine sandy loam    | 45.9169  | -87.3016  | 5                 | 36.4            | Aug-13         | 20.1 (1.8)      |
| 980004063    | Charlevoix coarse loam    | 45.8614  | -87.2543  | 6                 | 41.8            | Aug-13         | 22.3 (1.2)      |
| 980004047    | Onaway fine sandy loam    | 45.8638  | -87.2659  | 6                 | 33.0            | Jul-14         | 21.5 (1.6)      |
| 980010764    | Ensley-Angelica fine loam | 45.6644  | -87.2088  | 7                 | 44.3            | Aug-13         | 19.2 (2.3)      |
| 980002504    | Munising-Yalmer complex   | 46.8285  | -88.3289  | 11                | 10.8            | Jul-14         | 17.5 (0.9)      |
| 980002506    | Munising-Yalmer complex   | 46.8245  | -88.3240  | 11                | 26.2            | Jul-14         | 20.4 (0.9)      |

Numbers in parentheses indicate standard errors of the mean percentage of area of impact

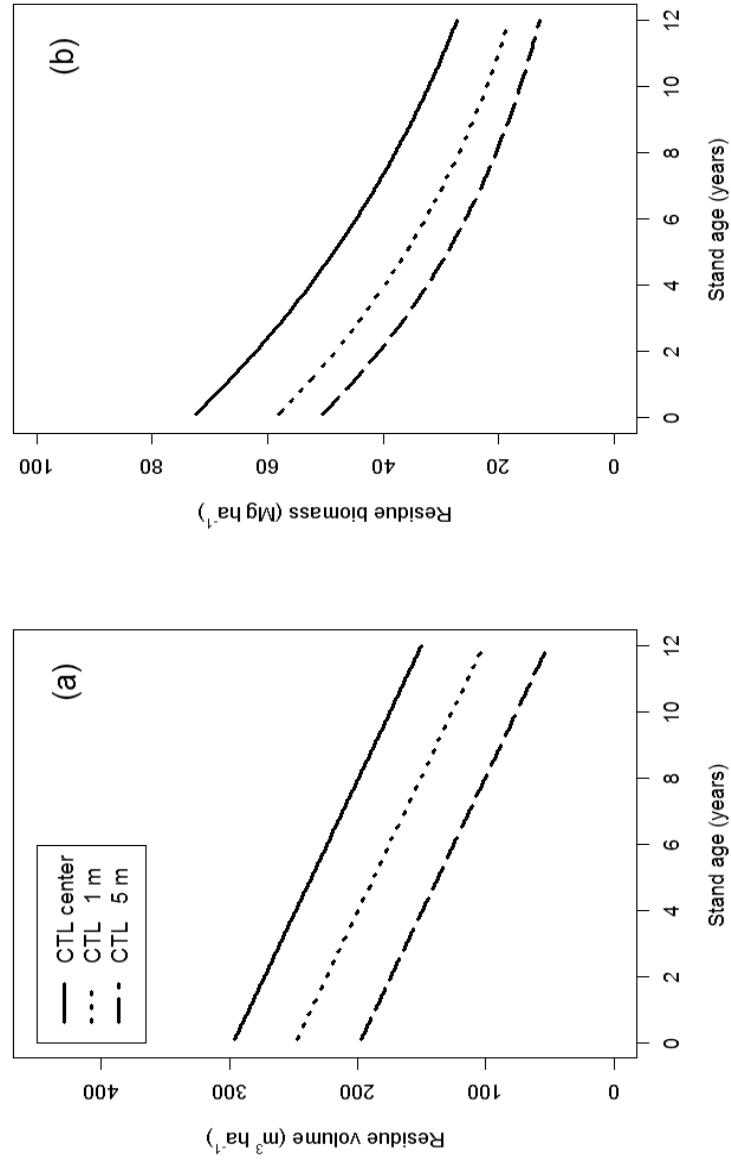


## 5.3 Results

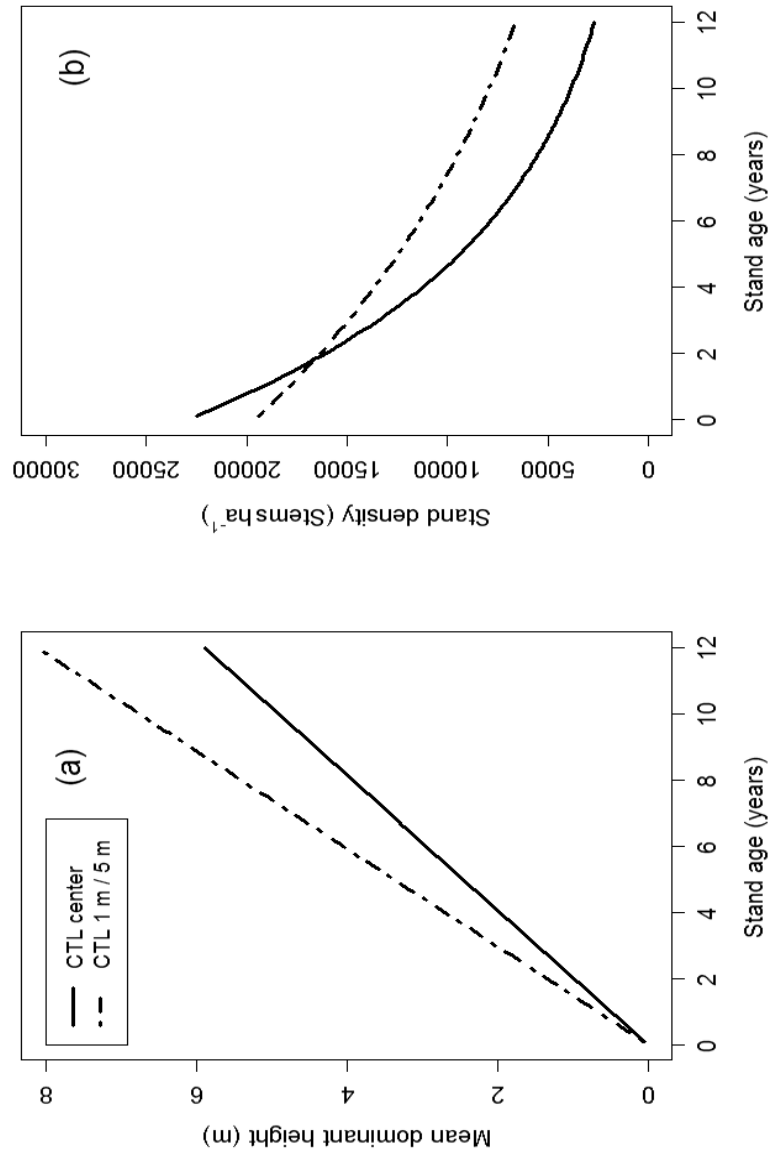
### 5.3.1 Within stand patterns of residues and regeneration

Results indicate a significant spatial correlation between residue volume and biomass and distance from CTL equipment trail ( $p < 0.001$ ). There was a clear temporal gradient in residue volume estimates, which varied in the intercept according to distance from trail ( $p = 0.049$ ) (Figure 5.1a). Immediately post-harvest, the highest levels of residue volume were found directly on the CTL trail at  $297.1 \text{ m}^3 \text{ ha}^{-1}$  ( $p = 0.044$ ). Subplots located 1 m off the CTL trail had the second highest residue volume estimates at  $248.1 \text{ m}^3 \text{ ha}^{-1}$ , which was greater ( $p < 0.001$ ) than those found 5 m off of trail at  $198.5 \text{ m}^3 \text{ ha}^{-1}$ . Models fit to observations suggest that these values decline at the same scale through time, as tests detected no difference in estimates of model slopes. The patterns observed for residue biomass were nearly identical to residue volume, with a strong influence of both distance to equipment trail ( $p < 0.001$ ) and time since harvest ( $p = 0.021$ ). Again, the highest residue biomass was found at the trail center at  $73.1 \text{ Mg ha}^{-1}$ , which was greater than biomass estimates at subplots 1 m ( $p = 0.098$ ) and 5 m ( $p < 0.001$ ). The declines in biomass through time were similar across subplots ( $p = 0.571$ ) (Figure 5.2).

Aspen sapling mean dominant height was significantly affected by distance to equipment trail ( $p < 0.001$ ) across all observed stands. A steeper slope in height growth across early stand development in areas located off of the CTL trail when compared to measurements taken directly on the trail ( $p < 0.001$ ), and there was no difference in mean dominant height between 1 or 5 m positions ( $p = 0.283$ ) (Figure 5.2a). Regeneration density was influenced by proximity to CTL trail ( $p < 0.001$ ). While density values immediately following harvest were initially higher in trail center ( $p = 0.002$ ), the slope on the center of the CTL trail was steeper ( $p = 0.015$ ) than regeneration density across the adjacent portions of the stand. These initially higher stem density values in CTL center and steeper declines through time are an artifact of the sampling design and fitting linear models to observations, rather than a lurking effect of self-thinning. From year 5 onward, I consistently detected lower stem density values in the center of the CTL trail than areas adjacent to the equipment trail (Figure 5.2b).



