FOREST DISTURBANCES: OCCURRENCES AND IMPACTS OF

RECREATIONAL, HYDROGEOMORPHIC, AND CLIMATIC DISTURBANCES

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ABSTRACT

Disturbances impact abiotic and biotic components within forested ecosystems. This dissertation identifies the impacts of recreational crossing disturbances on water quality, uses dendrochronological techniques to estimate sediment deposition and identify impacts hydrology and climate on radial growth in riparian forested wetlands, and quantifies influences of biotic and abiotic factors on the occurrence of frost-induced cambial damage.

In southwestern Virginia, modeled soil erosion rates for multiple-use (hiking, mountain biking, and horseback riding) recreational trail approaches to stream crossings were found to be 13 times greater than rates for undisturbed forests. Downstream changes in macroinvertebrate-based indices indicated water quality was negatively affected downstream from culvert and ford crossings. These findings illustrate recreational stream crossings have the potential to deliver sediment into adjacent streams, particularly where best management practices are not being rigorously implemented, and as a result can negatively impact water quality below stream crossings.

Impacts of hydrologic regime were apparent on sediment deposition and on green ash (*Fraxinus pennsylvanica* Marsh.) and water tupelo (*Nyssa aquatica* L.) radial growth along the Tensaw River in southwest Alabama. Annual dendrogeomorphic sediment accretion rates were significantly greater for a recent time period (25 years) when compared to longer term rates (131 years) along a natural levee and backswamp. Radial

growth in green ash along the natural levee and backswamp was found to be significantly correlated with days flooded and average daily stage level during April while water tupelo further in the backswamp appeared resistant to hydrologic and climatic fluctuations. Results illustrate the importance of riparian wetlands in trapping sediment from adjacent waterways and highlight the role hydrologic regime plays in bottomland succession and productivity.

Across northern lower Michigan, late spring frost-induced cambial damage in jack pine (*Pinus banksiana* Lamb.) occurred more frequently in younger trees and in trees with smaller diameters. Biotic and abiotic factors were found to influence the occurrence of earlywood frost rings. Frequent occurrences of frost-rings can be used to identify frost-prone environments and geographical boundaries for plant species. This dissertation bridged gaps in knowledge of recreational, hydrogeomorphic, and climatic disturbances in forested ecosystems which can be used to develop management strategies.

DEDICATION

To my many mentors at Campbell University, University of Arkansas at Monticello,

Auburn University, and Virginia Tech, who inspired/pushed me to pursue my PhD.

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CONTRIBUTION OF AUTHORS

Dr. W. Michael Aust had the idea for me to conduct a study at Pandapas Pond to examine the impacts of recreational stream crossing effects on water quality (Chapter 2). Dr. Aust and Dr. Carolyn A. Copenheaver assisted with multiple rounds of manuscript edits and provided direction for the introduction and discussion sections in Chapter 2. Dr. Aust and Dr. Copenheaver combined had the idea for me to incorporate flooding and sediment deposition disturbances into my doctoral research program (Chapter 3). Dr. Aust suggested I re-visit a site he formerly established in 1986 in southern Alabama to conduct this research. Previous data collected at this site was used to supplement the data I collected and analyzed in Chapter 3. Dr. Copenheaver and Dr. Aust will assist with any future edits associated with submission and publication of Chapter 3. Additional feedback will be provided by Dr. Audrey Zink-Sharp, Dr. Jay Sullivan, and Dr. Bryan Black during the defense of this work. Dr. Copenheaver guided the development of Chapter 4 and allowed me to supplement my jack pine tree cores collected from 6 sites with cores that she originally collected in 1997 at 5 sites in northern lower Michigan. Dr. Zink-Sharp provided laboratory equipment and supplies necessary to achieve images necessary to document the appearance of an earlywood frost ring in jack pine. Dr. Copenheaver and Dr. Zink-Sharp assisted with editing throughout the submission and publication of Chapter 4. Authorship order is presented below each chapter title, which indicates the level of contribution made by each author to the research manuscript. Specific publication information is presented in Chapter 1.

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CHAPTER 1. INTRODUCTION

1.1 Forest Disturbances

Forest disturbances have ecological and economic consequences (Holmes et al., 2008).
A disturbance is "any relatively discrete event in time that disrupts ecosystem, community, or
population structure and change resources, substrate availability, or the physical environment"
(Pickett and White, 1985). Previous research in the field of disturbance ecology, "the study of
interrelationships between biotic and abiotic components of an environment" (Rogers, 1996), has
primarily focused on the role disturbances play in forested ecosystems and the emulation of
disturbances to achieve desired forest conditions (Oliver, 1981; Franklin et al., 2002; Bergeron
and Harvey, 1997). Disturbances are frequently used as a management tool (i.e., natural
disturbance-based silviculture) to achieve desired outcomes in forested ecosystems based on
previously identified impacts (Oliver and Larson, 1990). Conversely, little attention has been
given in disturbance ecology to the identification of undesired outcomes and development of
management strategies to reduce the undesired disturbance occurrences and impacts.

Literature clearly indicates that an understanding of disturbance ecology is beneficial for
ecosystem management (Oliver, 1984; Attiwill, 1994; Rogers, 1996; Jõgiste et al., 2007; DeRose
and Long, 2014). However, gaps in knowledge of disturbance impacts and occurrences exist,
leaving forest managers and society oblivious to some disturbances and their consequential
impacts. This dissertation identifies the impacts of recreational crossing disturbances on water
quality, illustrates the use of a dendrogeomorphic method to estimate sediment deposition
patterns in riparian forested wetlands, identifies relationships between radial tree growth and

hydrology and climate, and quantifies the influence of biotic and abiotic factors on the likelihood of cambial damage occurrence during late spring frost disturbances.

1.2 Forest Recreational Stream Crossing Disturbances

Sediment is the leading cause of impaired waterways in the United States (US Environmental Protection Agency, 2000). In forested ecosystems, stream crossings have been identified as points of sediment entry into adjacent streams (Lane and Sheridan, 2002; Aust et al., 2011; Wear et al., 2013). Stream crossing approaches, road or trail segments which lead to a respective stream crossing structure, are a major source of sediment due to soil erosion along the approaches and the ability of the approaches to transport sediment-laden runoff into adjacent streams (Aust et al., 2011). Research on the impacts of stream crossings on water quality in forested ecosystems has, in majority, been limited to forest operational crossings and should be expanded to identify impacts of recreational stream crossings. Forest trail-based recreational opportunities have expanded over recent decades (Cordell and Overdevest, 2001), which raises management concerns over potential impacts of non-motorized recreational stream crossing disturbances on downstream water quality (Bosworth, 2007; Olive and Marion, 2009). Visual observations of sediment entry in adjacent streams have been made along recreational trail approaches to stream crossings (Olive and Marion, 2009; Wilkerson and Whitman, 2009). Along a stream crossing approach used by off road vehicles (ORVs), soil erosion rates were modeled, and downstream total suspended solids were found to be greater downstream than upstream from the ford crossing (Ayala et al., 2005). Results from this ORV approach study were limited to a single stream crossing approach used by motorized traffic. Negative downstream impacts on water quality have been demonstrated below ORV trail and operational forest skid trail and road stream crossings (Chin et al., 2004; Ayala et al., 2005; Neal et al., 2007;

Aust et al., 2011; Wear et al., 2013), but have not been evaluated below non-motorized recreational trail stream crossings.

1.3 Forest Hydrogeomorphic Disturbances

Riparian forested wetlands filter and trap sediment from adjacent waterways during overbank flooding events (Boto and Patrick, 1978; Walbridge, 1993; Kleiss, 1996). Thus, riparian forested wetlands improve water quality, a valuable ecosystem service. Quantification of sediment accretion rates over time allows determination of the quantity of sediment captured or removed from water ways. Previous research has identified the integral role sediment deposition and hydrology play in succession and site productivity in bottomland hardwood forests (Hodges, 1997; Mitsch et al., 2014). Spatial patterns of sediment deposition have been linked to a complexity of factors (e.g., elevation, distance from river, microtopography, density of vegetation) associated with hydrologic connectivity to sediment-laden waters (Hupp, 2015). Few studies have estimated long-term rates and compared them over different time periods (Hupp and Morris, 1990; Kleiss, 1996; Heimann and Roell, 2000). One reason for this gap is the amount of resources (e.g., time, money) required to conduct repeated measurements of sediment deposition through time. A previously validated dendrogeomorphic technique (Hupp and Morris, 1990) could be used to minimize site visits and provide data across time and space. This technique uses the establishment date of targeted bottomland trees and total vertical deposition values immediately adjacent to selected trees to derive sediment accretion rates. Changes in upstream disturbance regimes (e.g., land use) can alter sediment loads entering waterways and thus, the quantity of sediment trapped by forested wetlands during overbank flooding events.

Alterations in hydrologic regime can influence sediment deposition and inflow patterns, and as a consequence, can alter tree growth and productivity in forested wetlands (Conner and

Day, 1976; Mitsch and Rust, 1984). Previous studies have identified periodic growth response to hydrologic regime in either lower elevated swamp dominant species (Dicke and Toliver, 1990; Conner and Day, 1992; Keeland and Sharitz, 1995) or in a collection of less flood-tolerant bottomland hardwood species (Reily and Johnson, 1982; Mitsch and Rust, 1984; Anderson and Mitsch, 2008; Predick et al., 2009; Keim and Amos, 2012), but rarely both (Conner et al., 2014). Few studies have used annual resolution data to evaluate the relationship between radial growth and hydrologic variables, and radial growth and climatic variables. Identification of these relationships is essential to understanding how future alterations in hydrology (e.g., dam construction or removal) and climate (e.g., climate change) will impact site productivity in riparian forested wetlands.

1.4 Forest Climatic Frost Disturbances

Late spring frost disturbances have significant ecological and physiological impacts on forests. Previous research has identified reduced radial and vertical tree growth (Dittmar et al., 2006) and increased susceptibility of individual trees to dieback and damage-inducing pathogens (Pomerleau, 1971; Reich and van der Kamp, 1993) as impacts of growing season frosts. Growing season frost disturbances have been shown to influence the geographic range limits of individual species (Kollas et al., 2014). Frost-induced cambial damage that occurs when cells are actively dividing can result in the formation of frost rings within the annual growth rings of an impacted tree. Frost rings are abnormal modifications to wood anatomy that form as a result of an abrupt freeze during the growing season; thus, frost rings can record the occurrence of growing season frost disturbances. Previous studies have used frost rings to reconstruct and confirm dating of historically significant climatic and biological events (Bailey, 1925, Hantemirov et al., 2000; Gurskaya and Shiyatov, 2006; Payette et al., 2010). Many of these

studies assume a direct link between frost-ring formation and freeze events. However, a thorough examination of the relationship between a tree's biological characteristics and the likelihood of frost-ring formation in varying environmental conditions has yet to be conducted.

Therefore, the overall goals of this research were to determine impacts of recreational stream crossing disturbances on adjacent water quality, reconstruct and identify differences in long- and short-term sediment accretion rates using a dendrogeomorphic technique which minimizes site visits, determine impacts of hydrologic and climatic variables on radial growth in riparian forested wetlands, and quantify the influence of biotic and abiotic factors on the occurrence of frost-induced cambial damage.

1.5 Organization and Objectives

This dissertation is organized into five chapters. Chapter 1 provides an introduction to disturbances in forested ecosystems and describes the chapter organization and research objectives. Chapters 2 through 4 are stand-alone manuscripts that are related because they focus on the occurrences and impacts of forest disturbances at increased spatial scales. Chapters 2 and 4 have been peer-reviewed and published. Chapter 3 will be submitted for peer-review and future publication. Chapter 5 summarizes the implications of our findings, future research needs, and overall conclusions. Chapters are ordered based on the spatial scale at which each study was conducted. Chapter 2 was conducted at individual stream crossings within a recreational trail system, Chapter 3 across natural levee and backswamp riparian forests, and Chapter 4 in 11 jack pine dominated stands dispersed across northern lower Michigan.

Forest disturbance occurrences and impacts were identified using previously proposed methods in disturbance ecology (Lorimer, 1985; Rogers, 1996). Chapter 2 uses two models to estimate soil erosion disturbance rates and uses macroinvertebrates as bioindicators of

downstream changes in water quality (Rogers, 1996). In Chapters 3 and 4, physiological

evidence of disturbance in trees was used to identify the impacts of hydrologic and climatic

disturbances (Lorimer, 1985). Physiological evidence in Chapter 3 includes annual tree growth

patterns, and in Chapter 4 includes the occurrence of modifications to normal wood anatomy

within annual growth rings (Lorimer, 1985; Rogers, 1996). Additionally, in Chapter 3, sediment

deposits were used to reconstruct temporal and spatial patterns of sediment deposition as

influenced by hydrologic and anthropogenic disturbances.

Chapter 2 identifies the occurrence and impacts of forest recreational trail disturbances on

water quality at multiple stream crossing locations within the Poverty Creek watershed in

southwestern Virginia. The objectives in Chapter 2 were to: 1) use soil erosion models to

provide soil erosion rate estimates for recreational trail approaches to selected stream crossings

as potential sources of sediment inputs to the Poverty Creek watershed, a 2,145 ha watershed

located in the Southern Appalachian Mountains; 2) validate sediment yield estimates using

macroinvertebrate sampling as biological indication of altered water quality below stream

crossings; and 3) discern whether water quality impacts vary between culvert and ford stream

crossing types. Chapter 2 was written by Kathryn R. Kidd, with contributions by Dr. W.

Michael Aust and Dr. Carolyn A. Copenheaver and was published in *Environmental

Management* (Kidd et al., 2014a). The right to include this work published by Springer in this

dissertation is retained by the authors (Appendix A.1).

Chapter 3 illustrates the use of a dendrogeomorphic technique in estimation of temporal

and spatial sediment deposition patterns and identifies the impacts of hydrology and climate on

green ash and water tupelo radial growth in riparian forested wetlands in the Mobile-Tensaw

Delta in southwestern Alabama. Therefore, the objectives of Chapter 3 were to 1) estimate and

compare short- (25 years) and long-term (131 years) sediment accretion rates for a natural levee (35 m from river) and backswamp (75 m from river) adjacent to the Tensaw River in southern Alabama using a dendrogeomorphic technique; 2) determine the influence of distance from the Tensaw River on sediment accretion rates for the 1986 to 2012 time period using estimates from the dendrogeomorphic technique, sediment pin measurements, and repeated elevation surveys; and 3) identify the role of overbank flooding and climate on radial growth in green ash (*Fraxinus pennsylvanica* Marsh.) and water tupelo (*Nyssa aquatica* L.). Chapter 3 was written by Kathryn R. Kidd, with contributions by Carolyn A. Copenheaver and W. Michael Aust and will be submitted for peer-review and future publication.

Chapter 4 examines the occurrences and impacts of late spring frost disturbances on the cambium of jack pine trees through time across a spatial scale of northern lower Michigan. Therefore, the objectives of this study were to 1) quantify earlywood frost-ring formation in jack pine (*Pinus banksiana* Lamb.) across cambial age and diameter classes at 11 sites across northern lower Michigan and 2) determine the influence of biotic and abiotic factors on the probability of earlywood frost-ring formation. Chapter 4 was written by Kathryn R. Kidd, with contributions by Dr. Carolyn A. Copenheaver and Dr. Audrey Zink-Sharp and was published in Écoscience (Kidd et al., 2014b). The right to include this work in the dissertation is retained by the authors (Appendix A.2).

Chapter 5 provides an overview of findings from Chapters 2 through 4 and recommends management strategies where needed. This chapter provides the overall implications of research findings from this dissertation in a broader sense and direction for future research.

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CHAPTER 2. RECREATIONAL STREAM CROSSING EFFECTS ON SEDIMENT DELIVERY AND MACROINVERTEBRATES IN SOUTHWESTERN VIRGINIA, USA[1]

Kathryn R. Kidd, W. Michael Aust, Carolyn A. Copenheaver

2.1 Abstract

Trail-based recreation has increased over recent decades, raising the environmental management issue of soil erosion that originates from unsurfaced, recreational trail systems. Trail-based soil erosion that occurs near stream crossings represents a non-point source of pollution to streams. We modeled soil erosion rates along multiple-use (hiking, mountain biking, and horseback riding) recreational trails that approach culvert and ford stream crossings as potential sources of sediment input and evaluated whether recreational stream crossings were impacting water quality based on downstream changes in macroinvertebrate-based indices within the Poverty Creek Trail System of the George Washington and Jefferson National Forest in southwestern Virginia, USA. We found modeled soil erosion rates for non-motorized recreational approaches that were lower than published estimates for an off-road vehicle approach, bare horse trails, and bare forest operational skid trail and road approaches, but were 13 times greater than estimated rates for undisturbed forests and 2.4 times greater than a two-year old clearcut in this region. Estimated soil erosion rates were similar to rates for skid trails and horse trails where best management practices (BMPs) had been implemented. Downstream changes in macroinvertebrate-based indices indicated water quality was lower downstream from

crossings than in upstream reference reaches. Our modeled soil erosion rates illustrate recreational stream crossing approaches have the potential to deliver sediment into adjacent streams, particularly where BMPs are not being implemented or where approaches are not properly managed, and as a result can negatively impact water quality below stream crossings.

Keywords: recreation ecology; trail erosion; water quality; Universal Soil Loss Equation; Water Erosion Prediction Project; soil erosion

2.2 Introduction

Trail-based recreation has experienced a recent upsurge in popularity, presenting an environmental disturbance threat to forested ecosystems (Bosworth 2007) and demanding increased management resources for trail maintenance and erosion prevention (Lynn and Brown 2003; Olive and Marion 2009). Trails in forested recreation areas provide passage to all-terrain vehicles, bike riders, cross-country skiers, hikers, horseback riders, pack animals, and snowmobilers (Deluca et al. 1998; Törn et al. 2009; Pickering et al. 2010). Recreational trails often intersect streams or rivers at water crossings which creates pathways for eroded soil to enter adjacent streams and decrease water quality. Erosive disturbances that introduce sediment into streams such as traffic, crossing installation, and approach runoff that occur adjacent to and within stream crossings can alter total suspended solids, conductivity, temperature, and pH in streams, particularly if crossings are improperly designed or inadequately managed (Lane and Sheridan 2002; Aust et al. 2011; Wear et al. 2013). Increased fine sediment loads from recreational trails can also reduce fish and macroinvertebrate habitat and alter watershed processes downstream from trail crossings (Brown 1994; Chin et al. 2004; Arp and Simmons 2012). Increased fine sediment loads in waterways result in higher costs for downstream water treatment plants and increased frequency of dredging ditches, streams, lakes, and harbors (Clark

1985). User surveys indicate eroded trails decrease recreational opportunities and reduce the level of enjoyment experienced by recreationists, for whom these trails are ultimately managed (Manfredo et al. 1996; Lynn and Brown 2003).

Soil erosion along trails is a multiple-step process that begins with hikers and other recreational traffic trampling vegetative cover and scuffing away litter and duff layers (Manning 1979). The underlying soil is then compacted by trail traffic resulting in an impervious soil surface that is more prone to surface water runoff and creates conditions that increase the likelihood of soil erosion (Olive and Marion 2009). Although the amount and type of trail traffic influences the rate of erosion, even remote, infrequently traveled trails have measurable signs of soil erosion (Weaver and Dale 1978; Bratton et al. 1979). Type of trail usage influences soil erosion with horseback riding resulting in soil erosion rates that are two to eight times as high as rates caused solely by hikers (Wilson and Seney 1994; Deluca et al. 1998; Olive and Marion 2009). Trails with steeper grades, silty or sandy soils, and low levels of organic matter also result in erosion-prone conditions (Olive and Marion 2009). Another factor that influences erosion risk is the timing of trail use, with trail use during wet and muddy conditions and heavy rain events resulting in trail widening and higher suspended sediment loads downstream from trail stream crossings (Bayfield 1973; Ayala et al. 2005). An initial step for land managers is to identify existing stream morphologies, trail conditions, and trail use patterns with the highest soil erosion potentials, and then to balance soil erosion reducing best management practices (BMPs) with maintaining access and enjoyment for recreational trail users.

Previous research on recreational trail erosion has primarily focused on identifying underlying factors such site condition, user type, traffic frequency, user behavior, and trail design with the majority of research conducted in areas other than areas adjacent to stream crossings

(Bratton et al. 1979; Wilson and Seney 1994; Deluca et al. 1998; Olive and Marion 2009; Törn et al. 2009; Pickering et al. 2010). Sedimentation has been visually observed in streams adjacent to stream crossing approaches used by motorized and non-motorized traffic (Olive and Marion 2009; Wilkerson and Whitman 2009), and deposition rates have been quantified in streambeds below off-road vehicle ford crossings (Brown 1994). Few studies have evaluated soil erosion rates for approaches to recreational stream crossings (Ayala et al. 2005). Negative downstream impacts on water quality have been demonstrated below off-road vehicle trail and operational forest skid trail and road stream crossings (Chin et al. 2004; Ayala et al. 2005; Neal et al. 2007; Aust et al. 2011; Wear et al., 2013). Two commonly used stream crossing structures include culverts (pipe structures that allow for the diversion and passage of water underneath trails) and fords (low water crossings that allow for traffic to pass directly on top of the streambed or on supportive material such as geotextiles or hardened bottoms, allowing the passage of fish and other biota upstream) (Virginia Department of Forestry 2009). Research evaluating downstream changes in water quality below operational forest skid trail and road crossings has indicated crossing type can influence the degree to which water quality is impacted (Aust et al. 2011).

Additional research to evaluate impacts of recreational approaches and associated sediment delivery potentials into adjacent stream crossings and to quantify consequential downstream impacts on water quality is needed to better understand the implications and magnitude of trail erosion at stream crossings. Therefore, the objectives of this study were to: (1) use soil erosion models to provide soil erosion rate estimates for recreational trail approaches to selected stream crossings as potential sources of sediment inputs to the Poverty Creek watershed, a 2,145 ha watershed located in the Southern Appalachian Mountains, (2) validate sediment yield estimates using macroinvertebrate sampling as biological indication of altered

water quality below stream crossings, and (3) discern whether water quality impacts vary between culvert and ford stream crossing types.

2.3 Methods

2.3.1 Study Area

This study was conducted along the recreational trails in the Poverty Creek Trail System (Pandapas Pond Recreation Area) located within the George Washington and Jefferson National Forest in Montgomery County, Virginia (Figure 2.1). The trail system is maintained by the Eastern Divide Ranger District of the US Forest Service and is contained within the 2,145 ha Poverty Creek watershed (USDA 1972). The Poverty Creek watershed, which is a tributary to the New River, contains approximately 7.6 km of perennial streams and 47.5 km of intermittent streams, which typically carry flow from late fall through late spring/early summer. This recreational area contains 27.4 km of unsurfaced multiple-use, single-tracked trails that receive year-round, non-motorized traffic from hikers, mountain bikers, and horseback riders. Although the trail system is open year-round, user traffic is seasonal, with less trail use occurring during winter months and the heaviest traffic occurring in the months of May and October (Kratzer 1993). Construction of recreational trails first began in 1976. However, all trails used in this study were constructed after 1991, when the Pandapas Pond Recreation Area expanded recreational opportunities (Kratzer 1993). Many of the recreational trails originate from the Pandapas Pond Day-Use Area, which includes picnic areas, two parking lots, and a 3.3 ha pond that was constructed in 1951 and is stocked for fishing. Several first and second order intermittent streams supply water to the pond. Poverty Creek Trail is the longest trail within the Poverty Creek Trail System. This trail meanders along the floodplain of Poverty Creek, a first-order perennial stream originating at Pandapas Pond, and crosses the stream in multiple

locations. Several additional trails merge with the Poverty Creek Trail and lead to the ridges of

Gap and Brush Mountains after crossing several first- and second-order intermittent streams. In

total, there are approximately 47 total recreational trail stream crossings in the Poverty Creek

watershed.

