University of Tennessee, Knoxville
TRACE: Tennessee Research and Creative Exchange

Doctoral Dissertations | Graduate School

8-2005

Sedimentation, Hydrology, and Bottomland Hardwood Forest Succession in Altered and Unaltered Tributaries of the Hatchie River, TN

Aaron R. Pierce
University of Tennessee, Knoxville

Follow this and additional works at: https://trace.tennessee.edu/utk_graddiss

Part of the Natural Resources and Conservation Commons

Recommended Citation

Pierce, Aaron R., "Sedimentation, Hydrology, and Bottomland Hardwood Forest Succession in Altered and Unaltered Tributaries of the Hatchie River, TN. " PhD diss., University of Tennessee, 2005.
https://trace.tennessee.edu/utk_graddiss/4338

This Dissertation is brought to you for free and open access by the Graduate School at TRACE: Tennessee Research and Creative Exchange. It has been accepted for inclusion in Doctoral Dissertations by an authorized administrator of TRACE: Tennessee Research and Creative Exchange. For more information, please contact trace@utk.edu.

To the Graduate Council:

I am submitting herewith a dissertation written by Aaron R. Pierce entitled "Sedimentation, Hydrology, and Bottomland Hardwood Forest Succession in Altered and Unaltered Tributaries of the Hatchie River, TN." I have examined the final electronic copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Natural Resources.

Sammy L. King, David Buehler, Major Professor

We have read this dissertation and recommend its acceptance:

Carol Harden, Jake Weltzin, Jennifer Franklin, Ray Albright

Accepted for the Council:

Carolyn R. Hodges

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)

To the Graduate Council:

I am submitting herewith a dissertation written by Aaron R. Pierce entitled "Sedimentation, Hydrology, and Bottomland Hardwood Forest Succession in Altered and Unaltered Tributaries of the Hatchie River, TN". I have examined the final paper copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Natural Resources.

Sammy L. King, Co-Major Professor

David Buehler, Co-Major Professor

We have read this dissertation and recommend its acceptance:

Carol Harden

Jake Weltzin

Jennifer Franklin

Ray Albright

Accepted for the Council:

Vice Chancellor and Dean of Graduate Studies

Agvetmed
Thesis
2005b
.P54

Sedimentation, Hydrology, and Bottomland Hardwood Forest Succession in Altered and Unaltered Tributaries of the Hatchie River, TN

A Dissertation
Presented for the
Doctor of Philosophy
Degree
The University of Tennessee, Knoxville

Aaron R. Pierce
August 2005

Acknowledgements

I thank Dr. Sammy L. King for introducing me to wetlands and for his support, encouragement, and enthusiasm throughout this study. Dr. King's knowledge, work ethic, and passion for wetlands have been and will continue to an inspiration to me. I thank Dr. Dave Buehler, Dr. Jennifer Franklin, Dr. Ray Albright, Dr. Carol Harden, and Dr. Jake Weltzin for their support and advice throughout this study.

I also thank Alex Wyss and Glenn Gallien of the Nature Conservancy for their help and support of my study. The U.S. Fish and Wildlife Service provided support and funding for this project. I especially appreciated the assistance from the staff at the Hatchie River National Wildlife Refuge, Don Orr, Marvin Nichols, and Randy Kipley of the U.S. Fish and Wildlife Service. I thank Carl Wirwa of the Tennessee Wildlife Resources Agency for his support and assistance with this study. I also thank all of the landowners in western Tennessee that granted me permission to conduct my research on their land and have encouraged me throughout this study.

I thank the Department of Forestry, Wildlife, and Fisheries for funding and support throughout this study. I especially thank Dr. Mark Fly, Dr. Dave Ostermeier, and Dr. Don Hodges for their support and funding provided by the USDA IFAFS program. I also thank Dr. Frank van Manen and Laura Thompson for their assistance and guidance with the geospatial analysis. I also thank Dr.

Henry Grissino-Mayer and his lab for their assistance in the Laboratory of Tree Ring Science at the University of Tennessee.

I thank the numerous field assistance and graduate students who have assisted and encouraged me during this study, including: Justin Nolte, Eric Sawyers, Chuck Yoest, Nick Wirwa, B.J. Wilson, Wes Cochran, Jimmy Fox, Jeff Roberson, Michelle Pfeffer, Shelton Whittington, Serigo Pierluissi, Nick Winstead, Brian Hemal, and Aaron Wells.

I thank my immediate and extended family for the unending support and encouragement. I especially thank my parents, Harold and Claudette Pierce, who have always supported me and encouraged me to pursue my goal. They have also instilled me the work ethic needed to attain those goals.

Finally, and most importantly, I thank Marla Gomez, for her unparalleled love and support, both emotionally and financially. I thank her for tolerating my work schedule, stressful moods and for reading countless drafts of this dissertation and suggesting helpful revisions.

Abstract

Hydrologic and sedimentation processes are critical in determining floodplain site conditions and the distribution of bottomland hardwood (BLH) forest communities (Hodges 1997). Channelization of streams associated with BLH wetlands has occurred extensively throughout the southeastern United States, altering the hydrologic and sedimentation processes that sustain these systems. In western Tennessee, channelization and past land-use practices have resulted in drastic geomorphic and hydrologic changes, including excessive sand deposition on floodplains, and in extreme cases, the formation of valley plugs and shoals.

Our understanding of the processes associated with valley plugs and shoals, their rates and variability, and their impacts on BLH forest succession is limited but required for conservation and restoration efforts to be successful. The objectives of this study were: (1) to quantify the deposition rates and determine the temporal and spatial patterns of overbank sedimentation associated with different geomorphic features including valley plugs, shoals, and unchannelized systems (natural meandering channels), (2) to determine differences in surface and sub-surface hydrology associated with the three geomorphic features, (3) to experimentally determine the effects of hydroperiod and deposition rate and texture on germination and early growth of three BLH tree species, and (4) to quantify differences in floodplain forest communities as a result of altered

hydrologic and sedimentation processes associated with the three geomorphic features.

This study was conducted in the Hatchie River watershed, located in western Tennessee, from 2001 to 2005. The Hatchie River is the longest unchannelized stretch of river in the Lower Mississippi Alluvial Valley; however, extensive channelization of the Hatchie River tributaries has occurred. Channelization, the geology of the region, and past land-use practices have resulted in the formation of valley plugs and shoals within many of the altered tributaries (Diehl 2000).

Field studies conducted at three unchannelized sites, two shoal sites, and four valley plug sites indicated that overbank sedimentation was dramatically influenced by geomorphic features. At valley plug sites, deposition rates ($\bar{x}$ = 5.46 ± 0.44 cm/yr) were 10 times greater than at unchannelized ($\bar{x}$ = 0.46 ± 0.05 cm/yr) and shoal sites ($\bar{x}$ = 0.57 ± 0.24 cm/yr). At valley plug sites, sediment deposition contained significantly more coarse sands than at shoal and unchannelized sites and a larger extent of the floodplain was affected by high deposition rates. Sedimentation rates at both valley plug and shoal sites were variable because of other factors such as channel recovery processes and anthropogenic disturbances. Dendrogeomorphic analysis indicated that there had been a significant increase in deposition rates at valley plug sites since 1970, corresponding to the time period of channelization of most western Tennessee streams (Hupp and Bazemore 1993).

Field studies conducted at three unchannelized sites, two shoal sites, and three valley plug sites also indicated that both surface and sub-surface hydrology were affected by channelization and subsequent formation of valley plugs and shoals. Contrary to previous research, surface flooding at valley plug sites was less than at unchannelized and shoal sites. This result demonstrated the variability in hydrologic responses to valley plug formation. Water tables were also lower at valley plug and shoal sites, possibly as a result of channel bed lowering during channelization. However, even though water tables were lower at valley plug sites, root systems of trees at these sites were inundated for extended periods of time ($\bar{x}$ = 32.75 ± 12.76 days) during the growing season.

A greenhouse experiment was conducted to determine the effects of hydroperiod and sedimentation rates and textures on germination and growth of red maple (*Acer rubrum*), swamp chestnut oak (*Quercus michauxii*), and overcup oak (*Q. lyrata*). Poor germination of red maple prevented reliable testing of its germination or growth response to the treatments. Germination of swamp chestnut oak and overcup oak were most affected by hydroperiod, with sediment rate and texture being a secondary factor. The most important effect of the 8 cm deep sediment was reduced overall height, which may reduce a seedling's ability to compete for resources.

Floodplain forest species composition and structure and the environmental factors important in structuring plant communities were investigated at three unchannelized sites, two shoal sites, and three valley plug sites. Hydrologic and sedimentation conditions associated with channelized streams and valley plug

formation were the main processes influencing site conditions, including soil characteristics, resulting in extensive changes to the floodplain forest communities. At valley plug sites, typical BLH forest associations of oak species (*Quercus* spp.) and baldcypress/water tupelo (*Taxodium distichum/Nyssa aquatica*) are being replaced by several disturbance-tolerant species including boxelder (*Acer negundo*), black willow (*Salix nigra*), and red maple (*Acer rubrum*). This study, however, has demonstrated that there is considerable temporal and spatial variability in hydrologic and sedimentation processes associated with valley plugs. Thus, there is an interrelated temporal and spatial variability in forest response. The lack of predictability of abiotic processes associated with valley plugs makes the future composition of these forests uncertain.

Table of Contents

PART I: INTRODUCTION.....1

Introduction.....2

Western Tennessee.....5

Problem Identification.....9

Study Area – Hatchie River Watershed.....18

Appendix 1.....22

PART II: OVERBANK SEDIMENTATION DYNAMICS ALONG ALTERED AND UNALTERED TRIBUTARIES OF THE HATCHIE RIVER.....26

Introduction.....27

Methods.....32

Results.....40

Discussion.....48

Conclusion.....60

Appendix 2.....65

PART III: HYDROLOGICAL RESPONSES TO CHANNELIZATION AND THE FORMATION OF VALLEY PLUGS AND SHOALS.....96

Introduction.....97

Methods.....102

Results.....106

Discussion.....112

Conclusion.....119

Appendix 3.....121

PART IV: THE EFFECTS OF FLOODING AND SEDIMENTATION ON SEED GERMINATION OF THREE BOTTOMLAND HARDWOOD TREE SPECIES..146

Introduction.....147

Methods.....150

Results.....153

Discussion.....156

Conclusion.....160

Appendix 4.....163

PART V: THE EFFECTS OF ALTERED HYDROLOGIC AND SEDIMENTATION PROCESSES ON BOTTOMLAND HARDWOOD FOREST COMMUNITIES.....169

Introduction.....170

Methods.....177

Results.....181

Discussion.....187

Conclusion.....194

Appendix 5.....196

PART VI: CONCLUSIONS.....216

Conclusions.....217

Restoration.....222

Needs.....225

LITERATURE CITED.....229

VITA.....244

List of Tables

Table Page

1-1. Study sites with identification of site type, tributary, the upstream basin size at the study site and total basin size of the tributary.....23

2-1. Study tributaries with identification of channel type and feature studied (Diehl 2000).....66

2-2. Variogram models, direction of maximum spatial continuity, relative nugget effect and index of spatial dependence of sediment deposition at valley plug (Hickory, Bear, and Jeffers) and shoal sites (Piney and Porters).....67

3-1. Study tributaries with identification of geomorphic feature studied (Diehl 2000), number of groundwater wells per site, and sampling years.....122

5-1. Study tributaries with identification of geomorphic feature studied (Diehl 2000).....197

5-2. Importance values (relative density + relative dominance) and frequency of overstory species occurring at unchannelized sites.....198

5-3. Importance values (relative density + relative dominance) and frequency of overstory species occurring at shoal sites.....199

5-4. Importance values (relative density + relative dominance) and frequency of overstory species occurring at valley plug sites.....200

5-5. Relative density (number of individuals of species x/ total number of individuals of all species) of the most abundant seedlings at valley plug, shoal, and unchannelized sites.....201

5-6. Results of indicator species analysis.....202

5-7. Percentage of plots that were grouped into each of the eight forest communities by site type.....203

5-8. Correlations of environmental variables with ordination axes. Variables with the highest correlations are listed in bold text.....204

5-9. Mean (1 s.e.) of important environmental variables grouped by species associations.....205

List of Figures

Figure Page

1-1. Illustration of the ecological processes of a tributary system that contains a valley plug formation.....24

1-2. Map of Hatchie River Watershed and location of study streams (EPA 2005).....25

2-1. Sampling design for short-term and long-term sedimentation of (a) unchannelized sites, (b) valley plug sites, and (c) shoal sites.....68

2-2. Example of a variogram.....71

2-3. Mean sedimentation rates (+1 standard error) for all sites by type (a) unchannelized sites, (b) channelized sites, (c) shoal sites, and (d) valley plug sites.....72

2-4. Mean sedimentation rates (+1 standard error) for valley plug, shoal, channelized, and unchannelized sites based on sediment pad measurements from 2002-2004.....76

2-5. Percentage of plots with different soil textures for valley plug, shoal, channelized, and unchannelized sites.....77

2-6. Mean sedimentation rates (+1 standard error) for (a) unchannelized sites and (b) valley plug sites, for 2003 and 2004.....78

2-7. Mean sedimentation rates (+1 standard error) for (a) unchannelized sites and (b) valley plug sites, by distance from the channel (m).....80

2-8. Mean sedimentation rates (+1 standard error) for (a) shoal sites and (b) valley plug sites, by longitudinal location.....82

2-9. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+1 standard error) for valley plug and unchannelized sites.....84

2-10. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+ 1 standard error) by tree-age group for trees sampled at (a) unchannelized sites and (b) valley plug sites.....85

2-11. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+1 standard error) for valley plug sites by distance from the channel.....87

2-12. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+1 standard error) for valley plug sites by longitudinal distance and direction relative to the valley plug.....88

2-13. Examples of (a) an omindirectional variogram of the Hickory Creek site and (b) a variogram adjusted for anisotropy of the Bear Creek site.....89

2-14. Prediction map, produced through kriging, of sediment deposition occurring at the Hickory Creek valley plug site.....90

2-15. Prediction map, produced through kriging, of sediment deposition occurring at the Bear Creek valley plug site.....91

2-16. Prediction map, produced through kriging, of sediment deposition occurring at the Jeffers Creek valley plug site.....92

2-17. Prediction map, produced through kriging, of sediment deposition occurring at the Piney Creek shoal site.....93

2-18. Prediction map, produced through kriging, of sediment deposition occurring at the Porters Creek shoal site.....94

2-19. Mean monthly precipitation for short-term deposition sampling periods from 2002 to 2004.....95

3-1. Sampling design for study plots used to measure sub-surface hydrology at (a) unchannelized sites, (b) valley plug sites, and (c) shoal sites.....123

3-2. Floodplain inundation graphs for the valley plug sites from 2002 to 2004: (a) Bear Creek, (b) Hickory Creek, and (c) Jeffers Creek.....126

3-3. Floodplain inundation graphs at the shoal site floodplains from 2003 to 2004: (a) Piney Creek and (b) Porters Creek.....127

3-4. Floodplain inundation graphs for the unchannelized sites from 2002 to 2004: (a) Spring Creek –GVL, (b) Lower Spring Creek, and (c) Spring Creek – Sain.....128

3-5. Mean flood depth (cm) (+ 1 standard error) and mean maximum flood depth (cm) (+ 1 standard error) by sites of the same type from 2002 to 2004: (a) valley plug sites, (b) shoal sites, and (c) unchannelized sites.....130

3-6. Mean flood depth (cm) (+ 1 standard error) and mean maximum flood depth (cm) (+ 1 standard error) by site type from 2002 to 2004.....133

3-7. Mean total number of days flooded per year (+1 standard error) and mean duration of each flood event (days) (+1 standard error) by site type.....134

3-8. Mean flood depth (cm) (+1 standard error) and mean maximum flood depth (cm) (+1 standard error) during the growing season (April to October) by site type.....135

3-9. Mean total number of days flooded per year (+1 standard error) and mean duration of each flood event (days) (+1 standard error) during the growing season (April to October) by site type.....136

3-10. Mean water table depths (m) at valley plug sites in 2003 and 2004: (a) Hickory Creek and (b) Jeffers Creek.....137

3-11. Mean water table depths (m) at shoal sites in 2003 and 2004: (a) Piney Creek and (b) Porters Creek.....138

3-12. Mean water table depths (m) at unchannelized sites in 2003 and 2004: (a) Spring Creek – GVL and (b) Lower Spring Creek.....139

3-13. Mean depth to water table (m) (+1 standard error) by site for each site type for 2003 and 2004.....140

3-14. Mean standard deviation of water table depth (m) (+1 standard error) by site for each site type for 2003 and 2004.....141

3-15. Examples of the cross-sectional channel profiles at (a) unchannelized site (Spring Creek-GVL), (b) a shoal site (Porters Creek), and (c) a valley plug site (Hickory Creek) in 2003.....142

3-16. Mean channel profile depth (m) (+1 standard error) by site type for 2003 and 2004.....145

4-1. Percent germination of overcup oak among flood and sediment treatment combinations after 10-weeks.....164

4-2. Swamp chestnut oak mean percent germination (+ 1 standard error) by flood treatment after 10-weeks.....165

4-3. Swamp chestnut oak mean percent germination (+1 standard error) by sediment treatment after 10-weeks.....166

4-4. Overcup oak seedling mean above-ground height (cm) and standard error by treatment combination.....167

4-5. Swamp chestnut oak seedling mean above-ground height (cm) and standard error by treatment combination.....168

5-1. Sampling design for vegetation and soil at a) unchannelized sites, b) shoal sites, c) valley plug sites.....206

5-2. Mean number of stems per hectare (+1 standard error) at valley plug, shoal, and unchannelized sites.....209

5-3. Mean basal area (m^2 per ha) (+1 standard error) at valley plug, shoal, and unchannelized sites.....210

5-4. Dendrogram of the cluster analysis and indicator species associations.....211

5-5. Non-metric multidimensional scaling ordination diagram for all plots – Axis 1 vs. Axis 2.....212

5-6. Mean (+1 standard error) of the percentage of sand, silt, and clay at valley plug, shoal, and unchannelized sites.....213

5-7. Mean (+1 standard error) of phosphorus, calcium, potassium, and magnesium found in the soils of valley plug, shoal, and unchannelized sites...214

5-8. Mean (+1 standard error) of the percent organic matter and cation exchange found in the soils of valley plug, shoal, and unchannelized sites.....215

PART I

INTRODUCTION

Introduction

Bottomland hardwoods (BLH) occur in floodplains of rivers in the southeastern United States including regions of the Piedmont, Gulf Coastal Plain, and Lower Mississippi River Alluvial Valley (LMAV). BLH forests are biologically diverse and remarkably productive ecosystems that are adapted to fluctuating water levels (Odum 1969), defined and sustained by a natural hydrology of alternating dry and wet periods (Wharton et al. 1982).

In BLH systems, the primary driving process responsible for the existence, productivity, and interactions of the major biota is periodic overbank flooding, also known as the flood pulse (Junk et al. 1989). The interrelated process of overbank sedimentation is also essential in maintaining biologically diverse and productive systems. The fertility of the floodplain soils depends on nutrient inputs from the main channel and the quality of deposited sediments from overbank flooding (Wharton et al. 1982, Junk et al. 1989, Stanturf and Schoenholtz 1998).

BLH wetlands provide numerous valuable functions to both society and nature, including water quality enhancement, flood control, erosion control, timber production, and wildlife habitat. However, since the 1700's the wetlands in the United States have been destroyed and degraded (Hefner and Brown 1985). Human alteration of wetland hydrology, including dams, impoundments, regulated water flow, wetland drainage, habitat fragmentation and groundwater extraction, has reduced wetlands in the United States by 50% (Pringle 2000). Palustrine vegetated wetlands, including BLH, have experienced the greatest net loss of all wetland types (Hefner and Brown 1985, Wilen and Frayer 1990).

The LMAV originally contained over 10 million ha of BLH, which has been reduced to a highly degraded 2.8 million ha (Hefner and Brown 1985). Nearly all BLH wetland loss has been attributed to clearing, draining, and other hydrologic alterations for agriculture (Wilen and Frayer 1990). Throughout the southeastern Coastal Plain, channelization has been a common approach to reduce flooding, mainly for agricultural purposes (Shankman 1993). In upstream reaches of a system, channelization causes the floodplain to be disconnected from the fluvial system, while causing lower reaches to experience increased peak flood stage and flood frequency (Shankman and Pugh 1992). Channelization also increases stream power that facilitates sediment transport, which can increase deposition rates in lower stream reaches (Happ et al. 1940, Shankman and Samson 1991). Channelization and dredging of stream channels has also been shown to reduce the water table levels in the floodplain (Tucci and Hileman 1992).

The altered hydrologic and sedimentation processes, as a result of channelization, have degraded much of the remaining BLH habitat. Examples of BLH degradation as a result of channelization include: altered surface and sub-surface hydrology (Shankman and Pugh 1992, Tucci and Hileman 1992), high sedimentation rates (Happ et al. 1940), reduced lateral channel migration that creates sloughs and oxbow lakes (Shankman 1993), loss of aquatic habitat (Hohensinner et al. 2004), reduced growth and premature mortality of BLH tree species (USDA 1986), loss of plant species diversity (Miller 1990), loss of economically valuable timber (Wells 2004), and changes in plant species composition (Oswalt and King *In press*) among others. Loss and degradation of

BLH wetlands have had negative effects on wildlife communities including Neotropical migratory songbirds (Hunter et al. 1993), waterfowl (Heitmeyer and Fredrickson 1981), and fish (Risotto and Turner 1985, Hoover and Kilgore 1997).

Degradation of BLH systems caused by channelization has been exacerbated in the loess belt region of the LMAV, which includes portions of western Tennessee and northern Mississippi (Saucier 1994). The geology of the region and past land-use practices have resulted in extreme rates of gully erosion in the uplands areas of this region. Increased transport capacity of channelized streams has facilitated the transport of large quantities of eroded sediment into lower reaches of altered systems. Degradation, head-cutting, and bank failure of channelized reaches has also contributed to greater sediment loads. These processes have led to the formation of valley plugs and shoals throughout many of the altered systems in western Tennessee (Diehl 2000) and northern Mississippi (Happ et al. 1940). Valley plugs are within-channel geomorphic features that completely block the channel with accumulating sediment, and shoals are within-channel geomorphic features, at the confluence of two streams, that are accumulating sediment causing a decrease in channel depth but not a complete blockage of the channel.

The Hatchie River, in western Tennessee, remains the longest unchannelized stretch of river in the LMAV. However, it has not escaped the problems of channelization and valley plug formation, as most of its tributaries were channelized. To protect the unaltered main-stem of the Hatchie River and the extensive BLH habitat and wildlife that it supports, conservation efforts are

focused on the restoration of its tributaries. There are few data available, however, on the influence of valley plugs and shoals on fluvial-geomorphic processes that are critical for structuring and sustaining BLH systems. This study was conducted, in part, to aid restoration and conservation efforts by providing a better understanding of the effects of valley plugs and shoals on sedimentation and hydrologic processes and the resulting impacts on BLH forests. The results of this study will be useful in determining what floodplain processes need to be restored and may provide base-line data for evaluation of restoration success.

The remainder of this introduction provides detailed background information on the geology, land-use, and channelization projects of western Tennessee; it also provides detailed information on how the above factors have contributed to the development of valley plugs and shoals and subsequent degradation of BLH forests. The introduction concludes with a list of the study objectives and description of the study sites. Other chapters in this dissertation are presented in manuscript format, with each chapter addressing one of the four main objectives of this study. The final chapter is a discussion of the major conclusions from all chapters and their significance for management and restoration of BLH forests.

Western Tennessee

Geology

The western Tennessee region is bordered by the Tennessee River on the east and the Mississippi River on the west. This region encompasses both the

Gulf Coastal Plain and the Lower Mississippi River Alluvial Valley (LMAV) (Saucier 1994). The parent material of this region is unconsolidated coarse sand deposited mainly during the Quaternary period (Saucier 1994). There is no bedrock control in this area for the base level of streams flowing into the Mississippi River. Thus, streams freely adjust their profiles to recover from disturbances such as dredging and channelization (Simon 1994).

The unconsolidated alluvial sands of the western Tennessee region are mostly covered by a thin layer of windblown loess deposits (silt and clay). These deposits range in depth from 1 m to 30 m and are cut through by most of the tributary systems of this region (Saucier 1994). A few tributaries of the Upper Hatchie River Watershed originate on block clay deposits known as Porters Creek Clay of the Midway Group, deposited during the Tertiary age (Miller et al. 1966). However, the main river systems of this region, including the Obion River, the Forked Deer, and Hatchie River, cut through the alluvial sands deposited during the Quaternary period (Saucier 1994).

Both the loess and alluvial sand deposits are highly erodible. Most of the erosion in western Tennessee is thought to occur in the loess-capped uplands (Saucier 1994). This erosion consists of both the loess cap and the alluvial sands beneath. The past 150 years of erosion in the uplands of western Tennessee exceeds the erosion that occurred in the last several thousand or tens of thousands of years (Saucier 1994).

Land-Use

The western Tennessee region was rapidly colonized in the early 1800s. Forested areas in the upland regions were quickly cleared for timber and replaced with agriculture fields of corn, cotton and tobacco (Wilder 1998). Before deforestation occurred, the rivers within the region were described as flowing at "constant good depths" (Ashley 1910). However, clearing of the upland areas resulted in erosion and gullying of the loess and sandy soils (Simon 1994). Before settlement, sediment deposition rates in western Tennessee ranged from 0.02 to 0.09 cm/year; post-settlement rates increased to 3 cm/year (Wolfe and Diehl 1993). The increased erosion after settlement resulted in deposition in the floodplains and stream channels, causing a decrease in channel flood capacity. Streams were stifled with sediment and debris, leading to frequent and prolonged flooding in the bottomlands (Mogan and McCrory 1910). Stream channel alteration was proposed to alleviate the flooding problems (Hidinger and Morgan 1912).

By 1926, several streams in western Tennessee had been channelized, resulting in over 132 km of stream alteration (Speer et al. 1965, Simon and Robbins 1987). The Hatchie River main stem was the exception to the widespread channelization. Most of the tributaries of the Hatchie River were channelized by the 1970s, although the exact dates of channelization are unknown (Simon and Hupp 1992, Simon 1994). Clearing and snagging of altered systems was necessary due to sediment aggradation and debris

accumulation. Altered systems required continued maintenance from the 1930s through the 1950s (Simon 1994).

By 1970, the sediment build-up within the tributary floodplains was so great that the U.S. Department of Agriculture (1970) initiated a channelization project called the West Tennessee Tributaries Project (WTTP). Although only 35% of the project was completed, 128 km of channel alteration occurred because of the project (Robbins and Simon 1983). Channelization in western Tennessee resulted in streams being shortened by 44%, lowered by 170%, and steepened by 600% (Simon and Hupp 1992). The Obion-Forked Deer River system had suffered considerable alterations to its hydrology (Simon 1994); impacts to the Hatchie River were restricted to its tributary system. A total of 33 of the 36 major tributaries to the Hatchie River were channelized (USDA 1986, U.S. Fish and Wildlife Service 2001).

Channelization projects in western Tennessee lowered the bed level in stretches of the channels by as much as 5 m (Simon and Hupp 1987, Simon 1994). Transition slopes were constructed to offset differences in bed elevations at the junction of the modified and natural channel reaches (Robbins and Simon 1983). Transition slopes were steeper than both modified and natural channel reaches and produced headcutting and degradation of upstream reaches. This degradation moved upstream at a rate of 2.6 km/year on the South Fork Forked Deer River and resulted in approximately 2.6 m of incision from 1966 to 1967 (Simon 1994). This degree of degradation occurred in all streams of western Tennessee that experienced bed level lowering. Downstream reaches

accumulated material eroded from upstream reaches. Deposition rates on downstream reaches of the Obion River ranged between 0.03 to 0.12 m/year, while the South Fork Forked Deer River filled in with 2.2 m of sediment over a 12-year period (Simon 1994). Deposition events of 0.61 m were also observed in the Reelfoot lake area (Shelford 1954).