**Figure 5.2** Residue volume ( $\text{m}^3 \text{ha}^{-1}$ ) (panel a) and residue biomass ( $\text{Mg ha}^{-1}$ ) (panel b) observations by subplot and across observed stand age



**Figure 5.3** Aspen regeneration mean dominant height (m) (panel a) and stand density (stems  $\text{ha}^{-1}$ ) (panel b) observations by subplot and across observed age.

### *5.3.2 Spatial effects of residues to forest floor and mineral soil*

The retention of logging residues in distinct rows showed varying effect on forest floor and mineral soil conditions across all measured sites (Table 4.2). There was no consistent pattern between distance to CTL trail and forest floor mass across observed stands; however estimates of soil bulk density were highest in the track of the CTL trail in three of the seven tested sites. Models detected higher levels of soil C and N at trail center on two and four of the observed stands, respectively.

### *5.3.3 Residues and forest mineral soil effects on stand regeneration height and stocking density*

Similar to results in residue patterns, tests detected significant declines in regeneration density and mean aspen sucker height on the equipment trail across all stands included in this study. These trends suggest that regeneration height is consistently reduced up to 12 years post-harvest. However, results suggest that these patterns are reflective of increases in bulk density, rather than the physical impediment of confined harvest residues ( $w = 0.9985$ ; Table 4.4). Likewise, stand density models revealed that the addition of bulk density estimates to stand age to be the best predicting variable ( $w = 0.9974$ ). The addition of harvest residue biomass to the regeneration models failed to improve model quality.

**Table 5.2** Mixed effects analysis of variance results for the influence of distance to CTL trail on forest floor carbon, soil bulk density, and soil total carbon and nitrogen

|                         | Forest floor |         |                | Soil bulk density |         |                | Soil carbon |         |                | Soil nitrogen |         |                |
|-------------------------|--------------|---------|----------------|-------------------|---------|----------------|-------------|---------|----------------|---------------|---------|----------------|
|                         | df           | F       | p              | df                | F       | p              | df          | F       | p              | df            | F       | p              |
| Onaway - Age 1          | 3, 27        | 5.610   | <b>0.004</b>   | 3, 27             | 0.357   | 0.785          | 3, 27       | 3.426   | <b>0.031</b>   | 3, 27         | 312.320 | < <b>0.001</b> |
| Onaway - Age 5          | 3, 26        | 2.833   | <b>0.058</b>   | 3, 26             | 5.687   | <b>0.004</b>   | 3, 27       | 1.664   | 0.198          | 3, 27         | 211.829 | < <b>0.001</b> |
| Onaway - Age 6          | 3, 27        | 2.810   | <b>0.058</b>   | 3, 27             | 3.312   | <b>0.035</b>   | 3, 26       | 824.790 | < <b>0.001</b> | 3, 26         | 455.030 | < <b>0.001</b> |
| Charlevoix - Age 6      | 3, 26        | 961.361 | < <b>0.001</b> | 3, 26             | 385.967 | < <b>0.001</b> | 3, 26       | 732.042 | < <b>0.001</b> | 3, 26         | 8.997   | < <b>0.001</b> |
| Ensley/Angelica - Age 7 | 3, 25        | 407.930 | < <b>0.001</b> | 3, 25             | 19.141  | < <b>0.001</b> | 3, 23       | 1.552   | 0.228          | 3, 25         | 0.213   | 0.649          |
| Munising - Age 11       | 3, 27        | 404.022 | < <b>0.001</b> | 3, 27             | 284.406 | < <b>0.001</b> | 3, 26       | 676.060 | < <b>0.001</b> | 3, 26         | 799.038 | < <b>0.001</b> |
| Munising - Age 11       | 3, 27        | 15.980  | < <b>0.001</b> | 3, 27             | 27.311  | < <b>0.001</b> | 3, 27       | 3.861   | <b>0.059</b>   | 3, 27         | 0.060   | < <b>0.001</b> |