Mean annual precipitation in the study area is 1039 mm (NOAA 2011). Average January

temperature is 5.6 °C and average July temperature is 24.3°C (NOAA 2011). Dominant soils

within the trail system include Craigsville series cobbly sandy loam (loamy-skeletal, mixed,

superactive mesic Fluventic Dystrudepts), Jefferson series gravelly silt loam (fine-loamy,

siliceous, semiactive, mesic Typic Hapludults), and Berks-Weikert association composed of

Berks channery loam (loamy-skeletal, mixed, active, mesic Typic Dystrudepts) and Weikert

channery silt loam (loamy-skeletal, mixed, active, mesic Lithic Dystrudepts) (USDA and NRCS

2013).

2.3.2 Field Methods

We selected eleven recreational stream crossings from the 47 total crossings present

within the Poverty Creek Trail System so that there was a representation of culvert (N = 6) and

ford (N = 5) crossing structure types (Figure 2.1). Three of the selected ford crossings (F1, F3,

and F5) were located on the first-order perennial Poverty Creek with the remaining crossings

situated on first and second-order intermittent streams (Table 2.1). A distance of 1.2 km

occurred between the F1 and F3 crossings while the F3 and F5 were 1.3 km apart (Figure 2.1).

All stream crossings occurred in areas characterized by forested overstory with rhododendron

frequently present in the understory along stream banks, typical of mountainous streams in the

Southern Appalachian Mountains. Trails crossing streams were all single-tracked and designated

for multiple-use recreational activities including horseback riding, mountain biking, and hiking.

Average trail widths ranged from 0.9 to 3.6 m for trail segments approaching stream crossings (Table 2.1).

At each stream crossing, slope length was measured as the distance along each approach that runoff would directly flow into the adjacent stream crossing without being diverted or stored (Dissmeyer and Foster 1984). Slope percent was measured along the approach length using a clinometer and slope shape was classified as linear, concave, convex, or s-shaped. Percent bare soil was observed along each trail approach at 100 points on random transects and percent canopy cover was estimated with a spherical densiometer above points with bare soil. The percentage of slope length in steps, covered with invading vegetation, and containing potential woody debris, gravel, or other on-site depression storage features was estimated for each stream approach.

Streams crossed by trails were evaluated for substrate composition in streambed, channel width, and channel slope. Substrate material in each downstream and upstream reach was visually observed and classified as either silt (< 0.06 mm), sand (< 2 mm), gravel (2 to 64 mm), cobble (65 to 256 mm), or boulder (257 to > 1024 mm) (adapted from Wolman 1954). Channel width was measured at each stream crossing (Table 2.1). Channel slope was measured over the upstream and downstream reaches combined (Table 2.1).

A rapid bioassessment using benthic macroinvertebrate sampling was used to detect changes in water quality downstream from each crossing. Upstream reaches provided reference conditions or a control to which to compare below crossing, downstream conditions. Macroinvertebrates were collected along a 100 m reach upstream and downstream from each of the 11 recreational stream crossings according to standard rapid bioassessment protocols for benthic macroinvertebrates (Barbour et al. 1999). Samples were collected over a three day

period in late April 2013, the month when Pandapas Pond Recreation Area receives the third highest number of visitors (Kratzer 1993); with upstream and downstream reaches for each crossing being sampled on the same day. We acknowledge there could potentially be additional sources of non-point sediment input within the watershed other than the trail segments adjacent to the stream crossings and as a result we intentionally sampled and analyzed upstream and downstream reaches separately and used the upstream reach as a reference in order to minimize the influence of any other potential sediment inputs on downstream reach observations.

Using a 10 m buffer between the stream crossing and the upstream and downstream reaches, we collected 20 kick samples with a 0.3 m^2 D-frame dipnet with a 0.5 mm mesh filter (Barbour et al. 1999). Kick samples were taken along each reach such that habitat type was sampled proportionally to its presence in the stream thus allowing us to associate changes in channel morphology with potential shifts in taxonomic richness. For example, if a reach was characterized by 50% pools, 25% riffles, and 25% woody debris then 10 kicks were conducted in pools, 5 in riffles, and 5 along or under woody debris. Sampling was conducted at the furthest downstream location first and then progressed upstream to avoid cross-contamination of samples. Materials collected from the kick samples were preserved with 70% ethyl alcohol, placed in a cooler, and transported to the laboratory.

2.3.3 Laboratory Work

A total of 22 macroinvertebrate samples (11 crossings, upstream and downstream reaches sampled separately) were filtered and rinsed to remove excess sediment using a No. 30 sieve (0.595 mm). Large organic matter was rinsed and removed from the composited samples following an inspection for presence of macroinvertebrates. All macroinvertebrates were identified to family-level, as possible, and enumerated (Voshell and Wright 2002). Identification

to family-level has proven to provide valuable information in analyzing macroinvertebrate-indices of water quality (Vowell 2001; Roy et al. 2003). Reference specimens were preserved in 95% ethyl alcohol.

2.3.4 Data Analysis

Soil erosion rates were estimated for trail stream crossing approaches using Universal Soil Loss Equation for forestry (USLE-Forest; Dissmeyer and Foster 1984) and Water Erosion Prediction Project (WEPP, Windows interface version 2012.8) prediction models. USLE and WEPP models were originally developed for agricultural applications, but have since been modified for forestry applications. USLE was the first widely accepted erosion prediction model and as a result has been widely used and modified in the past. WEPP was originally developed to replace USLE and to address areas where USLE was not intended for use, such as steep forested slopes. WEPP is physically based on hydrological and soil processes and requires additional input parameters than the USLE-Forest model, which is based on previously established empirical relationships between independent variables and quantified soil loss (Dun et al. 2009). Although, WEPP was designed to replace USLE, we used both models in order to evaluate how soil erosion estimates on recreational stream crossing approaches modeled using USLE compare to other erosional studies that used USLE in the past and to conduct similar comparisons using WEPP estimates. Previous comparisons of USLE and WEPP estimated erosion rates with actual sediment trap data have illustrated that these models can vary in accuracy, but have the capacity to effectively rank erosion rates from different land uses or BMP treatments (Tiwari et al. 2000; Wade et al. 2012; Brown et al. 2013).

The USLE for calculating potential soil loss is as follows:

$$A = R \times K \times LS \times CP$$

where, *A* represents the computed soil loss per unit area per unit time, *R* = rainfall and associated runoff impact, *K* = soil erodibility factor, *LS* = combined slope-length and slope-steepness factor, and *CP* = a cover-management factor (Dissmeyer and Foster 1984). A rainfall index value (*R*) of 150 erosion index units (EI) was obtained for the Poverty Creek Trail System area from Wischmeier and Smith's isoerodent lines published in Dissmeyer and Foster (1984). For whole-soil erodibility factors (*K*), we used 0.28 for stream approaches on soils from the Jefferson series, 0.24 on Berks and Weikert soils, and 0.17 on Craigsville soils (USDA and NRCS 2013). For each approach, the slope-length and slope-steepness factor (*LS*) was calculated as:

$$LS = \left(\frac{\lambda}{72.6}\right)^{m} \times (65.41 \sin^2 \theta + 4.65 \sin \theta + 0.065)$$

where, λ = slope length in feet, θ = slope angle in degrees, and *m* = 0.2 for slopes < 1%, 0.3 for slopes 1 to 3%, 0.4 for slopes 3.5 to 4.5%, and 0.5 for slopes ≥ 5% (Dissmeyer and Foster 1984). Recreational trail approaches were treated as tilled soils due to the absence of topsoil and heavy trafficking. Therefore, the cover-management practice factor (*CP*) was a product of the relevant subfactors for tilled soils including: (a) bare soil, residual binding, and soil reconsolidation, (b) canopy above bare soil, (c) percentage of slope length in sediment trapping steps, (d) presence of fine roots associated with invading vegetation, and (e) percentage of slope with onsite-depression storage features (Dissmeyer and Foster 1984).

Soil erosion rates were also estimated for each trail crossing approach using WEPP software which was downloaded from the National Soil Erosion Research Lab (http://www.ars.usda.gov/Research/docs.htm?docid=10621). WEPP associated climate, soil, and land cover files were also obtained from the National Soil Erosion Research Lab. Simulations predicting soil loss were conducted following input of slope characteristics and selected climate, soil, and land cover or management files. Local climate data were obtained through the CLIGEN (version 4.30) parameter file for station Blacksburg 2 VA which was located approximately 6 km from the crossing locations. Appropriate soil files for each Virginia based soil series were determined based on each approach location, and the associated parameters were used for the WEPP simulations (USDA and NRCS 2013). Approaches situated on the Craigsville series had an interill erodibility 4.8 x 10^6 kg s m^{-4}, rill erodibility of 0.006 s m^{-1}, critical shear of 2.9 Pa, and a hydraulic conductivity of 8.6 mm h^{-1}. Parameter values for approaches on the Jefferson series were 5.4 x 10^6 kg s m^{-4} for interill erodibility, 0.020 s m^{-1} for rill erodibility, 3.5 Pa for critical shear, and 5.11 mm h^{-1} for hydraulic conductivity. Approaches located on the Berks-Weikert soils had an interill erodibility 5.1 x 10^6 kg s m^{-4}, rill erodibility of 0.015 s m^{-1}, critical shear of 3.5 Pa, and a hydraulic conductivity of 3.8 mm h^{-1}. Recreational trails are currently not an existing management type in the WEPP model; therefore, we used the "Forest Bladed Road" management type as it was most similar to the unsurfaced trail approaches and despite the name, represented initial road conditions and did not include a recent blading treatment for trail segments. Modeled soil erosion rates were multiplied by the area of each approach to determine the potential sediment yield at each crossing. Modeled soil erosion rates and sediment yields estimated by USLE and WEPP models for the two approaches at each crossing were averaged so that one value represented each selected crossing. USLE-Forest and

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WEPP estimated soil erosion rates and sediment yields along the approaches were characterized and compared between culvert and ford crossing types using the NPAR1WAY procedure, two-sample Wilcoxon rank sum exact tests, in SAS version 9.3 (SAS 2012).

Macroinvertebrate data from each reach were entered into the Ecological Data Application System (EDAS) obtained from the Virginia Department of Environmental Quality (http://www.deq.virginia.gov/Programs/Water/WaterQualityInformationTMDLs/WaterQuality Monitoring/BiologicalMonitoring.aspx) to calculate two macroinvertebrate-based water quality indices: Family-level Hilsenhoff Biotic Index (FHBI) and Family-level Virginia Stream Condition Index (FVSCI). The FHBI is calculated (adapted from Lenat 1993) as:

$$FHBI = \sum \left(\frac{TV_i N_i}{N} \right)$$

where, *FHBI* is the water quality index value that ranges from 0 (excellent) to 10 (poor); $TV_i =$ the family-level tolerance value (0 to 10; least to most tolerant of pollution), N_i abundance of ith family or taxa, and *N* total number of individuals in the sample (Hilsenhoff 1988). The FVSCI is a multi-metric index that incorporates 8 metrics: (a) number EPT (pollution-sensitive Ephemeroptera, Plecoptera, and Trichoptera) taxa, (b) number total taxa, (c) % Ephemeroptera, (d) % Plecoptera and Trichoptera minus Hydropsychidae, (e) % Chironomidae, (f) % top two dominant taxa, (g) FHBI, and (h) % scrapers (Burton and Gerritsen 2003). FVSCI values range from 0 to 100, with FVSCI < 61 classified as 'impaired' according to Virginia's DEQ monitoring standards. Based on these indices, if water quality decreases below stream crossings, we would expect to observe increased FHBI levels and decreased FVSCI based on the respective scale of each index. In addition to the calculated indices, we also noted changes that occurred in

the percent of each sample composed by macroinvertebrates from the family Chironomidae and order Oligochaeta as these have been found to be more tolerant of increased sediment levels in streams (Longing et al. 2010). Clinger organisms, species that "cling" to boulders, cobble, and woody debris, were also observed for change in downstream reaches as they are less tolerant of increased fine sediment and thus, typically decline in number and richness in streams with higher sediment loads and turbidity levels (Longing et al. 2010; Pollard and Yuan 2010).

Correlations were examined to evaluate the relationship between estimated potential sediment yield and downstream change in the FHBI and FVSCI macroinvertebrate-based index values as biological indication of altered water quality below stream crossings. Specifically, Spearman's rank sum correlation coefficients (ρ) were evaluated between the percent change in FHBI and FVSCI indices downstream, (downstream minus upstream value/upstream value)*100%, and predicted sediment yield calculated by multiplying each respective approach area by the predicted soil erosion rate by USLE and WEPP models. To evaluate whether stream crossing structure type is influential to macroinvertebrate communities, downstream changes in the FHBI and FVSCI indices were compared between culvert and ford crossing types using the NPAR1WAY procedure in SAS (SAS 2012). All statistical analyses were performed at a significance-level $\alpha = 0.05$.

2.4 Results

2.4.1 Modeled Soil Erosion Rates and Sediment Yields for Approaches

Modeled soil erosion rates were higher for unsurfaced recreational trail approaches leading to fords than to culvert crossings (Figure 2.2; Table 2.2). Predicted erosion rates ranged from 1.2 to 8.8 tonnes ha^{-1} yr^{-1} for culvert and 2.0 to 9.7 tonnes ha^{-1} yr^{-1} for ford approaches based on the USLE-Forest model estimates. Annual erosion rates predicted by USLE-Forest

were not significantly different between approaches to culvert and ford crossing types (P = 0.429; Figure 2.2). WEPP estimated erosion rates ranged from 2.5 to 10.0 tonnes ha^{-1} yr^{-1} for culvert approaches and 10.7 to 20.5 tonnes ha^{-1} yr^{-1} for ford approaches. Annual soil erosion rates modeled using WEPP were significantly higher (P = 0.004) for approaches to fords than to culvert crossings (Figure 2.2).

Predicted sediment yields, determined by multiplying the approach area by the respective modeled erosion rate, ranged from 4.0 to 20.3 kg yr^{-1} for culvert and 1.8 to 50.8 kg yr^{-1} for ford approaches when calculated using USLE-Forest (Figure 2.3; Table 2.2). Predictions from the WEPP model for approaches to culvert crossing types ranged from 7.7 to 30.4 kg yr^{-1}; whereas ford approaches ranged from 12.0 to 137.5 kg yr^{-1} in estimated sediment yield. Sediment yield values along the approaches were not significantly different between culvert and ford crossing types based on modeled estimates from USLE-Forest (P = 0.792) or WEPP (P = 0.177).

2.4.2 Stream Characterization for Macroinvertebrate Habitat

Streams crossed by culverts were classified as either first or second-order intermittent streams, while fords crossed a mixture of first and second-order intermittent as well as the first-order perennial Poverty Creek (Figure 2.1; Table 2.1). Streambeds were primarily composed of cobble and gravel which is typical of first-order mountainous streams in the southern Appalachian Mountains. The three fords (F1, F3, F5) crossing the first-order perennial Poverty Creek were also characterized by boulders in the streambed. Smaller particles, sand and silt, were observed more frequently in downstream reaches than in upstream reference reaches.

2.4.3 Stream Crossing Impacts on Macroinvertebrates and Water Quality

Downstream reaches had higher Family-level Hilsenhoff Biotic Index (FHBI) and lower Family-level Virginia Stream Condition Index (FVSCI) values which reflected a decrease in

water quality below stream crossings, downstream, than in upstream reaches (Table 2.3). As previously described, the FHBI values range from 0 (excellent) to 10 (very poor quality), while FVSCI values range from 0 to 100, with values < 61 indicating impairment. The majority (75%) of upstream reaches were classified as having 'excellent' water quality according to FHBI and were 'not impaired' (58%) as determined by FVSCI. However, when downstream reaches were examined, a different trend surfaced. Fewer downstream reaches were considered to have 'excellent' (17%) water quality with more downstream reaches falling into the 'very good' (42%), 'good' (33%), and 'fair' (8%) FHBI categories. Based on the FVSCI, the majority of reaches downstream from crossings had 'impaired' (83%) water quality.

Evaluation of Spearman's rank sum correlation coefficients indicated our USLE-Forest sediment yield estimates along the crossing approaches were significantly correlated (ρ = -0.69; P = 0.019) with percent downstream change in FVSCI values. However, the relationships between percent downstream change in FHBI and USLE-Forest sediment yield estimates (ρ = 0.20; P = 0.555) or between the percent changes in FHBI (ρ = 0.25; P = 0.467) and FVSCI (ρ = -0.22; P = 0.518) and WEPP sediment yield estimates were not significant. Crossing type did not appear to have a significant influence on the observed percent change in downstream values for FHBI (P = 0.177) and FVSCI (P = 0.662) between culvert and ford crossing types. Mean downstream changes in FHBI and FVSCI appeared to be similar between crossings located on perennial streams (n =3; FHBI = +1.0 and FVSCI = -15.5) when compared to crossings located on intermittent streams (n = 8; FHBI = +0.7 and FVSCI = -9.0; Table 2.3).

A total of 29 families representing 9 orders were observed during macroinvertebrate sampling in upstream and downstream reaches (Table 2.4). Of particular importance were families of the more pollution sensitive order Ephemeroptera: Baetidae, Ephemerellidae,

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Ephemeridae, and Heptageniidae and order Plecoptera: Chloroperlidae, Nemouridae, Perlidae, and Perlodidae. Although the majority of these families were present in both upstream and downstream reaches, changes in the numbers, proportion, and dominance of each taxa found in reach samples occurred, thus creating change to the calculated macroinvertebrate-based water quality indices (Table 2.3). As an additional indication of change: total number of taxa; percent EPT (excluding the more pollution tolerant Hydropsychidae); and the percent of macroinvertebrates classified as clingers decreased below most recreational stream crossings (Table 2.3). The proportion of macroinvertebrates of the more pollution tolerant Chironomidae and Oligochaeta organisms was higher in nearly all downstream reaches (75%).

2.5 Discussion

2.5.1 Modeled Soil Erosion Rates at Stream Crossings

Estimated soil erosion rates and corresponding sediment yield potentials predicted by WEPP were consistently higher than estimates derived from the USLE-Forest (Table 2.2). USLE-Forest and WEPP are models and were used in this to study as a tool to provide multiple soil erosion rate estimates and to allow for comparison of rates published for other forestland disturbances to characterize the magnitude of soil erosion along recreational trail approaches to stream crossings. Similar discrepancies between USLE-Forest and WEPP models have been reported for skid trails, forest roads, and forest road approaches to stream crossings (Aust et al. 2011; Wade et al. 2012; Brown et al. 2013). However, in each of these studies, despite accuracy issues, USLE-Forest and WEPP models proved capable of ranking conditions with greatest erosion potentials. Estimates associated with the WEPP model were based on the management file "Forest Bladed Road" which potentially contributed to higher erosion rate predictions in WEPP than in the USLE-Forest Model. This management type was chosen as it was most

similar to the conditions along the unsurfaced, high-traffic recreational trails. Differences in model parameters also contributed to estimate differences between the two models. USLE-Forest incorporates one overall erodibility factor for each soil series while WEPP includes different rill and interill erodibility factors, rate of effective hydraulic conductivity, and critical shear values for each soil type. Due to these differences between models, predicted soil erosion rates from this study were only compared with estimates from other studies that used the same erosion prediction model.

Soil erosion rates predicted by USLE-Forest were higher for recreational trail approaches (an average of 4.1 tonnes ha^{-1} yr^{-1} for all stream crossings; Figure 2.2) than published USLE-Forest estimates for adjacent forests. In the same forest type and physiographic region as this study, average estimated erosion rates were found to be 0.3 tonnes ha^{-1} yr^{-1} in undisturbed forests and 1.7 tonnes ha^{-1} yr^{-1} in two-year old clearcuts (Hood et al. 2002). Erosion estimates from this study were found to be higher than published erosion estimates (0.0 to 0.7 tonnes ha^{-1} yr^{-1}) for eastern forests (Patric et al. 1976). If trail surfaces are highly eroded, as in the case of the Poverty Creek Trail System, recreationists' enjoyment of the trail-use experience may be significantly decreased (Lynn and Brown 2003).

Unsurfaced recreational trails examined in this study were used for multiple non-motorized (horseback riding, mountain biking, and hiking) activities. Average WEPP erosion rate estimates for non-motorized trail approaches to stream crossings in this study were lower (average of 9.7 tonnes ha^{-1} yr^{-1} for all crossings; Figure 2.2) than WEPP estimates for motorized off-road vehicle (ORV) recreational trail approaches to a single ford stream crossing which reached 126.8 tonnes ha^{-1} (Ayala et al. 2005). USLE-Forest erosion estimates in this study were also lower than USLE-Forest erosion estimates reported along bare horse trails (34.6 to 69.2

tonnes ha^{-1} yr^{-1}) but, within the range of estimates where gravel had been applied (2.0 to 16.3 tonnes ha^{-1} yr^{-1}) (Aust et al. 2005). Due to the limited number of recreation-based soil erosion studies, erosion estimates for the unsurfaced trail approaches were also compared with erosion rates along forest operation skid trails. Soil erosion estimates from the USLE-Forest model along the multiple-use recreational trail stream approaches were lower than USLE-Forest predicted erosion rates along bladed forest skid trails where minimal, water bars only, BMPs were implemented (63.1 tonnes ha^{-1} yr^{-1}) (Wade et al. 2012). Erosion estimates in this study were more similar to USLE predicted soil erosion rates for forest skid trails where hardwood slash (4.3 tonnes ha^{-1} yr^{-1}), mulch (3.2 tonnes ha^{-1} yr^{-1}), and pine slash (1.6 tonnes ha^{-1} yr^{-1}) had been applied, according to BMPs to reduce soil erosion rates, than to erosion rates on bare trails (Wade et al. 2012).

Approaches leading to ford crossings had greater predicted USLE-Forest and WEPP soil erosion rate estimates than approaches to culverts (Figure 2.2). Differences in approach erosion estimates between the two crossing types were significant when the WEPP erosion estimates were evaluated, but not when based on USLE-Forest estimates. Observed differences were most likely attributed to site conditions on which the approaches occurred. For instance, slopes along approaches to fords (9.1%) were greater on average than those to culverts (3.5%) (Table 2.2). Steeper slopes contributed to greater combined slope-length and slope-steepness factor (*LS*) values in the USLE-Forest model for ford approaches (0.73) when compared to culvert approaches (0.27) and had a similar impact on predicted estimates in the WEPP model. Trails and roads characterized by steeper grades or approach lengths have been shown to have higher actual and modeled soil erosion rates (Dissmeyer and Foster 1984; Olive and Marion 2009; Brown et al. 2013).

2.5.2 Stream Crossing Impacts on Macroinvertebrates and Water Quality

Downstream changes in macroinvertebrate-based index and associated metric values observed in this study illustrated a detectable decrease in water quality below trail stream crossings (Table 2.3). Previous research conducted below road and trail stream crossings has illustrated the negative impacts of soil erosion along crossing approaches and associated crossing traffic disturbances can have on water quality (Lane and Sheridan 2002; Ayala et al. 2005; Aust et al. 2011; Wear et al. 2013). Unsurfaced approaches have been cited as the major source of sediment deposition downstream from low-water ORV ford crossings (Brown 1994). Similar to water quality impacts at forest operational crossings, water samples collected downstream from an ORV ford crossing were found to contain higher suspended sediment loads than samples from upstream controls (Ayala et al. 2005). In this study, a negative relationship was found between increased potential sediment yields and downstream change in the FVSCI, illustrating that as estimated sediment yield increased, the FVSCI value decreased indicating decreased water quality. Similarly, a positive relationship was identified between sediment yields and FHBI indices illustrating that as sediment yield increased the FHBI value increased, also indicating a trend of decreased water quality with increased estimated sediment yield levels.