By 1971, the completed channelization projects had directly and indirectly reduced the BLH habitat along the affected reaches by 60% (Barstow 1971). Systems not directly impacted by the channelization projects experienced losses of BLH forest due to clearing of the floodplain and construction of drainage ditches by individual landowners (Barstow 1971). During this period of drastic hydrologic alteration, BLH forests in western Tennessee were reduced from 404,000 ha in 1940 to 291,00 ha in 1970 (Turner et al. 1981).

Problem Identification

The rivers in western Tennessee historically supported large tracts of BLH forests along rivers in the Upper East Gulf Coastal Plain and the LMAV. However, the geology of the region, channelization, and past land-use practices have directly (i.e. timber harvests) and indirectly (i.e. alteration of hydrologic and sedimentation processes) reduced and degraded the BLH habitat in western Tennessee (Barstow 1971). Past land use practices, driven by agricultural objectives, have led to erosion of the thin loess cap of the region and have exposed and eroded the coarse alluvial sands that lie beneath the loess cap, resulting in massive gully erosion. Historically, the high meandering rates and low gradients of the rivers did not allow for transport of the sand. However,

channelization of all rivers, except the Hatchie River, greatly increased their stream power and has led to dramatic geomorphic changes (Diehl 2000, Oswalt 2003). Although the Hatchie River has not been channelized, 92% of its tributaries were channelized. As a result, an estimated 580 million kilograms of sediment is accumulating in the Hatchie River every year (USDA Soil Conservation Service 1986).

Channelization has contributed to the problem in two main ways. First, it alters the channel morphology by shortening, straightening and increasing the slope of the channel. As previously discussed (Robbins and Simon 1983, Simon and Hupp 1987, Simon 1994) such alterations cause a degradation of the stream channel leading to channel erosion. Several studies on sediment dynamics in western Tennessee suggest that channelization leads to bed-level lowering and stream degradation (Hupp and Simon 1986, Darby and Simon 1999, Diehl 2000). It is unclear at this point the degree to which both gully and channel erosion are contributing sediment to the systems.

The second major impact of channelization on these systems is the facilitation of sediment transport. The channel alterations produced by channelization combine to increase stream velocity and stream power (Gilvear and Bravard 1996). The increased stream power facilitates sediment transport downstream through suspended fine sediments and coarse bedload movement while also increasing channel bank and bed erosion (Happ et al. 1940).

Bedload transport is a result of the stream power or transport capacity of a given stream flow (Knighton 1998). Bedload transport entails the movement of

grains in groups referred to as a "wavelike movement of coarse particles" (Meade 1985). The increase in stream power, as a result of channelization, enables bedload transport to increase. Bedload transport is temporally intermittent at a cross-section due to the time it takes for one wave of sediment to pass the section and the arrival time of the next wave (Gomez 1991). The transport is also spatially variable across the channel's cross-section. The thalweg, which is the location of highest velocity in the channel, is usually associated with the highest bedload discharge, and its location can vary during flood events with fluctuating river stages.

Bedload deposition takes place when stream power decreases below the transport threshold. This typically occurs in altered systems at woody debris jams, where the steep slopes of tributaries with high sand bedloads meet with lower slopes of downstream sections, and at the confluence of tributaries and the main channel (Happ et al. 1940, Diehl 2000). Change in slope and debris jams reduce stream velocity; as a result, sediment deposits in the main stem of the channel. Excessive bedload deposition can result in the formation of valley plugs or shoals. Valley plugs are areas where the channel becomes completely filled with sediment, thus floodwater and sand bedload are forced out into the floodplain (Happ 1975). The roughness of the floodplain reduces the water velocity, causing additional deposition of sediment throughout the floodplain. This process spreads sediment throughout the floodplain as the stream braids out from the main channel, forming anastomosing streams throughout the floodplain (Happ et al. 1940, Diehl 2000). Shoals are points in the channel where

the depth decreases downstream due to bedload deposition; these features usually form at the confluence of tributaries and the main stem of the river (Diehl 2000).

The processes discussed above are consistent with the valley plugs and shoals located within the Hatchie River Watershed (Diehl 2000). Since the channelization of the tributaries, the main channel of the Hatchie River has become shallower and flooding has increased (USDA 1986). This change has been concentrated near the mouths of several tributaries (Diehl 2000). Increased flooding in these areas is thought to inhibit growth and increase mortality of BLH tree species. Unfortunately, few quantitative data exist on the types and rates of processes surrounding valley plugs and shoals or their impacts on BLH forests, but several hydrologic and geomorphic processes are expected to vary spatially around the geomorphic features (Figure 1-1).

Gully and channel erosion may be increased in the headwater areas due to geology, land use, and channel alterations. Erosion in these areas cause instability of channel banks and incised channels. These areas are considered the main contributors to sediment into the Hatchie River watershed and other watersheds in the region (Saucier 1994).

Channel deposition will occur where flow velocity decreases, such as where the channel gradient declines or where debris jams occur. This will cause bedload deposition or channel filling, thus reducing the flood volume capacity of the stream. Aggradation in the channels can create areas within the channel that are actually higher than the surrounding floodplain (Happ et al. 1940). Channel

deposition causes increased overbank flooding, resulting in increased sediment deposition in the floodplain (Happ et al. 1940). The water table in the floodplain is expected to increase because of the connectivity of the stream channel and the water table (Happ et al. 1940). The permeability of sand deposits that occur in both the channel and floodplains allows for a rapid response of groundwater levels to fluctuations in river stage (Brinson 1990). Tucci and Hileman (1992) showed a lowering of the water table in the floodplain as a result of channel dredging; thus, an aggrading channel would be expected to raise the water table.

Valley plugs and shoals form as a result of excessive bedload deposition (Happ et al. 1940, Diehl 2000). Channel filling, sand splays, and vertical accretion are all associated with valley plugs in much greater quantities than "natural" floodplains (Happ et al. 1940). Impacts from the processes associated with valley plugs include frequent overbank flooding due to the reduced channel capacity, excessive sand deposition in the floodplain, accelerated formation of natural levees due to deposition, excessive flooding and ponding of timber and an increase in the water table elevation (Happ et al. 1940). Immediate downstream sections of the stream are abandoned as a result of the valley plug. Stream flow is diverted into the floodplain by valley plugs, leaving an abandoned channel that may experience little periodic flooding and a lower water table. Lowered water table has been found to significantly decrease the growth of several BLH tree species (Reily and Johnson 1982). The above processes and impacts will vary spatially across the floodplain due to variables that influence these processes (climate, geomorphology, soils, land use and vegetation).

Shoal sites may be impacted in similar ways as valley plug sites, but to a lesser extent. Floodplains adjacent to shoals may experience frequent overbank flooding and a rise in the water table due to channel filling effects. Because the stream channel at shoal sites is not completely filled with sediment, less sediment deposition in the floodplain would be expected. However, shoal sites may also be subject to excessive sedimentation in restricted areas of low elevation that are subject to high flow velocities during overbank flooding. These high velocity flows can transport sediment into the floodplain, causing crevasse splays.

A clear understanding of the spatial variability of the processes associated with excessive sedimentation is necessary to gain insight on how these processes affect BLH forest succession. Presently, there is a lack of information on these processes, their rates and variability, and their implications for BLH forest succession. However, broad-scale predictions can be made on the influences that processes associated with excessive sedimentation may have on BLH forest succession.

Areas experiencing excessive deposition are buried by infertile sand, which covers the productive silt-clay deposits. If the sand deposits reach a depth of 15 cm, there is a significant decrease in the productive potential of the affected area (Happ et al. 1940). Excessive sedimentation could reduce the germination potential of typical BLH tree species by burying seed sources in infertile sand. Burial of established trees may also cause stress and mortality to mature BLH tree species due to a lack of available nutrients and other effects of burial.

Effects of excessive sedimentation on BLH tree species germination potential, growth, and survival have not been addressed in the literature. However, burial of several freshwater lowland plant species to a depth of 5 cm, 10 cm, and 15 cm produced an average reduction in shoot density ranging from 10% to 56% (van der Valk et al. 1983). Rate and texture of sediment deposition may be important factors in determining the response of the vegetation (van der Valk et al. 1983).

In addition to increasing sediment deposition, valley plugs and shoals can also influence hydrology. Increased overbank flooding events, as a result of channel deposition and reduced drainage capacity of the floodplain, will increase the frequency, depth and duration of flooding. Surface hydrologic effects associated with excessive sedimentation may have negative impacts on BLH tree species regeneration, growth, and survival.

BLH tree species exhibit different germination responses to prolonged flooding (Hosner 1957). Germination tolerance to flooding has been studied for only a few BLH tree species. Flooding seeds of red maple, silver maple, sycamore, elm, cottonwood, and willow for 32 days had little effect on their germination potential (Hosner 1957). However, establishment of BLH tree species may be affected by phenology of germination and hydrology (Leck et al. 1989). Spatial and temporal variation in tree species composition has been found to be influenced mainly by the abilities of species to reproduce rapidly during periods of low stress and to germinate under the influence of stress (Streng et al. 1989).

Swamping or ponding in the floodplain as a result of increased overbank flooding and reduced drainage capacity of the floodplain has caused total mortality of the original timber stands; allowing growth of only willow and "other relatively worthless types of vegetation" (Happ et al. 1940). There are substantial differences in the tolerance levels of BLH tree species to flooding. Frequent flooding, high water, and/or long periods of inundation places a "selective killing effect on reproduction and thereby affects the makeup of individual stands" (Hosner 1960). Hosner (1960) showed that relative tolerances to flooding varied among selected BLH tree species, with most tolerant to least tolerant as the following: silver maple, buttonbush, boxelder, black willow, cottonwood, green ash, American elm, pin oak (*Q. palustris*), sycamore, red maple, shumard oak (*Q. shumardii*), redgum *(L. styraciflua*), hackberry, and cherrybark oak.

Tree recruitment and survival has also been identified as being controlled mainly by floods during the growing season (Johnson 2000). Flooding regimes have been correlated with regeneration, mortality, and stress of overcup oak (King 1995). Hydrologic factors have also been found to be determinants of BLH tree species growth and survival (Harms et al. 1980, Hosner and Boyce 1962, Keeland and Sharitz 1997).

A rise in the water table within the floodplain also has direct implications on BLH forest succession. Aggradation of stream channels can cause such dramatic increases in the water table that it can even be above some floodplain surfaces (Happ et al. 1940). Not only does the rise in water table contribute to increased swamping and ponding of the floodplain, but it can also inundate root

systems of BLH tree species throughout the growing season. This process has the same effect on BLH tree species as flooding but without the surface water.

Overbank flooding and sedimentation are normal process in wetland ecosystems that provide several benefits including replenished nutrients, fertile soil, and water recharging. However, human interactions have accelerated this process resulting in negative impacts on functional processes of wetland ecosystems (Happ et al. 1940). Understanding how human alterations of hydrologic and geomorphic processes have affected successional processes of BLH forests is critical for restoration and management of forests and the wildlife it supports. The rapid response of geomorphic and ecological attributes to valley plugs and shoals provides a unique opportunity to study such relationships that are usually long-term in their development. This study will better our understanding of how human activity influences these processes. This information is necessary for the proper utilization of restoration approaches and management techniques needed to sustain healthy functioning BLH forests. The objectives of this study were to:

1) quantify the deposition rates and determine the temporal and spatial patterns of overbank sedimentation associated with valley plugs, shoals, channelized streams, and unchannelized streams;
2) determine differences in floodplain surface and subsurface hydrology associated with the valley plugs, shoals, and unchannelized streams;

3) determine the effects of hydroperiod and sediment rate and texture on germination and early growth of three BLH tree species; and
4) determine the vegetation communities associated with valley plugs, shoals, and unchannelized streams and determine the environmental factors important in structuring those communities.

Study Area - Hatchie River Watershed

The Hatchie River watershed is located in southwestern Tennessee and northern Mississippi (Figure 1-2). The Hatchie River originates from the Upper East Gulf Coastal Plain Coercion in Mississippi flowing northwest into Tennessee and the LMAV until draining into the Mississippi River north of Memphis. The Hatchie River drainage area is approximately 177 km long and averages 39 km wide to include over 673,654 ha, 72% of which are located in Tennessee (USDA 1986).

The Tennessee portion of the Hatchie River watershed includes 55,848 ha of BLH habitat, which includes 16% of the total BLH habitat in Tennessee (Schweitzer 2000 a,b, and c). The Hatchie River watershed is one of the most biologically diverse systems within the Upper East Gulf Coastal Plain Ecoregion including over 100 species of fish, 35 species of mussels, 250 species of birds, 50 species of mammals, and a vast number of amphibians and reptiles (U.S. Fish and Wildlife Service 2001). The Hatchie River has also been designated

one of 13 State Scenic Rivers and one of 75 "Last Great Places" by The Nature Conservancy.

Flooding in the Hatchie River watershed typically occurs from late fall through spring and is mainly dependent upon local rainfall events and water levels in the Mississippi River. Peak discharge usually occurs during January, February, and March, with flows averaging over 133 m^3/sec (USGS 2003). Highest recorded discharge flow for the Hatchie River occurred in 1973 with a flow of 1745 m^3/sec at a stage height of 6.6 m (USGS 2003).

Study Reach

The study reach is the stretch of the Hatchie River that encompasses all tributaries investigated in this study. The Hatchie River study reach is located in Haywood, Madison, and Hardeman counties in Tennessee, stretching from the Hatchie River National Wildlife Refuge in Brownsville south to Hickory Valley (Figure 1-2). Study sites were located along seven tributaries within the defined reach of the Hatchie River (Figure 1-2). The tributaries consisted of one unchannelized stream and six channelized streams (Table 1-1).

This project included four types of study sites: unchannelized sites, valley plug sites, shoal sites, and channelized sites. Three unchannelized sites were located along Spring Creek, starting at its confluence with the Hatchie River (Lower Spring Creek site), and spaced a minimum distance of 2 km between sites. Spring Creek is an unchannelized, meandering tributary of the Hatchie River that contains extensive BLH forests. It is one of only three unaltered major tributaries in the Hatchie River basin (USDA 1986). There has been relatively

little disturbance at the Spring Creek-Sain site at least during the past 50 years, however, bottomland forests of the Lower Spring Creek site and the Spring Creek-GVL have experienced some high-grading.

Valley plug sites have been identified on several tributaries of the Hatchie River including four tributaries used in this study: Bear Creek, Jeffers Creek, Hickory Creek, and Clover Creek (Tim Diehl, personal communication). All four tributaries were channelized before 1970 (Hupp and Bazemore 1993); my study sites were located adjacent to the valley plug formations located along those streams. It should be noted that approximately two years before this study was initiated, the stretch of channel adjacent to my study site at Jeffers Creek was dredged to remove the previous valley plug. However, within two years of dredging, the channel filled back in substantially with sediment. One year into my study, a bridge upstream of the Jeffers Creek study site was rebuilt. This construction also disturbed the channel system, removing sediment that had accumulated within the channel. These channel disturbances should be considered when drawing conclusions specifically associated with this site. According to landowners, none of the valley plugs sites have experienced extensive logging for at least the past 50 years.

Two shoal sites were included in this study: Porters Creek and Piney Creek. These streams were also channelized before 1970 (Hupp and Bazemore 1993) and each contains a shoal at its confluence to the Hatchie River (Tim Diehl, personal communication). Shoal study sites were located in the floodplain adjacent to the shoal formations at each stream. The Porters Creek shoal site

has not been logged for at least the past 50 years, however, the Piney Creek site has been high-graded periodically during the last 40-years.

Two channelized sites that did not contain either a valley plug or a shoal were located adjacent to the channelized streams of Jeffers Creek and Clover Creek. Both channelized sites were located at least 3 km downstream of the valley plugs located on both streams and at least 1 km upstream of the confluence of the tributaries and the Hatchie River. Neither the Jeffers Creek or Clover Creek channelized sites have experienced timber harvests during the past 50 years.

APPENDIX 1

Table 1-1. Study sites with identification of site type, tributary, the upstream basin size at the study site and total basin size of the tributary.

Site	Site Type	Tributary	Upstream Basin (Km^2)	Total Basin Size (Km^2)
Bear	Valley Plug	Bear Creek	97	105
Hickory	Valley Plug	Hickory Creek	42	47
Jeffers	Valley Plug	Jeffers Creek	78	89
Clover	Valley Plug	Clover Creek	108	269
Piney	Shoal	Piney Creek	152	152
Porters	Shoal	Porters Creek	162	162
Lower Spring	Unchannelized	Spring Creek	294	294
Spring-GVL	Unchannelized	Spring Creek	176	294
Spring-Sain	Unchannelized	Spring Creek	272	294
Clover	Channelized	Clover Creek	259	269
Jeffers	Channelized	Jeffers Creek	86	89

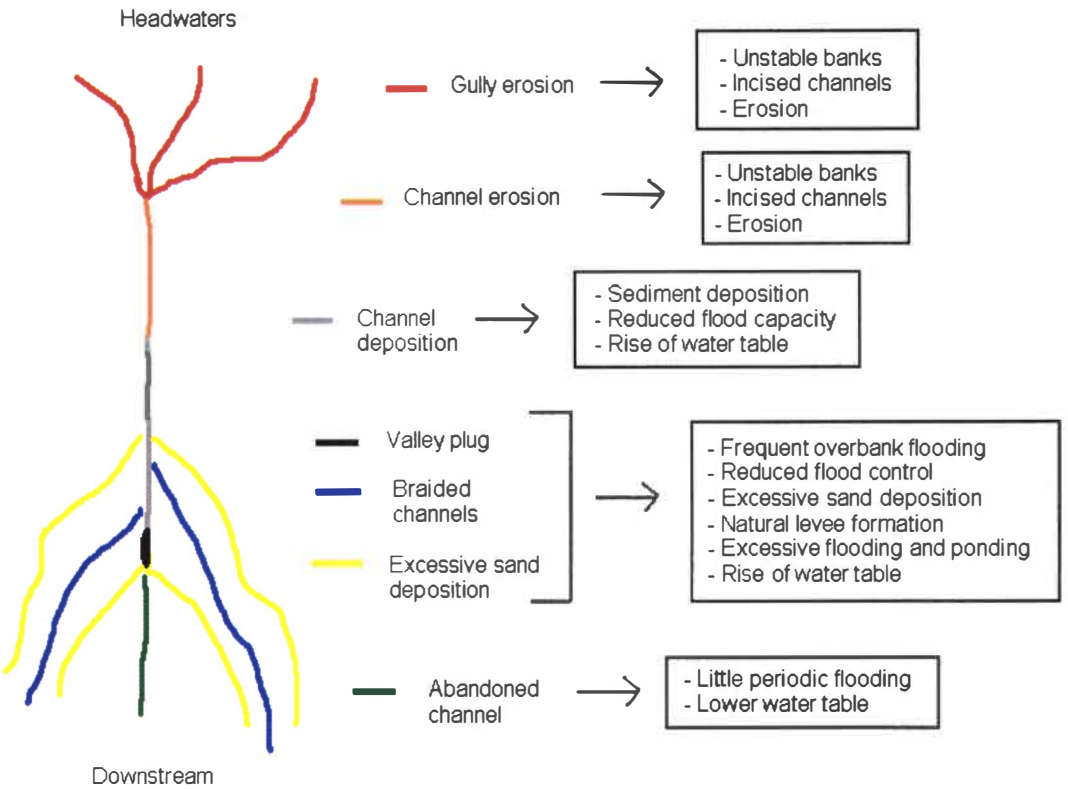
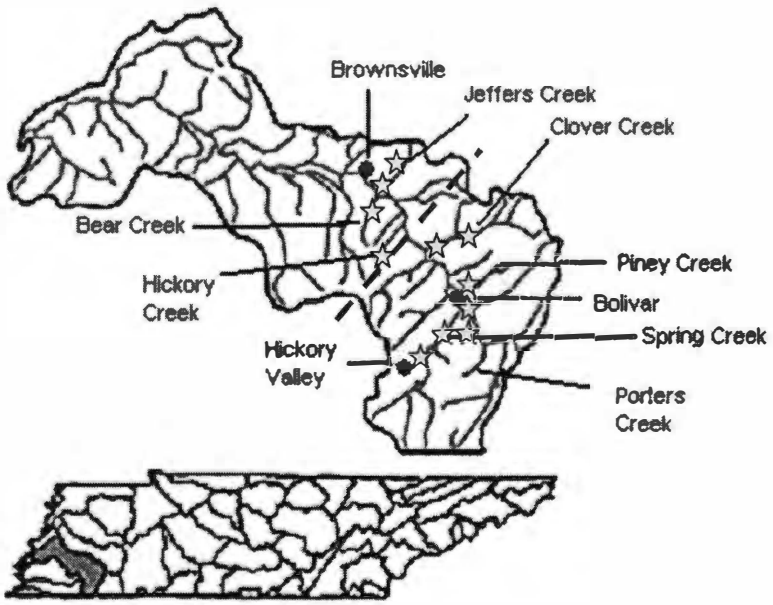


Figure 1-1. Illustration of the ecological processes of a tributary system that contains a valley plug formation.



— — - Approximate dividing line between the LMAV to the west and the Gulf Coastal Plain to the east.

☆ Approximate locations of study sites.

Figure 1-2. Map of Hatchie River Watershed and location of study streams (EPA 2005).

PART II

OVERBANK SEDIMENTATION DYNAMICS ALONG ALTERED AND UNALTERED TRIBUTARIES OF THE HATCHIE RIVER

Introduction

One of the most commonly acknowledged functions and benefits of wetlands, including bottomland hardwood (BLH) forests, is sediment retention (Boto and Patrick 1979, Kleiss 1996, Hupp 2000, Mitsch and Gosselink 2000). Local variation in the quantity of stored sediment is due to variations in the balance between erosion and sedimentation, position within the fluvial system, climatic variations in hydrologic processes, and land use (Steiger et al. 2003). Therefore, depositional areas and the processes associated with them vary both spatially and temporally.

In most alluvial floodplain systems, overbank deposition is the primary process of floodplain development (Walling and He 1998). Floodplain deposition occurs during overbank flooding events when water flows and sediment are forced outward into the floodplain (Knighton 1998). Floodplain deposition can be highly variable and depends on the frequency and magnitude of floods, distance to the channel, sediment load, sediment texture, water velocity, floodplain morphology, and vegetation cover (Wharton et al. 1982, Knighton 1998). Highest rates of deposition typically occur near the channel because water flows into the floodplain are slowed by the roughness of the floodplain (Happ et al. 1940, Johnston et al. 1984, Hodges 1997). Depressions within the floodplain may also receive high rates of deposition because they typically have longer periods of inundation, which allow fine sediments to filter out of the water column (Hupp and Morris 1990, Kleiss 1996).

Past studies in the southeastern United States have shown deposition rates in floodplains to average less than 1 cm/yr (Hupp and Morris 1990, McIntyre and Naney 1991, Hupp and Brazemore 1993, Kleiss 1996, Heimann and Roell 2000). Long-term floodplain deposition has been estimated to range between 0.3 m and 0.6 m in 200 to 400 years (Wolman and Leopold 1957). Human actions, mainly channelization and land use, can have broad and systematic influences on sedimentation processes in some areas, potentially resulting in negative impacts on functional processes of wetland ecosystems (Simon 1989). Examples of such impacts include degraded aquatic habitats; reduced flood storage capacity of the stream; increased water table levels in the floodplain; accelerated development of natural levees; increased flooding and ponding of water in the floodplain, which affects the survival, growth and regeneration of BLH tree species; burial of fertile soils with infertile sand and gravel; and increased lateral erosion (Happ et al. 1940).

The effects of channelization on sediment dynamics have not been quantified or investigated thoroughly. Dendrogeomorphic techniques have been used to estimate long-term historic deposition rates and to relate changes in deposition rates to land-use changes and channel disturbances (Hupp and Morris 1990, Hupp and Bazemore 1993, Kleiss 1996, Heimann and Roell 2000). Hupp and Bazemore (1993) found that deposition rates were lower along the channelized Big Sandy River compared to the unchannelized Hatchie River. They attributed the lower deposition rates to the reduced hydroperiod that typically occurs in floodplains adjacent to channelized streams. However, some

evidence suggests that deposition rates may increase as a result of channelization. Deposition rates may be higher along channels that have greater lateral stability (Knighton 1998), which can result from channelization (Shankman 1993). As Hupp and Bazemore (1993) suggested, channelization has resulted in a reduced hydroperiod in the upper reaches of a system, but the peak stage of floods and flood frequency has been found to increase in the downstream reaches of channelized systems (Shankman and Pugh 1992). Thus, the influence of channelization on overbank sedimentation may depend on the location of the site within the system and the connectivity of the stream to the floodplain at the site, which may also be related to other channel evolution aspects (Schumm et al. 1984, Simon and Hupp 1987, Hupp and Simon 1991).

In western Tennessee river systems, channelization can also result in the formation of valley plugs and shoals (Happ et al. 1940, Diehl 2000). Valley plugs are areas where the channel becomes completely filled with sediment, forcing floodwater and sand bedload out into the floodplain (Happ 1975). This process spreads sediment throughout the floodplain as the stream braids out from the main channel forming anastomosing streams throughout the floodplain (Happ et al. 1940, Diehl 2000). Channel filling, sand splays, and vertical accretion are all thought to be associated with valley plugs in much greater quantities than unaltered systems (Happ et al. 1940). However, the rates and variability of these accretion processes associated with valley plugs have not been investigated or quantified.

Shoals are points in the channel where the depth decreases downstream due to bedload deposition; these features usually form at the confluence of tributaries and the main stem of the river (Diehl 2000). Shoal sites may be impacted in similar ways as valley plug sites, but even less is known about the sedimentation processes associated with shoals than valley plugs. Because the stream channel at shoal sites is not completely blocked with sediment, less sediment deposition in the floodplain at shoal sites may be expected. However, some areas corresponding to overflow channels and crevasse splays may be experiencing high deposition rates. During high flow events, water may be forced into the floodplain with high velocity flows occurring in isolated areas of the floodplain where the difference in elevation between the channel and floodplain is lowest. The high velocity flows carry stream bedload into isolated areas of the floodplain, where the flows slow and deposit large amounts of sediment, causing a crevasse splay effect (Knighton 1998).

The processes discussed above are consistent with the valley plugs and shoals located within the Hatchie River Watershed (Diehl 2000). Since the channelization of its tributaries, the main channel of the Hatchie River has become shallower and flooding has increased (USDA 1986). The increased flooding is thought to inhibit growth and increase mortality of BLH tree species. But, there is a paucity of data available on the effects of sedimentation on BLH tree species although sedimentation has been implicated in BLH forest succession (Hodges 1997).

The first objective of this study was to determine differences in short-term sediment deposition rates and textures in BLH floodplains adjacent to four different features: unchannelized channels, channelized channels, shoals, and valley plugs. I expected that deposition at valley plug sites would be the greatest, based on our current understanding of valley plugs (Happ et al. 1940), mainly their ability to force stream flow and bedload into the floodplain. I expected that deposition at shoal sites might be similar to that at channelized sites, but probably elevated because of the occurrence of crevasse splays. I anticipated that the texture of deposition would also differ, with coarser sands being deposited mostly at valley plug sites and finer sediments of silt and clay being deposited at the other sites.

The second objective was to determine differences in within-site spatial dynamics of deposition rates at unchannelized, channelized, shoal, and valley plug sites. Specifically, I determined whether deposition rates differed with increasing distance from the channel. I also investigated the effect of longitudinal distance from valley plugs and shoals on deposition rates. Based on our current understanding, I predicted that deposition rates at valley plugs and upstream of valley plugs would be greater because it is thought that valley plugs expand upstream with each flooding event (Happ et al. 1940).

The third objective was to determine temporal differences in long-term deposition rates at valley plug and unchannelized sites. I anticipated that increases in deposition rate would be detected and would correspond to channel disturbances. Hupp and Bazemore (1993) found a dramatic increase in

deposition rates along the Hatchie River around 1960 that corresponded to the time when many streams in West Tennessee were channelized.