**Table 5.3** Forest floor and mineral soil nutrient estimates. Values with similar letters are not statistically different at  $\alpha = 0.1$  within age and soil type groupings

|   |  | 980009574             | 980004569              | 980004047             | 980004063            | 980010764                | 980002504            | 980002506             |
|---|--|-----------------------|------------------------|-----------------------|----------------------|--------------------------|----------------------|-----------------------|
|   |  | Onaway - Age 1        | Onaway - Age 5         | Onaway - Age 6        | Charlevoix - Age 6   | Ensley/Angelic-a - Age 7 | Munising - Age 11    | Munising - Age 11     |
| <b>Forest floor carbon (kg ha<sup>-1</sup>)</b> |  |                       |                        |                       |                      |                          |                      |                       |
| Middle of trail                                 |  | 370.64 <sup>a</sup>   | 1073.76 <sup>a</sup>   | 1247.45 <sup>a</sup>  | 539.04 <sup>a</sup>  | 1019.73 <sup>a</sup>     | 1028.70 <sup>a</sup> | 1752.48 <sup>a</sup>  |
| Equipment Track                                 |  | 377.01 <sup>ab</sup>  | 1093.44 <sup>ab</sup>  | 1389.76 <sup>a</sup>  | 390.72 <sup>b</sup>  | 785.76 <sup>b</sup>      | 1219.93 <sup>b</sup> | 1861.44 <sup>b</sup>  |
| 1 meters off trail                              |  | 355.59 <sup>abc</sup> | 1173.33 <sup>abc</sup> | 1302.98 <sup>ab</sup> | 684.80 <sup>c</sup>  | 1125.00 <sup>c</sup>     | 1251.54 <sup>c</sup> | 1986.24 <sup>c</sup>  |
| 5 meters off trail                              |  | 439.52 <sup>bd</sup>  | 945.12 <sup>ac</sup>   | 1061.79 <sup>ac</sup> | 519.00 <sup>cd</sup> | 758.93 <sup>d</sup>      | 1139.92 <sup>d</sup> | 2319.84 <sup>bd</sup> |
| <b>Soil bulk density (g cm<sup>3</sup>)</b>     |  |                       |                        |                       |                      |                          |                      |                       |
| Middle of trail                                 |  | 1.14                  | 0.59 <sup>a</sup>      | 0.89 <sup>a</sup>     | 0.76 <sup>a</sup>    | 1.24 <sup>a</sup>        | 0.760 <sup>a</sup>   | 0.75 <sup>a</sup>     |
| Equipment Track                                 |  | 1.14                  | 0.58 <sup>a</sup>      | 0.99 <sup>ab</sup>    | 0.68 <sup>b</sup>    | 1.04 <sup>a</sup>        | 0.884 <sup>b</sup>   | 0.9 <sup>b</sup>      |
| 1 meters off trail                              |  | 1.25                  | 0.52 <sup>b</sup>      | 0.94 <sup>a</sup>     | 0.68 <sup>c</sup>    | 1.18 <sup>a</sup>        | 0.907 <sup>c</sup>   | 0.97 <sup>c</sup>     |
| 5 meters off trail                              |  | 0.97                  | 0.54 <sup>b</sup>      | 0.76 <sup>b</sup>     | 0.88 <sup>d</sup>    | 0.88 <sup>b</sup>        | 0.850 <sup>d</sup>   | 0.94 <sup>cd</sup>    |
| <b>Soil carbon (Mg ha<sup>-1</sup>)</b>         |  |                       |                        |                       |                      |                          |                      |                       |
| Middle of trail                                 |  | 31.34 <sup>a</sup>    | 131.22                 | 38.70 <sup>a</sup>    | 127.09 <sup>a</sup>  | 208.46                   | 57.29 <sup>a</sup>   | 54.96                 |
| Equipment Track                                 |  | 37.95 <sup>b</sup>    | 142.95                 | 38.31 <sup>b</sup>    | 157.87 <sup>b</sup>  | 237.40                   | 50.05 <sup>b</sup>   | 61.16                 |
| 1 meters off trail                              |  | 37.55 <sup>ab</sup>   | 143.19                 | 38.93 <sup>c</sup>    | 149.47 <sup>c</sup>  | 197.05                   | 40.98 <sup>c</sup>   | 45.28                 |
| 5 meters off trail                              |  | 33.51 <sup>ab</sup>   | 137.89                 | 35.52 <sup>d</sup>    | 163.71 <sup>d</sup>  | 271.66                   | 34.35 <sup>d</sup>   | 45.91                 |
| <b>Soil Nitrogen (Mg ha<sup>-1</sup>)</b>       |  |                       |                        |                       |                      |                          |                      |                       |
| Middle of trail                                 |  | 2.19 <sup>a</sup>     | 7.91 <sup>a</sup>      | 2.68 <sup>a</sup>     | 6.51 <sup>a</sup>    | 9.95                     | 3.40 <sup>a</sup>    | 2.88 <sup>a</sup>     |
| Equipment Track                                 |  | 2.59 <sup>b</sup>     | 9.04 <sup>b</sup>      | 2.54 <sup>bcd</sup>   | 8.05 <sup>b</sup>    | 11.13                    | 2.48 <sup>b</sup>    | 3.12 <sup>ab</sup>    |
| 1 meters off trail                              |  | 2.19 <sup>c</sup>     | 8.64 <sup>c</sup>      | 2.60 <sup>bc</sup>    | 7.13 <sup>bc</sup>   | 9.41                     | 1.75 <sup>c</sup>    | 2.26 <sup>ab</sup>    |
| 5 meters off trail                              |  | 2.08 <sup>d</sup>     | 9.03 <sup>d</sup>      | 2.44 <sup>d</sup>     | 8.06 <sup>abc</sup>  | 12.79                    | 1.41 <sup>d</sup>    | 1.99 <sup>ac</sup>    |