Stream crossing type, culvert or ford, did not appear to impact changes in the macroinvertebrate-based indices illustrating that water quality was impacted by stream crossings in general rather than by specific crossing type in this study. Further, although downstream changes in macroinvertebrates were detected indicating water or habitat quality changes were occurring, many of the changes were not significant from a statistical standpoint. Other studies have reported that there may be a time lag between decreased water quality and detection of these differences in macroinvertebrate populations (Kaller and Hartman 2004). Given the evidence we found of changes in streambed substratum composition below crossings (silt and

sand were more frequent; Table 2.1) the full implications of stream crossing erosion may have yet to become detectable in the macroinvertebrate populations. Changes in streambed conditions may translate into changes in macroinvertebrate behaviors rather than detectable changes in macroinvertebrate populations (Rosenberg and Wiens 1978). Further, changes in water quality in this study were observed primarily based on changes in calculated index values; these values serve to indicate change and are not an absolute measure of abiotic factors associated with water quality. In this study, spatial location of the crossings within the watershed may have also influenced water quality and macroinvertebrate assemblages as topography, hydrology, and in-stream habitat can influence presence of macroinvertebrate assemblages (Longing et al. 2010). For example, Culvert 6, located lower in the watershed than the other crossings, had an 'impaired' upstream reach according to FVSCI criteria (Figure 2.1). This upstream impairment may have masked downstream changes in macroinvertebrate-based index and metric values (Table 2.3).

2.6 Conclusions

We modeled soil erosion rates that were 13 times higher along trail approaches than in previously evaluated adjacent control forestlands (Hood et al. 2002). Estimated soil erosion rates were within the same range as skid trails and horse trails where BMPs had been implemented. Despite crossing type, downstream changes in FHBI and FVSCI values indicated reduced water quality with increased estimated sediment yield potentials despite whether approaches were to a culvert or ford crossing type. Downstream changes in macroinvertebrate-based water quality index values illustrated recreational stream crossings are having an impact on conditions in streams crossed by recreational trails in the Poverty Creek Trail System. Results from this study further support the need for implementing BMPs to reduce soil erosion and protect water quality.

Increased usage rates and expansion of recreational trails justifies the timeliness of

evaluating the impacts of recreational trail approaches and stream crossings on water quality in

other recreational areas and should be considered in designing or re-locating trails. Future

research should be aimed at quantifying trail-eroded sediment that is actually delivered into

streams at stream crossings, measurement of below crossing changes in stress-related water

quality variables such as turbidity, TSS, and nutrient-levels, and evaluation of the effectiveness

of recreational BMPs at reducing soil erosion and ameliorating sediment entry into waterways.

2.7 Acknowledgments

2.8 References

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Table 2.1 Stream and trail width characteristics at selected culvert (C) and ford (F) stream crossings. Streams crossed by recreational trails are abbreviated as I (intermittent) and P (perennial).

Crossing	Stream class & order	Streambed substrates[a] Upstream	Downstream	Channel slope (%)	Channel width (m)	Trail width (m)
C1	I (1)	Cobble	Cobble	1.0	3.7	3.6
C2	I (2)	Cobble, gravel, sand	Gravel, sand, silt	3.0	1.8	3.4
C3	I (2)	Cobble, gravel, sand	Gravel, sand, silt	4.0	1.7	3.4
C4	I (1)	Cobble, gravel	Sand, silt	4.0	4.8	1.7
C5	I (1)	Cobble, gravel	Cobble, gravel	4.0	2.6	2.7
C6	I (2)	Cobble, gravel	Cobble, gravel	8.0	3.7	0.9
F1	P (1)	Boulders, cobble, gravel	Boulders, cobble, gravel, sand	1.0	4.4	2.4
F2	I (2)	Cobble, gravel	Cobble, gravel	3.0	2.1	1.1
F3	P (1)	Boulders, cobble, gravel	Gravel, sand, silt	1.0	6.9	3.4
F4	I (1)	Cobble	Cobble	4.0	1.5	1.5
F5	P (1)	Boulders, cobble, gravel	Boulders, cobble, gravel, sand	0.0	8.2	1.4

[a] Based on visual observations of pebble size classes (Wolman 1954)

Table 2.2 Average slope, modeled soil erosion rate (tonnes ha^{-1} yr^{-1}), and potential sediment yield (kg yr^{-1}) for trail approaches to selected culvert (C) and ford (F) stream crossings in the Poverty Creek Trail System in Southwestern Virginia. Values were estimated using Universal Soil Loss Equation for Forestry (USLE-Forest) and Water Erosion Prediction Project (WEPP) models. Presented crossing values are the average of the left and right approach values with the standard deviation in parentheses.

Crossing	Slope (%)	Estimated soil erosion rate (tonnes ha^{-1} yr^{-1})		Potential sediment yield (kg yr^{-1})	
		USLE-Forest	WEPP	USLE-Forest	WEPP
C1	2.0 (1.4)	1.2 (0.9)	2.7 (0.4)	5.9 (5.3)	12.9 (5.1)
C2	1.0 (0.0)	2.3 (0.0)	2.5 (0.0)	7.1 (0.5)	7.7 (0.5)
C3	4.0 (2.8)	5.2 (4.8)	7.8 (7.1)	20.3 (19.3)	30.4 (28.2)
C4	7.5 (0.7)	8.8 (1.7)	10.0 (2.5)	19.6 (2.6)	22.2 (4.2)
C5	2.0 (1.4)	1.7 (0.6)	2.9 (0.6)	8.4 (9.3)	13.0 (13.5)
C6	4.8 (0.4)	2.7 (1.9)	6.4 (1.0)	4.0 (3.4)	8.5 (0.5)
F1	9.0 (7.1)	4.6 (3.4)	10.7 (10.6)	14.2 (10.7)	33.3 (33.3)
F2	12.0 (0.0)	2.4 (0.4)	20.5 (8.9)	1.8 (1.7)	20.9 (24.8)
F3	9.5 (3.5)	6.7 (2.6)	19.8 (8.7)	50.8 (28.5)	137.5 (34.9)
F4	4.5 (2.1)	2.0 (0.2)	10.7 (6.0)	4.1 (0.1)	23.1 (15.1)
F5	10.5 (2.1)	9.7 (7.8)	14.7 (4.5)	6.8 (2.0)	12.0 (3.4)

Table 2.3 Downstream changes (downstream minus upstream) in Family-level Hilsenhoff Biotic Index (FHBI) and Family-level Virginia Stream Condition Index (FVSCI) values and along with some of the metrics reflected in the FVSCI: number of total taxa; percent of sample composed by Ephemeroptera, Plecoptera, and Trichoptera minus Hydropsychidae (EPT), Chironomidae, Oligochaeta, and clingers. Positive FHBI and negative FVSCI values indicate a decrease in associated water quality for downstream reaches when compared to upstream reference reaches at for the 11 crossings examined in the Poverty Creek Trail System. Change in metric values were negative if metric values decreased downstream. Values in italics represent average change for culvert and ford crossing types. Streams crossed by recreational trails are abbreviated as I (intermittent) and P (perennial).

Crossing	Stream class & order	Index FHBI	Index FVSCI	Taxa (#)	EPT (%)	Chironomidae (%)	Oligochaeta (%)	Clingers (%)
C1	I (1)	0.3	-7.0	-4	-14	9	9	-13
C2	I (2)	0.4	-14.8	-1	-25	-4	33	-14
C3	I (2)	0.5	-8.8	0	-11	-2	-1	-3
C4	I (1)	1.2	-23.7	-4	-33	3	37	-29
C5	I (1)	1.8	-13.5	-4	-54	14	36	-37
C6	I (2)	0.0	-4.5	-1	1	2	-2	-7
Average		*0.7*	*-12.1*	*-2.3*	*-22*	*4*	*19*	*-17*
F1	P (1)	0.9	-26.8	-4	-42	1	50	-25
F2	I (2)	0.9	-1.7	0	-33	1	22	-28
F3	P (1)	1.1	-11.7	1	-40	5	31	-22
F4	I (1)	0.5	3.0	0	11	0	-2	2
F5	P (1)	1.1	-7.8	1	-23	10	13	-22
Average		*0.9*	*-9.0*	*-0.4*	*-25*	*3*	*23*	*-19*

Table 2.4 List of macroinvertebrates (order: family) found in reaches above (upstream) and below (downstream) the 11 selected stream crossings along the Poverty Creek Trail System.

Order	Family	Upstream	Downstream
Amphipoda	Unknown	x	x
Coleoptera	Unknown	x	
	Dytiscidae		x
	Elmidae	x	x
	Psephenidae	x	x
Diptera	Athericidae	x	x
	Ceratopogonidae	x	x
	Chironomidae	x	x
	Unknown	x	x
	Empididae	x	x
	Simuliidae	x	x
	Stratiomyidae	x	
	Tabanidae	x	x
	Tipulidae	x	x
Ephemeroptera	Baetidae	x	x
	Ephemerellidae	x	
	Ephemeridae	x	x
	Heptageniidae	x	x
Megaloptera	Corydalidae	x	x
	Sialidae		x
Odonata	Gomphidae	x	x
	Libellulidae		x
Oligochaeta	Unknown	x	x
Plecoptera	Chloroperlidae	x	
	Nemouridae	x	x
	Perlidae	x	x
	Perlodidae	x	x
Trichoptera	Hydropsychidae	x	x
	Lepidostomatidae	x	x
	Limnephilidae	x	x
	Philopotamidae	x	x
	Polycentropodidae	x	x
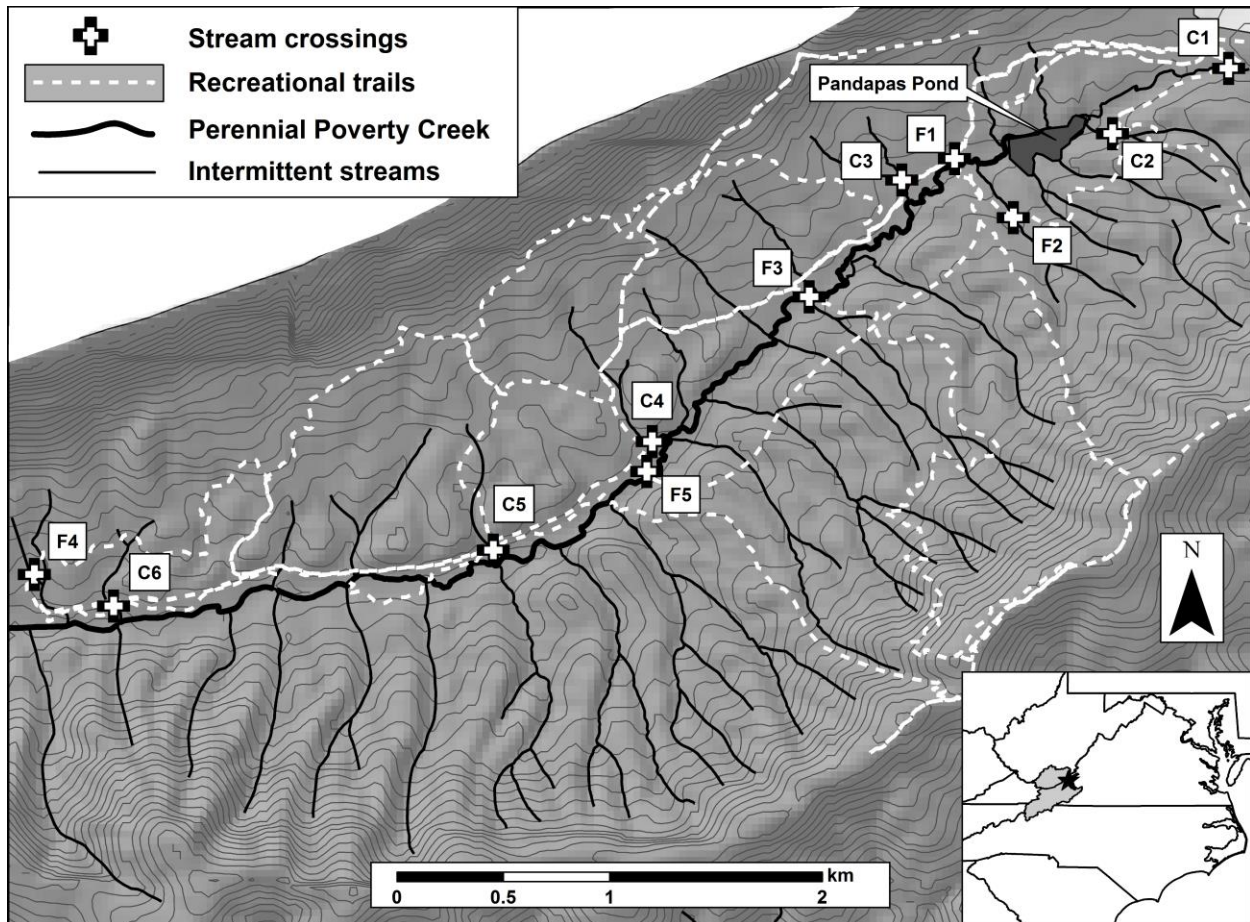
Figure 2.1 Recreational stream crossings located throughout the Poverty Creek trail system in the Washington and Jefferson National Forest in Southwestern Virginia. Crossings are labeled as C (culvert) and F (ford) followed by the respective crossing number. Contour lines represent 10 m of change in elevation. Shaded areas in inset illustrate the upper and middle New River sub-basins.
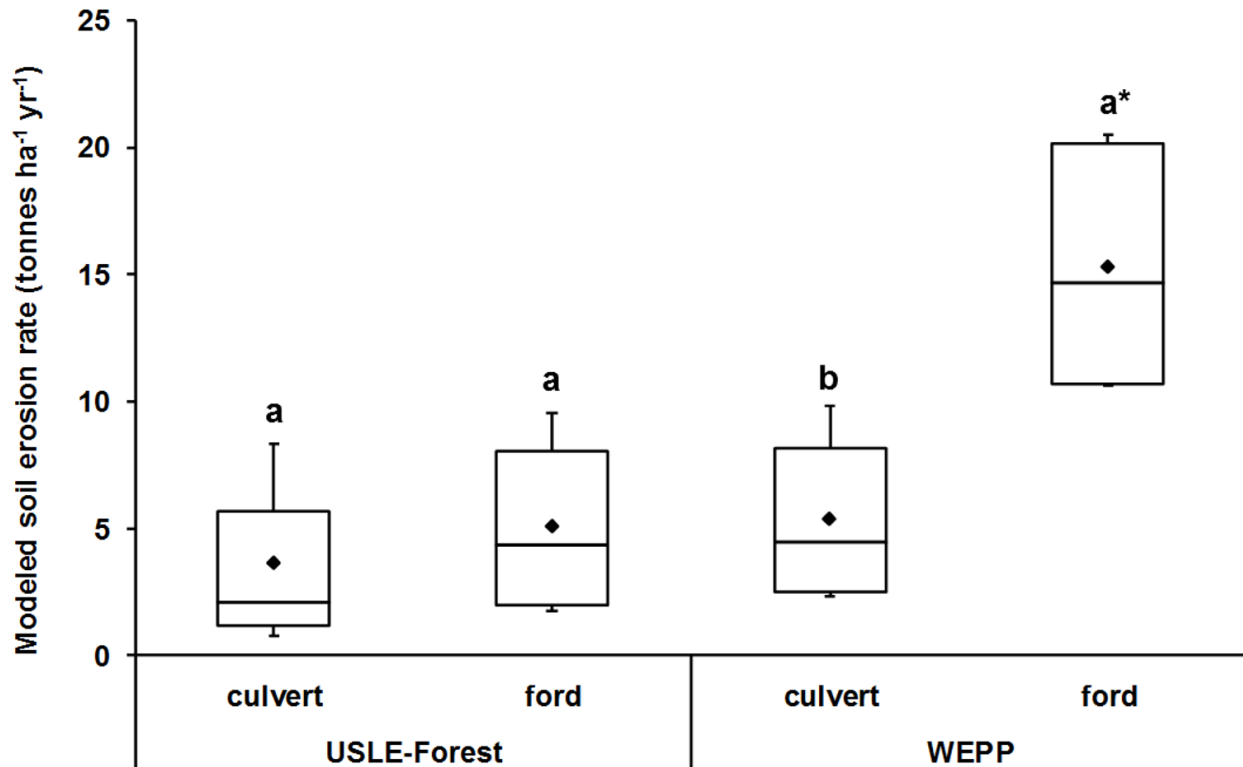
Figure 2.2 Estimated soil erosion rates (tonnes ha^{-1} yr^{-1}) for approaches to culvert and ford crossing types modeled using USLE-Forest and WEPP prediction models. Crossing types with different letters within each model were significantly different at $\alpha = 0.05$. Differences were determined using Wilcoxon rank sum exact tests. Diagonal data points represent means for culvert (N = 6) and ford (N = 5) crossings.
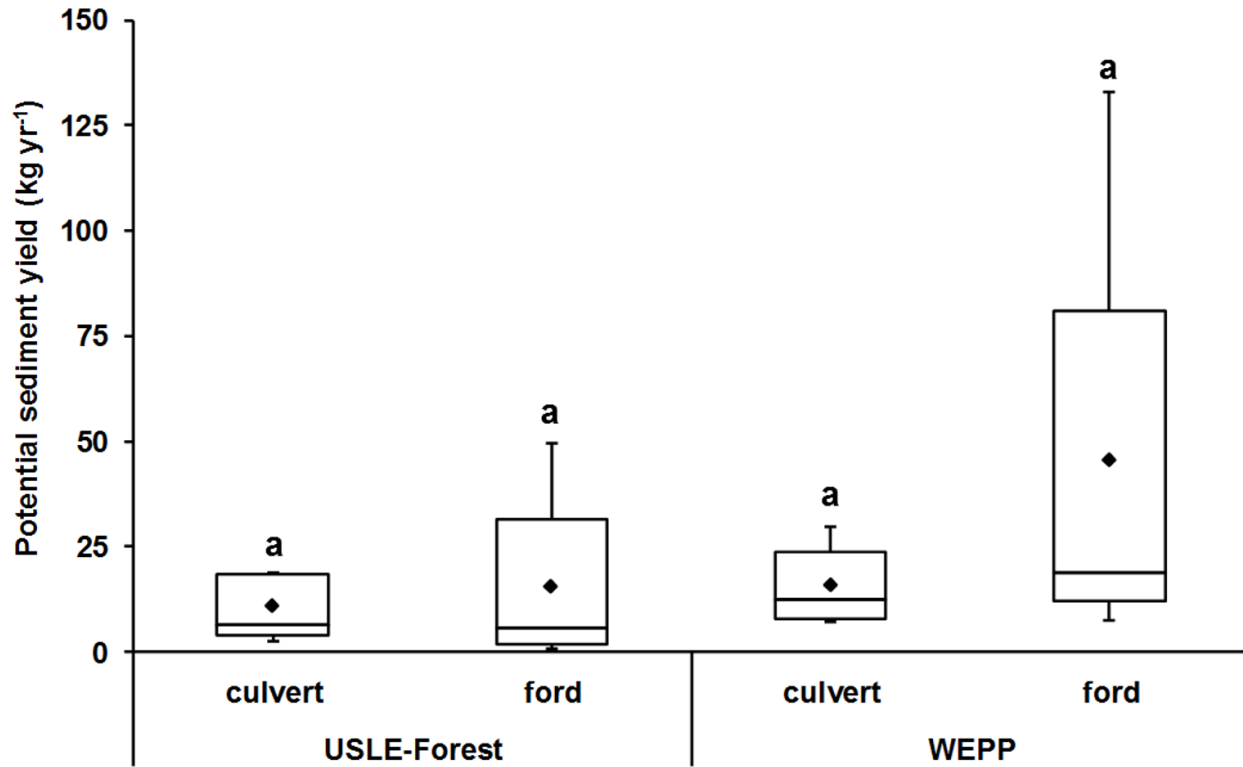
Figure 2.3 Potential sediment yields (kg yr^{-1}) for approaches to culvert and ford crossing types using USLE-Forest and WEPP prediction models. Crossing types with different letters within each model were significantly different at $\alpha = 0.05$. Differences were determined using Wilcoxon rank sum exact tests. Diagonal data points represent means for culvert (N = 6) and ford (N = 5) crossings.

CHAPTER 3. RECONSTRUCTION OF SHORT- AND LONG-TERM SEDIMENT ACCRETION RATES AND IMPACTS OF OVERBANK FLOODING ON GREEN ASH AND WATER TUPELO RADIAL GROWTH IN RIPARIAN FORESTED WETLANDS ALONG THE TENSAW RIVER, ALABAMA, USA

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3.1 Abstract

Riparian forested wetlands improve downstream water quality by trapping suspended sediment from adjacent waterways during overbank flooding. Overbank floodwaters also transport nutrients into stagnant backswamps enhancing site productivity. In this study, we used a dendrogeomorphic technique with green ash (*Fraxinus pennsylvanica* Marsh.) to estimate short- (25 years) and long-term (131 years) sediment accretion rates along a natural levee (35 m from river) and backswamp (75 m from river) riparian forested wetland along the Tensaw River in southern Alabama. We identified the influence of hydrologic and climatic fluctuations on ring width index for green ash and water tupelo. Sediment accretion estimates were higher for the natural levee (1.6 cm yr^{-1}) than the backswamp (1.2 cm yr^{-1}) during the short-term period. Dendrogeomorphic estimates of sediment accretion rates were higher on the natural levee than sediment pin and elevation survey estimates further from the river (160 to 330 m) in the backswamp. Conversion of sediment accretion depths to weights indicated these riparian forests trapped between 100.4 Mg ha^{-1} yr^{-1} (35 m from river) and 37.8 Mg ha^{-1} yr^{-1} (250 m from river) during the short-term period (1987 to 2012). Positive, significant correlations were identified between green ash radial growth and the number of days above flood stage and average daily river stage during April. Green ash trees along the natural levee and backswamp appeared more

responsive to changes in hydrology and climate than water tupelo (*Nyssa aquatica* L.) trees located further in the backswamp. This study highlights the important role riparian forested wetlands play in trapping sediment and how trapping efficiency can change through time. Results from this study illustrate the use of dendrochronological techniques to identify changes in sediment accretion through time and impacts of hydrology and climate on tree growth with minimized site visits.

Keywords: Forested wetlands; Sediment accretion; Hydrologic regime; Dendrochronology; Green ash; Water tupelo

3.2 Introduction

Bottomland hardwood forests, which are one of the more well-known types of riparian forested wetlands, improve water quality by trapping sediment from adjacent waterways during overbank flooding events (Boto and Patrick, 1978; Walbridge, 1993). Riparian forested wetlands reduce the velocity and peaks of overbank floodwaters by providing area in which surface roughness is increased by vegetation and other vertical structures, allowing suspended particles to fall out of suspension and to be deposited. Bottomland hardwood forests are highly productive systems with hydrologic cycles of wet and dry periods created through variations in overbank flooding, precipitation, and evapotranspiration rates (Broadfoot and Williston, 1973; Wharton et al., 1982, Hupp, 2000). Forested wetland morphology and function are highly influenced by hydrologic regime and sediment deposition patterns (Hunter et al., 2008). Thus, alterations in riverine hydrology and associated changes in sediment deposition impact these adjacent riparian wetland systems. Dendrochronology can be used to estimate previous sediment accretion rates and assess the impacts of altered hydrology through examination of the variation in annual tree-ring growth patterns (Reily and Johnson, 1982; Anderson and Mitsch, 2008; Gee,

2012). This information can help predict future impacts of hydrology and sediment deposition patterns on forest productivity, regeneration, carbon storage, bottomland restoration, and water quality (Conner and Day, 1976; Hodges, 1997; Hunter et al., 2008).