The final objective was to use geospatial statistics to determine the spatial continuity of deposition rates at valley plug and shoal sites. If deposition was occurring as classical fluvial geomorphology would predict, then there should be a high degree of spatial correlation and the direction of spatial dependence should be in the direction of the stream flow. However, the valley plug obstruction may be forcing enough floodwater and sediment into the floodplain to change the direction of spatial dependence. Excessive deposition occurring in isolated areas, such as crevasse splays, may also reduce the degree of spatial dependence by breaking up the spatial continuity of deposition over the entire floodplain.

Methods

Study Reach

The study reach is the stretch of the Hatchie River that encompasses all tributaries investigated in this study. The Hatchie River study reach is located in Haywood, Madison, and Hardeman Counties in Tennessee, stretching south from the Hatchie River National Wildlife Refuge in Brownsville to Hickory Valley (Figure 1-2). Study sites are located along seven tributaries of the Hatchie River. The tributaries consist of one unchannelized stream and six channelized streams (Table 2-1).

This study focused mainly on three types of study sites: unchannelized sites, valley plug sites, and shoal sites. However, two channelized sites (Jeffers

Creek and Clover Creek (Table 2-1) that did not contain a valley plug or shoal were also included to separate the effects of channnelization and the formation of valley plugs and shoals on deposition rates. Three unchannelized sites were located along Spring Creek at a minimum distance of 2 km between each site. Spring Creek is a natural meandering tributary of the Hatchie River and contains extensive BLH forests. It is one of only three unaltered major tributaries in the Hatchie River basin (USDA 1986), and the only unchannelized tributary within the study reach. Valley plug sites have been identified on several tributaries of the Hatchie River, including four tributaries chosen for this study: Bear Creek, Jeffers Creek, Hickory Creek, and Clover Creek. Two shoal sites were also included in this study: Porters Creek and Piney Creek. These streams each contain a shoal at their confluence to the Hatchie River.

Short-Term Deposition

Sediment deposition rates were measured in the spring and fall of each year from 2002 to 2004 at two unchannelized sites (Spring GVL, and Lower Spring), two valley plug sites (Hickory Creek and Jeffers Creek), and two channelized sites (Jeffers Creek and Clover Creek) (Table 2-1). Sediment deposition rates were measured from 2003 to 2004 at additional sites, including one unchannelized site (Spring-Sain), two valley plug sites (Bear Creek and Clover Creek), and both shoal sites (Piney Creek and Porters Creek) (Table 2-1). At channelized and unchannelized sites, I measured deposition at plots spaced 100 m apart along transects perpendicular to the stream channel and spaced 200 m apart (Figure 2-1a). At valley plug sites, I measured deposition at plots

spaced 50 m apart along transects perpendicular to the stream channel and spaced 50 m apart (Figure 2-1b). Transects were centered at the head of the valley plug and placed every 50 m in both directions (upstream and downstream) for a distance of 200 m. Both sides of the tributaries were sampled at unchannelized, channelized, and valley plug sites. At shoal sites, I measured deposition on only one side of the streams because of a lack of landowner permission. Deposition measurements were conducted at plots spaced 50 m apart along transects perpendicular to the channel and spaced 50 m apart (Figure 2-1c). Transects started at the confluence of the tributaries and the Hatchie River and extended 300 m upstream. At all sites, the length of each transect depended on floodplain morphology and landowner permission. A total of 188 plots at valley plug sites, 57 plots at shoal sites, 19 plots at channelized sites, and 28 plots at unchannelized sites were sampled.

Short-term deposition rates at all sites were estimated by two different approaches, depending on the situation. First, sedimentation pads made of white feldspar clay (Baumann et al. 1984) were placed at sampling plots that, based on field observations, experience little deposition. Pads were 2-3 cm thick and approximately 1 m in diameter and were marked with a PVC pipe on the north side of the pad. In addition, I marked a nearby tree and measured the direction and distance from the tree to the pad center to facilitate pad relocation (Hupp and Bazemore 1993, Hupp 2000).

At locations experiencing high deposition rates, I used PVC pipe to determine deposition rates. A 1.5 m section of PVC pipe was driven into the

ground and I recorded the distance from the top of the pipe to the ground surface. This distance was re-measured when sediment pads were checked to determine the amount of deposition. These two methods were used to measure annual sediment deposition rates.

Texture

I used the texture-by-feel analysis to evaluate sediment deposition texture in the field (Thein 1979) from 2002 to 2004. Sediment texture was estimated at 188 valley plug plots, 57 shoal plots, 19 channelized plots, and 28 unchannelized plots. Textures were classified as silt, sand/silt, or sand. These measurements were laboratory checked by obtaining sediment samples next to each sediment pad or pole and submitting them to A&L Laboratories (Memphis, TN) for texture analysis.

Long-Term Deposition

The dendrogeomorphic technique was used to estimate long-term historical deposition rates (Hupp and Morris 1990, Bazemore et al. 1991, Hupp and Bazemore 1993). Sites sampled with this technique were limited because of lack of landowner permission; sites included all four valley plug sites and one unchannelized site along Spring Creek (Table 2-1). An additional unchannelized site, located along the Hatchie River on the Hatchie River National Wildlife Refuge, was also used for this portion of the study. At each site, at least two trees of different ages and canopy positions were sampled at plots spaced 100 m apart (Figure 2-1). Some plots could not be sampled because all trees located

within the plot were dead. A total of 32 plots were sampled at unchannelized sites and 116 plots were sampled at valley plug sites.

The dendrogeomorphic technique involved measuring the distance from a tree's original lateral roots to the current soil surface and dividing the depth by the tree age. Depth of sediment to the lateral roots was used because the lateral roots are a marker of the original ground surface at time of germination (Hupp and Morris 1990). Trees were cored within 0.5 m of the ground surface to determine their age. Tree cores were transported back to Dr. Henri Grissino-Mayer's tree-ring laboratory at the University of Tennessee, where I dried, mounted, sanded, and aged the tree cores. A total of 295 trees were sampled from both unchannelized and valley plug sites.

Analysis

To validate grouping sites of the same type together, I conducted ANOVA or t-tests, depending on the number of sites, to determine differences in short-term deposition rates among sites of the same type. Site-level analysis of short-term sediment deposition included ANOVA tests to determine differences in sedimentation rates, determined from sediment pad measurements, among all site types (valley plug, shoal, channelized, and unchannelized). I examined temporal differences in sedimentation by using a t-test to compare deposition rates of site types from 2003 and 2004. ANOVA tests were also used to determine differences among site types in deposition rates by distance from the channel. Differences in sediment texture were tested using randomization tests.

ANOVA tests were also used to determine within-site differences in short-term deposition rates, determined from sediment pads at valley plug and shoal sites. Comparisons were made among three valley plug transect groups based on longitudinal distance and direction of transects to valley plugs: transects at the valley plug, transects above the valley plug, and transects below the valley plug. A similar analysis was performed for shoal sites but grouped by transects of increasing longitudinal distance from the confluence to the Hatchie River.

A t-test was used to determine differences in mean long-term sedimentation rates, determined with the dendrogeomorphic technique, between unchannelized and valley plug sites. I used ANOVA to test for differences in long-term sedimentation rates by distance from tributaries. Long-term sedimentation rates were separated by tree age into six age classes for unchannelized sites and seven age classes for valley plug sites. Mean long-term sedimentation rates were then tested by site type with ANOVA tests to determine differences among age classes.

Kruskal-Wallis tests were used in cases where ANOVA assumptions were not valid, and Tukey-Kramer multiple comparison tests were used to distinguish differences among groups (Alpha = 0.05) (Sokal and Rohlf 1995). Statistical analyses were conducted with SAS Version 9.1 (SAS Institute Inc. 2004) and NCSS (Hintze 2001).

Geostatistical Analysis

To determine spatial patterns of sedimentation rates, I performed a geostatistical analysis on short-term deposition rates at valley plug and shoal

sites using the Geostatistical Analyst in ArcGIS Version 9.0 (ESRI 2004). I first performed an exploratory spatial data analysis to verify assumptions of the geostatistical analysis (Rossi et al. 1992). Data sets were examined for normal distributions and local stationarity (Isaaks and Srivastava 1989). Local stationarity was tested using Voronoi diagrams (Aurenhammer 1991) constructed on each data set using the Geostatistical Analyst in ArcGIS. Non-normal data were corrected using a normal score transformation that ranks the data and matches the ranks to ranked values of a normal distribution.

Semivariograms or variograms were used to determine the average degree of similarity between sample plots separated by a given distance (h) in space (Le Corre et al. 1998). The variogram is estimated by the semivariance, $\gamma(h)$, as follows:

$$\hat{\gamma}(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [z(x_i) - z(x_i - h)]^2,$$

where $N(h)$ is the number of pairs of locations separated by distance h and z is the measure of interest at location x_i (Rossi et al. 1992). Variograms typically reach a plateau where pairs of locations are no longer spatially correlated (Isaaks and Srivastava 1989; Figure 2-2). The height of the plateau is known as the "sill" and the distance it takes the variogram to reach the plateau is called the "range". Typically, variograms do not intercept the semivariance axis at zero; this discontinuity at the origin is referred to as the "nugget effect". The "nugget effect"

is caused by either sampling error or spatial variability at scales too small to measure (Isaaks and Srivastava 1989, Rossi et al. 1992).

First, omnidirectional variograms (no directionality) were calculated using the Geostatistical Analyst in ArcGIS. Variograms were calculated using distance classes (Barbujani 1988), such that the number of distance classes (lags) multiplied by the size class (lag size) is less than one-half the maximum distance between pairs of sample plots (Le Corre et al. 1998). Number of lags and lag size were adjusted for each study site to validate the distance class rule.

Several models (spherical, exponential, etc.) were used to calculate variograms and selection of the appropriate model was determined by the model fit. Cross-validation on the residuals of the predicted values was used to determine model fit (Isaaks and Srivastava 1989). Unbiased residuals have a standardized mean near zero and a root-mean-squared standardized error near one.

Variograms constructed with the appropriate model were examined for spatial dependence, or an increase in variogram value as distance increased, and for presence of a sill, indicating that spatial patterns differ beyond a certain distance (Isaaks and Srivastava 1989, Rossi et al. 1992, Le Corre et al. 1998). The relative nugget effect (ratio of nugget effect to the sill) was calculated to determine the proportion of total variance that cannot be used for modeling due to sampling errors and spatial variability at small scales (Isaaks and Srivastava 1989, Rossi et al. 1992). I also calculated an index of spatial dependence that is

the inverse of the relative nugget effect or the ratio of the partial sill to the total sill (Gross et al. 1995).

Variograms were then adjusted for anisotropy to determine the maximum direction of spatial continuity. Anisotropy is when values of the variogram change more progressively in a particular direction, causing the variogram to reach the sill quicker (Isaaks and Srivastava 1989). The directional axis for maximum spatial continuity was based on a 0 - 360° scale.

I then used Kriging to produce prediction maps of sediment deposition over each entire study site. Kriging uses the measured values in conjunction with the variogram to predict values at unsampled locations (Le Corre et al. 1998). The predicted values are linear combinations of the observed values taken from a defined search neighborhood (Isaaks and Srivastava 1989). The search neighborhood was ellipsoid with a maximum of five observed values and a minimum of two observed values. If a transformation was needed to normalize the data, a back-transformation was used to return the values to the original scale before the prediction maps were projected.

Results

Short-Term Deposition

The short-term deposition rate analysis between sites of the same type (Figure 2-3) indicated that there was little variation among sites of the same type. There were no differences among the three unchannelized sites (N = 28, df = 2, F = 1.80, P = 0.189) (Figure 2-3a), between the two channelized sites (N = 19, T = 0.712, P = 0.486) (Figure 2-3b), between the two shoal sites (N = 57, T =

1.944, P = 0.057) (Figure 2-3c), or among the four valley plug sites (N = 188, df = 3, F = 1.15, P = 0.329) (Figure 2-3d). The results of these analyses justified the grouping of sites by type for the remainder of the analysis.

The site-level analysis of short-term sedimentation rates estimated from sediment pads revealed that sedimentation rates from were greater at valley plug sites ($\bar{x}$ = 5.90 ± 0.49 cm/yr) than shoal ($\bar{x}$ = 1.26 ± 0.64 cm/yr), channelized sites ($\bar{x}$ = 0.61 ± 0.14 cm/yr), and unchannelized sites ($\bar{x}$ = 0.40 ± 0.09 cm/yr) (N = 292, df =3, F = 16.42, P < 0.001) (Figure 2-4). The largest depth of sediment deposited in one year occurred at the Hickory Creek valley plug site in 2003, measuring 79.5 cm. Valley plug sites had 37 deposition measurements over 10 cm. The largest deposition measured at a shoal site was 32 cm in 2004 at the Piney Creek shoal site. Both shoal sites only had two deposition measurements over 10 cm. The largest deposition measured at a channelized site was 2.28 cm and the Lower Spring site (1.65 cm) had the largest deposition of all unchannelized sites.

Randomization tests determined differences in the proportion of plots that experienced deposition of each texture classification by site type (Figure 2-5). A greater proportion of plots at valley plug sites contained sand and silt/sand deposits than at the shoal sites (N = 245, T2 = 5.634, P < 0.001), channelized sites (N = 207, T2 = 3.633, P < 0.001), and at the unchannelized sites (N = 216, T2 = 5.586, P < 0.001). However, there was no difference in the texture of sediment deposited at the shoal and channelized sites (N = 76, T2 = 0.216, P = 0.861) or unchannelized sites (N = 85, T2 = 1.708, P = 0.129). Nor was there a

difference in sediment texture between channelized and unchannelized sites (N = 47, T2 = 1.595, P = 0.225).

To investigate temporal differences in short-term sedimentation rates at the site level, I compared mean deposition rates of valley plug and unchannelized sites by sampling year, 2003 vs. 2004. Shoal sites could not be used for this analysis because they were only sampled in one year. Deposition rates at the unchannelized sites did not differ from 2003 ($\bar{x}$ = 0.52 ± 0.07 cm/yr) to 2004 ($\bar{x}$ = 0.45 ± 0.08 cm/yr) (N = 28, T = 0.69, P = 0.49) (Figure 2-6a). However, deposition rates were greater at valley plug sites in 2003 ($\bar{x}$ = 11.19 ± 2.21 cm/yr) than in 2004 ($\bar{x}$ = 4.98 ± 0.38 cm/yr) (N = 188, T = 2.77, P = 0.007 (Figure 2-6b).

The effect of distance from the tributaries on mean short-term deposition rates was investigated by site type. There was no difference in the deposition rates by distance from the channel at the shoal sites (N = 57, df = 5, F = 0.54, P = 0.74) or channelized sites (N = 19, df = 3, F = 1.27, P = 0.33). Deposition rates at unchannelized sites (N = 27, df = 2, F = 6.03, P = 0.004) (Figure 2-7a) and valley plug sites (N = 184, df = 4, F = 4.91, P < 0.001) (Figure 2-7b), however, did differ by distance from the channel. At unchannelized sites, deposition rates at the channel or 0 m ($\bar{x}$ = 0.55 ± 0.07 cm/yr) and 100 m ($\bar{x}$ = 0.66 ± 0.14 cm/yr) were greater than at 200 m ($\bar{x}$ = 0.04 ± 0.01 cm/yr). At valley plug sites, the highest deposition rates were found at the channel edge or 0 m ($\bar{x}$ = 6.29 ± 0.89 cm/yr) and 50 m from the channel ($\bar{x}$ = 8.93 ± 1.27 cm/yr).

A within-site analysis of shoal and valley plug sites was conducted by comparing the short-term deposition rates of grouped transects based on their longitudinal distance and direction to a shoal or valley plug. There were no within-site differences of the shoal site transect groupings (N = 57, df = 2, F = 1.33, P = 0.27) (Figure 2-8a). The within-site analysis did show differences in deposition rates for valley plug sites where location was based on proximity to a valley plug (N = 188, df = 2, F = 6.97, P = 0.001)(Figure 2-8b). Transects located at valley plugs ($\bar{x}$ = 8.33 ± 1.17 cm/yr) and upstream of valley plugs ($\bar{x}$ = 6.44 ± 0.72 cm/yr) had greater deposition rates than transects downstream of a valley plug ($\bar{x}$ = 3.08 ± 0.51 cm/yr).

Long-Term Deposition

At unchannelized sites, trees sampled with the dendrogeomorphic technique ranged in age from 19 to 153 years with a mean age of 65.54 ± 3.53 years. At valley plug sites, trees sampled ranged in age from 8 to 99 years with a mean age of 31.01 ± 0.95 years. The range in depth to lateral roots at the unchannelized sites was from 5 cm to 51 cm and the range for valley plug sites was 10 cm to 126 cm. Mean long-term sedimentation rates determined from the dendrogeomorphic method differed between the unchannelized ($\bar{x}$ = 0.31 ± 0.02 cm/yr) and valley plug sites ($\bar{x}$ = 1.65 ± 0.08 cm/yr) (N = 295, T = 16.69, P < 0.001) (Figure 2-9).

Mean long-term sedimentation rates appeared to be similar for all trees at unchannelized sites regardless of the germination date. However, there were

some differences in mean long-term deposition rates at unchannelized sites among the age groups (N = 63, df = 5, F = 8.97, P < 0.001)(Figure 2-10a). Deposition rates in the youngest four age groups (20 to 59 years) were not different from each other and ranged in mean deposition from 0.28 ± 0.05 cm/yr to 0.51 ± 0.07 cm/yr. Only the deposition rates for the 60 to 69 year age grouping ($\bar{x}$ = 0.29 ± 0.04 cm/yr) and 70 years old or older ($\bar{x}$ = 0.19 ± 0.02 cm/yr) differed from those of the 20 to 29 year age grouping.

Long-term deposition rates measured from trees at all valley plug sites showed significant increases by tree age groupings (N = 232, df = 6, F = 14.99, P < 0.001) (Figure 2-10b) since 1970. The two youngest age groupings, 0 to 9 years ($\bar{x}$ = 3.53 ± 0.93 cm/yr) and 10 to 19 years ($\bar{x}$ = 2.47 ± 0.19 cm/yr), had the highest deposition rates and differed from all other age groupings. Similar results of long-term deposition rates by tree age groupings were found when valley plug sites were analyzed separately. At the Bear Creek valley plug site, long-term deposition rates differed among the age groupings (N = 69, df = 6, F = 14.14, P < 0.001), with the two youngest age groupings, 0 to 9 years ($\bar{x}$ = 3.79 ± 1.54 cm/yr) and 10 to 19 years ($\bar{x}$ = 2.92 ± 0.28 cm/yr), having greater deposition rates from all other age groupings, except the 20 to 29 year age group ($\bar{x}$ = 2.02 ± 0.17 cm/yr). Long-term deposition rates at the Clover Creek valley plug site also differed by tree age groupings (N = 18, df = 3, F = 5.75, P = 0.009). The youngest age grouping, 10 to 19 years ($\bar{x}$ = 1.57 ± 0.29 cm/yr) had greater deposition rates than all other groups, which were all less than 0.73 cm/yr. At the

Hickory Creek valley plug site, long-term deposition rates also differed among tree age groupings (N = 80, df = 4, F = 9.57, P < 0.001), with the two youngest age groupings, 10 to 19 years ($\bar{x}$ = 2.42 ± 0.32 cm/yr) and 20 to 29 years ($\bar{x}$ = 1.92 ± 0.26 cm/yr), having greater deposition rates from all other age groupings, except the 30 to 39 year age group ($\bar{x}$ = 1.29 ± 0.30 cm/yr). In contrast to all other valley plug sites, the Jeffers Creek site did not differ in long-term deposition rates among the tree age groupings (N = 64, df = 4, F = 1.24, P = 0.30). The range of long-term deposition rates at the Jeffers Creek valley plug site was from 1.64 ± 0.17 cm/yr (20 to 29 years) to 1.23 ± 0.09 cm/yr (40 to 49 years).

Within-site differences of long-term deposition rates at both unchannelized and valley plug sites were evaluated by comparing deposition rates by distance from the channel. There was no difference in mean long-term deposition rates by distance from the channel at the unchannelized sites (N = 63, df = 7, F = 1.17, P = 0.34). However, long-term deposition rates at valley plug sites (N = 214, df = 2, F = 11.01, P < 0.001) (Figure 2-11) were greater along the stream channel or 0 m ($\bar{x}$ = 2.06 ± 0.15 cm/yr) than at both 100 m ($\bar{x}$ = 1.48 ± 0.09 cm/yr) and 200 m ($\bar{x}$ = 1.14 ± 0.15 cm/yr). Within-site differences at valley plug sites were also investigated by grouping transects based on longitudinal distance and direction from valley plugs, the same groupings that I had used to analyze the short-term deposition, and comparing deposition rates among locations. Unlike the short-term deposition analysis, there was no difference in long-term deposition rates

among the three valley plug locations (N = 232, df = 2, X^2 = 5.25, P = 0.07) (Figure 2-12).

Geostatistical Analysis

For all study sites, the best-fitting variogram models produced standardized mean residuals ranging from 0.007 to 0.147 and root-mean-squared standardized errors from 0.913 to 1.404, indicating a good model fit. All variograms showed an increase in value with increased distance and presence of a sill, demonstrating spatial dependence (Figure 2-13a). Anisotropy was evident among variograms of each study site (Figure 2-13b), but the direction of spatial dependence varied among sites. The index of spatial dependence varied among study sites from 23.9% to 100% (Table 2-2). All study sites had a major range near the maximum of distance classes examined.

The exponential model used for the variogram of Hickory Creek had the best model fit, with a standardized mean residual of 0.007 and a root-mean-squared standardized error of 1.06. The major range of dependence was 275.57 m and the direction of spatial continuity was 284.2° (Table 2-2). The variogram calculated for the Hickory Creek valley plug site had the highest index of spatial dependence at 100%. The prediction map produced through kriging (Figure 2-14) illustrates that most of the study site is experiencing deposition rates exceeding 1.50 cm/yr and that the direction of spatial dependence was perpendicular to stream flow. The projection map includes approximately 240,391 m^2 of the Hickory Creek floodplain. Approximately 71% of the area

included in the projection map was receiving more than 1.52 cm/yr of sediment deposition, and 27% of the area was receiving more than 5.80 cm/yr.

A rational quadratic model was the best fit for variogram construction of the Bear Creek valley plug site. The major range of spatial dependence was 278.57 m, with a direction of spatial dependence at 6.6°. The index of spatial dependence was lower at the Bear Creek site than the other two valley plug sites at 55% (Table 2-2). Figure 2-15 shows the prediction map produced through kriging and illustrates that the direction of spatial continuity was at a 30° angle to the stream flow of Bear Creek. The prediction map includes an area approximately 169,983 m^2 in size. Approximately 97% of this area was receiving more than 1.64 cm/yr of deposition, and 74% of the area was receiving more than 4.43 cm/yr.

As with Hickory Creek, the best model fit for the variogram of Jeffers Creek was an exponential model. The major range of dependence at Jeffers Creek was 348.21 m with a direction of dependence at 348.6° (Table 2-2). The index of spatial dependence was greater than the Bear Creek site but lower than Hickory Creek, at 66.1%. The prediction map (Figure 2-16) showed a distinct area of separation between areas of high and low deposition. Similar to Bear Creek, the direction of spatial dependence at Jeffers Creek was at a 40° angle to the stream flow. Approximately 94% of the 244,006 m^2 area included in the prediction map was subject to deposition greater than 1.14 cm/yr and 69% of the area was receiving more than 3.43 cm/yr.

A gaussian model was used to calculate the variogram of deposition at Piney Creek. The major range of spatial dependence was 208.93 m and the direction of spatial continuity was 25°. Piney Creek had the lowest index of spatial dependence at 23.9% due to the large relative nugget effect (Table 2-2). The prediction map produced through kriging (Figure 2-17) showed that the direction of spatial dependence was at a 55° angle to the stream flow of Piney Creek. The prediction map included approximately 100,744 m^2 of the Piney Creek floodplain. Approximately 33% of this area was receiving more than 2.21 cm/yr of sediment deposition.

A spherical model was the best fit for the variogram of Porters Creek. The major range was 208.93 m with a spatial dependence direction of 305.6° (Table 2-2). The index of spatial dependence for Porters Creek was high at 70.4%. The prediction map (Figure 2-18) showed that no areas experienced deposition rates exceeding 0.50 cm/yr and that the direction of spatial continuity was the same as the stream flow of Porters Creek. The Porters Creek prediction map area included approximately 168,000 m^2 and the entire area was receiving less than 0.5 cm/yr of deposition.

Discussion

Deposition rates, types of deposited sediment, and spatial patterns of sediment deposition have been strongly affected by channelization and the subsequent formation of valley plugs and shoals in the Hatchie River watershed. Several of the predictions I made about sedimentation processes at the four site types were supported by the results. The results also suggest, however, that

there is considerable variability in sedimentation responses to the formation of valley plugs and shoals.

Short-Term Deposition

Mean short-term deposition rates, measured at sediment pads from 2002 to 2004, at unchannelized, channelized, and shoal sites were within the range of deposition rates reported in previous studies (Mitsch et al. 1979, Johnston et al. 1984, Hupp and Morris 1990, McIntyre and Naney 1991, Hupp and Brazemore 1993, Kleiss 1996, Heimann and Roell 2000). Although the shoal, channelized, and unchannelized sites did not differ in short-term deposition rates, the results suggest that the presence of a shoal may be influencing sedimentation rates. For example, the Piney Creek shoal site experienced excessive sedimentation (two deposition measurements greater than 10 cm/yr) in certain distinct areas. These areas correspond to low areas in the spoil bank along the channel where crevasse splays have formed during flood events. Shoals may encourage the formation of crevasse splays by increasing within-channel deposition upstream of the shoal as a result of reduced water flows, slowed by the presence of the shoal at the confluence. Channel deposits would reduce the distance between the channel bed and low spots in the natural levee and increase the frequency of high velocity overbank flows that transport and deposit sediment in the floodplain.

The short-term deposition measured at valley plug sites was significantly greater than the short-term deposition at shoal, channelized, and unchannelized sites. The greater deposition rate at valley plug sites compared to channelized and shoal sites suggest that the greater deposition rates may be the result of the

valley plug, not just channelization of the stream. This result supports my prediction and suggests that formation of a valley plug within the stream channel forces floodwater and sediment into the floodplain and causes greater rates of sedimentation.

A confounding factor is, however, the location of the channelized sites downstream of the valley plug sites. It is reasonable to suggest that deposition rates at channelized sites were reduced by the upstream formation of valley plugs. However, the channelized sites were located at least 3 km downstream of the valley plug sites and water flows had returned to the channelized channels. Deposition rates at channelized sites also did not differ from shoals sites, which did not have a valley plug upstream, suggesting that the deposition rates measured at channelized and shoal sites of this study may accurately depict the deposition rates associated with channelized streams in the Hatchie River watershed. Although channelized streams typically carry heavier sediment loads than unchannelized streams (USDA 1970, Simon 1994, Hopkinson and Vallino 1995), lower rates of deposition have been associated with channelized streams because of the reduced lateral connectivity between the stream and the floodplain. Hupp and Bazemore (1993) reported higher deposition rates in floodplains of the unchannelized Hatchie River compared to the channelized Big Sandy River in western Tennessee. They attribute their results to the reduced lateral connectivity of the channelized system to the floodplain. My results suggest that the formation of valley plugs has reconnected the channelized stream to the floodplain and caused the greater deposition rates in the floodplain.

The texture analysis confirmed my predictions that the texture of the sediment being deposited consisted of more coarse sand than the deposited sediment at shoal, channelized, and unchannelized sites. Sediment being deposited at valley plug sites was composed almost equally of the three texture classifications (sand, silt/sand, and silt). The deposition at shoal, channelized, and unchannelized sites was composed mostly of silt. Sand is usually transported as bedload and would only be deposited in the floodplain under high velocity flows of overbank flooding. This condition is probably much more common in valley plug systems where the channel is completely blocked and forces all stream flow into the floodplain (Happ et al. 1940). At unchannelized, channelized, and shoal sites, the channel is not blocked and overbank flooding only occurs during high flood events that typically inundate the floodplain for long periods of time allowing for the deposition of fine sediments like silt and clay (Happ et al. 1940, Hodges 1997). Overbank flows at unchannelized, channelized, and shoal sites are probably not powerful enough to transport large amounts of coarse sediment into the floodplain.