**Table 5.4** Mixed effects prediction models of aspen sucker mean dominant height and regeneration density. Models ranked by Akaike Information Criterion.

|  | Model                              | AIC    | Loglik | Delta AIC | <i>W</i> |
|--|------------------------------------|--------|--------|-----------|----------|
| <b>Regeneration mean dominant height (m)</b>                 | Stand age * bulk density           | 163.12 | -71.6  | 0         | 0.9985   |
|  | Stand age * biomass                | 176.15 | -78.1  | 13        | 0.0015   |
|  | Stand age * biomass * bulk density | 211.17 | -91.6  | 35        | 0.0000   |
| <b>Regeneration stocking density (stems ha<sup>-1</sup>)</b> | Stand age + bulk density           | 262.28 | -121.1 | 0         | 0.9974   |
|  | Stand age + biomass                | 274.54 | -127.3 | 12        | 0.0022   |
|  | Stand age * bulk density           | 281.00 | -124.5 | 19        | 0.0000   |
|  | Stand age + biomass * bulk density | 285.92 | -131.0 | 24        | 0.0000   |
|  | Stand age * biomass                | 288.93 | -133.5 | 27        | 0.0000   |
|  | Stand age * bulk density * biomass | 316.51 | -143.3 | 54        | 0.0000   |
|  |                                    |        |        |           |          |

#### 5.3.4 Long-term projections

Comparing temporal patterns in residues and mean dominant sapling height of CTL to WTH methods is limited to the early phase of stand development prior to stem exclusion and canopy closure. Residues and DDW measured along the WTH rotation were fit with a third degree polynomial with the equation form;

$$[1] \quad DDW \text{ biomass (Mg ha}^{-1}\text{)} = 17.087 - 1.953_{\text{stand age}} + 0.11_{\text{stand age}}^2 - 0.00169_{\text{stand age}}^3$$

All three subplots measured on CTL harvested stands have significantly higher values of residues immediately post-harvest when compared to WTH stands ( $p < 0.001$ ). Residue estimates on subplots 1 m and 5 m from the CTL trail were estimated had significantly different declines in biomass through time than WTH stands ( $p < 0.007$ ) (Figure 5.4a). Extrapolating these patterns past observations of CTL stands suggests that residues may persist for 20 years. Models detected significant differences in early height growth on stands harvested with WTH methods and saplings located on CTL trail center ( $p < 0.001$ ) in the early phases of stand development. However, there were no differences in sapling height on the 1 m and 5 m subplots when compared with WTH stands ( $p = 0.156$ ). I calculated the difference between WTH and CTL-1 mean dominant height at age 12 to adjust SI<sub>50</sub> model estimates and project height patterns through rotation to SI base age using exponential function height models developed by Lundgren (1970). Projected height from ages 12-40 on the CTL – 1 subplot were estimated with the model;