Repeated sediment deposition over time creates elevated, better-drained microhabitats that allow less flood-tolerant species to become established. Thus, sediment deposition plays an integral role in succession in bottomland forests (Hodges, 1997; McKee et al., 2012; Mitsch et al., 2014). Hydroperiod length, overbank flooding occurrence, and stream flow characteristics have been found to influence sediment accumulation in forested wetlands (Kleiss, 1996; Hupp, 2000; Hupp et al., 2008; Hupp et al., 2015). Coastal rivers transport sediment that originates upstream. When rivers are elevated, overbank floodwaters transport this suspended sediment into adjacent riparian forested wetlands (Kleiss, 1996). As the velocity of the floodwaters is reduced, sediment falls out of suspension and is deposited. Vertical structures such as vegetation, microtopographical features, and woody debris increase surface roughness which contributes to the decreased velocity of overbank floodwaters (Hupp, 2000). Some of the common microtopography features found in riparian areas that are influenced by hydrology and sediment deposition include natural levees (depositional areas immediately adjacent to streams), sloughs (abandoned stream channels filling with sediment), and flats or backswamps (sediment deposition influenced by distance from stream and vegetation) (Hodges, 1997).

Distributions and growth rates of individual species have been shown to vary among microtopographic landforms (e.g., levees vs. backswamps) in bottomland forests due to differences in flood tolerance among species (Broadfoot and Williston, 1973; Wharton et al., 1982). Sediment deposited from overbank floodwaters provides an allochthonous, external, source of nutrients to palustrine forested wetlands (Johnston et al., 1984). These additional

nutrients allow species such as water tupelo to achieve maximum growth, enhancing above ground productivity levels (Keeland and Sharitz, 1995). Changes in hydrologic regime can alter sediment deposition and inflow patterns and, as a consequence, can alter tree growth and productivity in forested wetlands (Conner and Day, 1976; Mitsch and Rust, 1984). Sediment deposition was also identified as an important potential factor in the recovery of bottomlands disturbed by harvesting equipment (McKee et al., 2012).

Dendrochronological techniques can be used to identify the occurrence and impacts of altered site hydrology on tree growth (Anderson and Mitsch, 2008). Growth responses to climatic (e.g., precipitation and temperature) and environmental conditions (e.g., light, water, and nutrient availability) create year-to-year variability in tree-ring widths which are reflective of growth conditions experienced by the tree (Fritts, 1976).

Previous research has concluded hydrologic connectivity for a spatial location to sediment-laden waters is the single most comprehensive factor controlling temporal and spatial patterns of sediment deposition (Hupp, 2015). Above ground productivity in forested palustrine wetlands is dependent on several factors including: silvical characteristics for individual species, hydrologic regime, nutrient availability, edaphic conditions, and previous disturbance history (Broadfoot and Williston, 1973; Conner and Day, 1976; Reily and Johnson, 1982; Wharton et al., 1982). The majority of previous studies have focused on annual growth response to hydrologic regime in either lower elevated swamp dominant species (Dicke and Toliver, 1990; Conner and Day, 1992; Keeland and Sharitz, 1995) or in a collection of bottomland hardwood species (Reily and Johnson, 1982; Mitsch and Rust, 1984; Anderson and Mitsch, 2008; Predick et al., 2009; Keim and Amos, 2012), but rarely both (Conner et al., 2014). Forested wetlands along meandering coastal plain rivers in the southeastern United States are generally

characterized by a gradient of less flood tolerant bottomland hardwood species along the higher

elevated natural levee and transitional areas and by more flood-tolerant species at lower

elevations (Wharton et al., 1982; Hupp, 2000).

Therefore, the objectives of this study were to 1) compare short- (1987 to 2012) and long-

term (1881 to 2012) sediment accretion rates for a natural levee (35 m from river) and

backswamp (75 m from river) adjacent to the Tensaw River in southern Alabama using a

dendrogeomorphic technique; 2) determine the influence of distance from river and recent

harvest activity on sediment accretion rates; and 3) identify impacts of hydrologic and climatic

fluctuations on radial growth in green ash (*Fraxinus pennsylvanica* Marsh.) and water tupelo

(*Nyssa aquatica* L.).

3.3 Material and Methods

3.3.1 Study Area

This study was conducted along the western bank of the Tensaw River within the Mobile-

Tensaw River Delta at approximately 1 m msl (Figure 3.1). The Mobile-Tensaw Delta is formed

by the Mobile, Tensaw, and Middle Rivers and is the second largest delta system in the United

States (Smith, 1988). This delta contains approximately 43,000 ha of wetlands, with 75%

forested (Smith, 1988). The Mobile-Tensaw Delta is located within the Mobile River Basin,

situated below the confluence of the Alabama and Tombigbee River Basins. The Alabama and

Tombigbee River Basins combined contain approximately 11.6 million ha (Smith, 1988).

Drainage from this area contributes to the estimated 4.3 million Mg yr^{-1} of suspended sediment

that is transported downstream into the Mobile Bay and then potentially into the Gulf of Mexico

(Figure 3.1; Ryan and Goodell, 1972).

The study site consists of a natural levee and water tupelo-baldcypress (*Taxodium distichum* (L.) Rich.) backswamp (30º57' N, 87º53' W) located in Baldwin County, Alabama. The site is approximately 4.5 km southwest of the community of Stockton and 50 km north of the Mobile Bay (Figure 3.1). A natural levee, or a narrow ridge of sediment, occurs from the riverbank to approximately 50 m away from the river. This natural levee receives greater sand deposition due to its proximity to the river. Thus, the natural levee provides microsite conditions favorable to vegetative species that are less flood-tolerant and grow best in better drained soils. The overstory and midstory on the natural levee is characterized by green ash, overcup oak (*Quercus lyrata* Walt.), and water oak (*Q. nigra* L.) with fewer occurrences of swamp tupelo (*Nyssa sylvatica* var. *bicolor* (Walt.) Sarg.), sandbar willow (*Salix exigua* Nutt.), American elm (*Ulmus americana* L.), and hornbeam (*Carpinus caroliniana* Walt.) (Aust and Lea, 1991).

The backswamp is lower in elevation than the natural levee. Overbank floodwaters pond and persist into summer months until evapotranspiration reduces standing water levels. During the summer months, the water table ranges from 25 cm above the surface to 50 cm below the surface (Aust et al., 2006). Very poorly drained soils of the Levy (fine, mixed, superactive, acid, thermic Typic Hydraquents) series characterize the backswamp (Aust et al., 2012). Plots were primarily composed of naturally regenerated flood-tolerant water tupelo and baldcypress, but also contained a small component of Carolina ash (*Fraxinus caroliniana* P. Mill.), pumpkin ash (*F. profunda* (Bush.) Bush.), black willow (*Salix nigra* Marsh.), red maple (*Acer rubrum* L.), and water elm (*Planera aquatica* J.F. Gmel.) (McKee et al., 2012). Buttonbush (*Cephalanthus occidentalis* L.) and dwarf palmetto (*Sabal minor* Jacq. Pers.) were common shrubs in the understory for the backswamp and natural levee, respectively. Additional site information is available (Aust and Lea, 1991; Aust et al., 2006; Aust et al., 2012; McKee et al., 2012).

Overbank flooding is more common on site from December to April when average daily river stage is above flood stage-level (Figure 3.2). The Tensaw River experiences semidiurnal tides that can cause the river to fluctuate 20 to 50 cm (Aust et al., 2006). Climate in the Mobile-Tensaw Delta region is classified as subtropical. On average, 1750 mm of precipitation is recorded annually with the greatest precipitation recorded during July (Figure 3.2; NOAA, 2011). Temperatures generally remain mild throughout the year. The average daily temperature is 19.2° C with the highest temperatures observed during June, July, and August (Figure 3.2). Only a few weeks of overnight freezing temperatures are observed each year providing an extended growing season for vegetation in this region.

3.3.2 Field Methods

During the summer of 2012, total sediment accretion was measured immediately adjacent to 20 green ash trees of seed origin along transects parallel to the river. Ten co-dominant trees were sampled on the natural levee and ten along the edge of the backswamp. Ash trees were selected with a "telephone pole" like appearance along the base (i.e., lacking basal flare) (Hupp and Morris, 1990). Areas adjacent to historical pull boat channels from the 1920 era logging operations were avoided due to potential influences on sediment deposition totals (Mader, 1990). Using a dendrogeomorphic technique that was previously validated by Hupp and Morris (1990), the original lateral roots were located on two sides of each ash tree. These roots initially formed as rootlets at the point of germination and in mature trees, provide support (Alestalo, 1971). Adventitious roots, formed in response to episodic sediment deposition disturbances, were smaller in diameter, shorter in length, buried at shallower depths, and occurred in common zones or "layers" (Shroder, 1980; Hupp and Morris, 1990; Walls et al., 2005). Once roots were located, the vertical distance between the lateral roots and current ground line was measured to

the nearest 0.25 cm. Depths were measured 0.5 to 1.0 m away from the tree base to avoid any direct interference of the base on deposition totals. To determine the year in which each tree was established, a tree core was extracted below ground level using an increment borer.

An additional two cores were extracted at diameter at breast height (DBH, 1.4 m above ground level) from the 20 green ash trees along the natural levee and backswamp. Fifteen water tupelo trees in the control plots (160 to 330 m from river) were cored at DBH. A minimum of two cores were taken from each tree to account for within-tree variation and to aid in identification of false rings, particularly common to water tupelo.

3.3.3 Laboratory Methods

Tree cores were air dried and glued to wooden mounts. Progressively finer sand paper was used to surface cores until cellular structures became visible in the cross-sectional view under magnification. Two tree-ring chronologies were developed: 1) green ash using DBH and ground level cores and 2) water tupelo using DBH cores from the control plots. Cores from each chronology were visually cross-dated using the list method, in which narrow growth rings common among samples were identified and used as signature years to ensure proper alignment of dating (Yamaguchi, 1991). Annual tree ring-widths were measured using the LinTab™ 5 RINNTECH® measuring system and TSAP-Win™ software (v. 4.69). Accuracy of measurements was validated to the nearest 0.1 mm using a standard microscope stage micrometer calibration slide. Dated tree-ring width measurement values were verified to ensure quality of visual cross-dating using COFECHA software (Holmes, 1983). Dating errors detected by COFECHA caused by misdated, missing, or false rings were corrected. False rings commonly occur in water tupelo which can make it difficult to accurately date without proper visual cross-dating and statistical verification in COFECHA (Phipps, 1985). Each dated series

was de-trended using in the computer program ARSTAN to remove negative age-related growth trends (Cook, 1985). Through interactive de-trending, a flexible cubic smoothing spline was fitted to each series. Cubic smoothing spline lengths ranged from 25 to 50 years. Over each spline length interval 50% of the variance is left in the chronology. This de-trending method allowed for removal of long-term growth trends and variation associated with changes in stand dynamics common to closed canopy forests. Index values, actual ring-width value divided by the predicted value, were computed for each series in ARSTAN. Ring width index values were averaged together to develop a master chronology for green ash and water tupelo. The arstan chronology was output for use in correlation analysis. Mean series length, series intercorrelation, and mean sensitivity statistics were computed for each chronology.

The two vertical sediment deposition values measured at each sampled tree were averaged together. Average total sediment deposition was divided by the tree age to estimate sediment accretion rates for the time period in which the tree was living. Sediment accretion values were averaged for trees on the natural levee and for trees along the backswamp. Average sediment accretion rates were calculated for short- (1987 to 2012) and long- (1881 to 2012) term periods based on the age of trees used along the natural levee and backswamp.

3.3.4 Existing Datasets

Measurement data from repeated elevation surveys were available for control plots located 160 to 330 m from the river (Aust et al., 2012; McKee et al., 2012). Periodic vertical sediment accumulation data at sediment pins (70 to 81 pins) were available from the same locations for six measurement periods between 1986 and 2010 (Aust et al., 1991; Aust et al., 1997; Warren, 2001; Gellerstedt and Aust, 2004; McKee et al., 2012; Aust et al., 2012).

Instrumental stream gauge data were downloaded from the Barry Steam Plant for 1983 to 2012 (Figure 3.1; USACE, 2015). This gauge was established in September of 1983 and has been maintained by the Army Corps of Engineers in the Mobile District. Previous calibration between onsite stage recorders and the Barry Steam Plant gage (11 km northwest of site) indicated overbank flooding occurs at the study site when the Barry Steam gage reaches a stage of 1.4 m (Figure 3.1; Warren, 2001). The number of days above flood stage and average daily stage level were determined for each calendar month from April to September and for each water year (prior October through September for a given year). Monthly values for the prior calendar year were also determined. Stage data were missing for the water year 1992 because as the stage recorder was broken. Monthly total precipitation and average daily temperature data were available for the Alabama Gulf Division (08) for years 1895 to 2012 from the National Climatic Data Center (NOAA, 2014).

3.3.5 Data Analysis

Dendrogeomorphic sediment accretion rate estimates were compared between the natural levee and backswamp for time periods 1987 to 2012 and 1881 to 2012. Accretion rate estimates were also compared between the two time periods on the natural levee and within the backswamp. Comparisons were made in Wilcoxon-Mann-Whitney tests using the NPAR1WAY procedure in SAS 9.3 (SAS, 2012). Exact methods were specified for each nonparametric procedure. All statistical analyses were performed at a significance-level $\alpha = 0.05$.

Estimated sediment accretion rates were compared among distances from the river from 1986 to 2012. Specifically, differences among the dendrogeomorphic rate estimates at 35 and 75 m and sediment pin measurements in the control plots at 160, 250, and 330 m were evaluated using a Kruskal-Wallis test. If significant differences were detected, pairwise Wilcoxon-Mann-

Whitney tests were conducted between the distances. Differences among all five distances using the dendrogeomorphic and elevation survey data in the control plots were evaluated by the same method.

To quantify the impacts of overbank flooding and climate on annual tree growth, Pearson correlation coefficients (r) between the hydrologic and climatic variables and ring width index for green ash and water tupelo were evaluated. Specifically, the number of days above flood stage, average daily stage level, total precipitation, and average daily temperature variables were analyzed against annual ring width index for the two chronologies. Hydrologic (1983 to 2012) and climatic (1895 to 2012) data for the months of April to September, water year, and the prior year, for chronology years were included in the analyses. Months or years with missing data were excluded from the analyses.

3.4 Results

3.4.1 Dendrogeomorphic sediment accretion rate estimates for the natural levee and backswamp

Long-term (1881 to 2012) accretion rates on the natural levee (0.4 ± 0.0 cm yr^{-1}; mean $\pm$ standard error) were slightly lower than along the backswamp (0.5 ± 0.1 cm yr^{-1}), but were not significantly different ($p = 0.247$) (Figure 3.3). Dendrogeomorphic sediment accretion rates for the short-term period 1987 to 2012 were higher on the natural levee (1.6 ± 0.2 cm yr^{-1}) than along the backswamp (1.2 ± 0.2 cm yr^{-1}), but were not significantly different ($p = 0.286$). Mean sediment accretion rates on the natural levee were significantly ($p = 0.009$) higher (124%) for the 1987 to 2012 time period than the 1881 to 2012 period. Along the backswamp, sediment accretion rates were significantly ($p = 0.032$) higher (72%) for the 1987 to 2012 time period than the 1881 to 2012 period (Figure 3.3). When estimates for both time periods were combined, the

overall sediment accretion rates were identical (0.9 ± 0.2 cm yr^{-1}) for the natural levee and backswamp.

3.4.2 Sediment Accretion Rates at Increased Distances from the River

Sediment accretion rate estimates were higher along the natural levee and backswamp than in the water tupelo-bald cypress stand at 160 to 330 m from the river (Table 3.1). Kruskal-Wallis tests indicated sediment accretion rates estimated from dendrogeomorphic and sediment pins were significantly different ($p < 0.001$) for at least one distance among the 35, 75, 160, 250, and 330 m distances (Table 3.1; Warren, 2001; Aust et al., 2012; McKee et al., 2012). Pairwise Wilcoxon-Mann-Whitney tests illustrated sediment pin sediment accretion rate estimates at 160 m (0.8 ± 0.0 cm yr^{-1}) were significantly lower than dendrogeomorphic estimates at 35 m (1.6 ± 0.2 cm yr^{-1}; $p = 0.034$) and 75 m (1.2 ± 0.2 cm yr^{-1}; $p = 0.036$). Dendrogeomorphic sediment accretion rates at 35 m were significantly different from sediment pin estimates at 250 m (0.9 ± 0.0 cm yr^{-1}; $p = 0.034$) and 330 m (0.8 ± 0.0 cm yr^{-1}; $p = 0.047$). Analysis of the dendrogeomorphic and elevation survey estimates among the same five distances indicated values for at least one distance were significantly different ($p = 0.043$). Additional pairwise comparisons revealed dendrogeomorphic estimates at 35 m were significantly different from elevation survey estimates at 250 (0.6 ± 0.1 cm yr^{-1}; $p = 0.047$) and 330 m (0.7 ± 0.1 cm yr^{-1}; $p = 0.047$). Sediment accretion rates estimated from the dendrogeomorphic technique at 35 and 75 m were higher than the elevation survey estimates at 160 m (0.8 ± 0.4 cm yr^{-1}), but were not statistically different ($p = 0.250$; Table 3.1).

Sediment pin sediment accretion rate estimates were significantly different ($p = 0.004$) among the 160, 250, and 330 m distances within the backswamp using a Kruskal-Wallis test. However, pairwise comparisons, revealed no significant differences between the distances as all

comparisons of sediment pin estimates had an exact p = 0.100 (Table 3.1). Mean elevation survey sediment accretion rates did not significantly differ (p = 0.866) among 160, 250, and 330 m distances. Total deposition values followed a similar trend with greater values occurring on the natural levee and along the backswamp at 75 m than at the 160, 250, and 330 m distances (Table 3.1).

3.4.3 Influence of Overbank Flooding on Sediment Deposition

Periodic sediment accumulations in the backswamp control plots reflected the trend in number of days flooded within a given period (Figure 3.4). For the 24-year period, an average of 19.8 cm of sediment accumulated (0.8 cm yr^{-1}) at sediment pins in the backswamp (160 to 330 m from the river). A minimum of 3000 overbank flooding days occurred during this same period. Total sediment accumulations ranged from 1.2 cm for the 1997 to 1998 time period (292 overbank flooding days) to 6.2 cm for the 2002 to 2010 time period (1021 overbank flooding days).

3.4.4 Impacts of Hydrologic and Climatic Variables on Green Ash and Water Tupelo Growth

Diameter at breast height cores from 20 green ash trees were used to develop a growth chronology representative of green ash growth along the natural levee and backswamp. Fifteen water tupelo trees, dating to 1890, were used to develop a ring width chronology representative of water tupelo growth in the backswamp (Table 3.2). Average annual ring width for green ash (1.5 ± 0.0 mm) was slightly lower than in the water tupelo (2.9 ± 0.1 mm)

Days above flood stage and average daily stage level appeared to have a greater influence on radial growth in green ash than in the water tupelo (Table 3.3 and Figure 3.5). Green ash ring width index had a positive, but not significant, relationship with days above flood stage (r = 0.21; p = 0.289) during the water year. Days above flood stage during April (r = 0.43; p = 0.025) were

significantly correlated with green ash ring width index. Green ash ring width index was significantly correlated with average daily stage level during the month of April ($r = 0.51$; $p = 0.008$), but not with average daily stage for the overall water year ($r = 0.13$; $p = 0.556$). In the water tupelo, days above flood stage ($r = 0.04$; $p = 0.858$) and average daily stage level ($r = 0.05$; $p = 0.825$) for water years were not significantly correlated with annual ring width index values. Correlation coefficients were not significant (all p-values > 0.05) between water tupelo ring width index and days above flood stage for April through September (r ranged from -0.13 to 0.19) or average daily stage level for April through September (r ranged from -0.09to 0.14).

Annual total precipitation and average daily temperature were not significantly correlated with ring width index for green ash or water tupelo (Table 3.3). Green ash ring width index was not significantly correlated with average daily temperature for any month, but was significantly correlated with total precipitation during the months of April ($r = 0.24$; $p = 0.010$) and August ($r = -0.20$; $p = 0.046$). Water tupelo ring width index was not significantly correlated with total precipitation or average daily temperature in any month. Lagged values for prior year total days above flood stage, average daily stage level, total precipitation, and average daily temperature were not significantly correlated with green ash or water tupelo ring width index.

3.5 Discussion

3.5.1 Sediment Accretion Rates along the Natural Levee and Backswamp

Dendrogeomorphic sediment accretion rates (0.26 to 2.0 cm yr^{-1}) were within the range of rates published for forested wetlands (Figure 3.4; Kesel et al., 1974; Kleiss, 1996, Heimann and Roell, 2000; Phillips, 2001; Hupp et al., 2015). Adjacent to the Cache River in Arkansas, dendrogeomorphic sediment accretion rates ranged from 0.1 to 0.4 cm yr^{-1} on natural levees (50 m from river) and from 0.0 to 0.6 cm yr^{-1} along baldcypress-water tupelo swamps (149 m from

river; Hupp and Morris, 1990; Kleiss, 1996). These rates were similar to our long-term (1881 to 2012) dendrogeomorphic sediment accretion rates along the natural levee (0.4 cm yr^{-1}) and backswamp (0.5 cm yr^{-1}).

Our short-term (25 years) dendrogeomorphic rates were significantly higher than the long-term rates along the natural levee (1.6 cm yr^{-1}) and the backswamp (1.2 cm yr^{-1}; Figure 3.3). Differences between short- and long-term rates were possibly a result of timber harvest disturbances in adjacent backswamp stands in 1986. No harvesting disturbances occurred on the natural levee or in the backswamp areas sampled in this study. However, old pull boat channels were used to transport harvesting equipment into an adjacent backswamp stand. A couple of these pull boat channels run perpendicular to the natural levee (Mader, 1990). Areas immediately adjacent to pull boat channels were not sampled; however, this disturbance likely resulted in increased levels of light penetration and thus, increased vegetation density. Increased low, dense vegetation along pull boat channels which boarder the natural levee is likely the main reason for higher short- than long-term sediment accretion rates. This finding is important to understanding the role of riparian wetlands as it illustrates changes in trapping efficiency with stand structure through time.

Other studies have attributed higher short-term dendrogeomorphic estimates to recent changes in land use and upstream channelization (Hupp and Morris, 1990; Kleiss, 1996; Heimann and Roell, 2000). For instance, dendrogeomorphic sediment accretion rates for a recent 10-year period (1.8 to 2.1 cm yr^{-1}) were higher than rates for a previous 25-year period (1.1 cm yr^{-1}) along the Loco Bayou floodplains in Texas (Phillips, 2001). Higher sediment accretion rates were determined when the dendrogeomorphic technique was used on younger trees ($\leq$ 40 years) than on older trees (40 to 80 years) along the Cache River (Hupp and Morris,

1990; Kleiss, 1996). Similar to trends were reported in floodplains along Long Branch Creek in Missouri (Heimann and Roell, 2000). Erosion is another likely influence on long-term rates on the natural levee. Differences between gross and net sediment accretion rates were reported for a 15-year period in wetlands adjacent to the Olentangy River in Ohio (Mitsch et al., 2014). Natural levees are created through re-current sediment deposition adjacent to meandering black rivers in the coastal plains (Hupp, 2000). However, these levees also are in direct contact with overbank floodwaters which have the potential to erode exposed portions of the natural levee through time.