The temporal analysis of short-term deposition rates indicated that deposition rates at unchannelized sites did not differ between years, but that deposition at valley plug sites was greater in 2003 than in 2004. This may be attributed to the 27% greater total precipitation that occurred in 2003 compared to 2004 (Figure 2-19) (NOAA 2005). Nevertheless, the greater deposition rates at valley plug sites in 2003 may also reflect the upstream expansion of valley plugs, where high rates of deposition would be impacting new areas that were

not included in my study. It is probable that greater rates of deposition occurring at valley plug sites cannot be sustained in the same areas for long time periods due to increases in elevation that would reduce flooding and thus sediment deposition. This result demonstrates the temporal variability of deposition rates that occur at valley plug sites, unlike the more stable unchannelized sites.

Within-site analysis of short-term deposition rates was first investigated by examining deposition rates by site type with respect to distance from the stream channel. At shoal and channelized sites, there was no difference in deposition by distance from the channel suggesting that deposition rates were uniform and relatively low across the floodplain. The lack of variation in deposition rates with increasing distance from the channel and the relatively low rates of deposition may be a result of the reduced lateral connectivity between the stream and the floodplain that typically occurs after channelization (Hupp and Bazemore 1993). At unchannelized sites, deposition was greatest at a distance of 0 m and 100 m from the channel, with deposition significantly decreasing at 200 m. This result is consistent with previous research (Happ et al. 1940, Johnston et al. 1984, Walling and He 1998) and my prediction that the greatest deposition rates would occur near the stream channel because as floodwaters leave the channel, flows are slowed by the roughness of the floodplain causing the deposition of coarse material (Happ et al. 1940, Wharton et al. 1982, Knighton 1998). Distance from the channel also affected short-term deposition rates at valley plug sites. Similar to the deposition at unchannelized sites, deposition at valley plug sites was greatest near the channel (0 m and 50 m). However, the deposition rates

occurring along the channel of valley plug sites was much greater than unchannelized sites, suggesting the water flows were of greater velocity and were transporting greater amounts of bedload into the floodplain.

The within-site analysis conducted by grouping transects showed no difference in short-term deposition rates by longitudinal distance from shoals. Thus, shoals do not seem to be progressing upstream at a significant rate. In contrast, short-term deposition rates at valley plug sites were greater at locations adjacent to and upstream of valley plug formations. This result is consistent with our prediction that sediment is forced into the floodplain by valley plugs causing excessive deposition in the floodplain (Happ et al. 1940). The analysis also indicates that the excessive sedimentation caused by valley plugs progresses upstream of the plug. In fact, the head of the valley plug at the Hickory Creek valley plug site expanded 80 m upstream during the period from November 2003 to March 2004. The rate of expansion upstream may depend on several factors including sediment sources to the systems, climatic variations, and geomorphic thresholds (Schumm 1977).

Long-Term Deposition

The overall mean long-term rate of sediment deposition at unchannelized sites was within the range of previous sedimentation studies in BLH forests (Delaune et al. 1978, Boto and Patrick 1979, Mitsch et al. 1979, Johnston et al. 1984, Hupp and Morris 1990, McIntyre and Naney 1991, Hupp and Bazemore 1993, Kleiss 1996, Heimann and Roell 2000). However, the overall mean long-

term deposition rate at valley plug sites is greater than any other previous reported long-term average.

Unlike short-term deposition, long-term deposition at unchannelized sites did not differ with increasing distance from the channel. This suggests that over long time-scales, the floodplain at unchannelized sites may be developing at a consistent rate over the entire floodplain. This may be a result of large magnitude floods that are discrete events in time (Schumm 1977, Sparks and Spink 1998), but have the greatest impact on floodplain surfaces through both erosion and deposition (Wharton et al. 1982, Hupp 2000). Conversely, both short-term and long-term deposition at valley plug sites differed among distances from the channel. The long-term deposition analysis at valley plugs supported the results from the short-term deposition analysis, indicating that greater deposition rates occur near the stream channel where there is the greatest reduction in flood velocity. The differences in spatial patterns of long-term deposition rates at unchannelized and valley plug sites may also be attributed to lateral channel migration. Floodplain deposition rates may be greater along channels that have greater lateral stability (Knighton 1998), which can result from channelization (Shankman 1993). The greater lateral mobility of natural meandering channels may result in the development of the entire floodplain at a consistent rate over long time scales.

Interestingly, the long-term deposition rates at valley plug sites did not differ by longitudinal distance and direction from valley plugs. In the short-term analysis, deposition along transects at the valley plug and upstream of the valley

plug were greater than downstream of the valley plug. In the case of short-term sedimentation, the valley plug actually protected at least part of the downstream floodplain from excessive sedimentation. High long-term deposition rates in the downstream floodplain indicate that it was previously not protected from excessive deposition by the current valley plug. The lack of difference in long-term deposition rates by location suggests that the valley plug formation has progressed upstream in recent years, providing the current protection of the downstream floodplain from greater rates of deposition.

There is little temporal variability in long-term deposition rates at unchannelized sites. The differences that were found (Figure 2-10a) are most likely a result of soil compaction. Over longer time spans, soil is subject to compaction that appears as a lower rate of accumulation (Hupp and Bazemore 1993, Kleiss 1996). Although temporal differences in deposition rates were found, the differences were small (largest difference of 3 mm) and may be accounted for by compaction. Temporal variability in long-term deposition rates was more apparent at the valley plug sites. Comparisons of long-term deposition rates by tree age groupings indicate a dramatic increase in deposition rates at valley plug sites approximately 30 years ago. Similar results were found at each valley plug, when analyzed separately, except for Jeffers Creek. Although long-term deposition rates at Jeffers Creek were greater than at unchannelized sites, there was no significant increase by tree age. This may be a result of the past disturbances at the site such as dredging of the channel and valley plug by the landowner. Such activity, which has occurred at the Jeffers Creek valley plug

site, would have reset the development of the valley plug and resulted in greater long-term deposition rates but no apparent increase through time. At the other valley plug sites, the increase in deposition rates may reflect an increase of agricultural activity within the basin or geomorphic readjustments of channels from channelization that occurred during this time period (Hupp and Bazemore 1993). Hupp and Bazemore (1993) also found a dramatic increase in deposition rates around 1960 at sites located along the main-stem of the Hatchie River.

Geostatistical Analysis

The index of spatial dependence was extremely high at the Hickory Creek valley plug study site indicating that there is little spatial heterogeneity in a 284.2° direction over a range of 275 m. The most surprising result for this geostatistical analysis is the direction of spatial continuity, which is perpendicular to the stream flow. Based on the within-site analysis and current understanding of sedimentation, one would expect the direction of spatial continuity to be in the direction of stream flow. The stream is the conduit for sediment transport, and as the previous results showed, most deposition occurs along the channel, thus the direction of spatial dependence of deposition rates should be in the direction of the stream flow. However, at the Hickory Creek valley plug site, it is clear that the valley plug formation has changed the sedimentation dynamics at the site. The valley plug is forcing sediment into the floodplain on both sides of the channel, resulting in a direction of spatial dependence that is perpendicular instead of parallel to stream flow. The contour maps produced through kriging also demonstrate that the excessive sedimentation resulting from valley plug

formation can impact a large portion of the floodplain, not just isolated areas near the plug.

The index of spatial variability at the Bear Creek valley plug site was high (55%) but less than the Hickory Creek valley plug site (Table 2-2). This result suggests that there is more spatial variability in deposition rates at the Bear Creek study site. The direction of spatial dependence is similar to the direction of stream flow and the contour map illustrates that areas along the stream channel are experiencing the most deposition (greater than 8 cm/yr). The differences in spatial patterns found at the Hickory Creek and Bear Creek sites may be a result of past disturbances at the Bear Creek site. In the early 1990s, a flood event buried 16 ha of the Bear Creek floodplain in 1.2 m of sand, including most of my study site (Marvin Nichols, USFWS, personal communication). The excessive deposition that occurred during this flood caused rapid geomorphic readjustment of the Bear Creek tributary. The resulting adjustments produced several anastomosing channels throughout the floodplain, including my study area. Instead of one main channel transporting water and sediment through the floodplain, there are currently at least three channels of equal size that transport the majority of water and sediment flows for the Bear Creek tributary. Since there are three channels doing the work of the previous single channel, the water flows and sediment load are presumably less for each of the three channels than before the formation of the anastomosing channels. The reduction in flow and sediment load are most likely the reason why large amounts of sediment have not been spread evenly across the floodplain as at the Hickory Creek site and

may be the reason why the direction of spatial dependence is still in the direction of stream flow. However, similar to the Hickory Creek site, the excessive sedimentation resulting from valley plug formation is still impacting a significant portion of the floodplain as illustrated by the contour map (Figure 2-15).

The index of spatial dependence of deposition rates at the Jeffers Creek valley plug site is high (66%) but again less than at the Hickory Creek valley plug site. The direction of spatial dependence intersects with stream flow at a 40° angle and is skewed somewhat toward the floodplain. The difference in direction of spatial dependence and direction of stream flow is a result of the excessive sedimentation that has occurred over most of the floodplain (Figure 2-16). The differences in spatial patterns between the Hickory Creek and Jeffers Creek sites may be due to differences in the development of their valley plugs or past channel disturbances at the Jeffers Creek site. Approximately two years before this study was initiated, the landowner at the Jeffers Creek site obtained an EPA permit and dredged the channel of Jeffers Creek at my study site. The dredging was in an effort to remove the valley plug from the channel, however, within two years of dredging, the channel had already filled back in substantially with sediment. One year into my study, a bridge just upstream of the Jeffers Creek study site was rebuilt. This construction also disturbed the channel system because a small dam was constructed upstream of the study site while the bridge was under construction. The dam was removed after construction, which produced sufficient water velocities to remove most of the sediment that had accumulated in the channel throughout my study reach. These two channel

disturbances have disrupted the development of the valley plug that existed at the Jeffers Creek study site, and are most likely the reason that the sedimentation dynamics differ from the patterns seen at the Hickory Creek site.

The Piney Creek shoal site had the lowest spatial dependence (23.9%) of all study sites including the other shoal site (Porters Creek). This indicates that there is a large amount of spatial variability in deposition rates at the Piney Creek site. The lack of spatial dependence also means that the prediction map produced through kriging may be somewhat confounded, since much of the spatial variability could not be modeled. As discussed earlier in the short-term deposition section, most of the deposition occurring at the Piney Creek shoal site was within the range of previous studies. However, there were some specific areas corresponding to crevasse splays that experienced high sedimentation rates. The crevasse splays produced isolated areas of high deposition that break up the spatial continuity of the otherwise average deposition rates found in BLH forests. The impact of the crevasse splays is also evident by the skewed direction of spatial dependence that is at a 55° angle to the direction of stream flow. It is also important to note that crevasse splays can also cause high deposition rates over a significant portion of the floodplain, with one-third of the Piney Creek study area experiencing deposition rates over 2.2 cm/yr as a result of the crevasse splays.

The spatial patterns found at the Porters Creek shoal site were different from those at all other study sites. These patterns represent our current understanding of sedimentation along unaltered channels. There was a high

degree of spatial dependence (70.4%) in the direction of stream flow. There were some isolated areas of greater deposition rates (Figure 2-18), which correspond to depressional areas within the floodplain that experience longer periods of inundation than the rest of the floodplain. However, deposition rates over the entire prediction map are within the range of previous studies. This suggests that under some circumstances, deposition dynamics may not be influenced by the formation of a shoal, but this may change in the future as the shoal continues to develop.

The geospatial analysis was effective in determining the influence of valley plugs and shoals on the spatial dynamics of overbank sedimentation. However, the results also demonstrate the variability in responses among sites of the same type. Thus, although valley plugs and shoals seem to have a significant impact on overbank sedimentation dynamics, there are other confounding factors that can influence the sedimentation patterns. Recovery processes of channelization and anthropogenic disturbances seem to be important in understanding the spatial dynamics of sedimentation occurring at my study sites, but further research is needed to investigate these processes and their effect on overbank sedimentation.

Conclusion

Overbank sedimentation rates and processes in the depositional zone (Schumm 1977, Hupp and Simon 1991) can be influenced dramatically by geomorphic features that can form within the channel as a result of channelization, geology, and land-use practices. In the Hatchie River watershed,

the formation of valley plugs seems to occur in the lower reaches of the altered systems, or the depositional zone, where stream gradients decrease and sediment accumulates from eroding upstream reaches. Valley plugs can cause greater deposition rates over a large extent of the adjacent floodplain. The types of sediment that are being deposited at valley plug sites are also different from unaltered and other altered sites. The high proportion of coarse sand being deposited at valley plug sites suggest that high velocity overbank flows are created as a result of the valley plug blockage. The spatial dynamics of deposition rates at valley plug sites followed classical geomorphologic theory in the sense that deposition rates decreased as distance from the channel increased. However, the short-term deposition analysis also showed that valley plugs are protecting at least the immediate downstream sections from excessive sedimentation. The geospatial analysis also showed that valley plugs can strongly affect the spatial dynamics of deposition rates. Valley plugs changed the direction of spatial dependence from parallel to stream flow to perpendicular to the stream flow, however, responses were variable due to other factors such as channel recovery processes and anthropogenic disturbances. This change in direction has enabled high deposition rates to impact a large proportion of the floodplain. Over short-time periods, the rate of deposition at valley plug sites had greater temporal variability than in unaltered systems due to climatic variability and geomorphic thresholds. Geomorphic thresholds refer to progressive or constant changes in external variables that trigger abrupt and significant changes or failure within the affected system (Schumm et al. 1984). For example,

continued erosion in the upper reaches may occur for several years, but because of minor climatic events, transport of the sediment through the system may be limited until a major rainfall event occurs that exceeds a critical threshold and removal and transport of the sediment to lower reaches occurs.

Over long-time periods, the dendrogeomorphic analysis has shown that there has been a dramatic increase in sediment deposition rates at valley plug sites since 1970. This result not only corresponds to the time period of channelization of most western Tennessee streams but also supports previous findings (Hupp and Bazemore 1993) that showed an increase in deposition rates during this time period. The lack of difference in long-term deposition rates by longitudinal location to valley plugs suggests that valley plugs are progressing upstream and can impact new floodplain areas. Further study is needed to determine the rate of expansion upstream, which may be extremely variable.

The two shoal sites that were investigated demonstrated the range of influence that shoals can have on overbank deposition. The shoal at the Porters Creek site did not seem to influence the sedimentation dynamics. However, the Piney Creek site did have some areas with greater deposition rates that correspond to crevasse splays. The crevasse splays also disrupted the spatial continuity of deposition rates across the floodplain and influenced the direction of spatial dependence. Differences in the effects of shoals on deposition rates may be a result of shoal development, but more research is needed to understand the influence of shoals on floodplain sedimentation dynamics during the early stages of formation.

The results obtained from this study have important implications for the understanding of the development, evolution, and recovery of floodplains affected by channelization. The implications extend further into the understanding of the influence of sedimentation, especially excessive sedimentation, on surface and sub-surface hydrology (Happ et al. 1940, Brinson 1990) and BLH succession (Jones et al. 1994, Hodges 1997). The rates of deposition occurring at valley plug sites combined with their influence on hydrology may have substantial influences on floodplain microtopography and site quality that can influence the germination, establishment, and survival of BLH tree species.

This study is the first reported attempt to quantify deposition rates and textures and investigate spatial patterns of overbank sedimentation in BLH forests adjacent to valley plugs and shoals. The results have provided a better understanding of sedimentation processes in BLH forests and the influence of channel alterations and the subsequent formation of within-channel geomorphic features. However, further investigation is needed to understand the influence of other factors, such as channel recovery processes and anthropogenic disturbances, that may influence overbank sedimentation dynamics in conjunction with valley plugs and shoals. The results of this study clearly show the variability in responses of sedimentation to valley plug and shoal formation and indicate the complexity and lack of predictability of these systems. For BLH conservation and restoration efforts to be successful a clearer understanding of

overbank sedimentation associated with valley plugs, shoals, and other potential influential factors is needed.

APPENDIX 2

Table 2-1. Study tributaries with identification of channel type and feature studied (Diehl 2000). Sites with * were used in the dendrogeomorphic study.

Site - Tributary	Channel Type	Feature Type	No. Plots	Years
Spring Creek -GVL	Meandering	Unchannelized	11	2002-04
Lower Spring Creek	Meandering	Unchannelized	10	2002-04
Spring Creek – Sain*	Meandering	Unchannelized	7	2003-04
Bear Creek*	Channelized	Valley Plug	61	2003-04
Jeffers Creek*	Channelized	Valley Plug	49	2002-04
Hickory Creek*	Channelized	Valley Plug	60	2002-04
Clover Creek*	Channelized	Valley Plug	18	2003-04
Piney Creek	Channelized	Major Shoal	26	2003-04
Porters Creek	Channelized	Major Shoal	31	2003-04
Jeffers Creek	Channelized	Channelized	11	2002-04
Clover Creek	Channelized	Channelized	8	2002-04

Table 2-2. Variogram models, direction of maximum spatial continuity, relative nugget effect and index of spatial dependence of sediment deposition at valley plug (Hickory, Bear, and Jeffers) and shoal sites (Piney and Porters).

Site	No. plots	Variogram model	Direction	Spatial Dependence Index
Hickory	60	Exponential	284.2°	100%
Bear	61	Rational Quadratic	6.6°	55%
Jeffers	49	Exponential	348.6°	66.1%
Piney	26	Gaussian	25°	23.9%
Porters	31	Spherical	305.6°	70.4%

A) Unchannelized Sites

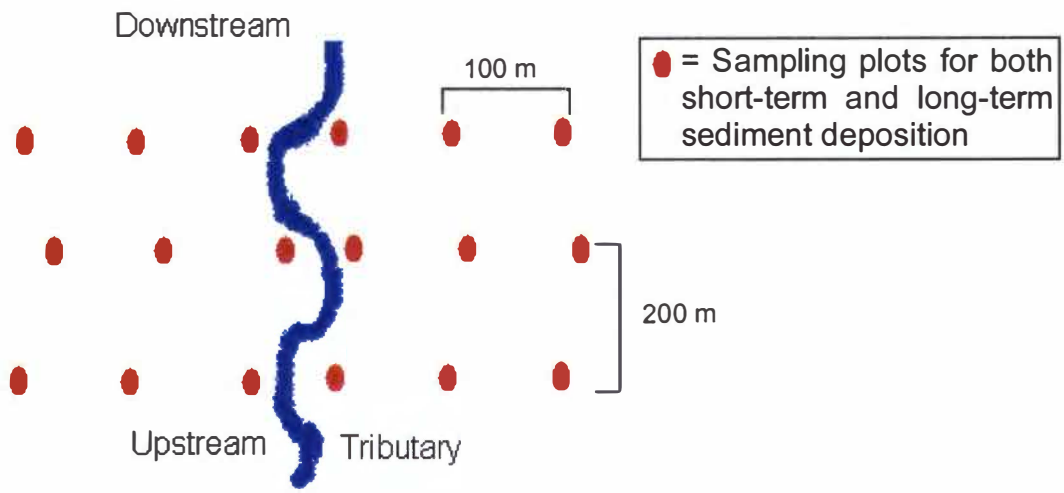


Figure 2-1. Sampling design for short-term and long-term sedimentation at (a) unchannelized sites, (b) valley plug sites, and (c) shoal sites. The sampling desing for channelized sites was the same as unchannelized sites except that long-term deposition was not measured at channelized sites.

B) Valley plug sites

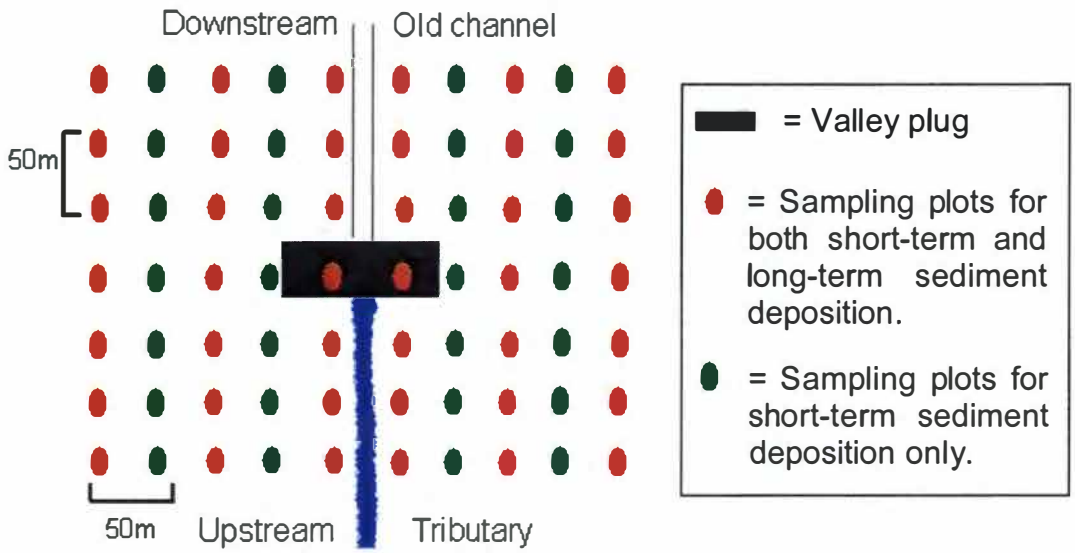


Figure 2-1. Continued

C) Shoal sites

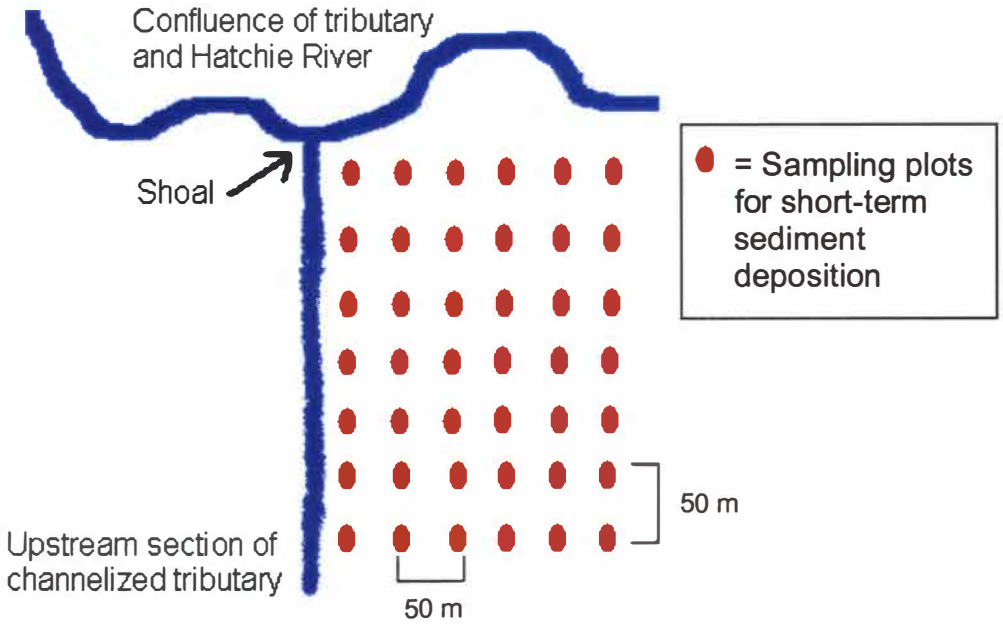


Figure 2-1. Continued

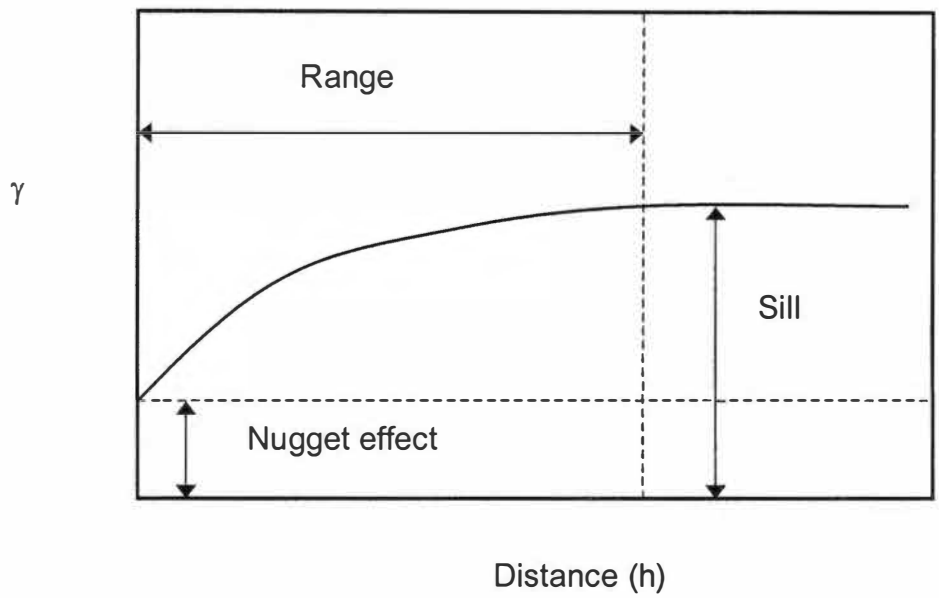


Figure 2-2. Example of a variogram. Semivariance values (γ) increase with distance (*h*) until pairs of locations are no longer correlated and a plateau or sill is reached. The distance to the sill is range. The nugget effect, representing discontinuity at the origin is a result of measurement error and small-scale variability.

A) Unchannelized sites

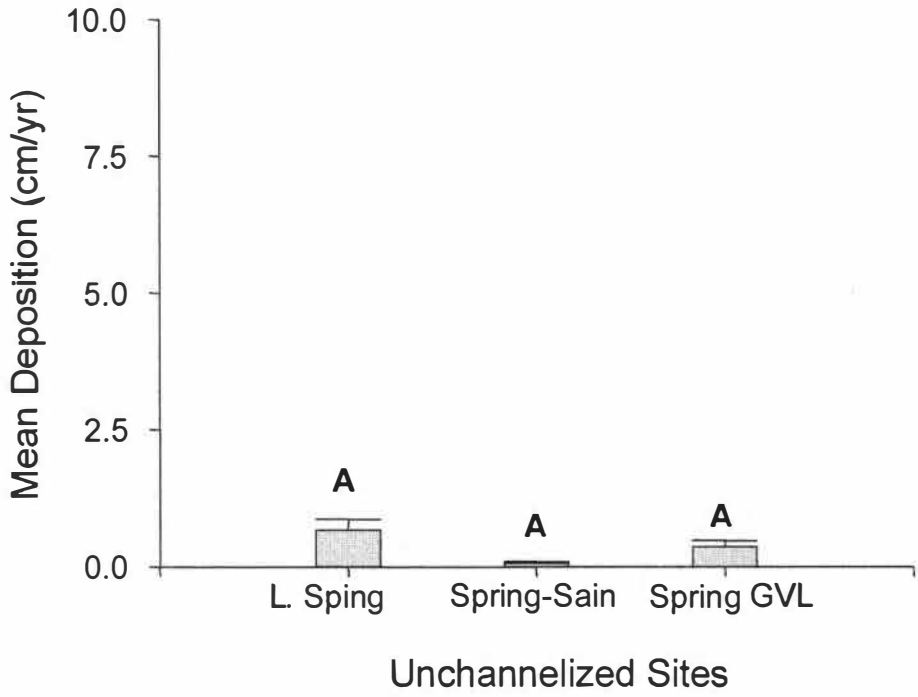


Figure 2-3. Mean sedimentation rates (+1 standard error) for all sites by type (a) unchannelized sites, (b) channelized sites, (c) shoal sites, and (d) valley plug sites.