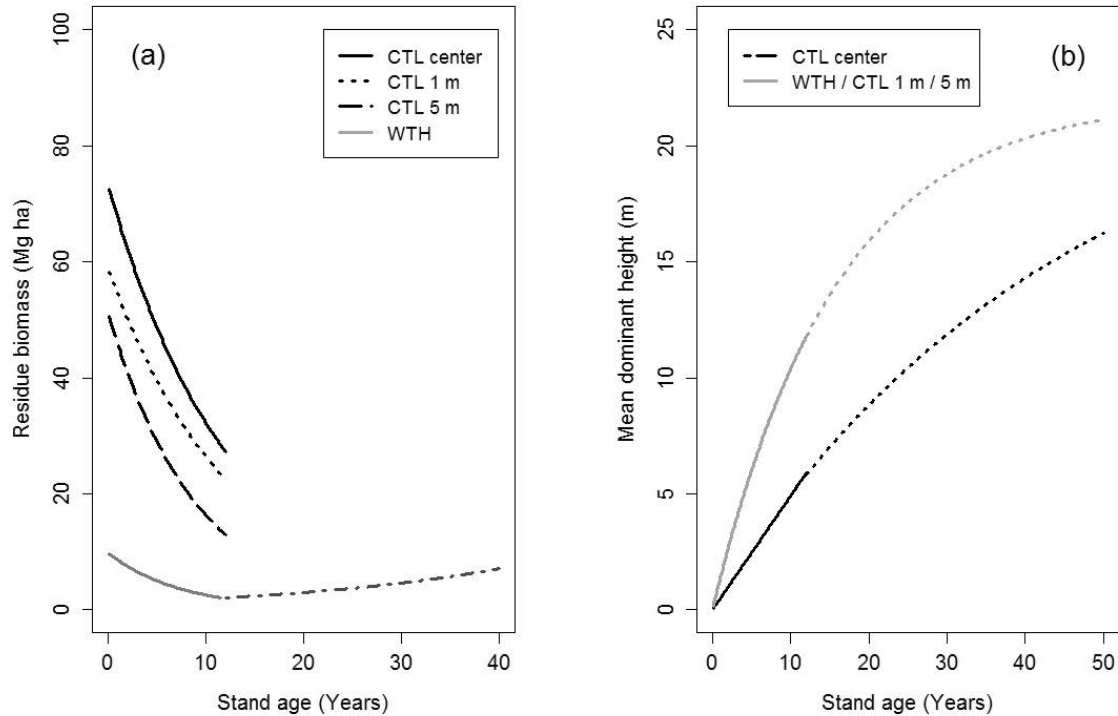
$$[2] \quad \text{Height (m)} = 62.5(1.25 - 1.24(1 - \exp^{-0.02274(\text{stand age})}) 0.3048$$

Due to the lack of difference between WTH height observations and those located at 1 and 5 m from CTL trail, I combined the data and fit observed height measurements with a Chapman Richards function with the model form;

$$[3] \quad \text{Height (m)} = 22.037(1 - \exp^{-0.085(\text{stand age})})^{0.75}$$

Long-term predictions through stand rotation based on patterns observed in early stand development suggest that saplings located on the CTL trail will decrease by approximately 6.03, or roughly a 29.7% loss in total height by the end of stand rotation (Figure 5.3b). Given the observed spacing between CTL trails on the study sites, coupled with and equipment specifications of CTL manufacturers, at least  $21.21 \pm 1.21$  % of the stand area will be directly impacted by equipment trails.





**Figure 5.4** Panel a - Long term trends in residue and DDW pools across the commercial aspen rotation by CTL subplot and stands harvested with WTH methods. Panel b - Projected height patterns through stand rotation of CTL and WTH stands.

## 5.4 Discussion

Findings indicate that residues are being concentrated and confined to distinct rows and create a distinctive spatial pattern within and across the stand in some cases where CTL harvest machinery is utilized. Models predict roughly a 50% increase in residue loads directly on the equipment trail when compared to areas between trails (Figure 5.1a and b). However, the area of impact was variable across all observed stands. Review of aerial images suggested estimates of approximately 17-27% of the stand directly in CTL trails, which is notably higher in proportion than heavily trafficked skid trails in WTH operations (Navratil, 1991; Zenner et al., 2007). These patterns directly correspond to equipment specifications of CTL processors. The area of the stand located in equipment trails is ultimately dependent upon several factors, including but not limited to: (i) specific CTL equipment utilized by the contractor; (ii) local topography and harvest unit layout, and; (iii) individual or patch tree retention. Local topography and live tree retention can have varying influences on equipment operability and residue distribution patterns, and have also been noted to result in heavy accumulations in piles (personal observation). During CTL harvests with residue retention, operators place residues directly in the path of the harvester to mitigate soil disturbance and comply with third party certification standards (SFI, 2015), as previous studies have noted increased

compaction and rutting by heavy equipment (Shetron et al., 1988; Navratil, 1991; Bates et al., 1992; Zenner et al., 2007).

The spatial patterns of residues were correlated with reductions in aspen sucker height and regeneration density, and these findings consistently detected lower values of these metrics in the CTL trails across all observed stands. Sucker mean dominant height in the CTL trail was found to be reduced by roughly 27% when compared to adjacent areas of the stand through observations. While these results detected initially higher density levels in the CTL trail center one year after harvest, this difference quickly reversed by year 2, and reflects the high variance in initial stand density in the first three years post-harvest (Bella, 1986). Observations indicate a decrease in sucker density levels directly on CTL equipment trails which becomes more pronounced through time. I expect differences in regeneration density to disappear by the end of the rotation due to higher self-thinning rates of more densely stocked areas of the stand (Sorenson 1968; Bella 1986). However, reductions in regeneration density early in development may be exacerbated by additional factors that may limit optimal regeneration stocking and threaten stand productivity. Previous work in temperate aspen forests has shown strong evidence that aspen regeneration is generally highest following winter harvesting, mainly due to (i) total non-structural carbohydrate reserves in parent root systems; (ii) minimal impacts of harvesting on forest soil conditions, and; (iii) relatively long growing season for newly emerged suckers.