3.5.2 Influence of Distance from River on 1986 to 2012 Sediment Accretion Rates

Mean sediment accretion rates decreased as the distance from the river increased from 35 m to 330 m for the 1986 to 2012 time period (Table 3.1). In our study, as distance from the Tensaw River increased, elevation also decreased from the natural levee to the lower elevation backswamp. Our findings of a negative trend in sediment accumulation with increased distance and elevation both agree (Kesel et al., 1974; Johnston et al., 1984; Asselman and Middlekoop, 1995) and disagree (Hupp and Morris, 1990; Hupp and Bazemore, 1993; Kleiss, 1996) with other studies due to differences in site-level connectivity to sediment-laden waters (Hupp et al., 2015). Particle size of sediment deposited may have contributed to higher sediment accretion rates along the natural levee from the river than in the backswamp. Larger particles fall out of suspension first as overbank floodwater velocity decreases (Boto and Patrick, 1978). Therefore, particle size could have influenced total vertical deposition values with change in distance from the river. Along the Mississippi River, the percentage of sand (0.063 to 2 mm) particles that were deposited on the natural levee (68%) and the levee backslope (47%) was greater than in the backswamp (3%) (Kesel et al., 1974). Smaller silt (0.002 to < 0.063 mm) and clay (< 0.002 mm)

particles made up more particles in the backswamp than along the natural levee and levee backslope (Kesel et al., 1974). Similar findings were reported along transects extending from natural levees into backswamps in the Atchafalaya Basin in Louisiana (Hupp et al., 2008) and in floodplains along Long Brank Creek in Missouri (Heimann and Roell, 2000). In our study, sediment accretion rates between the natural levee and backswamp were not statistically different (Figure 3.3). This may have been because at 75 m from the river, the backswamp resembled more of a levee backslope or a transition zone than a true depressional backswamp (Kesel et al., 1974; Hupp et al., 2008).

Although sediment rates decreased from the 75 to 330 m distances, values were not always statistically different (Table 3.1). Dendrogeomorphic sampling was not conducted at distances beyond 75 m from the river due to logistical difficulties when attempting to locate original lateral roots in areas where the water table was above the root systems. Estimation methods were not compared in our study due to the difference in distances from river among dendrogeomorphic, sediment pin, and elevation sampling locations. Other studies that have used multiple estimation methods to determine sediment accretion rates have found dendrogeomorphic estimates more conservative than estimates from other methods (Kleiss, 1996; Heimann and Roell, 2000). Despite potentially conservative dendrogeomorphic estimates, sediment accretion rates were still higher at 35 and 75 m than at 160 to 330 m.

3.5.3 Impacts of Hydrologic and Climatic Variables on Green Ash and Water Tupelo Growth

Average annual ring width for green ash in our study (1.5 mm) was similar to rates (1.6 mm yr^{-1}) along the Missouri River in North Dakota (Reily and Johnson, 1982), but was lower than values (6.3 mm) along the Kankakee River in Illinois (Mitsch and Rust, 1984). Differences in green ash growth rates among studies are likely attributed to variation in site elevation,

microtopography, soil texture, and flooding regime (Anderson and Mitsch, 2008). In our study, elevation was not measured for green ash locations. However, based on visual observation and sediment accumulation data, the natural levee and front edge of the backswamp (75 m) were relatively higher in elevation than the backswamp at 160 to 330 m. Stress induced by periodic flooding and sedimentation disturbances may have initially reduced growth of the green ash (Yanosky, 1982; Walls et al., 2005). Along the Wisconsin River, the differences in green ash radial growth were greater among topographic positions in an active floodplain than a floodwater excluded floodplain. Although green ash is classified as moderately flood-tolerant, it responds to periodic flooding events through formation of adventitious roots. Thus, belowground allocation of resources in response to flooding and sedimentation disturbances could have a negative effect on aboveground radial growth. Green ash trees sampled in our study were close to the river and thus, may have experienced recurring stress from flooding and sedimentation disturbances. Floodplain species along the Olentangy River in Ohio appeared to have an optimal number of flood days, after which reduced growth becomes a tradeoff due to increased stress from additional disturbances (Anderson and Mitsch, 2008).

Average annual ring width for water tupelo was 2.9 mm in the backswamp water tupelo. Annual radial growth was much higher than annual growth rates reported for water tupelo (0.7 mm) in a mature (63 year-old) baldcypress-water tupelo stand in the Atchafalaya River Basin in Louisiana (Dicke and Toliver, 1990). Growth rate and dominance of water tupelo over baldcypress in wetlands is highly related to flooding regime (Dicke and Toliver, 1990; Conner and Day, 1992; Keeland and Sharitz, 1995). Water tupelo growth rates were shown to be higher on continuously flooded sites than sites which only received seasonal floodwaters (Dicke and Toliver, 1990). Although overbank flooding is characterized as seasonal on the banks of the

Tensaw River, residual floodwaters typically pond in the lower backswamp creating conditions

which likely contributed to the higher radial growth rates in our study. A relatively high water

table and influences of Squirrel Bayou located 200 m to the west of the site may have

contributed to continuously hydric conditions in the backswamp. In addition, higher water

tupelo growth rates have been shown to coincide with deep periodic flooding (Keeland and

Shartiz, 1995). In our study, fluctuations in water-levels were created through periodic overbank

flooding events beyond the flooding season, which likely provided additional nutrients and

dissolved oxygen required to achieve maximum growth in water tupelo (Keeland and Sharitz,

1995).

Correlations between ring width index and hydrologic and climatic variables for water

years were not significant for either the green ash or water tupelo in our study (Table 3.3).

Examination of correlations on a monthly-scale revealed several growing season months had a

significant influence on annual ring width index. In our study, positive significant correlations

were identified between green ring width index and days above flood stage (April) and average

daily stage level (April). Similarly, positive correlations between green ash radial growth and

the mean monthly stage were significant for April, May, and June on the White River in

Arkansas (Gee, 2012). Positive relationships were identified between 5-year average annual

basal area increments and average daily streamflow and days of floodplain inundation along the

Kankakee River (Mitsch and Rust, 1984). Relationships between hydrologic variables and green

ash ring width index were likely positive in our study due to potential moisture limitations in

sandy soils with lower water holding capacities along natural levee and backswamp edge.

Correlations between ring width index and hydrologic variables for April, early in the growing

season, were most likely positive as they preceded the onset of annual radial growth in green ash.

Radial growth of water tupelo in the backswamp appeared to be less influenced by hydrologic and climatic variables than green ash. No significant correlations existed between water tupelo ring width index and the hydrologic or climatic variables (Table 3.3). Lack of relationships between hydrologic variables and water tupelo ring width index was unexpected as other studies have illustrated that water tupelo is highly dependent on periodic flooding to transport additional nutrients and dissolved oxygen, particularly in backswamps, which may become stagnant by ponded residual floodwaters (Dicke and Toliver, 1990; Conner and Day, 1992; Keeland and Sharitz, 1995). Detection of these relationships could have been influenced by defoliation by the endemic forest tent caterpillar (*Malacosma disstria*). Defoliation following the onset of radial growth causes xylem production to cease until a new flush of leaves is developed (Conner and Day, 1992). During more severe years, radial growth may stop and re-start up to three times, with the last leaves persisting until the first frost which could be as late as early November (Dicke and Toliver, 1990; Conner and Day, 1992). Another reason relationships between water tupelo ring width index and hydrologic or climatic variables were not detected could be related to changes in physiology as trees mature in age and increase in diameter. These changes in physiology may make mature trees more resistant to short-term alterations in environmental conditions than younger trees (Voelker, 2011; Liñán et al., 2012).

3.6 Conclusions

Temporal and spatial patterns of sediment deposition play an integral role in successional pathways in bottomlands (Hodges, 1997; McKee et al., 2012; Mitsch et al., 2014). Changes in hydrologic regime impact these depositional patterns and can thus, alter the functions and values of forested wetlands (Hunter et al., 2008). Site connectivity to sediment-laden waters appears to be the most comprehensive factor influencing temporal and sediment deposition patterns (Hupp,

2015). Site-specific connectivity to inflow is influenced by distance from water/sediment source, relative elevation, microtopography, vertical structure and density, on site water storage, and occurrence of overbank flooding (Hupp, 2000; Hupp, 2008). Upstream land use and input of sediment into waterways determines the supply of sediment available for deposition downstream (Hupp and Morris, 1990; Kleiss, 1996; Heimann and Roell, 2000). This study highlights the important role riparian forested wetlands play in trapping sediment from adjacent waterways. Further, results illustrate different sediment accretion rates for short- (25 years) and long-term (131 years) periods, which is attributable to trapping efficiency. Overbank flooding provides additional nutrients to backswamps, which can have significant influences on tree growth (Keeland and Sharitz, 1995).

Restoration of bottomland hardwood systems is heavily dependent on temporal and spatial patterns of sediment deposition and occurrences of overbank flooding. This study illustrates changes in sediment accretion rates over time and space and the role of hydrologic variables (days above flood stage and average daily stream stage) on tree growth. We used dendrochronological techniques to estimate short- and long-term sediment accretion rates and to determine impacts of hydrology and climate on tree growth. These techniques minimize site visits while still providing post-disturbance data that allow analysis of the impacts of system hydrology on sediment deposition and tree growth to better predict future impacts from altered hydrology.

3.7 Acknowledgements

3.8 References

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Table 3.1 Mean sediment accretion rates (cm yr^{-1}), total deposition (cm), and mean sediment trapped annually (Mg ha^{-1} yr^{-1}) and total trapped (Mg ha^{-1}) for the 1986 to 2012 time period as distance from the Tensaw River increased from the natural levee (35 m) to across the mature tupelo-cypress backswamp stands (75 to 330 m) based on dendrogeomorphic estimates (Dendro), previous sediment pin measurements (Pins), and elevation surveys (Elev). Same letters within a column indicates values were not significantly different based at α = 0.05.

Distance from river (m)	Estimation [a] method	N	Mean accretion rate (cm yr^{-1})	Mean total deposition (cm)	Mean rate [b] of sediment trapped (Mg ha^{-1} yr^{-1})	Mean total [b] sediment trapped (Mg ha^{-1})
35	Dendro	4	1.62[a]	36.0[a]	100.4	2232.0
75	Dendro	5	1.25[a,b]	29.6[a,b]	77.5	1835.2
160	Pins	3	0.76[c]	18.2[c]	47.1	1128.4
	Elev	3	0.78[a,b,c]	18.8[a,b,c]	48.4	1165.6
250	Pins	3	0.89[b,c]	21.3[b,c]	55.2	1320.6
	Elev	3	0.61[c]	14.6[c]	37.8	905.2
330	Pins	3	0.84[b,c]	20.1[b,c]	52.1	1246.2
	Elev	3	0.74[b,c]	17.8[b,c]	45.9	1103.6

[a] Previous sediment pin measurements and elevation survey data (Aust et al., 1991; Aust et al., 1997; Warren, 2001; Gellerstedt and Aust, 2004; McKee et al., 2012; Aust et al., 2012).
[b] Bulk density value (0.62 Mg m^{-3}) used was obtained from McKee et al. (2012).

Table 3.2 Tree ring chronology characteristics for *Fraxinus pennsylvanica* (FRPE) and *Nyssa aquatica* (NYAQ) chronologies developed in stands adjacent to the Tensaw River in southern Alabama.

| Species/stand | Distance from river (m) | Chronology span | Series lengths | | Number of trees | Number of radii | Total growth rings | Series intercorrelation | Mean sensitivity |
			Range (years)	Mean (years)					
FRPE[a]	35 to 75	1881 - 2011	18 to 131	55	20	58[a]	2032	0.466	0.424
NYAQ	160 to 330	1890 - 2011	54 to 122	80	15	30	2391	0.581	0.492

[a] Chronology included belowground cores used to determine tree age/establishment date in estimation of sediment accretion rates.

Table 3.3 Pearson correlation coefficients (r) between annual ring width index and hydrologic and climatic variables for *Nyssa aquatica* (NYAQ) backswamp stands and for *Fraxinus pennsylvanica* (FRPE) along the natural levee and backswamp. Bolded values indicates significant correlation at α = 0.05.

Variable/month	NYAQ	FRPE
Days above flood stage		
Water year[a]	0.04	0.21
April	0.19	**0.43**
May	0.06	0.23
June	0.01	0.31
July	0.05	0.16
Aug	-0.10	0.10
Sept	-0.13	0.02
Average daily stage level		
Water year[a]	0.05	0.13
April	0.13	**0.51**
May	-0.09	0.13
June	0.09	0.30
July	0.07	0.16
Aug	0.09	-0.02
Sept	0.14	0.02
Total precipitation		
Water year[a]	-0.00	0.10
April	-0.06	**0.24**
May	0.03	0.07
June	0.08	-0.01
July	0.05	0.18
Aug	-0.10	**-0.20**
Sept	-0.12	-0.04
Average daily temperature		
Water year[a]	0.03	0.06
April	0.02	-0.11
May	0.00	-0.11
June	0.10	-0.11
July	-0.11	-0.07
Aug	-0.08	-0.05
Sept	0.11	0.03

[a] Water year (October of previous year to September of current year).
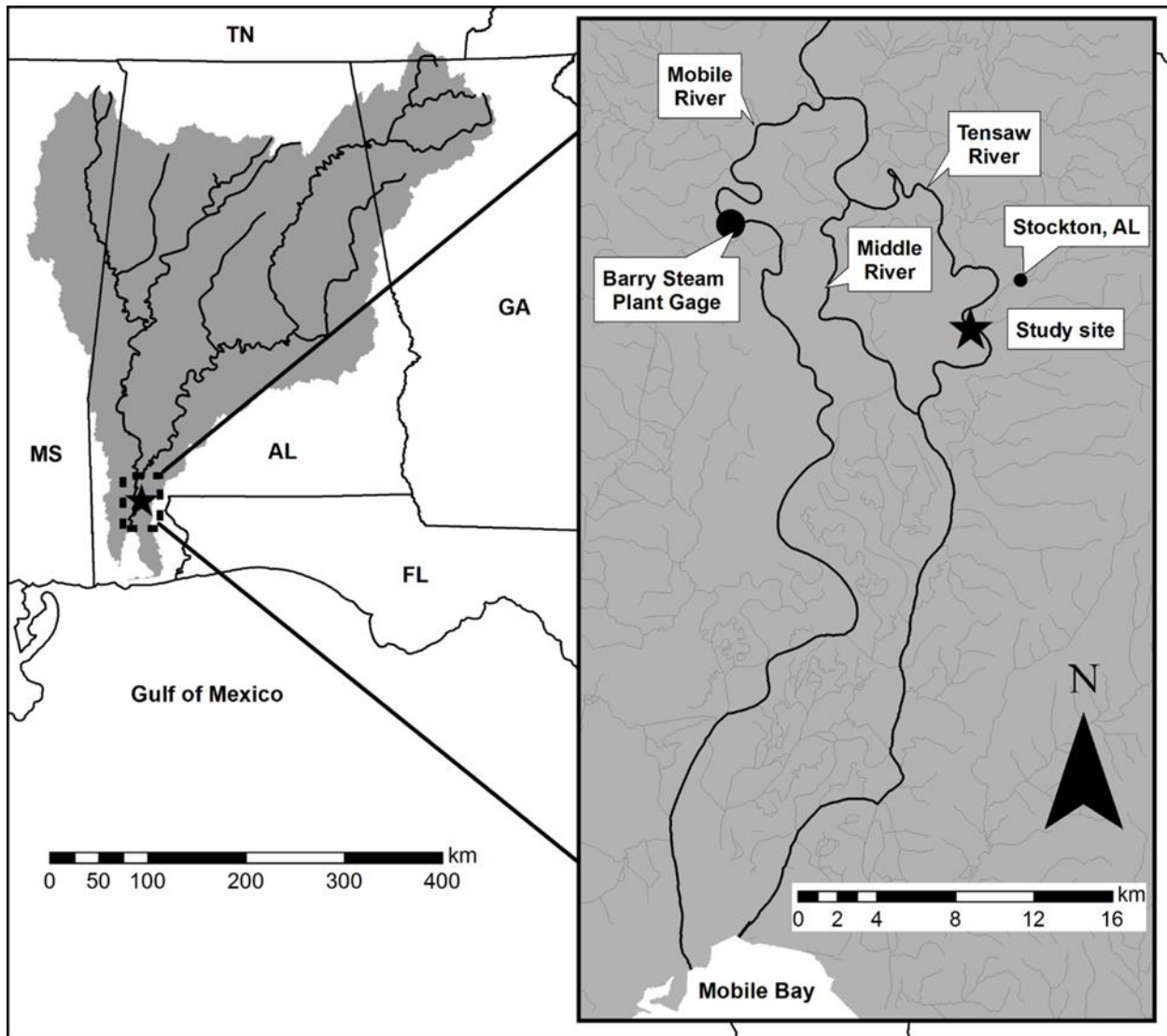
Figure 3.1 Study site (star) located in southwestern Alabama (AL) within the Mobile River Basin composed by the Mobile, Tombigbee, Black Warrior, Alabama, Tallapoosa, Coosa, and Cahaba sub-basins (shaded) that outflow into the Mobile Bay and subsequently the Gulf of Mexico. Inset (right) illustrates study site is situated on the west bank of the Tensaw River in the Mobile-Tensaw River Delta and the Barry Steam Plant gage is approximately 11 km northwest of the study site.
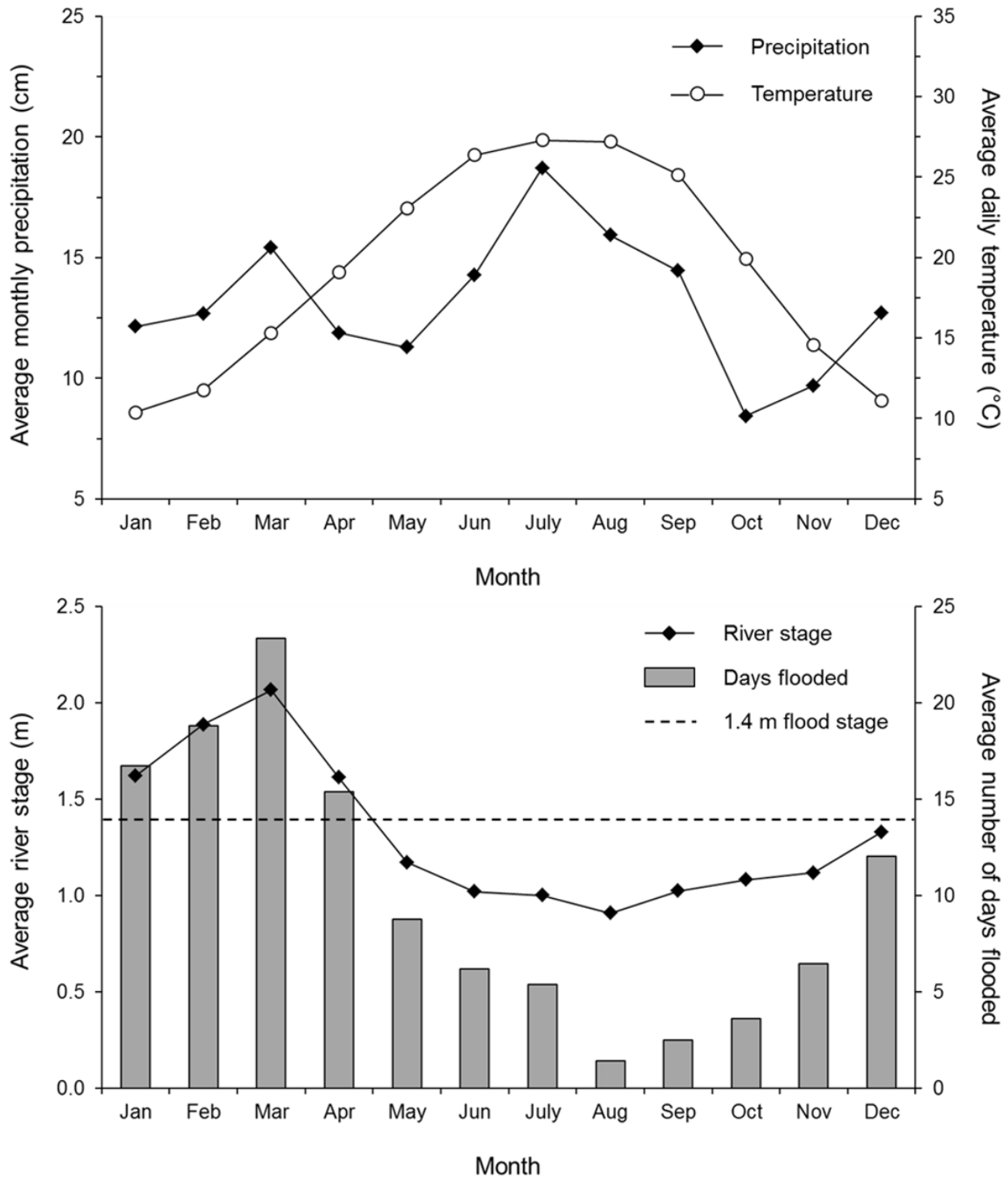
Figure 3.2 Average monthly precipitation (cm) and average daily temperature (°C) for the Gulf climate division in Alabama for 1895 to 2012 (top panel). Average river stage (m) at the Barry Steam Generating Electricity Plant in the Mobile-Tensaw Delta and number of days the site is flooded (bottom panel). Previous comparisons between on-site stage recorders and the Barry Steam Plant stage recorder indicate overbank flooding occurs when the Barry Steam Plant stage reaches 1.4 m.
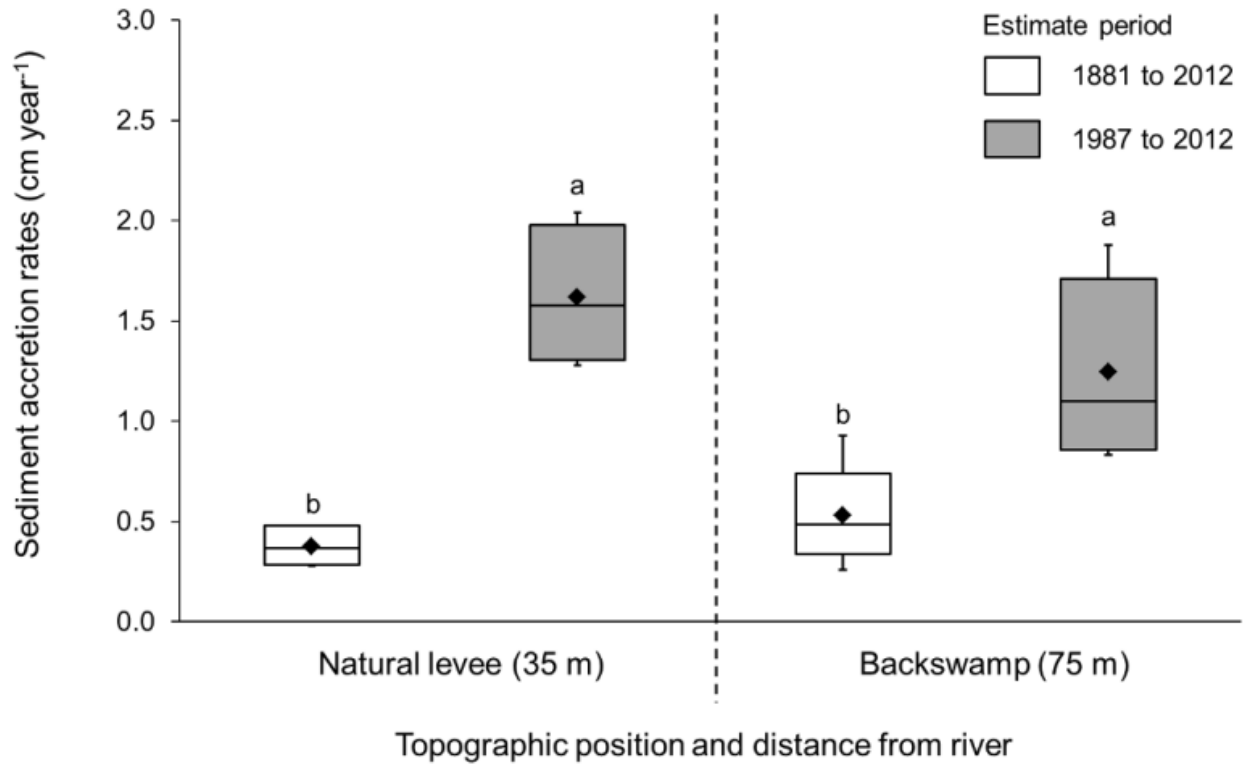
Figure 3.3 Sediment accretion rates (cm yr^{-1}) estimated using a dendrogeomorphic technique on green ash along an unharvested natural levee (35 m from river) and backswamp (75 m from river) adjacent to the Tensaw River. Estimates were derived for different time periods: 1881 to 2012 (tree ages ranged from 60 to 131 years) and 1987 to 2012 (tree ages ranged from 20 to 25 years). Mean estimates are indicated by closed markers. Different letters within same topographic position indicates estimates for time periods are significantly at $\alpha = 0.05$ based on a Wilcoxon-Mann-Whitney test.
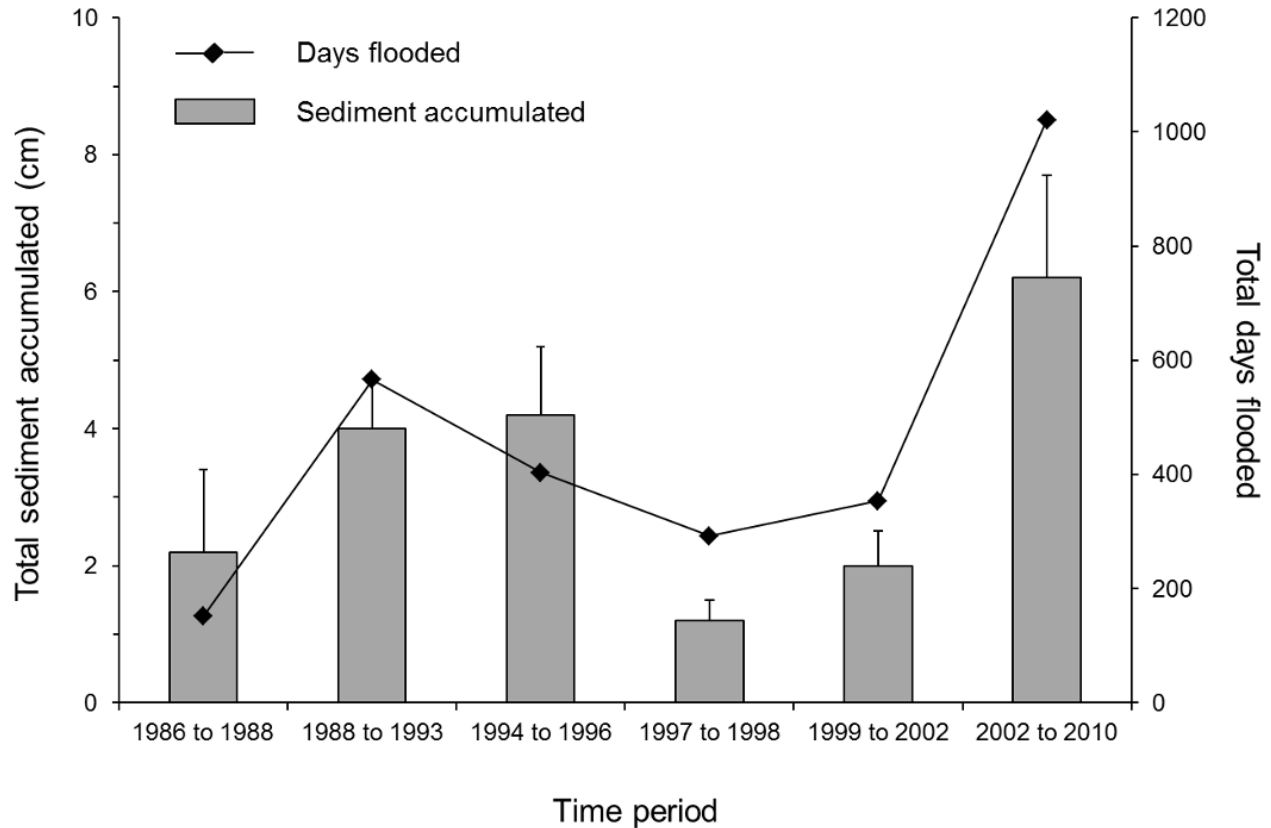
Figure 3.4 Total sediment accumulated (cm) compared with the total number of days flooded for water years (previous October to September for a given year) for different time periods across the unharvested control backswamp (160 to 330 m from the river). Sediment accumulation totals were based on periodic sediment pin measurements. Error bars represent 1 standard deviation. Total days flooded for 1992 was omitted from time period 1988 to 1993 as stage recorder was broken.
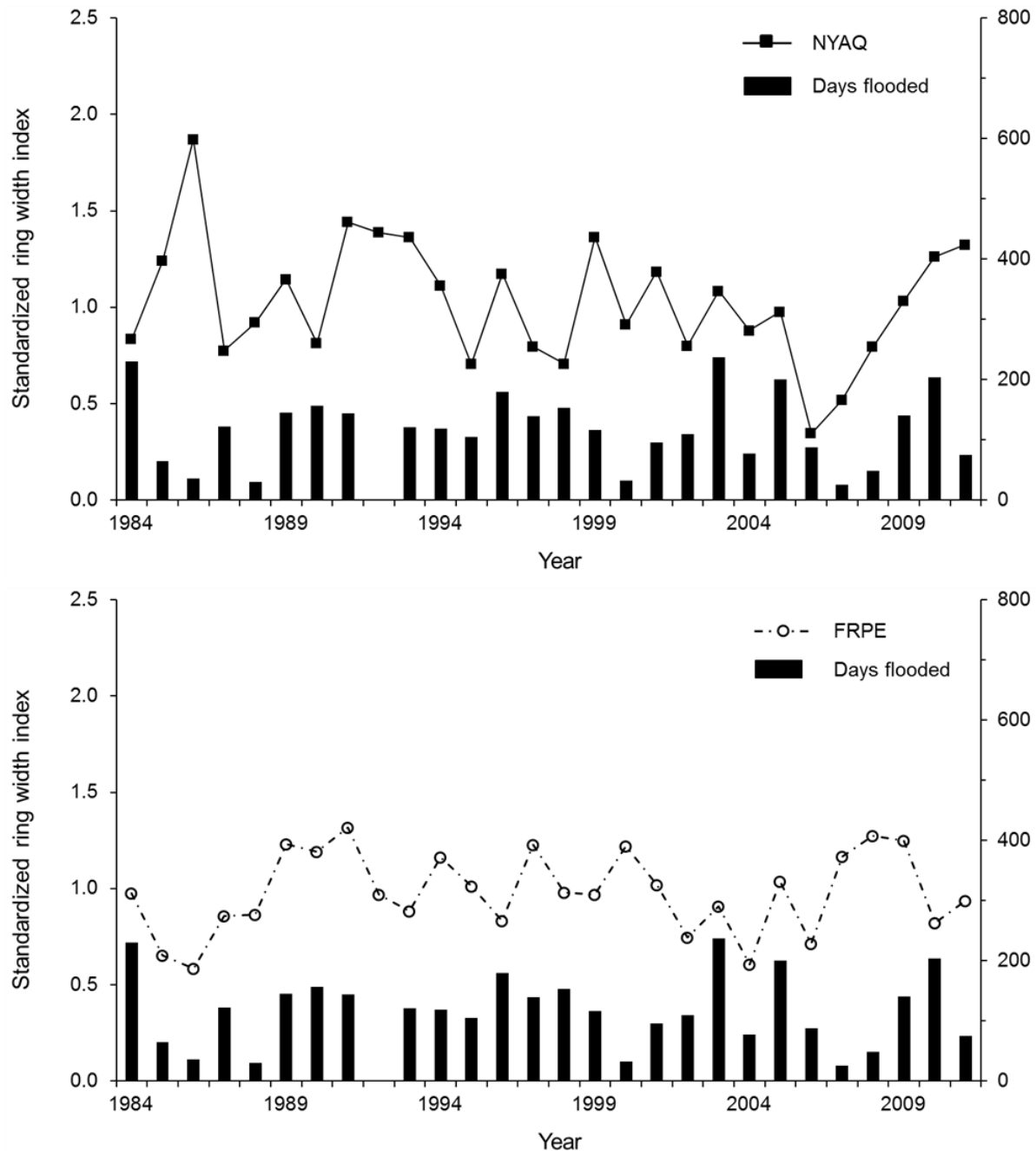
Figure 3.5 Standardized ring width index values for *Nyssa aquatica* (top panel) in backswamp stands and for *Fraxinus pennsylvanica* (FRPE) (bottom panel) along the natural levee and backswamp for the period 1984 to 2011 and compared with the number of days flooded (secondary axis) for each water year. Days flooded for 1992 were ommitted due to missing data.