B) Channelized sites

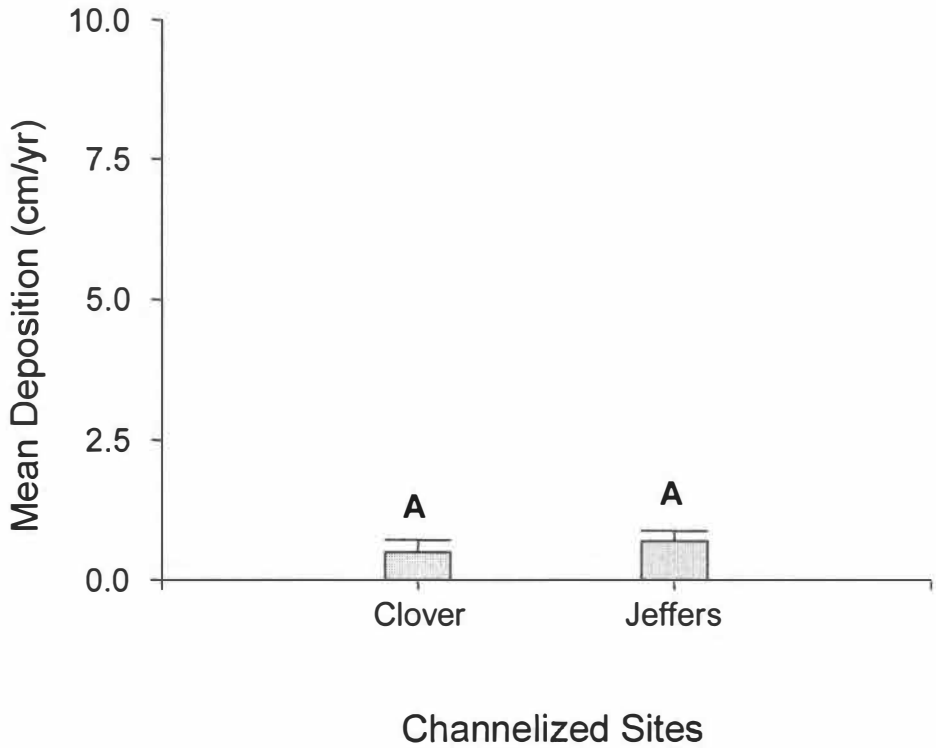


Figure 2-3. Continued

C) Shoal sites

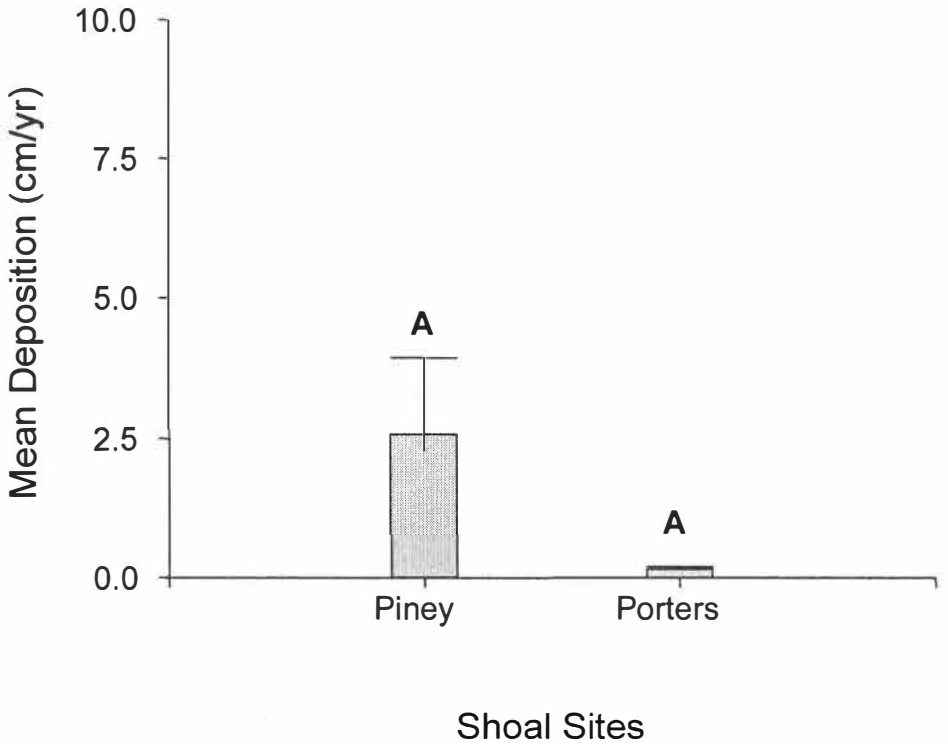


Figure 2-3. Continued

D) Valley plug sites

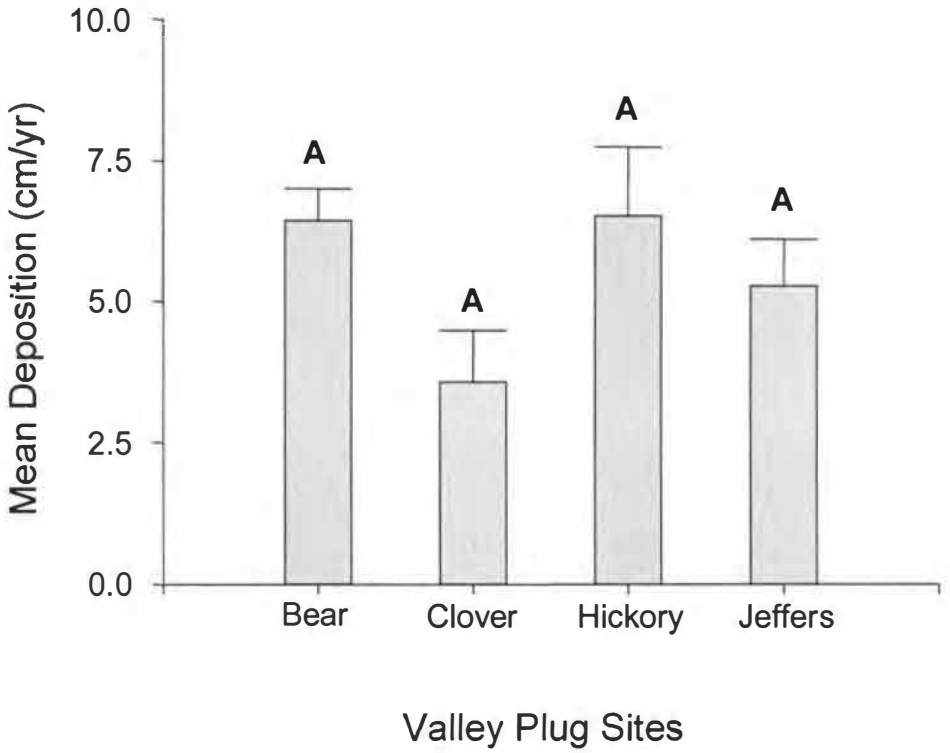


Figure 2-3. Continued

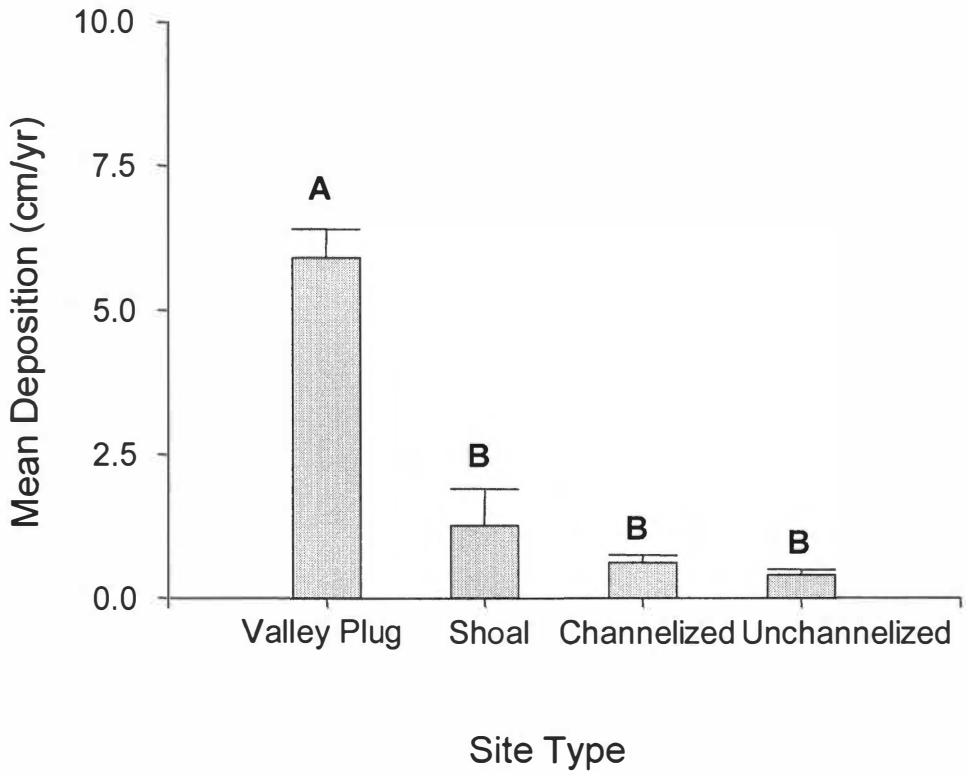


Figure 2-4. Mean sedimentation rates (+1 standard error) for valley plug, shoal, channelized, and unchannelized sites based on sediment pad measurements from 2002-2004. Bars with unlike letters are significantly different ($P < 0.05$).

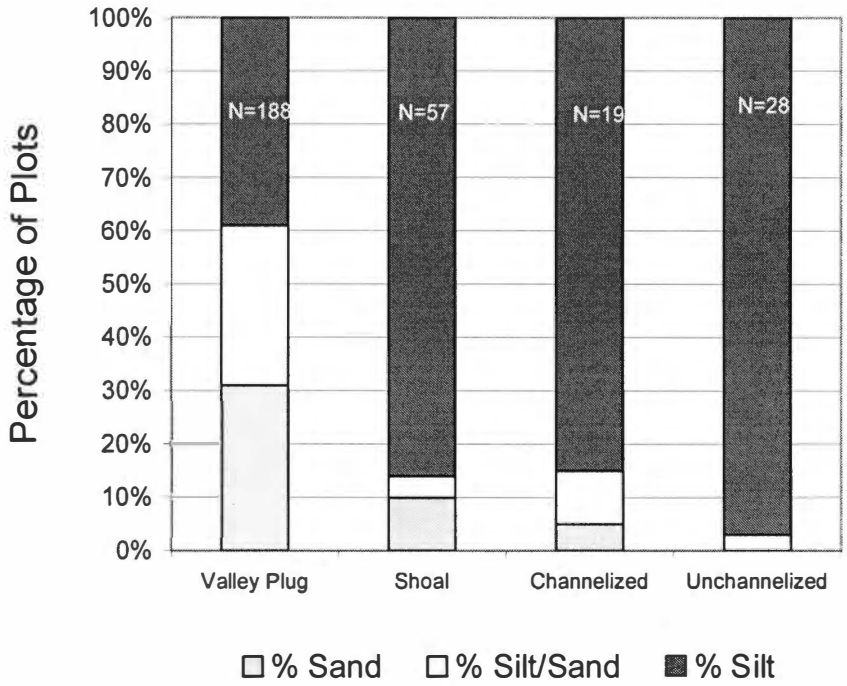


Figure 2-5. Percentage of plots with different soil textures for valley plug, shoal, channelized, and unchannelized sites. Based on sediment pad measurements from 2002-2004.

A) Unchannelized Sites

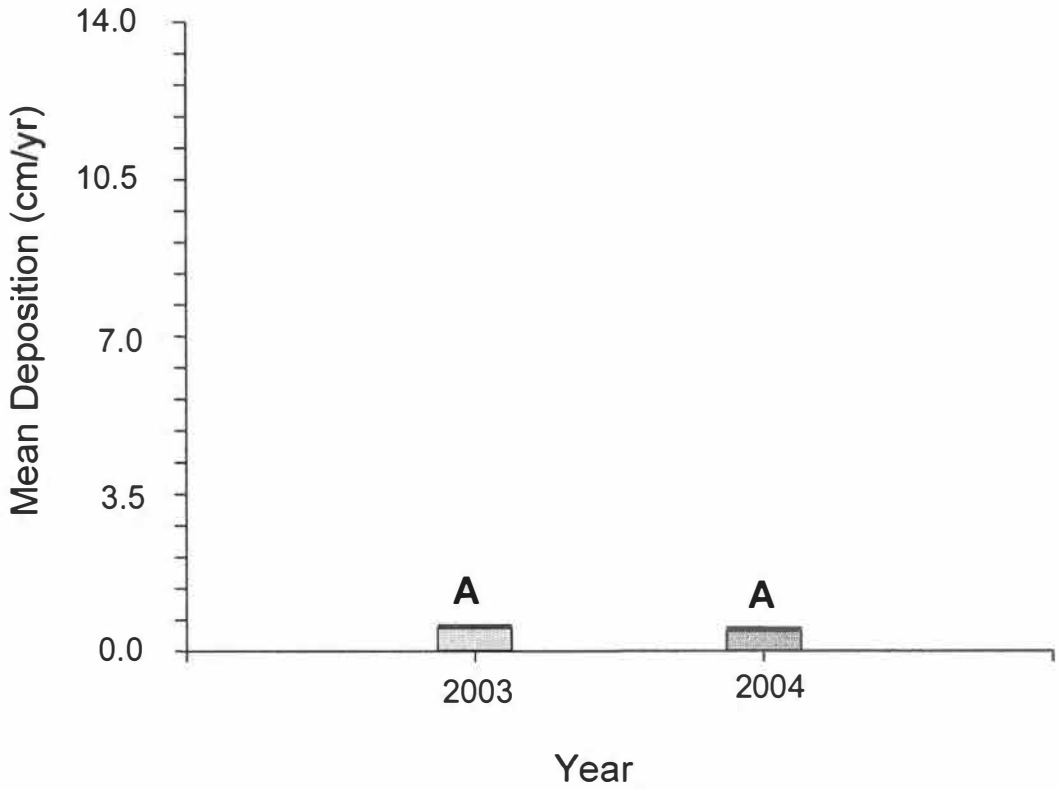


Figure 2-6. Mean sedimentation rates (+1 standard error) for (a) unchannelized sites and (b) valley plug sites, for 2003 and 2004. Shoal sites were excluded due to lack of data. Bars with unlike letters are significantly different ($P < 0.05$).

B) Valley plug sites

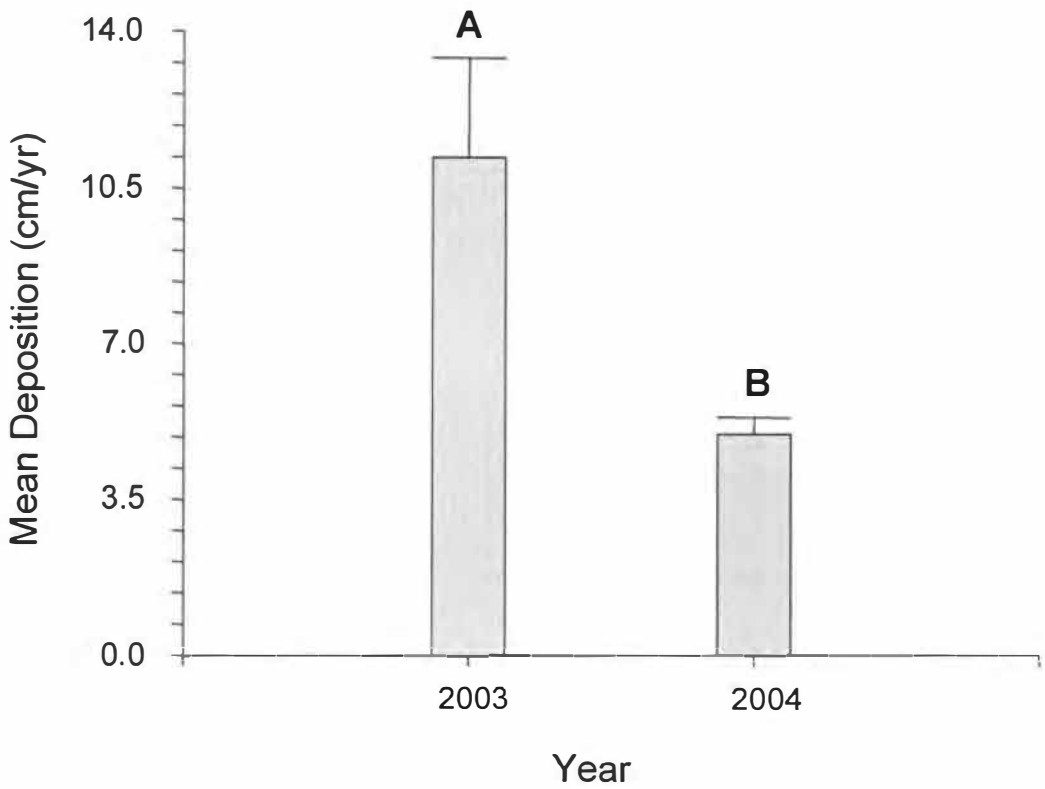


Figure 2-6. Continued

A) Unchannelized Sites

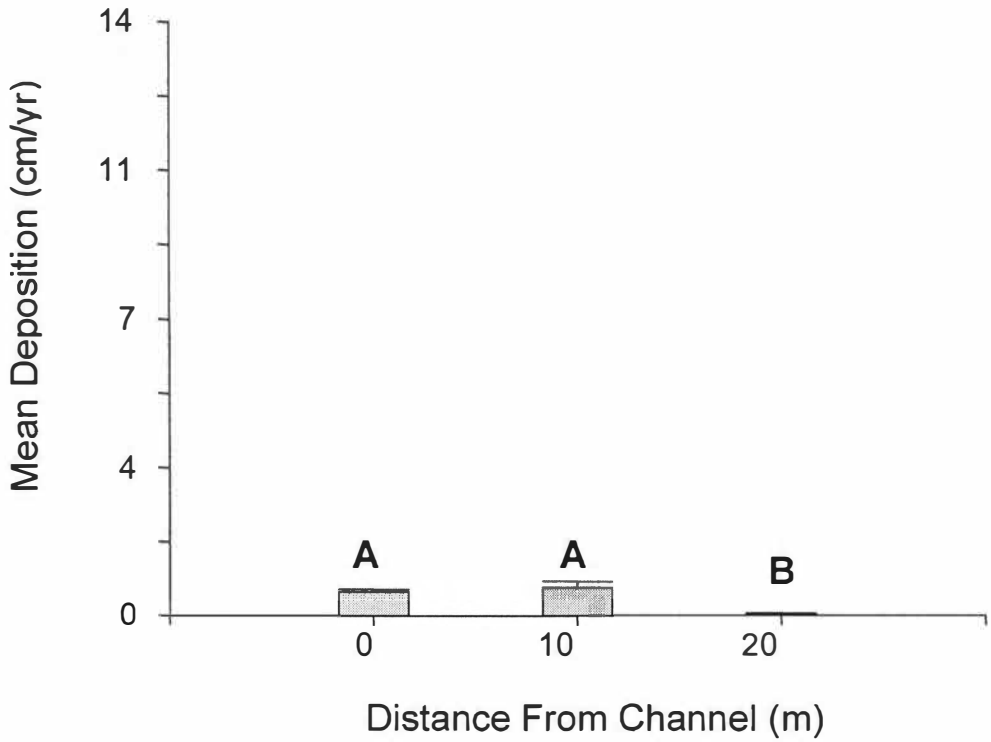


Figure 2-7. Mean sedimentation rates (+1 standard error) for (a) unchannelized sites and (b) valley plug sites, by distance from the channel (m). Based on sediment pad measurements from 2002-2004. Bars with unlike letters are significantly different ($P < 0.05$).

B) Valley plug sites

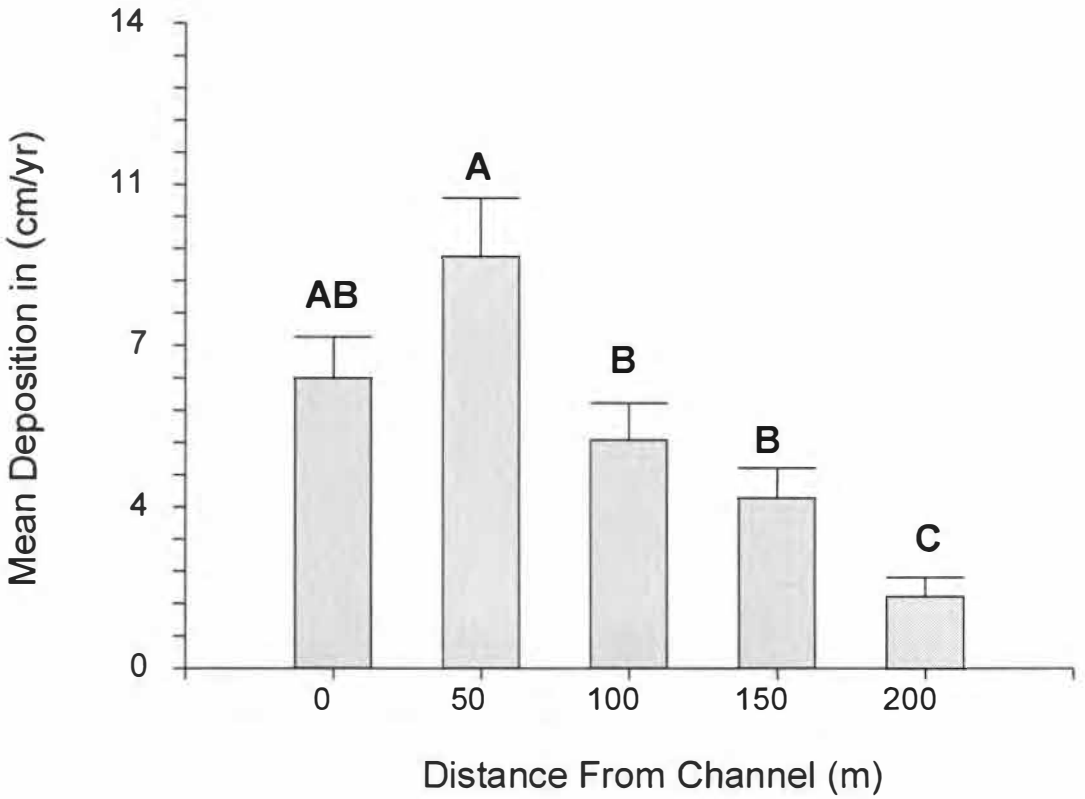


Figure 2-7. Continued

A) Shoal sites

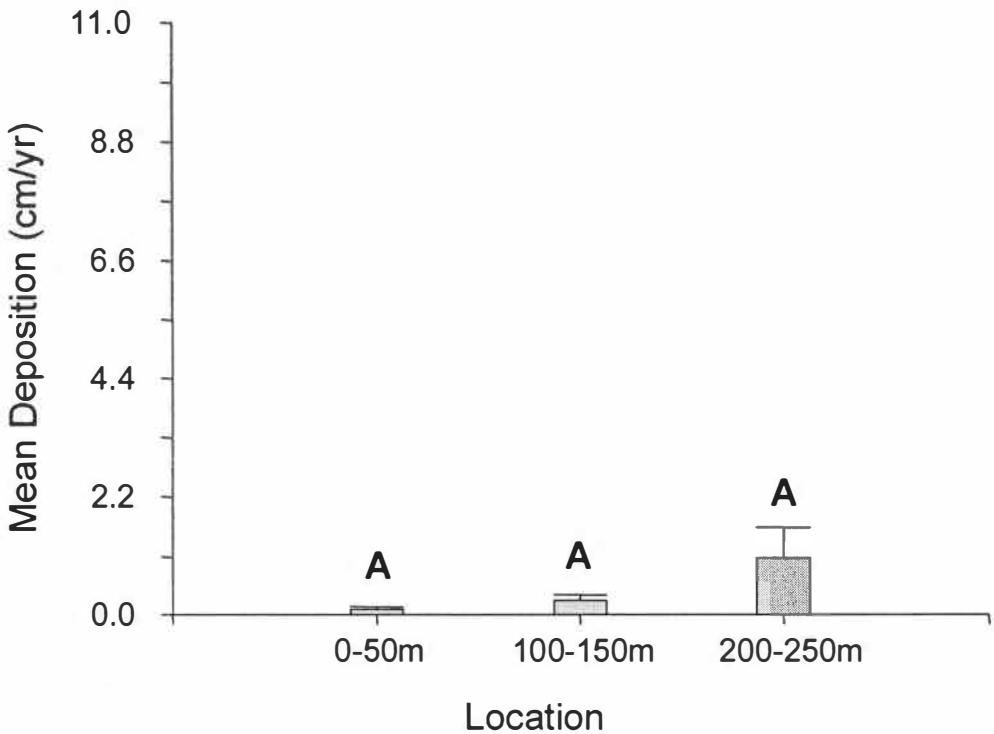


Figure 2-8. Mean sedimentation rates (+1 standard error) for (a) shoal sites and (b) valley plug sites, by longitudinal location. Shoal locations are transects grouped by upstream distance from the shoal, valley plug locations are transects grouped by longitudinal distance and direction from valley plugs. Based on sediment pad measurements from 2002-2004. Bars with unlike letters are significantly different ($P < 0.05$).

B) Valley plug sites

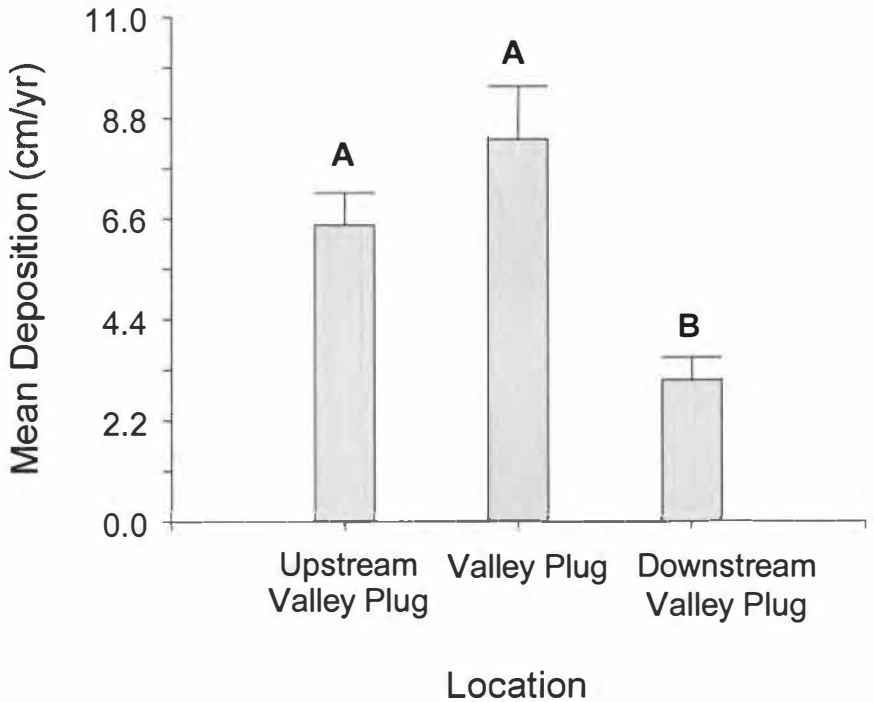


Figure 2-8. Continued

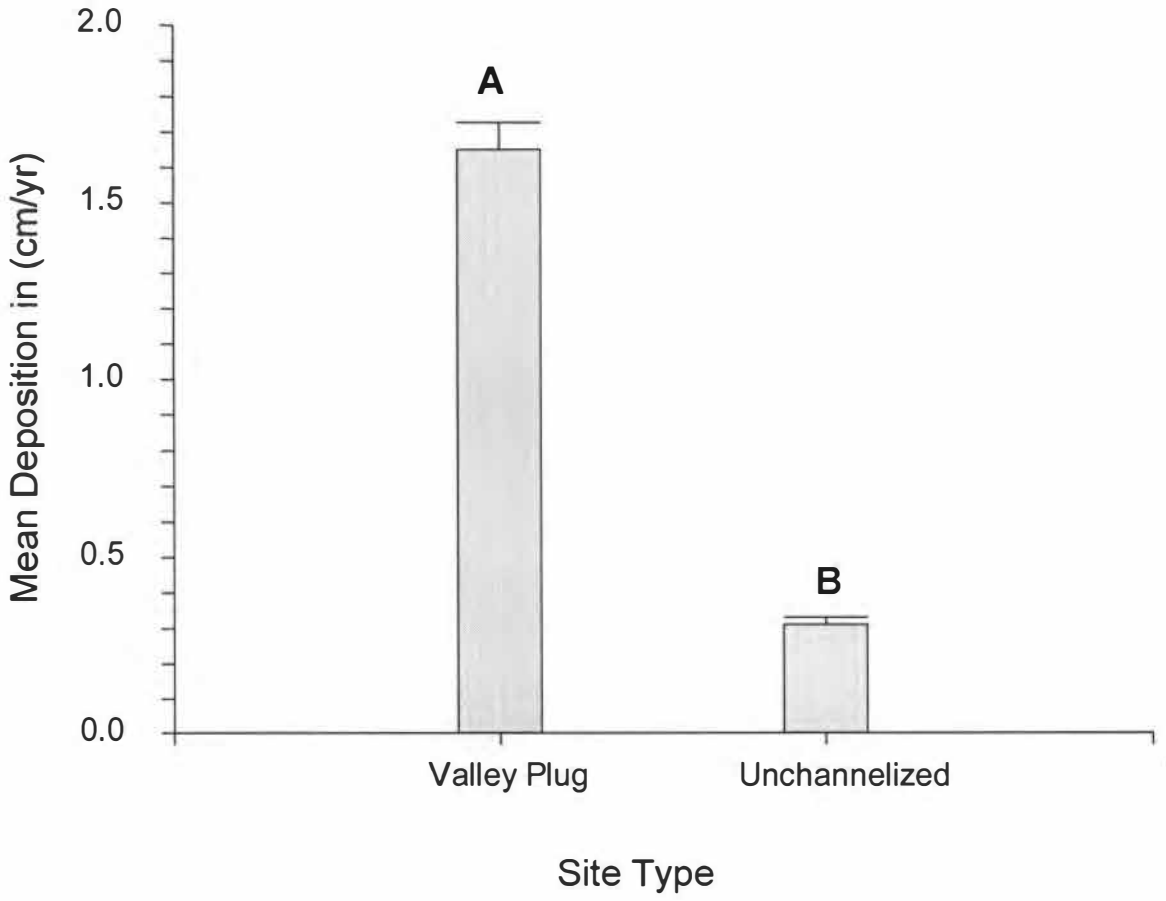


Figure 2-9. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+1 standard error) for valley plug and unchannelized sites. Bars with unlike letters are significantly different ($P < 0.05$).