Given harvesting equipment and operational logistics, results suggest that residue retention under these harvesting techniques can lead to an average 58 % decline in stem density in CTL trails before canopy closure across a mean of 21.2 % of the stand. Any reductions in stand regeneration due to direct or indirect seasonal factors, coupled with potential losses from equipment impacts, may increase susceptibility to forest health issues including but not limited to; *Hypoxylon mammatum* (Wahl. :Mill.), *Venturia macularis* (Fr.), and *Armillaria mellea* (Vahl: Fr) (Bates et al., 1989). These factors, when taken together with traditionally high ungulate concentration and associated browse common to the study area (Doepker et al., 1993), may threaten long-term production and require silvicultural intervention.

These patterns in stand regeneration observed in this study are similar in orientation to strip thinning operations in aspen stands of northern Minnesota as reported by Zasada et al., (2001). However, it should be noted that pre-commercial thinning in the aforementioned silvicultural programs occurs at age 10, after the site has been allowed to reach optimal regeneration density, and just prior to crown closure and increases in mortality. In contrast, observations reported here show that portions of the stand in CTL trails are severely understocked compared to adjacent areas after two years post-harvest. Further, and of particular note, is that despite differences in equipment types and residue treatments of CTL and WTH operations, results of regeneration patterns in CTL systems corroborate with findings in WTH systems in the region. Work conducted by Navratil (1991), Zenner et al., (2007), and Puettmann et al. (2008) reported reduced height growth and density on heavily trafficked skid trails (> 100 passes) in aspen stands of Alberta and

northern Minnesota, respectively. They attributed these patterns to soil compaction and physical disturbance to the aspen rooting systems from multiple equipment passes from harvesting operations.

The deliberate addition of residues to the forest floor is intended to mitigate soil compaction and damage to shallow root systems caused by heavy harvesting equipment. Multiple equipment passes from WTH skidding operations have been reported to increase soil bulk density to the magnitude of 159-220 % when compared to adjacent areas of the stand, and can reduce sucker density and height growth (Zenner et al., 2007; Puettmann et al., 2008). These results indicate that the addition of residues to the forest floor does appear to decrease soil compaction levels when compared with WTH operations, as tests detect increases in bulk density on the CTL trails at 8-28 % in several stands (Table 4.3). These trends corroborate with findings by Han et al., (2009) who reported high residue loads can effectively mitigate soil compaction. Further, despite the reduction in soil disturbance, observed reductions in sucker height are in parallel with those found in roughly 7.4 % of heavily trafficked WTH skid trails reported by Zenner et al., (2007). It should be noted that stands exhibiting higher bulk density off of the equipment trail also show higher carbon content in the trail, and reflect the incorporation of decayed residues into the mineral soil. Therefore, the reductions in regeneration height and density may be due to a lurking physical effect of the concentrated residue loads. Several instances were noted where residues were unable to be tallied, as they were incorporated into the forest floor and covered by subsequent detritus and regenerating vegetation, including dense mats of *Carex* spp. These factors reduced LIDS estimates of residues, and therefore likely underestimated the general influence of residue loads on sucker regeneration.

Previous work has suggested that logging slash concentrations can reduce aspen suckering through shading the soil surface and thereby resulting in lower soil temperatures for optimum suckering (Maini and Horton, 1966; Bates et al., 1989; Zabowski et al., 2000). Indeed, Slesak (2013) found that heavy slash loads in aspen forests of northern Minnesota can significantly reduce soil temperatures within the upper 30 cm of the soil horizon. Declines in soil temperature have been noted to inhibit or delay sucker initiation, which in turn result in a shorter effective growing season (Frey et al., 2003). Further, as CTL and forwarding equipment pass over the residues during harvest removal, residues are compressed into the forest floor, and may be creating a physical impediment to sucker growth. It should be noted that the area of the stand exhibiting significant reductions in stand regeneration due to CTL equipment is far greater than those documented with WTH methods (excluding roads and landings) (Navratil, 1991; Zenner et al., 2007). Results suggest an approximate increase of 186 % in area severely affected by CTL machinery when compared with studies investigating areal impacts of feller-bunchers and skidding equipment (Navratil, 1991; Zenner et al., 2007). Physical variation in forest units and silvicultural prescriptions will influence harvesting operations and subsequent area of impact.