CHAPTER 4. FREQUENCY AND FACTORS OF EARLYWOOD FROST-RING FORMATION IN JACK PINE (*PINUS BANKSIANA*) ACROSS NORTHERN LOWER MICHIGAN[2]

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4.1 Abstract

Late spring frost disturbances have significant ecological and physiological impacts on forests. Frost-induced cambial damage that occurs when cells are actively dividing can result in the formation of frost rings, abnormal modifications to wood anatomy within the annual growth rings of an injured tree. Frost rings are indicators of growing season frost damage to the cambium and therefore, have potential to be used as a proxy in the reconstruction of extreme climatic events as well as to identify frost-prone environmental conditions. In this study, we measured the occurrence of earlywood frost rings across cambial age and diameter class in 11 jack pine (*Pinus banksiana* Lamb.) populations of northern lower Michigan. Earlywood frost-ring formation was greater in younger trees and in trees with smaller diameters. Biotic (cambial age, diameter, & ring width) and abiotic factors (elevation, initial site-related growth rate, & minimum temperature) demonstrated significant influence on the probability of earlywood frost-ring formation. When using frost rings as a proxy of historical climate, the susceptibility to abrupt freezing temperatures during the growing season and thus, the ability of an individual tree to record a frost disturbance should be considered.

Keywords: climatic events, dendrochronology, frost disturbances, frost rings, wood anatomy, xylem.

4.2 Introduction

Late spring frosts can reduce radial and vertical tree growth (Dittmar, Fricke & Elling, 2006) and create zones of weakness in wood that are susceptible to breakage when processed (Kubler, 1983; Lee, Singh & Kim, 2007). Growing season frost disturbances also increase the susceptibility of individual trees to dieback and damage-inducing pathogens (Pomerleau, 1971; Reich & van der Kamp, 1993) and reduce the potential for a stand to regenerate (Reich & van der Kamp, 1993; Dy & Payette, 2007). As a consequence, growing season frost disturbances influence the geographic range limits of individual species (Kollas, Körner & Randin, 2014). Frost rings are modifications to normal wood anatomy that form within annual growth rings following frost-induced cambial damage during the growing season (Schweingruber, 2007). Thus, frost rings have the potential to be used both as a proxy for reconstructing historical, extreme climatic events and to identify environmental conditions most prone to frost-induced damage (Payette, Delwaide & Simard, 2010).

Frost rings are contained within annual growth rings and are the result of frost-induced damage to the cambium during the active growing season. Frost events occur during periods when the temperature suddenly drops to sub-freezing and cambial-damaging frost disturbances are typically caused by radiational cooling. This typically occurs during temperature inversions on clear nights as a result of the upward release of trapped, daytime thermal energy (Dy & Payette, 2007). Sudden exposure to freezing temperatures causes tissues within a tree to contract exerting excess pressure on the vascular cambium. During periods of growth, frost rings can form if the pressure exerted on the immature xylem cells results in deformation or collapse of xylem cells (Sorauer, 1922; Day & Peace, 1934; Glerum & Farrar, 1966). The majority of previous studies report that frost rings form when the temperature drops below 0° C during the growing season (Glock, 1951; Glerum & Farrar, 1966; Dy & Payette, 2007). A few studies

demonstrate formation at temperatures slightly above freezing (Lee, Singh & Kim, 2007; Gurskaya, 2014). The degree of cell deformation varies with the minimum temperature, length of freeze, and the level of cambial activity at the time of the freeze (Bailey, 1925; Glerum & Farrar, 1966; Reich & van der Kamp, 1993). Frost rings are more likely to form in the earlywood than in the latewood because abrupt freeze events are more common when the cambial initials are actively dividing and forming new xylem cells (Gurskaya & Shiyatov, 2006). Trees with cambial activity earlier in the growing season are most vulnerable to abrupt freeze damage (Glock, 1951; Glerum & Farrar, 1966).

Frost rings are marked by at least one row of collapsed under-lignified xylem cells. This damage zone has an inner layer of deformed thin-walled xylem cells (Bailey, 1925; Glerum & Farrar, 1966; Lee, Singh & Kim, 2007), bent ray parenchyma cells that appear curved and laterally offset near the row of collapsed xylem cells (Bailey, 1925; Glock, 1951; Glerum & Farrar, 1966), and the development of callus tissue in areas adjacent to the damaged ray parenchyma cells (Schweingruber, 2007) (Figure 4.1). Newly formed xylem cells are particularly vulnerable to frost damage as they are under-lignified and temporarily lack a secondary cell wall (Lee, Singh & Kim, 2007). Following a significant freeze event, abnormal xylem cell formation continues through at least the first row of xylem that is formed after the freeze event (Bailey, 1925; Glerum & Farrar, 1966; Fritts, 1976).

Previous studies have used frost rings to reconstruct and confirm dating of historically significant climatic and biological events (Bailey, 1925, Hantemirov, Gorlanova & Shiyatov, 2000; Gurskaya & Shiyatov, 2006; Payette, Delwaide & Simard, 2010). Frost-ring chronologies have been used to quantify the spatial impact and date of climatic cooling events following volcanic eruptions (LaMarche & Hirschboeck, 1984; Brunstein, 1996; Hantemirov, Gorlanova &

Shiyatov, 2000; Salzer & Hughes, 2007). Biological events with impacts on forest health, such as spruce budworm outbreaks (Bailey, 1925), Douglas-fir dieback and canker formation (Reich & van der Kamp, 1993), have been dated using frost rings. Many of these studies assume a direct link between frost-ring formation and freeze events; however, a thorough examination of the relationship between a tree's biological characteristics and the likelihood of frost-ring formation in varying environmental conditions has yet to be conducted. This is an important first step before a proxy record should be employed.

The biological and environmental conditions in which earlywood frost rings most frequently occur must be documented to understand an individual tree's ability to record a frost disturbance event. Therefore, the objectives of this study were to (1) quantify earlywood frost-ring formation in jack pine (*Pinus banksiana* Lamb.) across cambial age and diameter classes at 11 sites across northern lower Michigan and (2) determine the influence of biotic and abiotic factors on the probability of earlywood frost-ring formation. Jack pine was an ideal species for this study because it is a dominant and widespread species of the North American boreal forest and its southern temperate fringe.

4.3 Methods

4.3.1 Study Area

This study was conducted across northern lower Michigan, a landscape influenced by historical glaciation with moraines, till plains, outwash plains, and lake plains as common influences on the distribution of vegetative communities (Albert, 1995; Barnes & Wagner, 1996). Elevation ranges from 177 to 472 m asl, with the lowest areas occurring in the exterior lake-border plains and the highest elevations in the interior highlands (Sommers, 1977). Daily minimum and maximum temperatures vary during the early growing season months, reflecting

differences in latitude and elevation. Weather patterns are strongly influenced by the Great

Lakes (Sommers, 1977; Eichenlaub *et al.*, 1990). Temperatures in the lower elevation areas are

moderated by Lake Michigan and Lake Huron while further inland, more extreme temperatures

are observed. Thus, the interior regions experience a shorter growing season than the lakeshore

regions (Eichenlaub *et al.*, 1990; Barnes & Wagner, 1996; Figure 4.2). As a result of past glacial

deposits and current vegetation, the majority of soils are comprised of well-drained, acidic

Spodosols with the remaining soils mostly classified as Entisols (Sommers, 1977; Schaetzl,

2002; NRCS, 2011).

Jack pine dominates the glacial-origin, well-drained, sandy soils within northern lower

Michigan (Barnes & Wagner, 1996). This species is shade-intolerant and naturally establishes

on exposed mineral soil following fire disturbance, which also aids in the release of seeds from

its serotinous cones (Burns & Honkala, 1990).

4.3.2 Site Selection

Eleven study sites were selected based on dominant jack pine overstory composition,

proximity to a weather station (< 8 km), absence of recent (< 20 years) harvest or fire activity,

and were located on state or national forest properties (Table 4.1). All sites were naturally

regenerated jack pine with the exception of Alpena and Gaylord, which were pure jack pine

plantations. All stands were dominated by jack pine (> 90%) with patches of grasses and ferns

providing primary ground cover. The exterior-located sites (Locke Dam, Nurnberg), were also

characterized by a small hardwood component in the understory including species that are

frequently found within these pine communities such as white oak (*Quercus alba*), Northern pin

oak (*Q. ellipsoidalis*), black oak (*Q. velutina*), scarlet oak (*Q. coccinea*), and pin cherry (*Prunus

pensylvanica*) (Barnes and Wagner 1996). Sites located within central and eastern portions of

northern lower Michigan had characteristics of pine barrens; a vegetation community pre-dominantly composed of jack pine with common associates of red pine (*Pinus resinosa*) and white pine (*P. strobus*) with a diversity of grasses, forbs, and small shrubs. The Beaver Creek (1977) and Goose Creek (1970) sites were established following documented stand-replacing wildfires and a result these sites contained randomly dispersed supercanopy jack pine and red pine which survived the fires. All sites were located on relatively flat slopes (0 to 6%; NRCS, 2011). Site elevation ranged from 201 to 406 m asl (Table 4.1).

Due to prevailing west-to-east winds sites located in the western half of northern lower Michigan (Baldwin, Gaylord, Goose Creek, Jack Pine, Pellston, and Nurnberg) receive more precipitation on average than sites located in the eastern half of the peninsula (Somers, 1977). Mean annual precipitation for the sites ranged from 687 to 920 mm yr^{-1} (NOAA, 2011). All sites were located on sandy soils from the Au Gres (sandy, mixed, frigid Typic Endoaquods), Croswell (sandy, isotic, frigid Oxyaquic Haplorthods), Graycalm (isotic frigid Lamellic Udipsamments), Grayling (isotic frigid Typic Udipsamments), Lindquist (sandy, mixed, frigid Lamellic Haplorthods), and Rubicon (sandy, mixed, frigid Entic Haplorthods) series as well as sandy soils with a deep water table characteristic of Entic Haplorthods (Table 4.1; NRCS, 2011).

4.3.3 Field Methods

Five of the sites (Alpena, Baldwin, Gaylord, Houghton Lake, and Pellston) were sampled in the summer of 1997, and six (Beaver Creek, Cherry Creek, Goose Creek, Jack Pine, Locke Dam, and Nurnberg) in the fall of 2011. Two cores were extracted from a minimum of 20 jack pine trees at each site using a manual increment borer. Two cores were collected from each tree to account for within-tree variation in ring width and to aid in identification of frost rings and false rings. Cores were taken 0.5 m above ground because previous research had indicated frost

rings were most likely to occur lower in the stem (Fayle, 1981; Gurskaya, 2014). Sampled trees were selected along transects positioned to cover the largest continuous area of each stand using the point-centered quarter (in 1997) and nearest individual distance (in 2011) method (Barbour *et al*., 1998). Sampling points were established at random distances in 1997 and every 20 m in 2011 along transects. This procedure was chosen to ensure sampling occurred across the range of diameter and age classes present in the stands (Gurskaya & Shiyatov, 2006).

4.3.4 Tree-ring Chronology Development

Cores were air dried and glued to wooden mounts. Cores were surfaced using progressively finer sand paper until cellular structures became visible in the cross-sectional view under magnification. Once prepared, cores were visually cross-dated using the list method (Yamaguchi, 1991). Annual tree ring-widths were measured to the nearest 0.01 mm using the LinTab™ 5 RINNTECH® measuring system (Heidelberg, Germany) and TSAP-Win™ software (v. 4.69, Heidelberg, Germany). Dated tree-ring width measurement values were verified to ensure quality of visual cross-dating using COFECHA software (Holmes, 1983). Dating errors detected by COFECHA caused by misdated rings or missing rings were corrected. Mean series length, series intercorrelation, and mean sensitivity statistics were computed using the COFECHA software values for each site-level chronology.

4.3.5 Frost Ring Measurements

Frost rings were identified, marked, and dated within each core sample. Frost rings were identified under magnification and were based on the appearance of deformed, crumpled tracheids, bent ray cells, and the presence of callus tissue (Bailey, 1925; Glock, 1951; Glerum & Farrar, 1966; Lee, Singh & Kim, 2007; Schweingruber, 2007; Payette, Delwaide & Simard, 2010) (Figure 4.1). Sections within core samples with cellular deformations such as those

caused by fire damage, branch scars, or rot were omitted. Only earlywood frost ring occurrence was analyzed in this study.

Cambial age and diameter were recorded for each core to examine the influence of biological factors on frost-ring formation. Cambial age was determined for each annual growth ring as the difference between the year assigned through cross-dating and the first year of growth. Radial distance was measured from the center of the tree to the start of each annual growth ring. Diameter inside of the bark (referred to as diameter throughout this study) at which each annual growth ring was formed was calculated by summing the two radii measured for each core sample for a given year. In the case where only one radius was included in a chronology, the single measured radius was multiplied by two. Cores that were missing the inner rings or pith were not included in the frost ring analysis due to the potential for error in estimating age or diameter.

4.3.6 Temperature Data

Global Historical Climatology Network-Daily data for the frost-prone, early-growing season months of May and June were available from the National Climatic Data Center (NOAA, 2014) for the extent of the tree-ring chronologies at each site. Previous research in adjacent regions has indicated that on average the onset of xylem production in white pine (Fraser, 1952; Ahlgren, 1957) and jack pine (Heinrichs, Tardif & Bergeron, 2007) as well as other species (Ahlgren, 1957) occurs around the second week of May. Daily minimum temperature data were used to identify the last freeze event with daily minimum temperature below 0°C. The calendar date for the last freeze observed was then converted to a Julian date (days after January 1) for each year. Diurnal range in temperature was calculated as the maximum minus the minimum temperature (°C) observed for each day.

4.3.7 Data Analysis

Earlywood frost-ring formation in jack pine was quantified across cambial age and diameter classes for the 11 sites in northern lower Michigan. The total number of trees and annual growth rings containing at least one frost ring in the earlywood was determined. The proportion of annual growth rings that contained at least one earlywood frost ring was calculated for each one-year age class and each 0.5 cm diameter class. The range in cambial age, diameter, last freeze date, and May minimum temperature in which frost rings occurred was determined for annual growth rings at each site.

Nonlinear normal logistic regression models were estimated using the NLMIXED procedure in SAS 9.3 to quantify the influence of biotic and abiotic factors on the probability of earlywood frost-ring formation across the 11 sites. Specifically, biotic (cambial age, diameter, ring width) and abiotic (elevation, mean initial site-related growth at cambial age 5, Julian date of last freeze, May and June minimum temperatures, May and June maximum diurnal temperature range) predictor variables were used to model the dependent binary outcome (earlywood frost-ring formation) for a given annual growth ring (Table 4.2). Data were obtained through repeated observations within individual trees and therefore, a random effect was also included in each model to account for potential within tree correlation (Table 4.2; Collet, 1991; Wolfinger, 1999).

A logit link function (used in normal logistic regression) was specified within the NLMIXED procedure to estimate each model that predicted the probability of earlywood frost-ring formation in the annual growth rings observed. The correlation coefficient matrix revealed the potential for multicollinearity between cambial age and diameter ($r = 0.8$) and between cambial age and ring width ($r = -0.5$). Therefore, to quantify the influence of these variables on the probability of frost-ring formation, separate models were estimated. Starting parameter

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values for iterations were obtained using the LOGISTIC procedure in SAS 9.3 (SAS, 2012).