A) Unchannelized sites

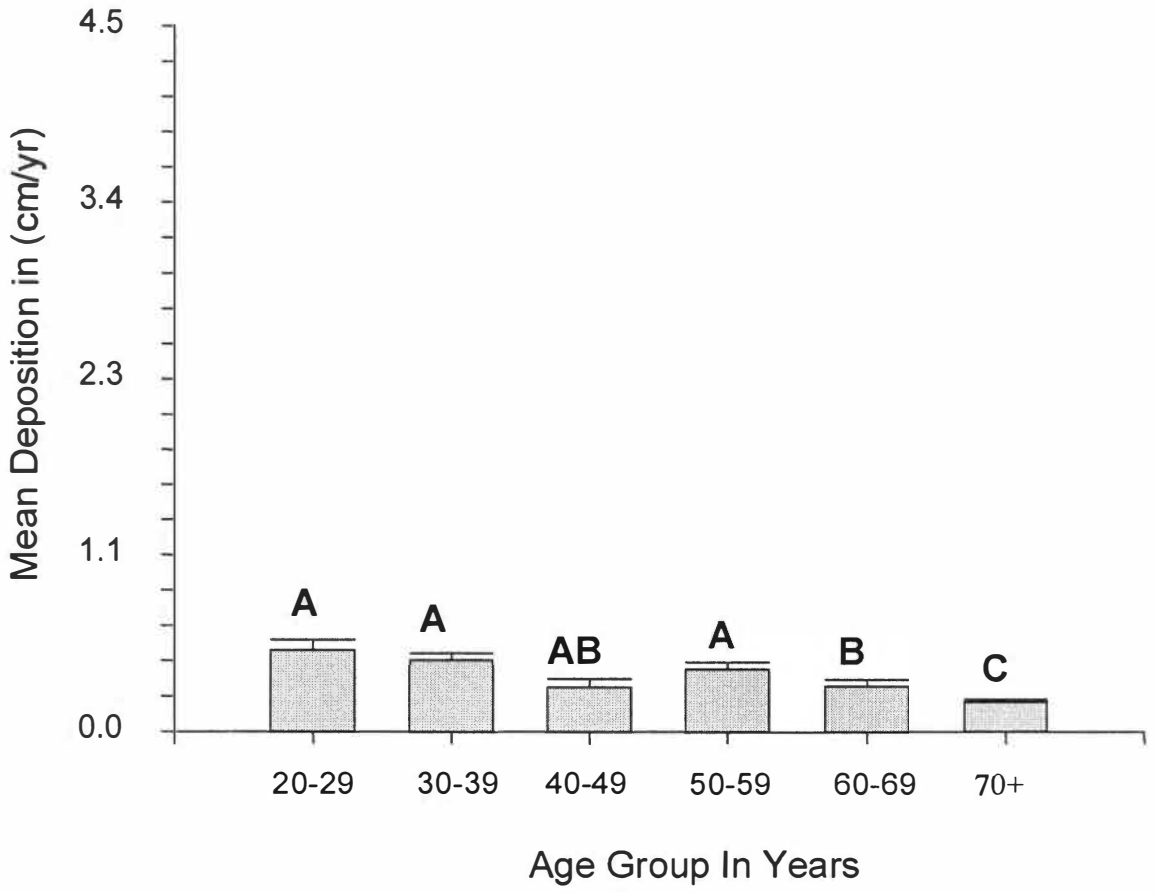


Figure 2-10. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+ 1 standard error) by tree-age group for trees sampled at (a) unchannelized sites and (b) valley plug sites. Bars with unlike letters are significantly different ($P < 0.05$).

B) Valley plug sites

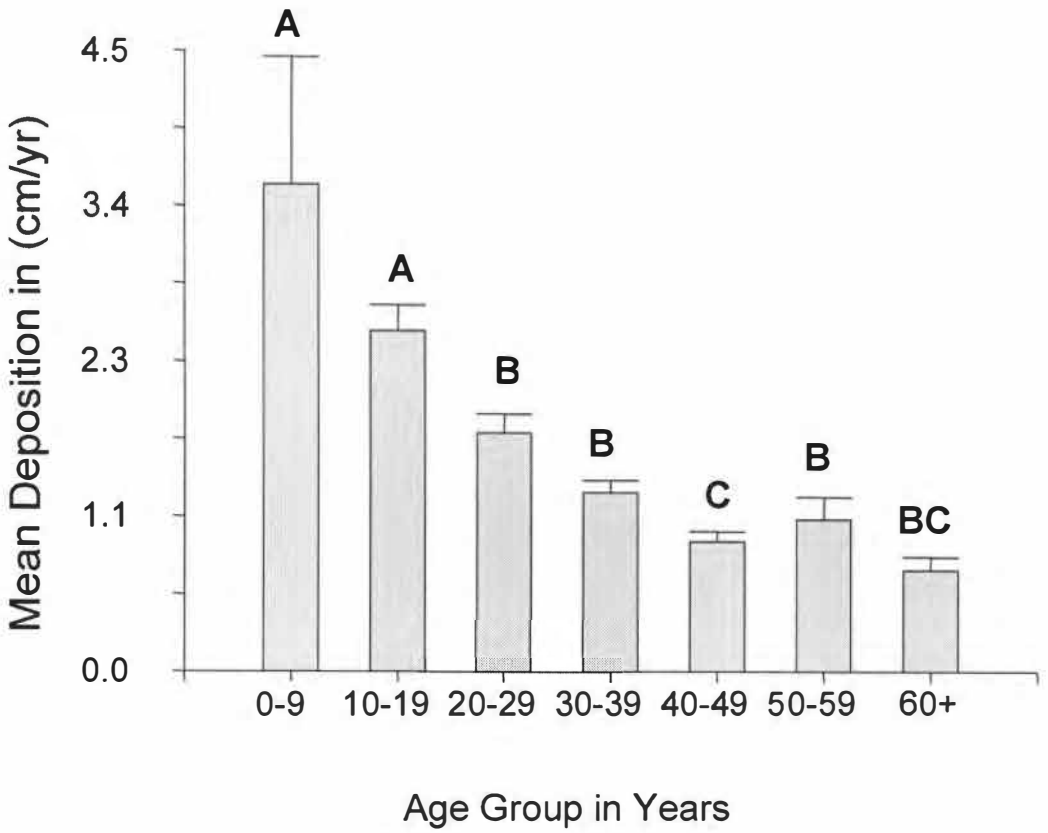


Figure 2-10. Continued.

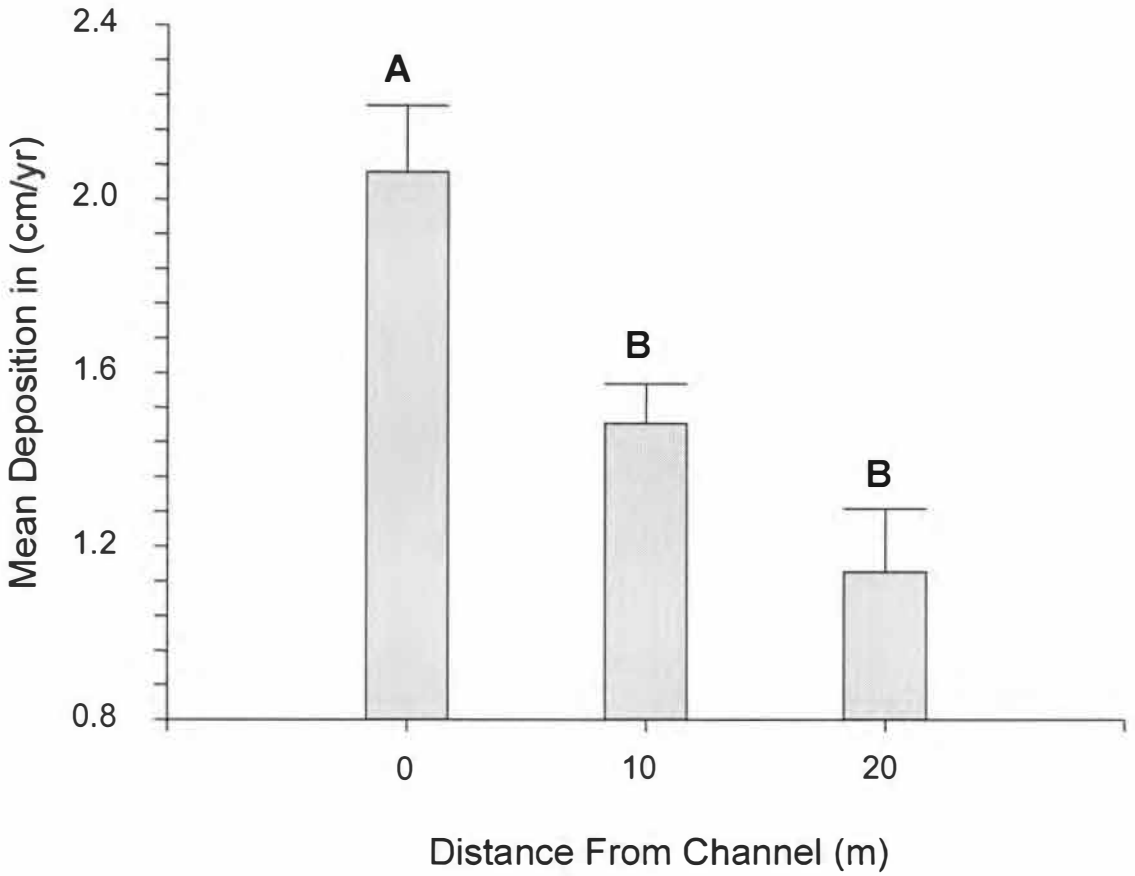


Figure 2-11. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+1 standard error) for valley plug sites by distance from the channel. Bars with unlike letters are significantly different ($P < 0.05$).

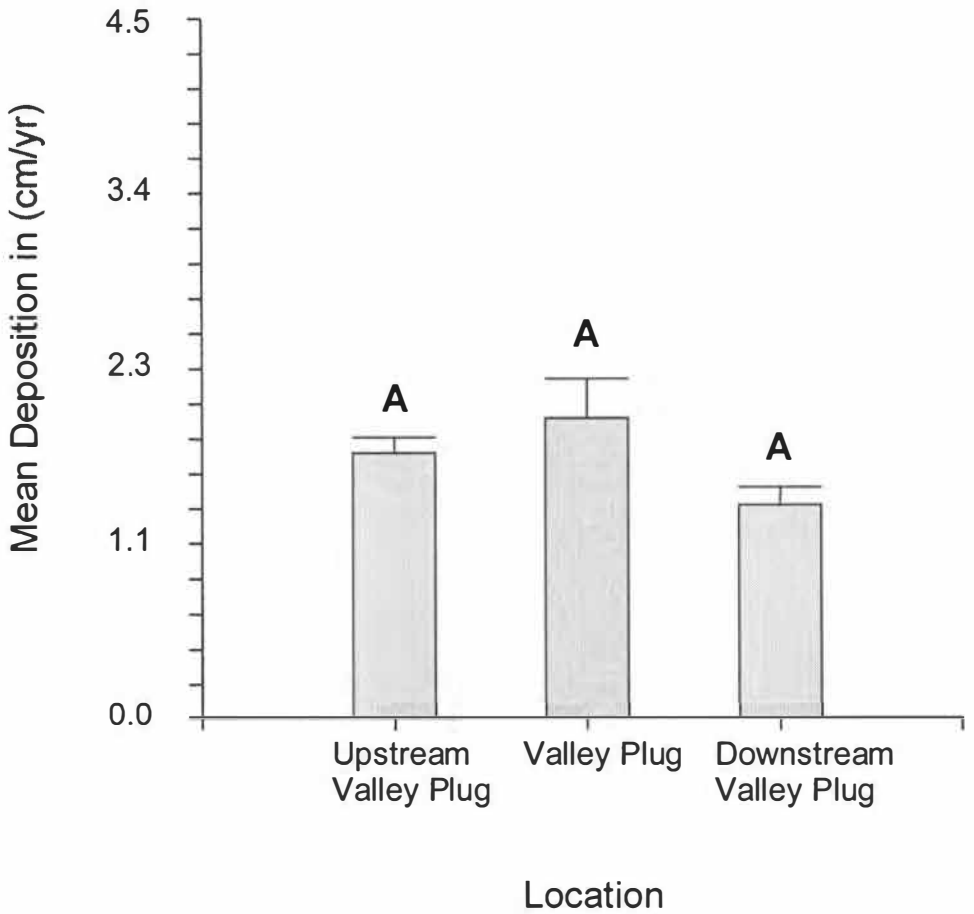
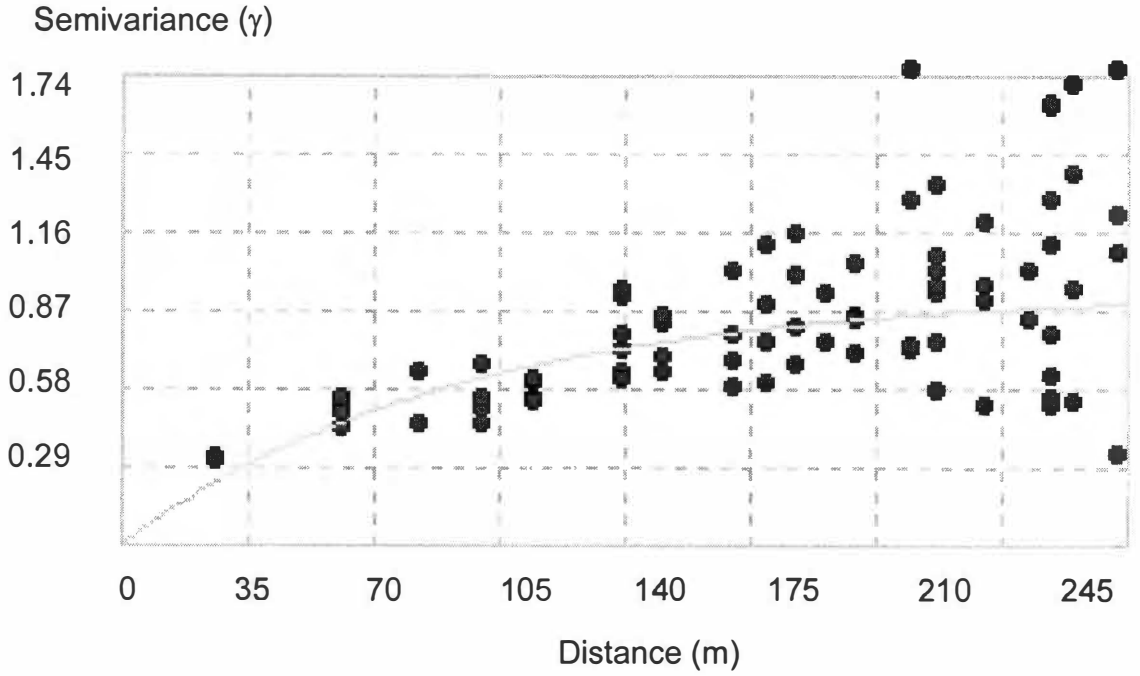


Figure 2-12. Results of dendrogeomorphic analyses of mean long-term sedimentation rates (+1 standard error) for valley plug sites by longitudinal distance and direction relative to the valley plug. Bars with unlike letters are significantly different ($P < 0.05$).

A) Omindirectional variogram



B) Variogram adjusted for anisotropy

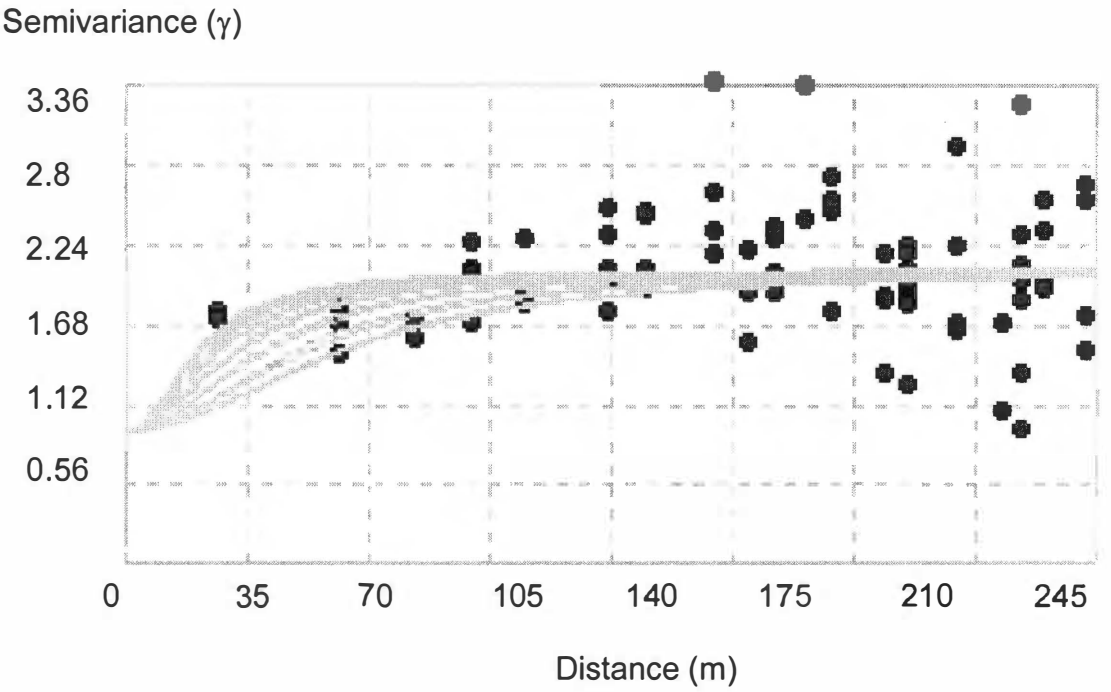


Figure 2-13. Examples of (a) an omindirectional variogram of the Hickory Creek site and (b) a variogram adjusted for anisotropy of the Bear Creek site.

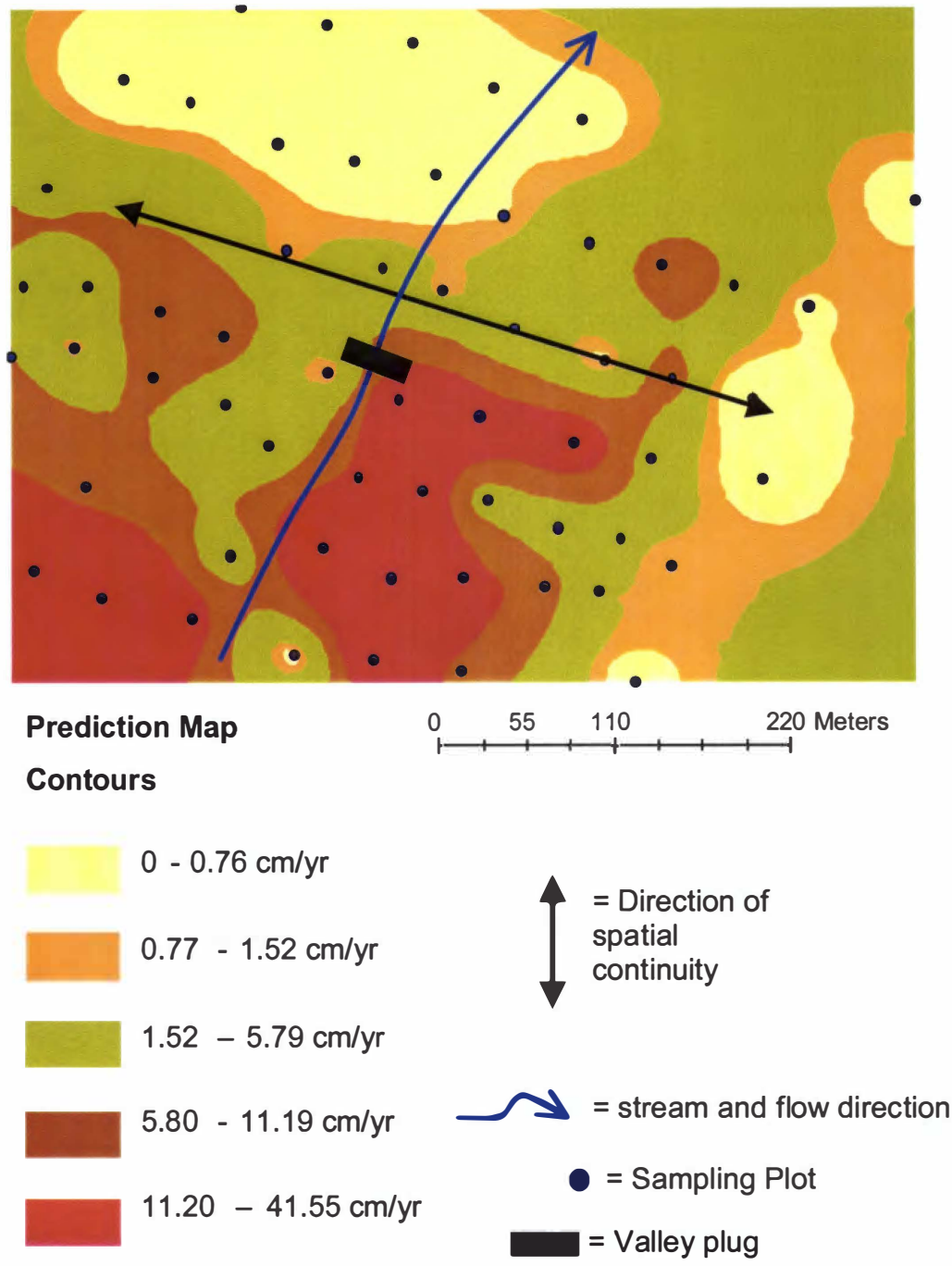


Figure 2-14. Prediction map, produced through kriging, of sediment deposition occurring at the Hickory Creek valley plug site. Illustrates areas of different deposition rates, location of sampling plots, direction of stream flow, and direction of spatial continuity.

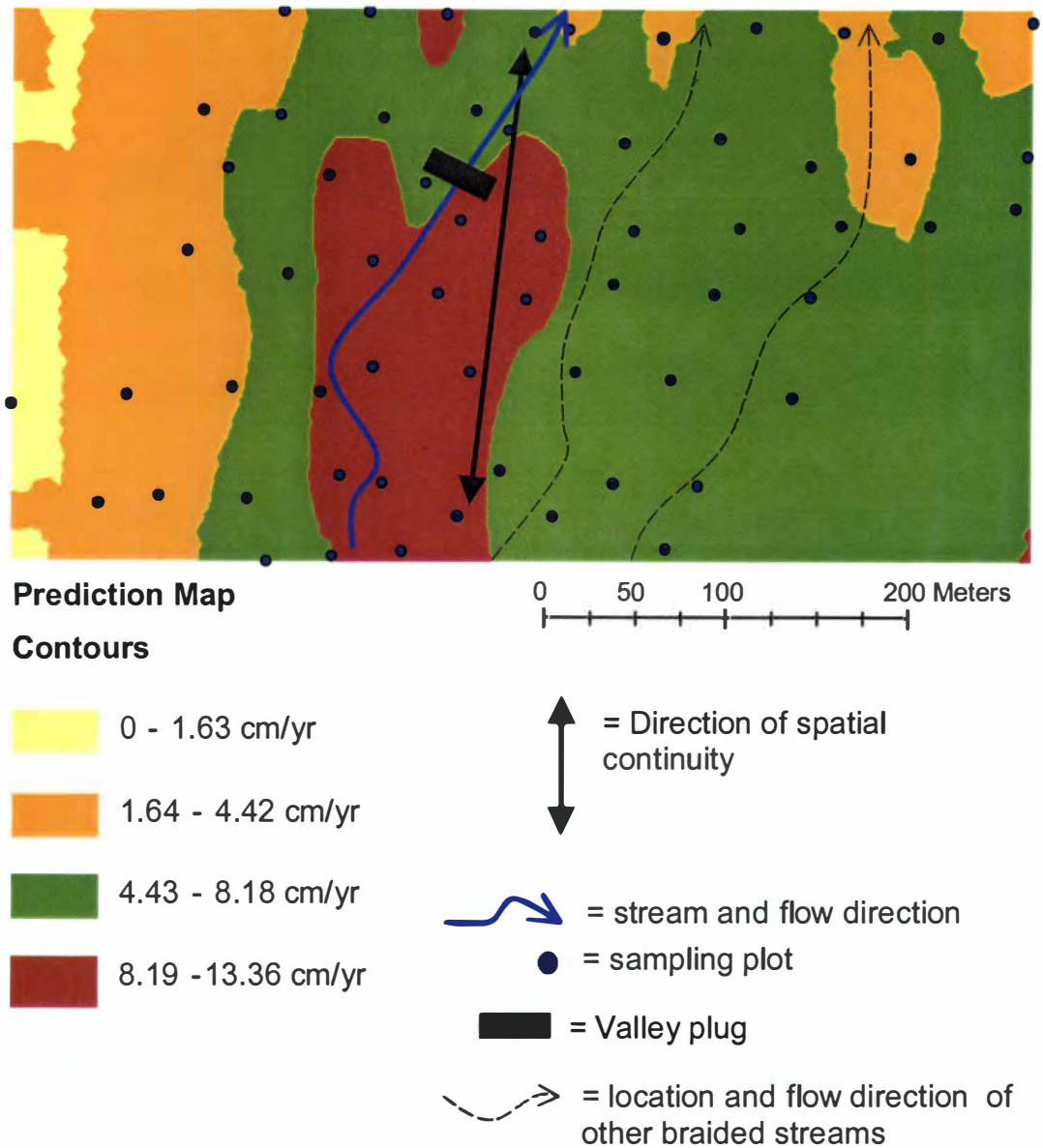


Figure 2-15. Prediction map, produced through kriging, of sediment deposition occurring at the Bear Creek valley plug site. Illustrates areas of different deposition rates, location of sampling plots, direction of stream flow, and direction of spatial continuity.