The concentration of residues into trails may have unintended influences on early stand development through spatial partitioning of residue-bound nutrients and soil conditions.

Given the relatively high levels of nutrients in harvest residues (Alban, 1986; Klockow et al., 2013), it is reasonable to expect post-harvest forest soil nutrients reflect spatial orientation of residues, which could in turn favor vegetation in close proximity to equipment trails. However, observations from this study found that high concentration of residues loads had a mixed level of impact on forest soil C and N. Only two of the observed sites exhibited higher values of soil C at CTL trail center. Unexpectedly, several sites had higher soil C values off of CTL trails. This may be due to fine root turnover and decomposition of coarse roots of the aspen parent system (Ruark and Bockheim, 1987; Alban and Perala, 1992; Pregitzer and Friend, 1996). Results found higher levels of soil N in CTL trail center in 4 of the 7 measured stands, which was unsurprising, given reported high N content in aspen twigs and shoots (Alban, 1985) and heavy residue loads (Figure 5.2). These findings may be due to decreased decomposition rates from lower soil temperatures under areas of high residue concentrations (Taylor and Parkinson, 1988) and testing for total soil N in the sampling design.

Given the strong and distinctive patterns of both harvest residues and corresponding sucker regeneration, it is important to consider long-term effects on overstory stand dynamics, FWD and DDW pools, and forest soil conditions. The temporal residence time of harvest residues warrants further investigation, given renewed interest in utilization as a bioenergy feedstock. Results suggest rapid decomposition rates of residues 12 years post-harvest, and if levels continue to decline in this manner, may limit residence time to 15-25 years. Observations from the rotation length chronosequence study exhibit less pronounced declines up to years 15-20, which can be attributed to both lower residue loads and the increased probability of including DDW ( $\leq 10$  cm) in addition to residues. As previously noted, I found intact residues under a thick layer of regenerating vegetation at 6 and 7 years post-harvest. I suspect that the residence time of harvest residues varies by soil type and associated nutrient cycling regimes.

While it is reasonable to expect initial variations in stand density due to CTL trails to diminish over time due to rapid self-thinning, it is expected that the observed decrease in height growth in trail center will result in these stems being suppressed at the time of canopy closure. As aspen has been noted as being heavily shade intolerant (Perala, 1990), it is probable that these trees will be the first to succumb to mortality once overtopped. In the unlikely case of survival, predictive models show a loss of roughly 30 % in aspen yields across the CTL impacted areas of the stand (Figure 5.1 and 5.4). This loss in growth may be minimized or balanced by increased growth of trees on the periphery of CTL trails through increased light levels and availability of nutrients; therefore, further investigation of long-term stand dynamics is required. The findings presented here provide another example of how cumulative effects of constraints on forest management may be resulting in decreased yields of wood fiber.

## **5.5 Conclusions and Management Implications**

Results suggest that the incorporation of residues into the forest floor during CTL harvesting operations is an effective tool to mitigate soil compaction. However, the

patterns in which residues are retained can have a distinctive effect on the stand, and can significantly reduce regeneration density in up to 20 % of the stand area. While this areal extent may be minimal by stand maturity, reductions in density of this magnitude early in stand development, when coupled with other inhibitory factors such as ungulate browse which is characteristic of the study region, can enhance susceptibility to forest health issues, potentially reduce stand productivity, and require intensive stand tending efforts.

It is recommended that using designated feller-buncher and tractor skidding machinery in the winter season to minimize areal impact of regeneration issues, mitigate forest floor disturbance and soil compaction issues, and create a more dispersed pattern of residues through incidental breakage, which has been shown to meet the suggested residue loads by state guidelines (MFRC, 2007; Klockow et al., 2013). Restricting skid trails to pre-designated areas of the stand coupled with moderate residue loads in medium to heavily trafficked areas may further minimize detrimental harvesting impacts. As of yet, there are no conclusive findings reporting productivity loss by residue removal in temperate aspen forests, however there have been noted declines in soil macronutrient levels (Voldseth et al., 2011; Premer et al., in preparation). It is further suggested that harvest residue management be site and harvest equipment specific. Sites with high productivity levels are likely to be less affected by residue removals; however the interaction of both spatial patterning of residues and associated nutrients on regenerating vegetation warrants further attention given potential influences on stand dynamics. Finally, continued investigation into the residence time of harvest residues is warranted, which may serve as either a viable bioenergy feedstock or source of long-term C pools.

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