Nine predictor variables and one random coefficient (tree id) were used to estimate a full model

(Table 4.2). Predictor variables with parameter values that were significantly different from zero

($P < 0.05$) were carried into reduced models. A total of 425 observations were omitted from the

total 10,334 observations to estimate the full models due to missing daily May or June

temperature data. In the reduced models, 351 observations were omitted due to missing daily

May temperature data leaving a total of 9,983 observations used to estimate the reduced models.

Fit statistics including -2 Log Likelihood (-2LL) and Aikaike Information Criterion (AIC) were

compared among models to determine the best fit model. Odds ratios, interpreted as the change

in odds of an earlywood frost-ring formation due to a one unit increase in a predictor variable

were examined to identify the specific influence of each predictor variables.

4.4 Results

4.4.1 Tree-ring Chronologies

Tree-ring chronologies were developed for each site to ensure accuracy of dating

associated with the presence or absence of frost rings. Chronology length varied across the 11

sites, but overall included years from 1923 to 2011 (Table 4.3). Although the distribution across

age classes varied at the time tree cores were taken, sampled trees ranged from 13 to 104 years

old across all sites. The average length in years for all cores was shortest (40 years) at the

Gaylord site and the longest (64 years) at the Nurnberg sites (Table 4.3). The average correlation

between each series and the master chronology, ranged from 0.497 at Goose Creek to 0.624 at

the Baldwin site. Series intercorrelation values provide an indication of the strength of the stand-

level signal for a site. Mean sensitivity values, which provide a year-to-year measure of

variability in ring width, were on average 0.322 across the 11 sites. The lowest mean sensitivity

value occurred at the Houghton Lake site (0.246), while mean sensitivity was greatest at Beaver Creek (0.392; Table 4.3).

4.4.2 Earlywood Frost-ring Formation

A frost ring was identified for 924 of the 10,334 (9%) total annual growth rings observed. Frost rings were observed when the cambial age was between 1 to 27 years, although tree ages ranged from 13 to 104 years (Table 4.4). Ninety percent of these frost rings occurred between cambial ages 1 to 10 years and 63% between cambial ages 1 to 5 (Figure 4.3). Frost rings did not occur when the corresponding diameter for a respective tree was greater than 10 cm (Table 4.4). Of the total frost rings observed, 87% were formed when diameters ranged from 0 to 3 cm and nearly all (98%) were observed when trees ranged from 0 to 5 cm in diameter (Figure 4.3). Earlywood frost-ring formation varied among the 11 sites across northern lower Michigan (Table 4.4). A significantly greater number of earlywood frost rings were observed at the Beaver Creek, Baldwin, Cherry Creek, Goose Creek, Jack Pine, and Locke Dam sites (112 ± 9.4; mean ± SE) than at the other five sites (51 ± 7.2) when compared using a two-sample t-test (P = 0.001). Earlywood frost rings were also observed in a significantly greater (two sample t-test; P = 0.003) proportion of the annual growth rings within trees at the six sites where the greatest quantities of frost rings were observed (12.2% ± 0.7; mean ± SE) than at the other five sites (5.5% ± 1.2) (Table 4.5). The six sites with a greater quantity and greater proportion of annual growth rings with a frost ring present also had more frost rings in older ages classes (Table 4.5). For instance, 33% of total observed frost rings occurred in trees older than 10 years in cambial age at the Locke Dam site which was much greater when compared to the Alpena, Gaylord, Houghton Lake, and Nurnberg sites at which no frost rings were observed after age 10.

4.4.3 Influences on the Probability of Frost-ring Formation

The influence of biotic and abiotic factors on the probability of frost-ring formation was examined through estimation of nonlinear normal logistic regression models (Table 4.5). To eliminate the potential influence of collinearity in our models, full and reduced models were estimated separately for age (full age model and Reduced Model 1) and diameter (full diameter model and Reduced Model 2) as the correlation coefficient matrix indicated a high degree of correlation ($r = 0.8$) between these two variables. Ring width was only included in the full diameter model and Reduced Model 2 as the degree of correlation was also high between ring width and age ($r = -0.5$). Convergence criteria were satisfied for the full diameter model following 29 iterations and after 20 iterations for the full age model. Convergence criteria were met for the Reduced 1 (age) model after 16 iterations and after 12 iterations for the Reduced 2 (diameter) model. T-tests on individual parameters in the full diameter model indicated diameter ($P < 0.000$), site mean diameter at age 5 ($P < 0.000$), site elevation ($P = 0.013$), ring width ($P < 0.000$), minimum temperature experienced during May ($P < 0.000$), and the random effect of tree ($P < 0.000$) significantly contributed to the model (Table 4.5). These same predictor variables (except for diameter, which was excluded) and the addition of age ($P < 0.000$) were found to be significant in the estimation of the full age model. Estimation of the two reduced models (Reduced Model 1 & 2) confirmed diameter ($P < 0.000$; Reduced Model 2), age ($P < 0.000$; Reduced Model 1), site mean diameter at age 5 ($P < 0.000$; Reduced Model 1 & 2), site elevation ($P = 0.000$, Reduced Model 1; $P = 0.004$, Reduced Model 2), ring width ($P < 0.000$; Reduced Model 2), May minimum temperature ($P < 0.000$; Reduced Model 1 & 2), and the random effect of tree ($P < 0.000$; Reduced Model 1 & 2) were significant predictors of earlywood frost-ring formation across northern lower Michigan (Table 4.5). A total of 351 observations were omitted

from the reduced model regression analyses due to missing May minimum temperature data, leaving a total of 9,983 observations.

Reduced Model 2 was selected as the best fit model based on lower -2 Log Likelihood (-2LL) and Aikaike Information Criterion (AIC) statistics. Reduced Model 2 correctly predicted the presence or absence outcome of 9,515 out of 9,983 (95%) events using a predicted probability cutoff value of 0.7 to evaluate model performance. Odds ratios indicated that as diameter increased by 1 cm, the odds that a tree would form a frost ring in the earlywood of an annual growth ring decreased by 68% (Table 4.5). Based on Reduced Model 2, as the mean site diameter at age 5 increased by 0.5 cm, the odds of earlywood frost-ring formation decreased by 35%. Odds of earlywood frost-ring formation increased by 35% as annual growth ring width increased by 1 mm. Each 1 m increase in site elevation resulted in an increase of 0.4% in the odds of earlywood frost-ring formation. With extrapolation, an increase of 50 m in site elevation increased the odds of earlywood frost-ring formation by 20%. A one unit increase (°C) in May minimum temperature experienced reduced the odds of earlywood frost-ring formation in a respective growing season by approximately 13% (Table 4.5). For each year that cambial age increased, based on Reduced Model 1, the odds that an earlywood frost ring would form decreased by 35%.

4.5 Discussion

4.5.1 Influence of Biotic Factors

Earlywood frost rings most commonly formed in jack pine that had a younger cambial age and smaller diameter (Figure 4.3). These findings concur with previous observations that younger trees and trees with smaller diameters are more susceptible to cambial damage and thus, frost-ring formation (Fayle, 1981; Hantemirov, Gorlanova & Shiyatov, 2000; Gurskaya &

Skiyatov, 2006; Payette, Delwaide & Simard, 2010). Similar age-based patterns have been documented in Siberian spruce (*Picea obovata*) and Siberian larch (*Larix sibirica*), where frost rings were not observed after age 40 in Siberian spruce and not after age 29 in Siberian larch (Gurskaya & Skiyatov, 2006). At a boreal forested site, the average maximum cambial age at which black spruce (*Picea mariana*) formed a frost ring was 37 (Payette, Delwaide & Simard, 2010). Similar to our size-related findings in jack pine, occurrence of frost-rings was greatest in Siberian spruce and Siberian larch when tree diameter was < 2.5 cm with the last frost ring observed between 5 and 6 cm in diameter (Gurskaya & Skiyatov, 2006). In red pine, earlywood frost rings were observed in the greatest density within annual growth rings located in trees < 3 cm in diameter and nearly absent after diameter exceeded 4 cm (Fayle, 1981).

Results from our NLMIXED analyses indicated biotic factors such as cambial age, diameter, and ring width had significant influence on the probability of earlywood frost-ring formation. Increased cambial age and diameter values resulted in decreased odds of earlywood frost-ring formation (Table 4.5). Larger diameter trees are capable of retaining heat absorbed during the daytime for longer periods of time than smaller diameter trees, making them less susceptible to sudden drops in temperature, particularly at night (Gurskaya & Skiyatov, 2006). This decreased occurrence of cambial frost damage with increased cambial age and diameter may also reflect the potential role of increased bark thickness that occurs with increased age and diameter. Bark thickness provides thermal protection to the cambium, thus reducing the likelihood of frost-ring formation with increased bark thickness (Hantemirov, Gorlanova & Shiyatov, 2000; Gurskaya & Skiyatov, 2006). Bark thickness measurements were not included in this study; however, the biotic factors cambial age and diameter, which are highly correlated with bark thickness, were significant predictors of earlywood frost-ring formation illustrating the

overall influence of biotic factors. Increased ring width resulted in increased odds of earlywood frost-ring formation. This finding may be a result of wider growth rings forming at earlier ages in some trees depending on environmental conditions at establishment. However, it may also provide support for the idea that more narrow annual growth rings are less likely to form frost rings due to fewer cells in the differentiation phase at any given time during growing season frost events (Schweingruber, 2007; Payette, Delwaide & Simard, 2010).

4.5.2 Influence of Abiotic Site-related Factors

Frost ring occurrence varied by site; trees at sites characterized by a greater proportion of annual growth rings with an earlywood frost ring appeared to be more susceptible to cambial damage at older ages and larger diameters (Table 4.4). Previous studies have shown the susceptibility to frost-induced cambial damage may vary by site as a result of differences in site characteristics such as, microtopography, aspect, stand exposure, soil moisture, and elevation (Langvall & Löfvenius, 2002; Bär, Bräuning & Löffler, 2006; Dy & Payette 2007; Payette, Delwaide & Simard, 2010; Hadad, Amoroso & Juñent, 2012; Panayotov, Zafirov & Cherubini, 2013; Gurskaya, 2014). Microtopography, aspect, and soil water holding capacities were very similar across our study sites, which reduced the variation in these factors. However, results from our NLMIXED analyses identified site-based elevation as a significant factor, influencing the probability of earlywood frost-ring formation (Table 4.5). Although the interpretation of this influence is limited to the range of our 11 site elevation values (201 to 406 m asl), the odds of earlywood frost-ring formation increased as site elevation also increased. This finding is in agreement with previous studies which report increased frost ring occurrence at increased elevations (Hadad, Amoroso & Juñent, 2012; Panayotov, Zafirov & Cherubini, 2013).

Variation in earlywood frost-ring formation among sites was also influenced by differences in initial site-related growth rates. Results from the NLMIXED analysis indicated greater mean diameter values at age 5 for trees at a given site resulted in decreased odds of earlywood frost-ring formation. This is likely due to the extended length in time it takes slower growing trees to reach a diameter where frost rings are less likely to form (Fayle, 1981; Gurskaya & Skiyatov, 2006; Payette, Delwaide & Simard, 2010). To illustrate, at the Alpena site (207 m asl), trees averaged 4.3 cm in diameter by age 5 and earlywood frost rings were only formed in 3% of annual growth rings, all within the first 5 years of growth. In contrast, at Locke Dam (229 m asl), trees averaged 1.7 cm in diameter at age 5 and had earlywood frost rings present in 11% of total growth rings, which occurred in trees up to 27 years old.

Differences in crown cover and stand density through time may have also contributed to the differences observed among sites in earlywood frost-ring formation. Dense, uniform crown cover reduces susceptibility to cambial frost damage because the closed canopy creates a greenhouse-like warming effect and traps heat during the day which provides increased thermal protection against cold temperatures at night (Langvall & Ottosson-Löfvenius, 2002; Dy & Payette, 2007). Sites with a more open, less dense overstory canopy, such as the Goose Creek barren, had a greater proportion of annual growth rings with an earlywood frost ring than the more uniformly, closed Alpena and Gaylord plantations, or the natural Nurnberg stand (Table 4.4, personal observation). Although stem density could not be measured through time, examination of age distributions revealed potential differences in the stand density and canopy cover among sites. For instance, age distribution of the sampled jack pine across age classes was unimodal for the Alpena, Gaylord, and Nurnberg sites, indicating a single age cohort whereas, trees were distributed across a greater range of age classes at the Beaver Creek, Cherry Creek,

and Locke Dam sites. Sites with unimodal age distributions were characterized by a lower proportion of growth rings with an earlywood frost ring than sites where trees were distributed across a greater range of age classes. Similar trends were documented across sites in boreal forests that varied in cambial age distribution (Payette, Delwaide & Simard, 2010).

4.5.3 Influence of Abiotic Climatic Factors

Frost-ring formation is dependent on the occurrence of a freeze event during the early growing season. Earlywood frost rings formed in jack pine when May minimum air temperatures ranged from -13.3° to 2.2°C, which is similar to values reported in other studies. Earlywood frost-ring formation in Scots pine (*Pinus sylvestris*) was observed when minimum ambient temperatures ranged from -11.0° to -7.8°C at 2,100 m asl and from -0.6° to 2.6°C at 800 m asl (Panayotov, Zafirov & Cherubini, 2013). Similarly, minimum temperature ranges associated with frost-ring formation also differed among sites (Panayotov, Zafirov & Cherubini, 2013). In our study, the range of May minimum temperatures associated with frost-ring formation varied among sites from -10.6° to -1.7°C at the Baldwin site (251 m asl), -9.4° to 1.1°C at the Beaver Creek site (373 m asl), -7.8° to -4.4°C at the Gaylord site (406 m asl), and from -2.2 to 2.2°C at the Nurnberg site (Table 4.4). Mean minimum air temperature associated with earlywood frost-ring formation in Siberian spruce and Siberian larch was 0.9°C, with upper thresholds of occurrence at 2.0°C and 2.9°C for the two species (Gurskaya, 2014).

May minimum temperatures associated with earlywood frost-ring formation were slightly higher at some sites than the minimum threshold value of 0°C cited by others (Dy & Payette, 2007). Temperatures higher than 0°C may have been associated with temperature differences between instrument height and height at which frost rings were observed as well as the distance between the site and the weather station at which temperature values were recorded. Other

studies have illustrated temperatures recorded at instrument height, typically 2 m, were slightly warmer than the minimum temperature at ground-level (Langvall & Löfvenius, 2002; Gurskaya & Skiyatov, 2006; Dy & Payette, 2007). Additionally, weather stations used to document daily or hourly weather data frequently occur in convenient, more urban areas such as airports, which are locations known to experience slightly higher temperatures than sites with increased forest cover. Despite these potential slight differences, our models clearly illustrated that as May minimum air temperature increased, the probability of earlywood frost-ring formation decreased.

Unexpectedly, the additional temperature variables did not significantly influence the likelihood of earlywood frost-ring formation (Table 4.5). Although May and June maximum diurnal temperature ranges were not significant predictors in our models, these values may have had a slight influence on the probability of earlywood frost-ring formation. The odds of earlywood frost-ring formation increased by 1% as the maximum range in daily diurnal temperature (daily maximum minus daily minimum) for May and 2% for June increased by 1°C. Last freeze date had no influence on the probability of frost-ring formation (Table 4.5). This was likely due to last freeze dates occurring after earlywood xylem formation began, the time when the cambium was most vulnerable to temperature-induced damage (Glerum & Farrar, 1966).

4.6 Conclusions

This study illustrates the influence of biotic and abiotic factors on the occurrence of frost-induced cambial damage to trees across multiple forested sites. Frost rings, a modification to wood anatomy, have the potential to aid in reconstructing historical events, including growing season frosts, as well as to indicate frost-prone environmental conditions (Fritts, 1976; Wimmer, 2002). As illustrated in northern lower Michigan, the susceptibility to frost damage and thus, the ability of an individual tree to record a frost disturbance should be considered when using frost

rings as such a proxy. Impacts of late spring frost disturbances are well known; however, understanding trends in frost-ring formation as influenced by biotic and abiotic factors can also provide forest managers with additional information relative to making management decisions aimed at minimizing the negative impacts of growing season frosts (Langvall & Löfvenius, 2002; Dy & Payette, 2007). Further, frost rings could also be used in the future as a physiological indicator of geographical boundaries that limit the spatial range of plant species as well as to indicate shifts in these ranges which may occur as a result of climate change disturbances (Kullman, 2002; Wimmer, 2002).

4.7 Acknowledgements

4.8 References

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Table 4.1 Description of the Alpena (AL), Beaver Creek (BC), Baldwin (BL), Cherry Creek (CC), Goose Creek (GC), Gaylord (GY), Houghton Lake (HL), Jack Pine (JP), Locke Dam (LD), Nurnberg (NU), and Pellston (PL) study sites in northern lower Michigan.

Site	Latitude & longitude	Elevation (m ASL)	Distance to [a] lakeshore (km)	Mean [b] annual precipitation (mm)	Location	Soil series [c]
AL	45º03' N, 83º33' W	207	12	715	Mackinaw State Forest	Au Gres, Croswell, Rubicon
BC	44º21' N, 84º20' W	373	67	797	AuSable State Forest	Graycalm
BL	43º51' N, 85º49' W	251	66	878	Pere Marquette State Forest	Grayling
CC	44º40' N, 84º15' W	315	89	687	AuSable State Forest	Grayling
GC	44º42' N, 84º50' W	350	67	854	AuSable State Forest	Grayling
GY	45º00' N, 84º43' W	406	48	920	Mackinaw State Forest	Lindquist
HL	44º20' N, 84º50' W	346	89	781	AuSable State Forest	Au Gres
JP	44º39' N, 85º21' W	295	22	840	Pere Marquette State Forest	Rubicon
LD	44º28' N, 83º28' W	229	17	796	AuSable State Forest	Grayling
NU	44º06' N, 86º21' W	201	7	881	Manistee National Forest	nd [d]
PL	45º34' N, 84º46' W	218	19	766	Mackinaw State Forest	Rubicon

[a] Distance to the closest Great Lake (Lake Huron or Lake Michigan) determined using Near distance tool in ESRI, ArcMap 10.1.
[b] Station Summary of Monthly Normals 1981-2010 (NOAA, 2011).
[c] Determined using Web Soil Survey (NRCS, 2011).
[d] Series (nd) not defined; soil taxonomy classified as Entic Haplorthods (NRCS, 2011).

Table 4.2 Description of variables included in full and reduced nonlinear normal logistic regression models.

Variable	Description
Frost ring	Dependent binary variable; presence (1) or absence (0) of at 1 least frost ring in a given annual growth ring with known year
Diameter	Diameter (rounded to nearest cm) of tree at which a given annual growth ring occurred
Site diameter age 5	Mean diameter (rounded to nearest 0.5 cm) for all trees at each site at cambial age 5
Elevation	Site elevation (meters above sea level)
Ring width	Mean ring width (rounded to nearest mm) of annual growth ring for a dated year in each respective tree
Age	Cambial age (years) at which a given annual growth is formed
May minimum	Lowest minimum temperature recorded (rounded to nearest °C) in May of a given year
June minimum	Lowest minimum temperature recorded (rounded to nearest °C) in June of a given year
May max. diurnal	Maximum range in daily diurnal temperature (daily maximum minus minimum temperature rounded to nearest °C) for May of a given year
June max. diurnal	Maximum range in daily diurnal temperature (daily maximum minus minimum temperature; rounded to nearest °C) for June of a given year
Julian last freeze	Julian calendar date corresponding to date of last freeze event during the growing season of a given year

Table 4.3 Tree ring chronology characteristics for the Alpena (AL), Beaver Creek (BC), Baldwin (BL), Cherry Creek (CC), Goose Creek (GC), Gaylord (GY), Houghton Lake (HL), Jack Pine (JP), Locke Dam (LD), Nurnberg (NU), and Pellston (PL) study sites in northern lower Michigan.

Site	Chronology span	Series lengths		Number of trees	Number of radii	Total growth rings	Series intercorrelation	Mean sensitivity
		Range (years)	Mean (years)					
AL	1947 - 1996	28 to 50	44	21	41	1814	0.572	0.258
BC	1936 - 2011	26 to 77	48	23	44	2122	0.572	0.392
BL	1923 - 1996	37 to 78	45	24	48	2147	0.624	0.289
CC	1937 - 2011	20 to 78	49	23	45	2201	0.556	0.382
GC	1909 - 2011	31 to 104	42	24	48	2032	0.497	0.322
GY	1955 - 1996	38 to 42	40	25	50	1989	0.610	0.296
HL	1920 - 1996	33 to 79	49	25	50	2452	0.583	0.246
JP	1929 - 2011	27 to 82	53	26	50	2687	0.585	0.386
LD	1935 - 2011	20 to 77	49	24	47	2289	0.579	0.386
NU	1943 - 2011	29 to 69	64	24	48	3055	0.514	0.329
PL	1938 - 1996	13 to 59	43	20	40	1721	0.541	0.260

Table 4.4 Percentage of trees and growth rings containing at least one earlywood frost ring and cambial age (years), diameter (cm), last freeze date (month, day), and May minimum temperature (C) range for years where a frost ring was present for the Alpena (AL), Beaver Creek (BC), Baldwin (BL), Cherry Creek (CC), Goose Creek (GC), Gaylord (GY), Houghton Lake (HL), Jack Pine (JP), Locke Dam (LD), Nurnberg (NU), and Pellston (PL) study sites in northern lower Michigan.

| | | Frost ring present | | | Range of frost ring occurrence | | |
Site	Number of trees analyzed	Trees (%)	Growth rings (%)	Cambial age (years)	Diameter (cm)	Last freeze	May minimum temp. (°C)
AL	20	75.0	3.0	1 - 5	0 - 3.0	May 10 - June 18	-6.1 to -2.2
BC	21	100.0	15.6	1 - 24	0 - 5.5	April 28 - June 20	-9.4 to 1.1
BL	20	100.0	11.3	1 - 16	0 - 8.5	May 9 - June 22	-10.6 to -1.7
CC	16	100.0	12.6	1 - 21	0 - 4.0	May 3 - June 18	-8.3 to 0.6
GC	20	95.0	11.7	1 - 12	0 - 6.0	May 2 - June 25	-7.8 to -1.1
GY	23	95.7	5.4	1 - 4	0 - 4.0	May 17 - June 14	-7.8 to -4.4
HL	22	90.9	4.8	1 - 9	0 - 10.0	April 28 - June 22	-7.2 to 1.1
JP	20	100.0	11.3	1 - 18	0 - 5.5	May 8 - June 30	-13.3 to -1.1
LD	18	94.4	10.8	1 - 27	0 - 6.0	April 20 - June 14	-6.7 to 1.1
NU	18	88.9	4.2	1 - 8	0 - 3.5	April 18 - May 28	-2.2 to 2.2
PL	16	100.0	9.9	1 - 12	0 - 3.5	May 8 - June 23	-11.7 to -2.2

Table 4.5 Full and reduced nonlinear normal logistic regression model estimates, t-test results, and -2 Log Likelihood (-2LL) and Aikaike information criterion (AIC) fit statistics for models using variables to predict earlywood frost-ring formation across northern lower Michigan sites.