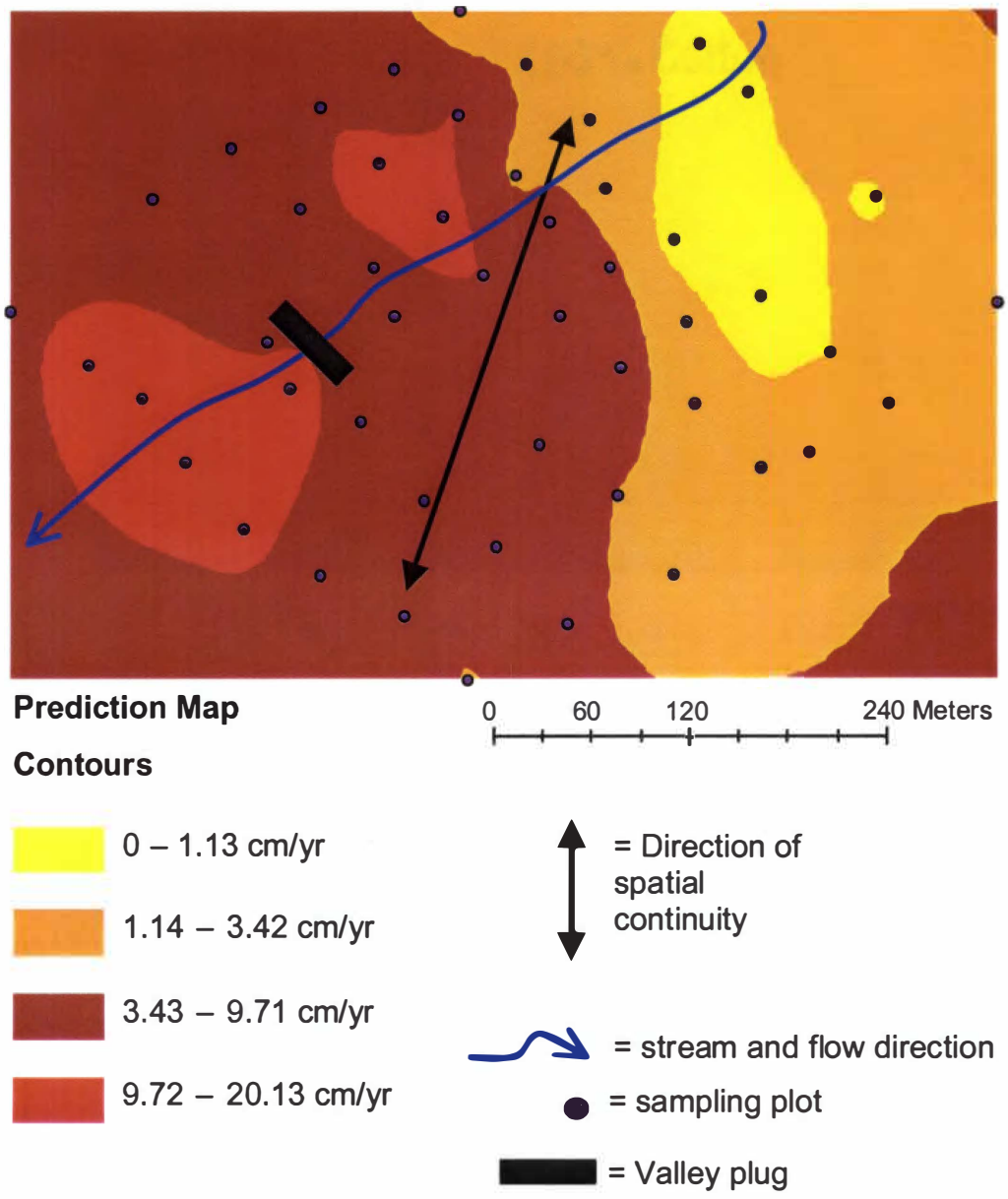


Figure 2-16. Prediction map, produced through kriging, of sediment deposition occurring at the Jeffers Creek valley plug site. Illustrates areas of different deposition rates, location of sampling plots, direction of stream flow, and direction of spatial continuity.

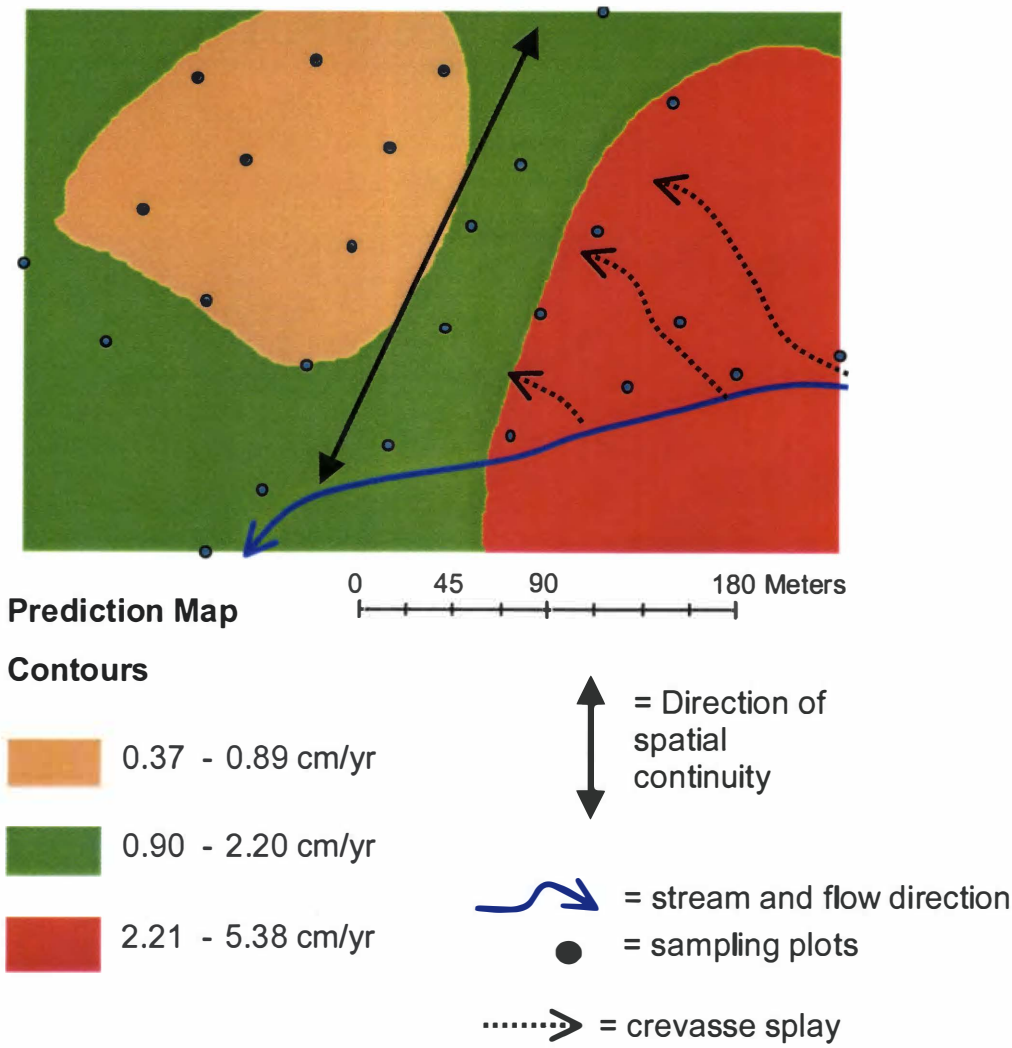


Figure 2-17. Prediction map, produced through kriging, of sediment deposition occurring at the Piney Creek shoal site. Illustrates areas of different deposition rates, location of sampling plots, direction of stream flow, and direction of spatial continuity.

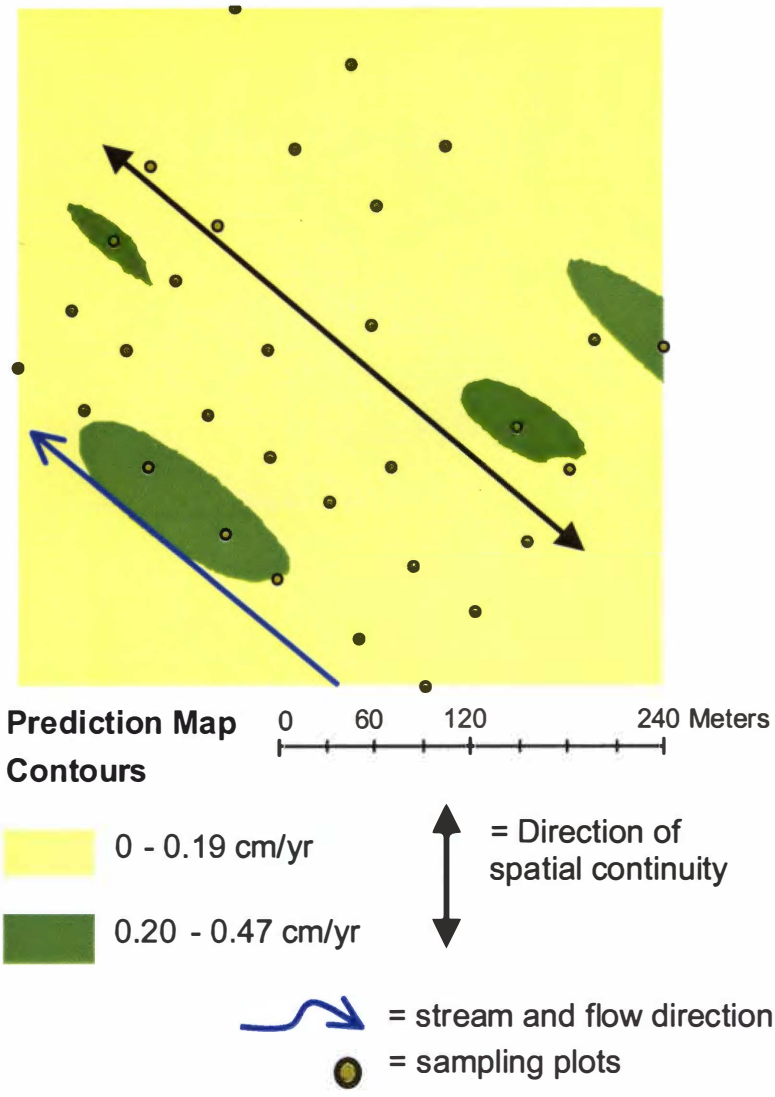


Figure 2-18. Prediction map, produced through kriging, of sediment deposition occurring at the Porters Creek shoal site. Illustrates areas of different deposition rates, location of sampling plots, direction of stream flow, and direction of spatial continuity.

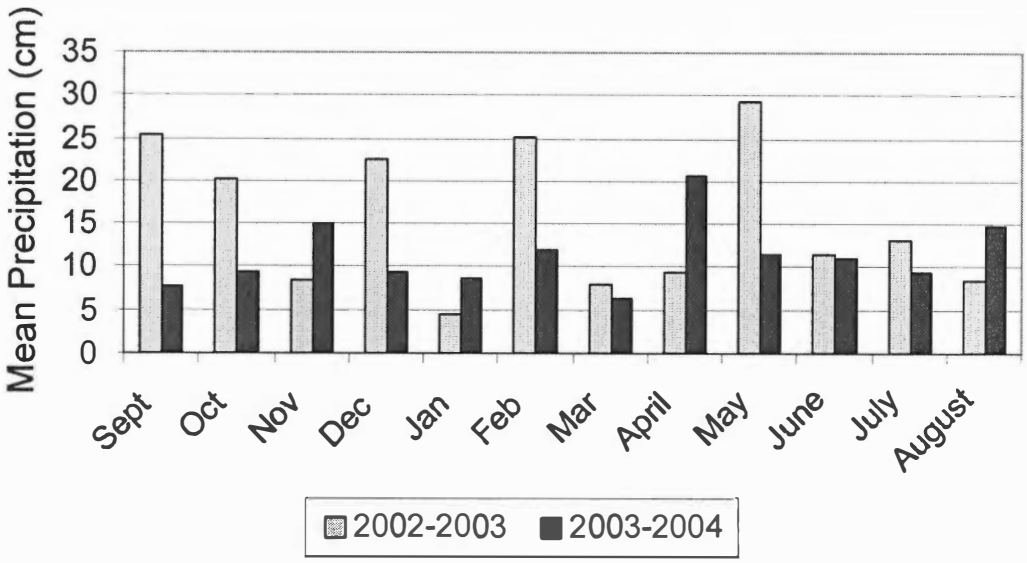


Figure 2-19. Mean monthly precipitation for short-term deposition sampling periods from 2002 to 2004. Monthly precipitation values are the mean amounts measured from weather stations in Brownsville and Bolivar, TN (NOAA 2005).

PART III

HYDROLOGICAL RESPONSES TO CHANNELIZATION AND THE FORMATION OF VALLEY PLUGS AND SHOALS

Introduction

Flood frequency, depth, timing and duration are primary determinants of floodplain plant species composition (Bedinger 1978, Huffman 1980, Hodges 1997). The bottomland hardwood (BLH) community level changes, described by Hodges (1997), are a result of differential survival of overstory trees, seedlings, and germination rates of seeds under prevailing hydrologic and sediment deposition conditions. The hydrological regime including timing, depth, frequency and duration of inundation has been shown to influence tree growth and mortality (Johnson and Bell 1976, Reily and Johnson 1982, Keeland and Sharitz 1997). The hydrologic regime is also the major factor affecting germination, seedling establishment, growth, and survival (Hosner 1957, Huffman 1980, Harms et al. 1980, Streng et al. 1989, Johnson 1994, Jones et al. 1994, Johnson 2000).

The influence of timing, depth, duration, and frequency of flooding on plant species composition, distribution and survival is most critical during the growing season. Floods during the dormant period have a modest effect on physiology and survival of plant species (Bedinger 1978, Wharton et al. 1982). In the southeastern United States, dramatic fluctuations in water levels are the result of high flow periods due to winter and spring rains and low flow periods attributed to dry summer and fall months with high evapotranspiration (Wharton et al. 1982). High runoff rates during the winter and spring months usually overflow the floodplain features. This high water level is sometimes sustained by the cumulative effect of many tributaries and isolated rainfall (Wharton et al.1982).

Meanwhile, infrequent rain events during late summer and fall coupled with high evapotranspiration can result in near zero discharge.

Plant species composition is also influenced by an array of factors within the floodplain including: sediment deposition, soils, elevation, and groundwater levels (Clark and Benforado 1981, Hodges 1997). Groundwater levels are interrelated with surface hydrology in floodplain systems (Burt et al. 2002). Groundwater levels have been shown to influence the establishment, growth, survival and diversity of BLH tree species (Scott et al. 1999, Scott et al. 2000, Hughes et al. 2000). Lowered water table has been found to significantly reduce the growth of several BLH tree species (Reily and Johnson 1982, Scott et al. 1999, Scott et al. 2000). A higher water table may inundate root systems of trees for long periods of time and have a similar effect on vegetation as surface inundation, causing stress and mortality of standing timber (Happ et al. 1940).

Human alterations including habitat fragmentation, wetland drainage, dams, impoundments, channelization, and groundwater extraction have severely affected both the surface and sub-surface hydrology of floodplain systems, especially in the southeastern United States. Throughout the southeastern Coastal Plain, channelization has been a common approach to reduce flooding, mainly for agricultural purposes (Shankman 1993). Channelization involves widening and deepening a channel to increase channel capacity, shortening stream length, and increasing the stream gradient to increase water transport.

The increased channel capacity and increased transport efficiency of channelized reaches causes channels to be disconnected from the adjacent

floodplain. Channelization has been shown to reduce flooding in upstream reaches of a system, while causing lower reaches to experience increased peak flood stage and flood frequency (Shankman and Pugh 1992). Channelization and dredging of stream channels has also been shown to lower the water table levels in the floodplain (Tucci and Hileman 1992) because of the connection between the stream channel and the floodplain water level (Happ et al. 1940).

Channelization can initiate further degradation of the stream channel and lead to bank failure and channel erosion (Robbins and Simon 1983, Simon and Hupp 1987, Simon 1994). Channelization also increases stream power, thus facilitating sediment transport, which can increase deposition in lower stream reaches. In western Tennessee, the effects of channelization have been exacerbated by the geology of the region and past land-use practices. In this region, channelization has led to dramatic changes in hydro-geomorphic processes that have caused the formation of valley plugs and shoals (Diehl 2000). Valley plugs are areas where the channel becomes completely filled with sediment, forcing streamflow and sand bedload out into the floodplain (Happ 1975). Shoals are points in the channel where the depth decreases downstream due to bedload deposition. Shoals usually form at the confluence of tributaries and the main stem of the river (Diehl 2000). For more information on valley plugs and shoals, see Chapter 2.

Valley plugs and shoals have been shown to influence overbank sedimentation dynamics (Chapter 2), however they may also influence both surface and sub-surface hydrology (Happ et al. 1940). Increased overbank

flooding events, as a result of channel deposition and reduced drainage efficiency of the floodplain, may increase the frequency, depth and duration of flooding (Happ et al. 1940, Diehl 2000). Surface hydrologic effects associated with excessive sedimentation can have negative impacts on BLH tree species regeneration, growth, and survival (Johnson and Bell 1976, Harms et al. 1980, Reily and Johnson 1982, Jones et al. 1994, Keeland and Sharitz 1997, Johnson 2000).

Aggradation of stream channels can cause the water table to rise so that it becomes elevated above some floodplain surfaces (Happ et al. 1940). Not only does the water table rise contribute to increased ponding on the floodplain, but it can also inundate root systems of BLH tree species throughout the growing season. The inundation of the roots may have the same negative effects on BLH tree species as flooding does, but without the surface water.

Although there has been some study on the effects of channelization on surface hydrology (Emerson 1971, Shankman and Pugh 1992) and groundwater levels (Tucci and Hileman 1992, Burt et al. 2002), presently little information exists on the effects of valley plugs and shoals on these hydrologic processes or on the effects of these features on BLH forest succession. The rapid response of hydro-geomorphic and ecological attributes to valley plugs and shoals provides a unique opportunity to study such relationships that are usually long-term in their development. This information is necessary for the development of restoration approaches and management techniques needed to sustain functioning BLH forests.

The first objective of the study was to determine differences in depth, duration, and frequency of flooding in floodplains adjacent to three features: valley plugs, shoals, and unchannelized channels. I expected that the depth, duration, and frequency of flooding would be greater at valley plug and shoals sites than at unchannelized sites due to reduced channel capacity resulting from channel filling and reduced floodplain drainage efficiency.

Since flooding during the growing season has the greatest effect on BLH tree species, my second objective was to determine differences in depth, duration, and frequency of flooding during the growing season (April through September) among the three geomorphic features. For the growing season, I expected that depth, duration, and frequency of flooding would be greater at valley plug and shoal sites than unchannelized sites due to the reduced channel capacity and accelerated development on natural levels that impede the recession of floodwaters (Happ et al. 1940, Shankman and Pugh 1992).

The third objective of this study was to determine if groundwater levels and variation in groundwater levels differed among valley plug, shoal and unchannelized sites. I predicted that groundwater levels and variation in groundwater levels would be greater at valley plug and shoal sites than unchannelized sites due to increases in the channel bed elevation and to the high porosity of sand, deposited at valley plug and shoal sites (Chapter 2). The high permeability of sand deposits in both the channel and in floodplains can reduce the lag time response of groundwater levels to fluctuations in river stage (Brinson 1990).

The final objective was to determine whether there were any relationships between groundwater levels and geomorphic attributes including sediment deposition rates, channel depth, channel aggradation or incision, and variation in channel depth. Based on previous research (Brinson 1990, Tucci and Hileman 1992, Burt et al. 2002), I expected channel depth and channel aggradation to have the strongest relationship to groundwater levels.

Methods

Study Reach

The Hatchie River study reach encompasses all tributaries that were investigated in this study and is located in Haywood, Madison, and Hardeman Counties in Tennessee, stretching south from the Hatchie River National Wildlife Refuge in Brownsville to Hickory Valley (Figure 1-2). Study sites are located along six tributaries of the Hatchie River. The tributaries consist of one unchannelized stream and five channelized streams (Table 3-1).

This study focused on three types of study sites: unchannelized sites (meandering channel), valley plug sites, and shoal sites. Three unchannelized sites were located along Spring Creek at a minimum distance of 2 km between sites. Spring Creek is a natural meandering tributary of the Hatchie River and contains extensive BLH forests. Spring Creek is the only unaltered major tributary within the study reach (USDA 1986). Valley plug sites have been identified on several tributaries of the Hatchie River including three tributaries chosen for this study: Bear Creek, Jeffers Creek, and Hickory Creek. Two shoal

sites were also included in this study: Porters Creek and Piney Creek. These streams contain a shoal at their confluence to the Hatchie River.

Surface Hydrology

One pressure sensor water level recorder (Infinities USA, Inc.) with an accuracy of ± 0.45 cm, was placed at each of the three valley plug sites (Bear Creek, Hickory Creek, and Jeffers Creek), the two shoal sites (Piney Creek and Porters Creek), and the three unchannelized sites (Spring Creek – GVL, Spring Creek – Sain, and Lower Spring Creek). Water level recorders were tied to trees located at the apparent lowest elevation on the floodplain at each site and they recorded instantaneous stage or surface inundation of the floodplain at 12:00 pm each day. The elevation of water level recorders was adjusted by deposition rates to compensate for changes in floodplain surface elevation because of sediment deposition. Some small overbank flooding events may have been missed because only one water level recorder could be placed at each site.

Measurements were taken at the valley plug and unchannelized sites from September 2002 to October 2004. Due to a malfunction in one water level recorder at the Jeffers Creek valley plug site, data could only be collected from September 2002 to July 2003. Shoal sites were not established until the summer of 2003, thus surface-water depths at shoal sites were collected from September 2003 to October 2004.

Sub-surface Hydrology

Water table monitoring wells were used to measure sub-surface hydrology at two sites of each site type (valley plug, shoal, and unchannelized). Sampling

plots used to measure sub-surface hydrology at the two unchannelized sites (Spring Creek –GVL and Lower Spring Creek) included three transects at plots located within 100 m of the stream channel, for a total of 9 wells per site (Figure 3-1a). At two valley plug sites (Hickory Creek and Jeffers Creek), I installed water wells at each of the sampling plots within 100 m of the stream edge, 150 m upstream of the valley plug, and 50 m downstream of the valley plug, for a total of 30 wells per site (Figure 3-1b). Water wells at both shoal sites (Piney Creek and Porters Creek) were located at every sampling plot within 100 m of the tributary edge and 200 m upstream of the confluence to the Hatchie River, for a total of 15 wells per site (Figure 3-1c).

The water wells were drilled with a 10 cm diameter auger to a depth of 1.5 m. PVC pipe of 10 cm in diameter was placed in the wells to a depth of 1.5 m below the ground surface. Holes of 7 mm in diameter were drilled into the PVC pipe to allow ground water to enter (Nakamura et al. 2002). The holes were covered with fine mesh screen to prevent sediment filling of the wells. Both ends of the PVC pipe were sealed with plastic caps. These wells were used to measure the water table level from June to October in 2003 and May to September in 2004. I collected approximately weekly measurements of sub-surface water levels, however, some floods prevented access to the sites for weekly measurements. Groundwater levels were measured using a float to determine the groundwater level and then I measured the depth from the ground surface to the water level.

At two valley plug sites (Hickory Creek and Jeffers Creek), two shoal sites (Piney Creek and Porters Creek), and one unchannelized site (Spring Creek – GVL), I determined the cross-sectional channel profiles according to methods detailed by Harrelson et al. (1994). One profile was measured at each transect (Figure 3-1). Each profile was measured twice, once in August of 2003 and once in August of 2004. These data were used to determine the average channel depth from the average bank height and the amount of channel elevation change from 2003 to 2004. This information was used to determine the relationship of channel depth, channel elevation change, and water tables.

Analysis

Graphs of surface flooding were constructed for each study site using Excel (Microsoft Corporation 2000). Individual floods were defined as surface inundation occurring at the site with at least two days of no surface inundation between flood events. Surface-water data were used to calculate the duration of each flood, frequency of floods, and the mean and maximum flood depths at all sites by year and during the growing season (April through September). The defined growing season was based on the average last and first occurrences of 0° C temperatures in spring and fall (Flowers 1964, AMS 2000) and patterns of tree species budbreak (McGee 1986). Site-level analyses of surface hydrology included ANOVA tests to determine statistical significant differences in flood duration, flood frequency, and flood depth among site types (valley plug, shoal, and unchannelized).

Graphs of sub-surface water levels were produced for each study site using Excel (Microsoft Corporation 2000). Mean depth to tree root collars, determined from the dendrogeomorphic study (Chapter 2), were compared to the graphs of groundwater level, to estimate time periods in which root systems were inundated to the root collar level. For valley plug sites and unchannelized sites, mean root collar depth was estimated as the mean of all samples at those sites. For shoal sites, the mean root collar depth was estimated as the mean from both valley plug and unchannelized site plots because no data existed for shoal sites.

Means and the standard deviations of sub-surface water levels were compared among site types using ANOVA. Data collected from all sites on sediment deposition rate (Chapter 2), mean channel depth, and mean change in channel depth from 2003 to 2004 were considered in a stepwise multiple regression model with mean water table level of each transect as the dependent variable.

Kruskal-Wallis tests were used in cases where ANOVA assumptions were not valid, and Tukey-Kramer multiple comparison tests or Dunnett's two-sided multiple comparison tests were used to distinguish differences among groups (Alpha = 0.05) (Sokal and Rohlf 1995). Statistical analyses were conducted with SAS Version 9.1 (SAS Institute Inc. 2004) and NCSS (Hintze 2001).

Results

Surface Hydrology

Graphs of floodplain surface inundation at valley plug sites (Figure 3-2) illustrate floods of shorter duration and lower depths than floods at either shoal

sites (Figure 3-3) or unchannelized sites (Figure 3-4). Shoal site inundation graphs (Figure 3-3) indicate floods of similar duration and depth to that of floods at unchannelized sites (Figure 3-4). There were no significant differences in the frequency of floods among valley plug ($\bar{x}$ = 12.6 ± 1.76 floods/yr), shoal ($\bar{x}$ = 15.5 ± 3.5 floods/yr), and unchannelized sites ($\bar{x}$ = 8.2 ± 1.93 floods/yr) (N = 12, df = 2, F = 2.57, P = 0.131).

Mean flood depth differed among the three valley plug sites (N = 63, df = 2, F = 5.81, P = 0.005) as did maximum flood depth (N = 63, df = 2, F = 6.10, P = 0.004). At the Bear Creek valley plug site, both the mean flood depth ($\bar{x}$ = 5.59 ± 0.61 cm) and maximum flood depth ($\bar{x}$ = 10.84 ± 1.39 cm) were greater than at the Hickory Creek or Jeffers Creek valley plug sites (Figure 3-5a). There was no difference in mean flood depth (N = 31, df = 1, T = 1.5, P = 0.14) or maximum flood depth (N = 31, df = 1, T = 0.62, P = 0.54) between the two shoal sites (Figure 3-5b). At unchannelized sites, mean flood depth differed among sites (N = 41, df = 2, F = 8.40, P < 0.001) but the maximum flood depth did not differ (N = 41, df = 2, F = 2.81, P = 0.07) (Figure 3-5c). At the Lower Spring Creek unchannelized site, the mean flood depth ($\bar{x}$ = 34.87 ± 7.69 cm) was greater than at both the Spring Creek-Sain site ($\bar{x}$ = 4.27 ± 0.89 cm) and the Spring Creek-GVL site ($\bar{x}$ = 9.95 ± 2.77 cm).