Model/variable[a]	Parameter estimate	Standard error	t-value	P-value	Odds ratio	-2LL	AIC
						\ Fit statistics	
Full						2063.2	2085.2
Intercept	0.98	1.26	0.78	0.435			
Diameter	-1.15	0.05	-24.42	<0.000*	0.318		
Site diameter age 5	-0.46	0.10	-4.71	<0.000*	0.630		
Elevation	0.00	0.00	2.51	0.013*	1.003		
Ring width	0.50	0.06	8.29	<0.000*	1.650		
May minimum	-0.13	0.03	-4.20	<0.000*	0.877		
June minimum	-0.02	0.03	-0.51	0.612	0.983		
May max. diurnal	0.01	0.03	0.38	0.707	1.010		
June max. diurnal	0.02	0.03	0.92	0.358	1.024		
Julian last freeze	-0.00	0.01	-0.74	0.463	0.996		
$\sigma^2\mu1$ [a]	0.78	0.18	4.27	<0.000*			
Reduced 1 [b]						2415.1	2427.1
Intercept	2.68	0.40	6.72	<0.000*			
Age	-0.43	0.02	-26.27	<0.000*	0.648		
Site diameter age 5	-0.80	0.06	-12.36	<0.000*	0.448		
Elevation	0.00	0.00	3.95	0.000*	1.005		
May minimum	-0.10	0.03	-4.04	<0.000*	0.903		
$\sigma^2\mu1$ [a]	0.69	0.05	19.66	<0.000*			
Reduced 2 [b]						2092.7	2106.7
Intercept	0.84	0.44	1.93	0.055			
Diameter	-1.14	0.05	-24.76	<0.000*	0.320		
Site diameter age 5	-0.44	0.10	-4.55	<0.000*	0.647		
Elevation	0.00	0.00	2.94	0.004*	1.004		
Ring width	0.50	0.06	8.39	<0.000*	1.647		
May minimum	-0.13	0.03	-4.99	<0.000*	0.872		
$\sigma^2\mu1$ [a]	0.74	0.17	4.30	<0.000*			

[a] A random effect ($\mu1$) was included in all models to account for potential within tree correlation effects on annual observations. Tree id (n=214) was specified as the subject; df =213.
[b] Two reduced models were estimated to examine the influence of age and diameter and due to a high degree of correlation between the variables age and diameter and between age and ring width. Only estimates for full diameter model presented as variables carried into reduced models were identical.
* Denotes significance at $\alpha = 0.05$.
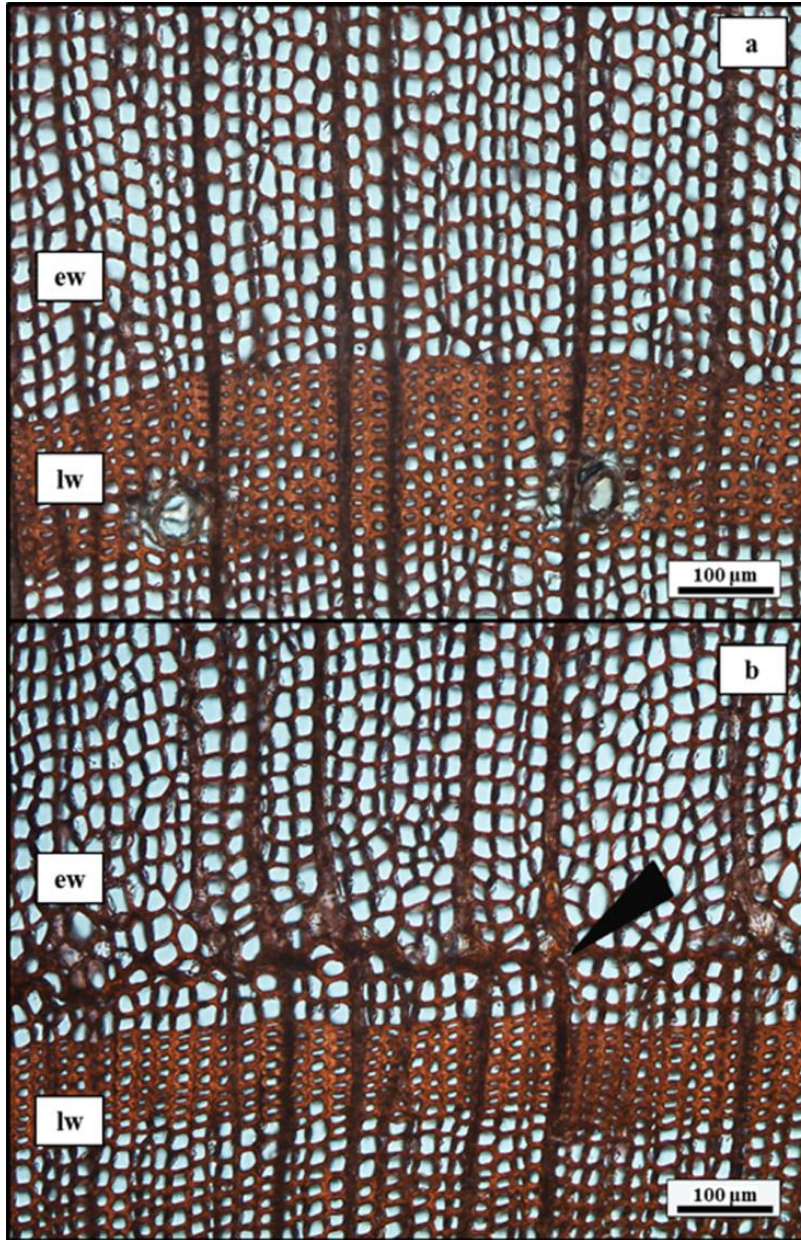
Figure 4.1 Cross sectional view of a jack pine annual growth ring with a) normal and b) abnormal wood anatomy. Normal growth ring is characterized by undamaged tracheids in the earlywood (ew) and prior year latewood (lw) zones. Abnormal growth ring contains a frost ring in the earlywood (black arrow) characterized by a line of deformed, collapsed tracheids intersected by bent ray parenchyma cells damaged by a late spring freeze event in northern lower Michigan. Cross section were 30-40 μm thick and stained with safranin for contrast.
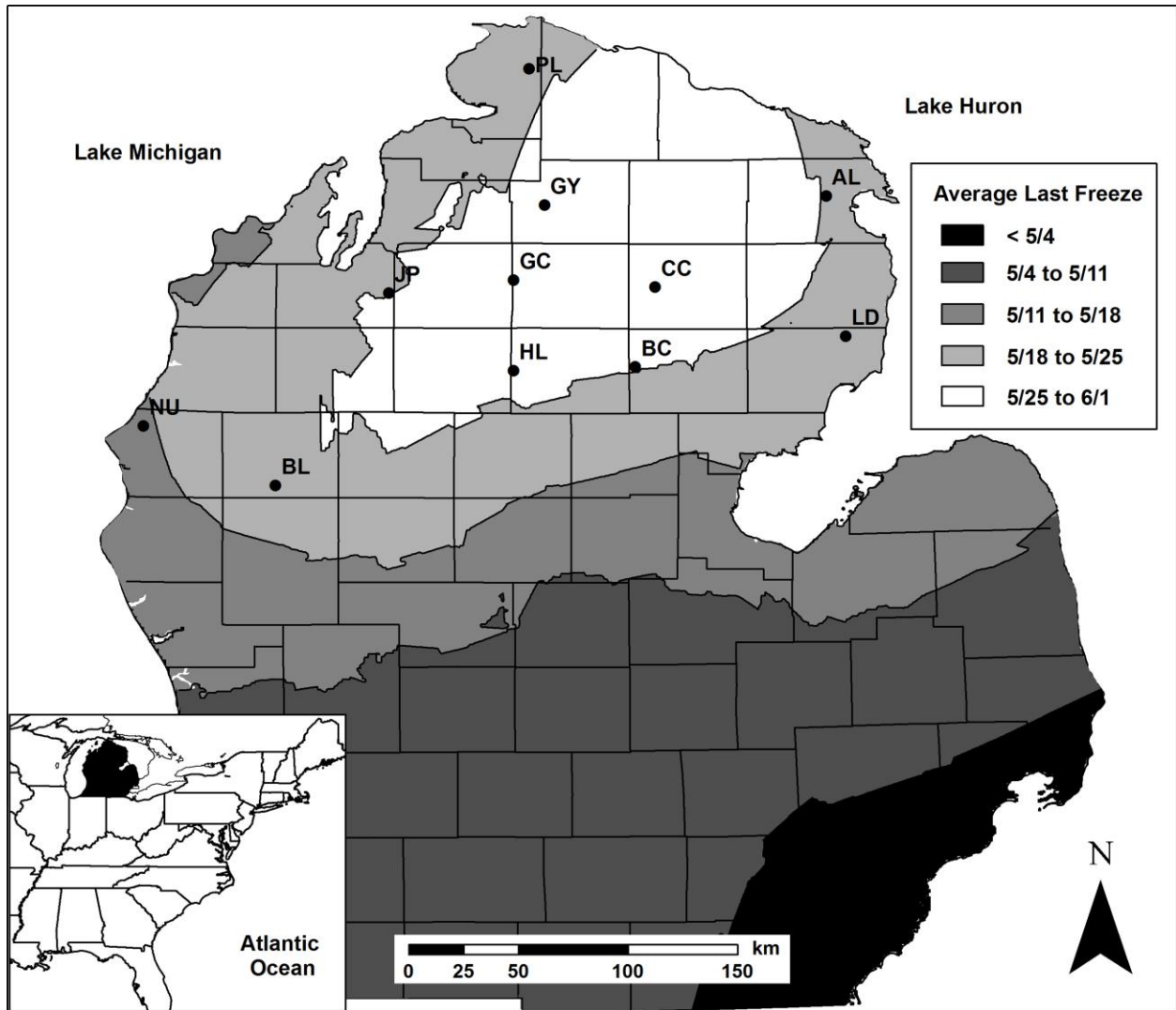
Figure 4.2 Study sites located in northern lower Michigan included Alpena (AL), Beaver Creek (BC), Baldwin (BL), Cherry Creek (CC), Goose Creek (GC), Gaylord (GY), Houghton Lake (HL), Jack Pine (JP), Locke Dam (LD), Nurnberg (NU), and Pellston (PL). Shading corresponds to ranges of average last freeze dates estimated using the ordinary kriging Gaussian interpolation method and data from 68 selected Global Historical Climatological Network stations in lower Michigan (NOAA, 2014).
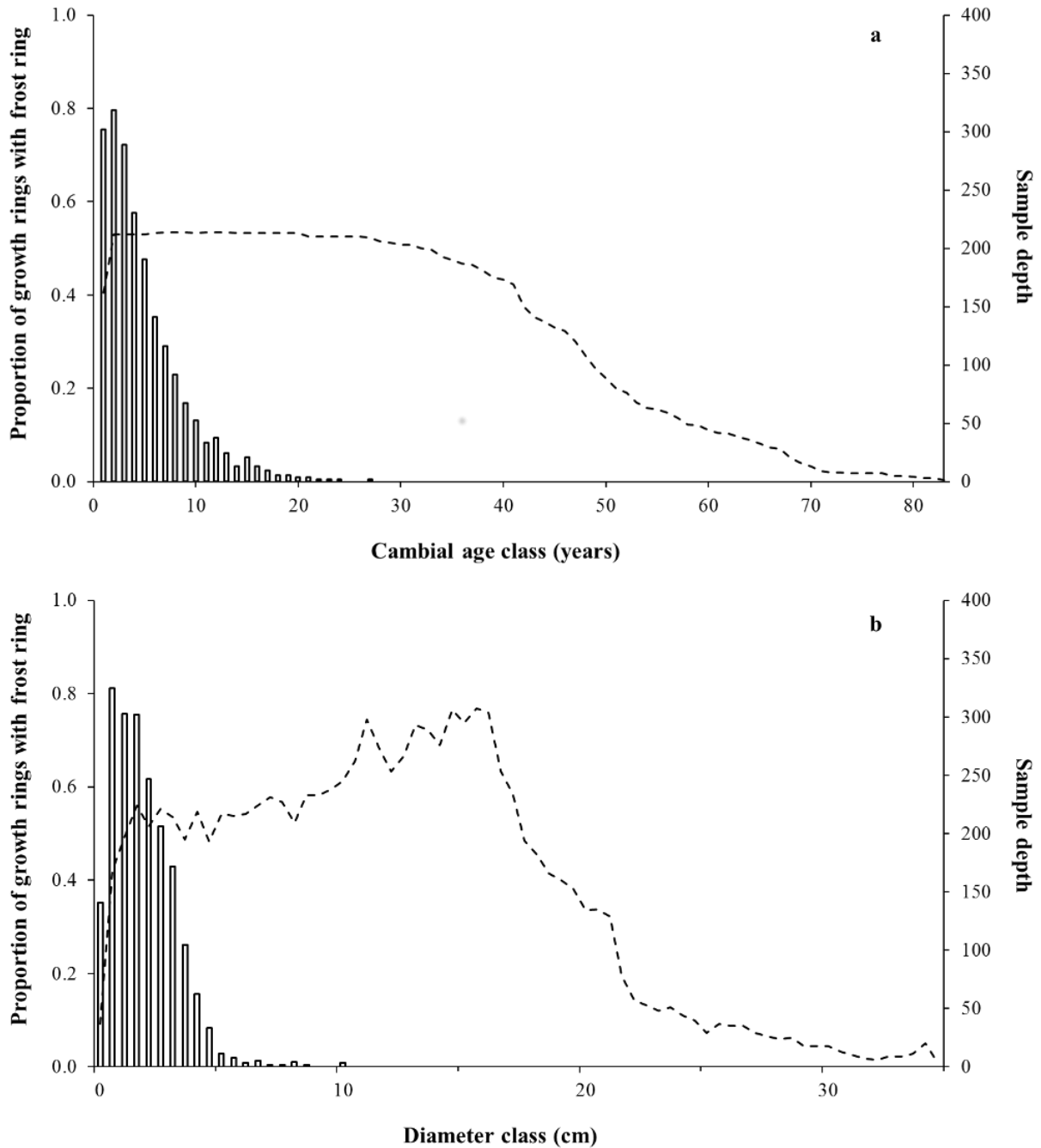
Figure 4.3 Proportion of growth rings that contained at least one earlywood frost ring across a) cambial age classes (years) and b) diameter classes (0.5 cm), including observations from all eleven sites in northern lower Michigan. Dashed lines represent sample depth for each class. Data occurring after age class 83 and diameter class 34 cm were not included in graphs due to decreased sample depth (n=1).

CHAPTER 5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Forest Disturbances

An understanding of disturbance processes and impacts proves beneficial to the management of forested ecosystems (Oliver, 1981; Attiwill, 1994; Rogers, 1996; Jõgiste et al., 2007; DeRose and Long, 2014). However, gaps in knowledge of disturbance impacts and occurrences remain, leaving forest managers and society oblivious to the occurrences of some disturbances and their consequential impacts. This dissertation provides knowledge on the impacts of recreational crossing disturbances on water quality, illustrates the use of a dendrogeomorphic method to estimate sediment deposition patterns through time in riparian forested wetlands, identifies relationships between radial growth and hydrology and climate, and quantifies the influence of biotic and abiotic factors on the likelihood of cambial damage during late spring frost disturbances.

5.2 Forest Recreational Stream Crossing Disturbances

To determine the impacts of recreational stream crossing disturbances, soil erosion rates were modeled using Universal Soil Loss Equation for Forestry and Water Erosion Prediction Project models along multiple-use (hiking, mountain biking, and horseback riding) recreational stream crossing approaches within the Poverty Creek Trail System of the George Washington and Jefferson National Forest in southwestern Virginia. Soil erosion estimates along crossing approaches were used as potential sources of sediment input at culvert and ford crossing locations. Changes in water quality below stream crossings were evaluated using two different macroinvertebrate-based indices (Family-level Virginia Stream Condition Index and Family-

level Hilsenhoff Biotic Index). Specifically, index values in upstream reference reaches were compared to respective downstream values below each crossing to determine change in water quality.

Modeled soil erosion rates were 13 times higher along trail approaches to stream crossings than in previously evaluated adjacent control forestlands (Hood et al., 2002). Modeled soil erosion rates illustrate that recreational stream crossing approaches have the potential to deliver sediment into adjacent streams, particularly where best management practices (BMPs) are not being implemented or where approaches are not properly managed. Downstream changes in macroinvertebrate-based water quality index values illustrated recreational stream crossings were having a negative impact on conditions in streams crossed by recreational trails in the Poverty Creek Trail System. Crossing type (ford vs. culvert) did not appear to impact the degree to which water quality was impacted in this study. Negative recreational disturbance impacts on soil erosion and water quality in Chapter 2 support the need for implementing BMPs to reduce soil erosion and protect water quality. Previous research along roads and skid trails used for forestry operations has resulted in continued development, evaluation, and implementation of forestry best management practices (Wade et al., 2012; Wear et al., 2013). Potentially effective recreational BMPs may include use of low cost bridges, better culvert stabilization, hardened fords, increased water diversion structures (e.g., water bars) along trails, application of cover material (e.g., gravel) along stream crossing approaches, or a combination of supportive material (e.g., geotextile) and cover material. Trail-based recreational opportunities continue to expand due to increased popularity; thus, a system emphasizing accountability of management practices to reduce soil erosion potentials along approaches to stream crossings is needed.

This study was the first to evaluate soil erosion potentials along non-motorized recreational stream crossing approaches and to evaluate the impact of this erosion on downstream water quality. Findings from Chapter 2 are critical to the way society views recreational land uses and their impacts on water quality. Trail-based soil erosion that occurs near stream crossings represents a non-point source of pollution to streams and based on Chapter 2 findings are significant enough to cause degradation of water quality. This study identifies the role recreational trail erosion can play in increasing organic pollution in waterways. Sediment is the leading cause of impaired waterways in the United States (US Environmental Protection Agency, 2000). Thus, recognition of recreational trail erosion and associated stream crossing disturbances as a source of sediment introduction into waterways is a critical first step to protecting water quality in the future. The Clean Water Act of 1972 was developed to protect water quality by regulating discharges of pollutants into the waterways of the United States. Trail-based recreation remains under the radar as a source of pollution; however, for water quality to be protected this must change. Results from this study provide an important first step in this process. Future research along recreational trail approaches to stream crossings should quantify the trail-eroded sediment that is actually delivered into streams at stream crossings and evaluate model accuracy associated with predictions; identify soil erosion and sediment introductions with specific traffic and precipitation event-based disturbances; measure below crossing changes in stress-related water quality variables such as turbidity, TSS, and nutrient-levels; and evaluate the effectiveness of recreational BMPs at reducing soil erosion and ameliorating sediment entry into waterways and user response to the BMPs.

5.3 Forest Hydrogeomorphic Disturbances

Sediment accretion rates were estimated for long- (131 years) and short-term (25 years) periods along a natural levee and backswamp riparian forested wetland along the Tensaw River in southern Alabama, using a previously validated dendrogeomorphic technique (Hupp and Morris, 1990). This technique was used to provide longer term estimates with only one site visit compared to other methods which require repeated site visits through time. Sediment accretion rates were compared between long- and short-term time periods. Short-term dendrogeomorphic rates on the natural levee (35 m from river) and backswamp (75 m from river) were compared values obtained from previous periodic sediment pin measurements and repeated elevation surveys at 160, 250, and 330 m from the river. Relationships between annual radial growth and hydrologic and climatic variables were identified through correlation analysis.

Short-term sediment accretion estimates were higher than long-term rates and were higher along the natural levee than backswamp. Sediment accretion rates were higher on the natural levee than estimates further from the river demonstrating the role of spatial location or site connectivity to sediment-laden waters. Overbank flooding disturbances during the growing season had positive impacts on green ash water tupelo growth. Positive, significant correlations were identified between green ash radial growth and the number of days above flood stage and average daily river stage during April. Management implications associated with hydrologic regime in delivery and deposition of sediment, integral to succession of bottomland hardwood systems and wetland restoration sites were realized (Hodges, 1997). Management efforts in artificially created forested wetlands should focus on connectivity to sediment-laden waters as a source of sediment input in these systems (Middleton, 1999). Overall findings from this study highlight the importance of wetlands to society, and therefore support retention and protection of these bottomland systems. Overbank flooding disturbances had positive impacts on tree growth.

Green ash and water tupelo trees were impacted by overbank hydrologic disturbances more than climatic disturbances which suggest management efforts to maximize productivity should be aimed at increasing periodic flooding events, where possible (e.g., impoundments or created wetlands), during growing season months.

Findings in Chapter 3 provide one of the few illustrations of the use of the dendrogeomorphic technique in detecting changes in sediment deposition through time and space. This study also demonstrates the use of dendrochronological techniques and annual resolution data in identifying relationships between radial growth and hydrologic and climatic variables in riparian forested wetlands. Long-term observations of sediment accretion rates provide information necessary to estimate the quantity of sediment being trapped by wetlands. Wetlands trap sediment that is filtered from adjacent waterways and thus, riparian wetlands improve water quality. This study highlights the usefulness of the dendrogeomorphic technique in providing such long-term estimates in understanding the importance of forested riparian wetlands in trapping sediment and pollutants. Through the use of the dendrogeomorphic technique, differences in sediment accretion rates and thus, changes in sediment trapping efficiency, through time were detectable which is critical to understanding how land use and resulting sediment inputs into waterways change through time. Changes in upstream land use can alter the amount of sediment being transported downstream. Examples of changes may include agriculture, urban/suburban development, land clearing, afforestation, or dam construction. Additionally, the trapping efficiency of riparian forested wetlands can also change through time. Therefore, the ability to detect such changes in sediment loads and wetland functions is extremely important to valuation of forested wetlands. Future research should be

aimed at determining sources of the sediment deposited in riparian wetlands and impacts of altered hydrologic regimes on temporal and spatial patterns of sediment deposition.

5.4 Forest Climatic Frost Disturbances

In Chapter 4, the occurrence of late spring frost-induced cambial damage was quantified among diameter and age classes in jack pine, across northern lower Michigan. Specifically, the occurrence of frost rings, abnormal modifications to wood anatomy within the annual growth rings, was documented in jack pine in trees with a known diameter and cambial age across 11 sites which varied in elevation. The influence of biotic and abiotic factors on frost-ring formation was quantified through estimation of nonlinear normal logistic regression models.

Findings in Chapter 4 indicate negative frost-induced impacts to the cambium occurred most frequently in smaller diameter trees and younger trees. The number and frequency of frost ring occurrence was lower at sites with faster initial growth rates, which implies management strategies aimed at minimizing frost-induced cambial damage should focus on increasing initial growth rates. Additionally, sites located in the higher-elevated interior sections of northern lower Michigan had fewer occurrences and lower frequencies of frost-ring formation. Spatial location relative to temperature modifying landscape features such as the Great Lakes should be considered when identifying frost-prone sites. Management strategies should focus on biotic and abiotic factors identified in Chapter 4 to gain additional information relative to making management decisions aimed at minimizing the negative impacts of growing season frosts (Langvall and Löfvenius, 2002; Dy and Payette, 2007).

This study is the first to quantify the influence of biotic and abiotic factors on the occurrence of frost rings, indicators of frost-induced cambial damage, in trees across multiple forested sites. Frost rings may serve as physiological indicator of shifts in geographical

boundaries which may occur as a result of climate change disturbances (Kullman, 2002; Wimmer, 2002). Previous studies indicate that as climate change occurs and early spring air temperatures increase, bud burst, and essentially, plant growth will begin earlier in the year (Cannell and Smith, 1986; Hänninen, 1991, Gu et al., 2008; Rigby and Porporato, 2008). Early cambial activity exposes premature vegetation to a greater risk of abrupt freeze due to greater temperature fluctuations in early spring (Cannell and Smith 1986, Gu et al., 2008). Therefore, it is likely that climate change will increase the frequency of cambial frost damage (Cannell and Smith, 1986; Hänninen, 1991; Gu et al., 2008; Rigby and Porporato, 2008) and thus, frost ring formation. Further, negative impacts on growth have been identified following growing season frost damage (Dittmar et al., 2006). Reduced growth following increased frequencies of growing season frost may have impacts on above ground biomass, and thus, pulp production in jack pine in the Great Lakes region. Future research should quantify the occurrence of frost rings as indication of physiological frost-induced cambial damage along the geographic boundaries of species expected to be impacted by climate change. This could provide a useful method to determine the occurrence of shifts in species distributions. The impacts of growing season frosts on future production should be further investigated.

Overall, this dissertation bridged gaps in knowledge of recreational, hydrogeomorphic, and climatic disturbances in forested ecosystems. Recognition and interpretation of the occurrence and impacts of disturbances provides managers with information which aids in the development and implementation of effective management strategies.

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