Mean flood depth also varied significantly among site types (N = 135, df =2, F = 13.39, P < 0.001) (Figure 3-6). At valley plug sites ($\bar{x}$ = 4.31 ± 0.40 cm), mean flood depth was lower than the flood depth at both shoal ($\bar{x}$ = 25.53 ± 5.35 cm) and unchannelized sites ($\bar{x}$ = 18.95 ± 4.06 cm). Maximum flood depth also

differed among site types (N = 135, df = 2, F = 11.45, P < 0.001) (Figure 3-6). Valley plug sites had the lowest mean maximum flood depth of 7.92 ± 0.89 cm and differed from both shoal ($\bar{x}$ = 48.45 ± 12.21 cm) and unchannelized sites ($\bar{x}$ = 41.83 ± 8.71 cm). To factor out the influence of potential flooding from the Hatchie at the confluence to tributaries, mean flood depth and maximum flood depth were compared between valley plug sites and the unchannelized sites of Spring Creek-Sain and Spring Creek-GVL, which are located at least 3 km upstream of the confluence to the Hatchie River. In this case, valley plug sites still had significantly lower mean flood depth (N = 86, df = 1, T = 2.12, P = 0.037) and maximum flood depth (N = 86, df = 1, T = 3.61, P < 0.001).

Flood duration did not differ among sites of the same type, however, total duration of flooding per year did differ among site types (N = 12, df = 2, F = 12.17, P = 0.003) (Figure 3-7). Valley plug sites had the lowest mean number of days flooded per year at 94 ± 34.18 days, which differed from both the shoal sites ($\bar{x}$ = 284.5 ± 13.5 days) and unchannelized sites ($\bar{x}$ = 256 ± 19.9 days). The mean duration of each flood event also differed among site types (N = 135, df = 2, X^2 = 12.43, P = 0.002) (Figure 3-7). Valley plug sites ($\bar{x}$ = 7.46 ± 1.91 days) had lower mean flood event duration than unchannelized sites ($\bar{x}$ = 31.22 ± 10.65 days). However, shoal sites ($\bar{x}$ = 18.35 ± 6.54 days) did not differ in mean flood duration from either valley plug sites or unchannelized sites.

During the defined growing season from April through September, the mean flood depth differed among site types (N = 73, df =2, F = 7.93, P < 0.001) (Figure 3-8). At valley plug sites, the mean flood depth during the growing

season ($\bar{x}$ = 5.09 ± 0.74 cm) was less than both shoal ($\bar{x}$ = 30.87 ± 6.62 cm) and unchannelized sites ($\bar{x}$ = 20.66 ± 5.05 cm). The maximum flood depth during the growing season (Figure 3-8) also differed among site types (N= 73, df = 2, F = 7.78, P = 0.001). The maximum flood depth at valley plug sites ($\bar{x}$ = 8.04 ± 1.21 cm) was significantly lower than both shoal ($\bar{x}$ = 50.86 ± 12.09 cm) and unchannelized sites ($\bar{x}$ = 32.77 ± 7.43 cm).

The total duration of flooding per year during the growing season also differed among site types (N = 12, df = 2, F = 14.5, P = 0.002) (Figure 3-9). Valley plug sites ($\bar{x}$ = 27.2 ± 11.52 days) were flooded the fewest number of days, while shoal sites ($\bar{x}$ =125 ± 5 days) and unchannelized sites ($\bar{x}$ = 101.6 ± 13.21 days) did not differ in total days flooded during the growing season. The mean duration of each flood event during the growing season differed among site types (N = 73, df = 2, X^2 = 6.41, P = 0.04) (Figure 3-9). A Dunnett's two-sided multiple comparison test showed that during the growing season, shoal sites ($\bar{x}$ = 10.87 ± 2.42 days) did not differ in flood duration from either valley plug sites ($\bar{x}$ = 5.04 ± 1.03 days) or unchannelized sites ($\bar{x}$ = 17 ± 5.15 days), but mean flood duration did differ between valley plug sites and unchannelized sites.

Sub-surface Hydrology

Graphs of water table depth for valley plug sites indicated that at the Hickory Creek valley plug site (Figure 3-10a) groundwater was above the mean root collar depth (40.5 cm) on 25 days in 2003 and 53 days in 2004. The Jeffers Creek Valley plug site (Figure 3-10b) had no days in 2003 and 52 days in 2004 in which the groundwater inundation was above the root collar depth (47.8 cm).

The groundwater level never reached the root collar depth (37 cm) in either year at either shoal site (Figure 3-11a and b). At the Spring Creek – GVL unchannelized site (Figure 3-12a), the groundwater reached the root collar depth (17.3 cm) for two days in 2003 and 41 days in 2004. The Lower Spring Creek unchannelized site (Figure 3-12b) was similar, with groundwater levels at or above the root collar depth (17.3 cm) for six days in 2003 and 25 days in 2004.

There was considerable variation in mean depth and standard deviation in water tables between sites of the same type. At valley plug sites, the Hickory Creek and Jeffers Creek sites differed in mean water table depth ($N = 60$, $df = 1$. $T = 13.64$, $P < 0.001$) and standard deviation in water table levels ($N = 60$, $df = 1$, $T = 2.88$, $P = 0.005$). The Jeffers Creek site had a greater mean depth to the water table ($\bar{x} = 1.26 \pm 0.03$ m) and a smaller standard deviation ($\bar{x} = 0.16 \pm 0.01$ m) than the mean water table depth ($\bar{x} = 0.57 \pm 0.04$ m) and standard deviation ($\bar{x} = 0.21 \pm 0.01$ m) at the Hickory Creek site. The two shoal sites did not differ in mean water table depth ($N = 30$, $df = 1$, $T = 0.24$, $P = 0.81$) but the standard deviation in water table depths did differ ($N = 30$, $df = 1$, $T = 2.54$, $P = 0.02$). The Porters Creek shoal site ($\bar{x} = 0.13 \pm 0.02$ m) had significantly lower standard deviation in water table levels than the Piney Creek shoal site ($\bar{x} = 0.21 \pm 0.02$ m). Unchannelized sites differed in both mean water table depth ($N = 18$, $df = 1$, $T = 3.62$, $P = 0.002$) and standard deviation in water table levels ($N = 18$, $df = 1$, $T = 5.58$, $P < 0.001$). The Lower Spring Creek unchannelized site had a greater mean depth to the water table ($\bar{x} = 0.79 \pm 0.10$ m) and a greater standard deviation in water table levels ($\bar{x} = 0.36 \pm 0.08$ m) than the mean water

table depth ($\bar{x}$ = 0.38 ± 0.07 m) and water table standard deviation ($\bar{x}$ = 0.15 ± 0.07 m) at the Spring Creek-GVL unchannelized site.

Mean depth to the water table also differed among site types (N = 108, df = 2, X^2 = 22.40, P < 0.001) (Figure 3-13). All site types differed in mean depth to the water table, with unchannelized sites having the shallowest water table at 0.58 ± 0.08m. Water tables at shoal sites were the furthest below the ground surface, at 1.15 ± 0.05 m. The standard deviation of water table levels also differed among site types (N = 108, df = 2, F = 5.19, P = 0.007) (Figure 3-14). The highest standard deviation in water table levels occurred at unchannelized sites ($\bar{x}$ = 0.25 ± 0.03 m), which differed significantly from the standard deviation in water tables at valley plug sites ($\bar{x}$ = 0.18 ± 0.01 m) and shoal sites ($\bar{x}$ = 0.17 ± 0.02 m).

There was a great deal of variability in channel depth among the site types (Figure 3-15) and among cross-sectional profiles at valley plug sites and shoal sites (Figure 3-15b and c). Mean channel depth differed among the three site types in both 2003 (N = 421, df = 2, X^2 = 239.78, P < 0.001) and 2004 (N = 421, df = 2, X^2 = 211.84, P < 0.001). Differences in mean channel depth were the same across both years, with shoals sites having lower channel depths than valley plug sites and unchannelized sites (Figure 3-16).

The effects of mean channel depth, mean channel depth change from 2003 to 2004, the standard deviation of channel depth change, and mean transect sediment deposition rate (Chapter 2) on mean water table depth per

transect were evaluated with stepwise multiple regression. The regression model that best fit the data was:

> Mean Water Table Depth = –0.1639 – (0.0233 • Deposition rate) – (0.8642 • Mean channel profile change from 2003 to 2004) + (0.3394 • Mean channel profile depth)

The model was significant ($N = 21$, $df = 3$, $F = 11.9$, $P < 0.001$) and the adjusted R^2 was 0.62. Most of the variation in water table depth was explained by mean channel profile depth with a partial R^2 of 0.63 ($P < 0.001$). The model was also improved with the addition of mean channel profile change (partial $R^2 = 0.45$, $P = 0.002$) and deposition rate (partial $R^2 = 0.23$, $P = 0.037$).

Discussion

The results of this study clearly support my predictions that valley plug sites have altered surface and sub-surface hydrology relative to unchannelized sites and that shoal sites have altered sub-surface hydrology. However, some of the changes were unexpected based on our current understanding of valley plugs and shoals (Happ et al. 1940, Miller 1990, Diehl 2000). Shankman and Pugh (1992) found that channelization increases flow effficency and reduces flooding in upstream reaches, while the depth and frequency of flooding increases downstream. Shankman and Pugh (1992), however, did not specifically address the effects of valley plug or shoal formation on hydrological processes. Several authors have suggested that flooding would increase around valley plugs as a result of decreased channel storage capacity (Happ et al. 1940, Miller 1990, Shankman and Samson 1991, Diehl 2000). Miller (1990) confirmed these findings as they found open water and marsh communities developing near

some valley plugs. My study, however, indicated that reduced flood duration and flood depth are also associated with valley plugs. Thus, my results emphasize the variability associated with hydrologic processes around valley plugs and our rudimentary understanding of the impacts of these geomorphic features.

Surface Hydrology

The lower flood depth and flood duration at valley plug sites relative to shoal and unchannelized sites may be a result of the age of the valley plugs examined and their stage of development or the variability in responses to valley plugs based on site specific conditions such as position in the basin, basin size, stream gradient or floodplain and channel morphology. Past research (Happ et al. 1940) and field observations suggest that as valley plugs develop, anastomosing channels form, creating new channels in the floodplain for stream flow. Once these channels are formed and floodplain surfaces have stabilized with respect to elevation and deposition rates, these areas typically become swamped with permanent water due to the reduced capacity of the anastomosing channels. My study sites may represent different stages of valley plug formation and may not have had time to develop permanently flooded conditions. For example, landowners have confirmed that the valley plug at Hickory Creek was at least 30 years of age and influenced sediment deposition patterns over 16 ha area of the floodplain. This suggests that the Hickory Creek valley plug is not in a earlier developmental stage and that other factors are likely involved in the reduced flooding that occurred at this site.

Basin size may also explain some differences between my study and others. Although Oswalt (2003) and Miller (1990) found permanent flooded conditions associated with valley plugs, their research was conducted within the middle or lower reaches of a much larger river system. Duration of overbank flooding has been shown to be related to the drainage area of the watershed (Wharton et al. 1982). Larger basins collect and transport more water than small basins, which results in longer duration and higher peak floods. Position in the basin is also important because it determines the effective drainage area, as only the upstream portion of the basin will contribute to possible flooding at a downstream site. Basin size may have also contributed to the differences in flood depth and duration among my site types. The upstream basin sizes of the valley plug sites were smaller than the upstream basin size of the shoal and unchannelized sites (Table 1-1). Thus, even though basin sizes were smaller upstream of the valley plug sites relative to the other sites, I still expected greater flooding because of the large amount of sediment being transported and deposited at the valley plug sites. Nevertheless, this was not found in this study. Instead, floods at valley plugs may be best characterized as floods of short duration, low depth and high velocity. These types of floods would enable the systems to transport and deposit the greater amounts of sediment that are being deposited at the valley plug sites.

The lower depth and duration of flooding that occurred at the valley plug sites may also be a result of increases in elevation of floodplain surfaces as a result of high deposition rates (Chapter 2). Thus, a progressively greater flow is

required to achieve overbank flooding. Furthermore, floodplain features like sloughs and backswamps fill with water to levels greater than most other floodplain surfaces (Wharton et al. 1982). Valley plug sites in this study lacked these depressional areas due to high rates of sedimentation, thus the lowest point of the floodplain, and the location of the water level recorders, was at a higher point on the floodplain at valley plug sites than at shoal and unchannelized sites. The reduced flooding in the growing season at valley plug sites may be a result of processes discussed above for annual flooding, but may also be influenced by the flashiness of these channelized systems and the lack of stream flow during the summer months.

There was no difference in flood depth or flood duration between shoal sites and unchannelized sites, which was not expected based on previous research on the effects of channelization (Emerson 1971, Shankman and Samson 1991, Shankman and Pugh 1992). Confounding factors in this study, however, are the proximity of some sites to the Hatchie River and the upstream basin size of the tributaries to each site. Both shoal sites and the Lower Spring Creek unchannelized site are located at the confluence with the Hatchie River. Mean flood depth at the Lower Spring Creek site was significantly greater than at either of the other two unchannelized sites (Figure 3-5c), and is most likely the result of flooding from the Hatchie River. Because the tributaries of both shoal sites are channelized and the shoal sites' proximity to the Hatchie River, which seems to increase flood depth, one would expect their to be greater flooding at the shoal sites than the unchannelized sites. The lack of difference in flooding

depth and duration at shoal and unchannelized sites may be a result of basin size (Table 1-1). The upstream basin sizes of the tributaries at both shoal sites are smaller relative to those of the unchannelized sites. Thus, while lower reaches of channelized tributaries may experience deeper flooding relative to unchannelized tributaries of similar basin size, large unchannelized basins may have similar floods because of the greater volume of water moving through the watershed.

Sub-surface Hydrology

The results of the sub-surface hydrologic analyses did not support my hypothesis that groundwater levels would be higher at valley plug and shoal sites due to increased rates of channel filling. In fact, groundwater levels were lowest at the shoal sites and groundwater levels at valley plugs were also significantly lower than at unchannelized sites (Figure 3-12). There was also considerable variability, however, in groundwater levels between sites of the same type. Previous work (Tucci and Hileman 1992) has shown that channelization and dredging results in lower groundwater levels in the adjacent floodplain. Channel depth profiles showed that shoal sites had lower channel elevations with respect to the floodplain than either valley plug or unchannelized sites. Rates of annual channel filling at both shoal sites ($\bar{x}$ = 6.18 ± 2.35 cm) may not be increasing channel elevation enough to compensate for the previous lowering of the channel bed caused by channelization, resulting in lower groundwater levels than at the unchannelized sites. Lower channel bed elevations and the lower stream flows

that typically occur during the summer may have also contributed to the lower variation in groundwater levels compared to the unchannelized sites.

Groundwater levels at the valley plug sites may have also been responding in a similar way as shoal sites to channelization and bed level lowering. However, the surface hydrologic analysis and field observations confirm the flashiness of the channelized streams at the valley plug sites. Burt et al. (2002) showed that in some cases, stream flow influenced groundwater levels in the floodplain and that during low stream flow periods there is an influx of groundwater to the stream. The low water flows in channels of the valley plug sites through most of the summer months may be causing a discharge of groundwater from the floodplain to the channel and contributing to the lower groundwater levels and the reduced variation in groundwater levels during the summer.

The estimated mean root collar depth varied among valley plug sites (N = 232, $\bar{x}$ = 42.37 ± 1.5 cm), shoal sites (N = 295, $\bar{x}$ = 37.02 ± 1.34 cm) and unchannelized sites (N = 63, $\bar{x}$ = 17.35 ± 0.98 cm), but there was little variation within the same site type. There was also considerable variation among site types, within site type, and between years in the number of days that groundwater levels reached root collar depths, but I was unable to detect significant differences in duration of root collar flooding by groundwater among site types. However, the number of days that groundwater levels inundated root systems at valley plug sites ($\bar{x}$ = 32.75 ± 12.76 days) was more than 50% greater than the number of days root collars were inundated at unchannelized sites, but

the variability within and among sites masked any significant differences. This may indicate that even though groundwater levels at valley plug sites were lower than at unchannelized sites, root systems at valley plug sites may be inundated for longer periods of time than unchannelized sites as a result of lower root collar depths and higher deposition rates (Chapter 2); however, more research is needed to determine if differences in root inundation really occur among the site types. Prolonged inundation of root systems can cause stress, low seed production, reduced growth, and mortality of some BLH tree species (Happ et al. 1940, Hosner and Boyce 1962, Kozlowski 2002).

In contrast to the groundwater levels at valley plug sites, the groundwater at shoal sites never reached the root collar depths of trees. Access to water by root systems may therefore be limited at shoal sites. Although the impacts of lower water tables has received limited study in BLH floodplains, lower water table levels as a result of channel incision have been shown to cause stress and mortality of some riparian tree species (Scott et al. 1999, Hughes et al. 2000, Scott et al. 2000).

The regression model of groundwater levels explained 62% of the variation in groundwater levels and indicated that mean channel depth accounted for most of the explained variation. These results reaffirm the importance of channel depth to groundwater levels in the floodplain and may be the reason for the variability in groundwater levels between sites of the same type. The model was improved with the addition of channel filling rate and sediment deposition rate, which suggests that sedimentation, both within the channel and on

floodplain surfaces, may also influence groundwater levels. Greater sedimentation rates in the channel may reverse the effects of dredging and increase water tables in the floodplain, and the permeability of sand deposits in the floodplain may contribute to higher water tables because water can fluctuate more freely.

Conclusion

Past research on hydrologic processes associated with BLH forests has focused on broad scale patterns (Wharton et al. 1982, Hupp 2000), reach responses to channelization (Emerson 1971, Shankman and Samson 1991, Shankman and Pugh 1992), and qualitative observations of valley plugs (Happ et al. 1940, Miller 1990, Diehl 2000, Oswalt 2003). This study has provided evidence that valley plugs can affect both surface and sub-surface hydrology in different ways than previously thought. The results of this study, which were unexpected based on previous studies, should not be viewed as contradictory to previous work, but rather as an expansion of our knowledge of the variability in hydrological responses to valley plug formation. Channelization and recovery processes may also be changing the hydrological conditions at valley plug and shoal sites, but our understanding of these relationships is still rudimentary.

The results also indicate that sedimentation rates (Chapter 2), surface, and sub-surface hydrology are highly variable within and among valley plug, shoal, and unchannelized sites. An integrated approach that considers basin size, age of geomorphic features, spatial location of geomorphic features in the watershed, and channel filling rates is needed to further our understanding of

these systems. The mechanisms involved in the creation of permanently flooded areas and in different developmental stages of valley plug formation are still poorly understood. Further research is needed to test hypotheses of surface and sub-surface hydrological response to valley plug formation related to stage of development and specific site conditions, which were beyond the scope of this study, in order to understand the factors influencing the variability of hydrological responses. This information will also be useful for understanding the implications of valley plug formation on BLH forests and important for management and restoration efforts.

APPENDIX 3

Table 3-1. Study tributaries with identification of geomorphic feature studied (Diehl 2000), number of groundwater wells per site, and sampling years.

Site - Tributary	Feature Type	No. Wells	Years
Spring Creek - GVL	Unchannelized	9	2002-04
Spring Creek – Sain	Unchannelized	9	2003-04
Spring Creek - Lower	Unchannelized	0	2002-04
Bear Creek	Valley Plug	0	2003-04
Jeffers Creek	Valley Plug	30	2002-04
Hickory Creek	Valley Plug	30	2002-04
Piney Creek	Shoal	15	2003-04
Porters Creek	Shoal	15	2003-04

A) Unchannelized sites

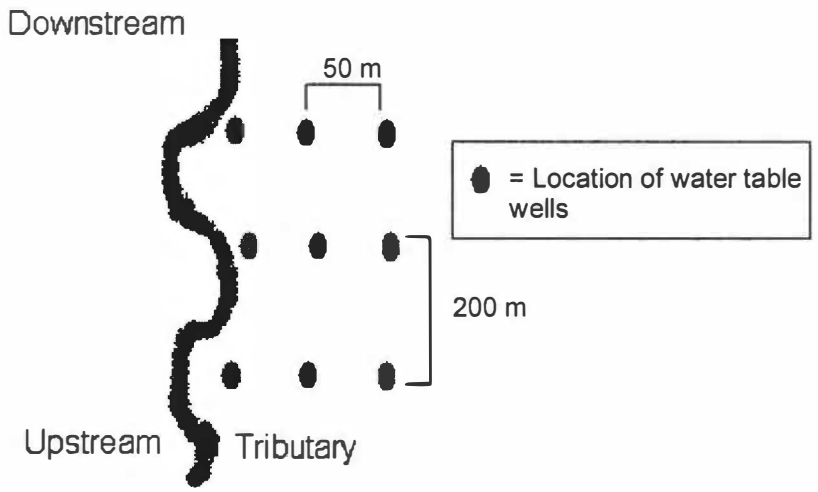


Figure 3-1. Sampling design for study plots used to measure sub-surface hydrology at (a) unchannelized sites, (b) valley plug sites, and (c) shoal sites.

B) Valley plug sites

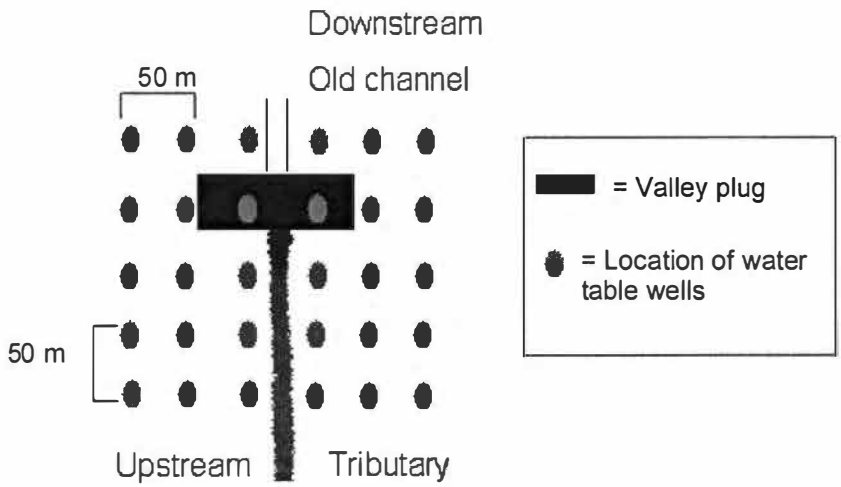


Figure 3-1. Continued

C) Shoal sites

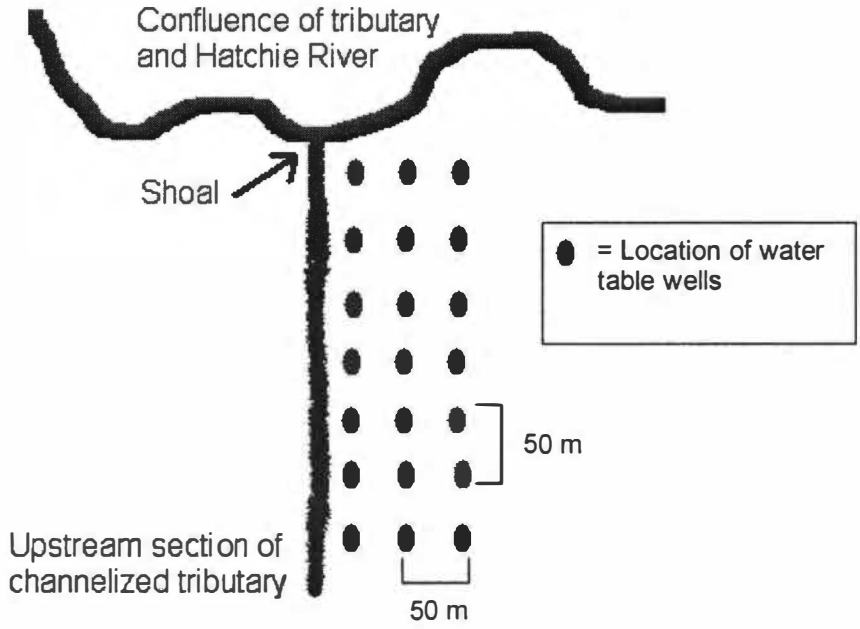


Figure 3-1. Continued

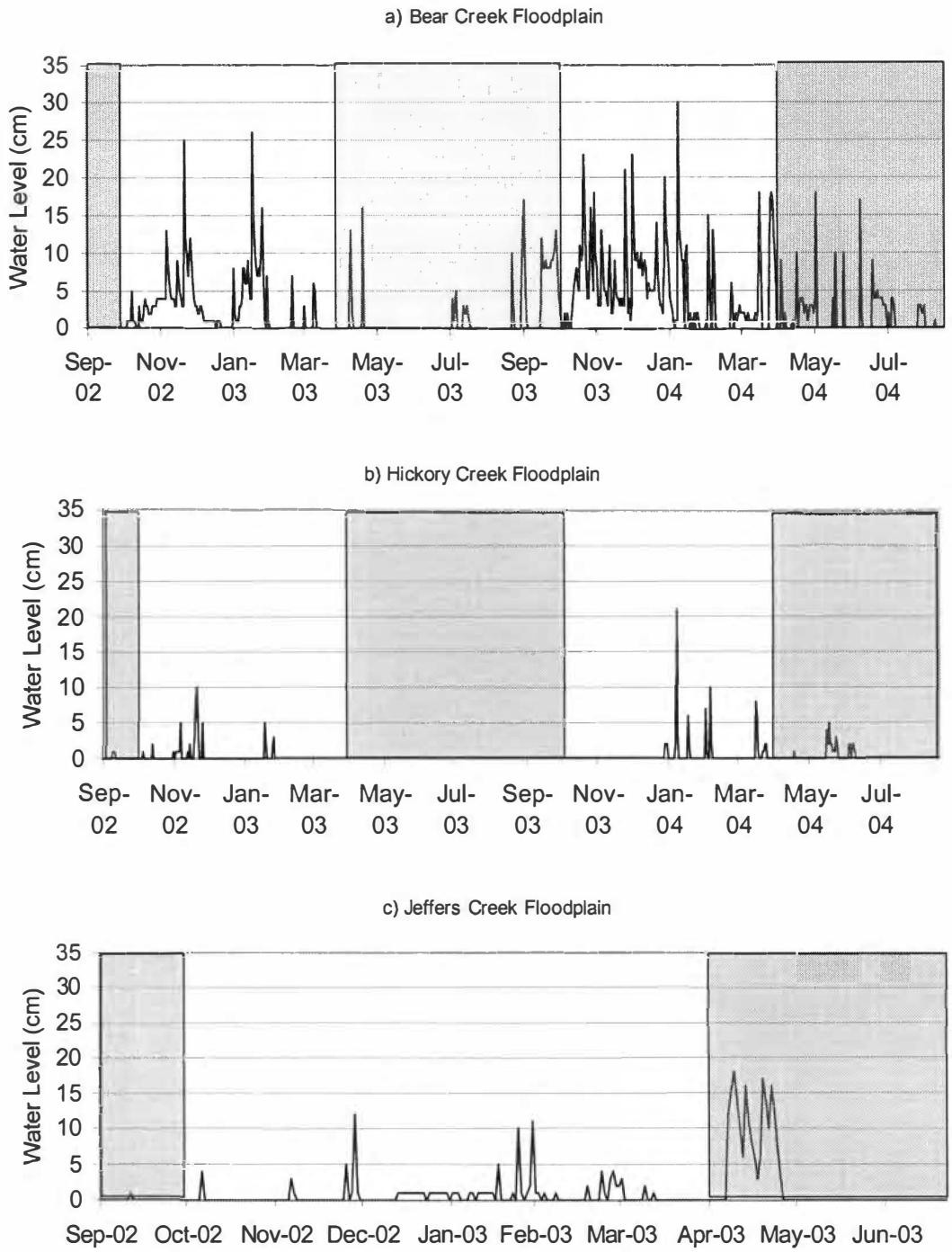


Figure 3-2. Floodplain inundation graphs at the valley plug sites from 2002 to 2004: (a) Bear Creek, (b) Hickory Creek, and (c) Jeffers Creek. Shaded areas indicate time during the growing season. Note that the Jeffers Creek floodplain data are only from 2002 to 2003.

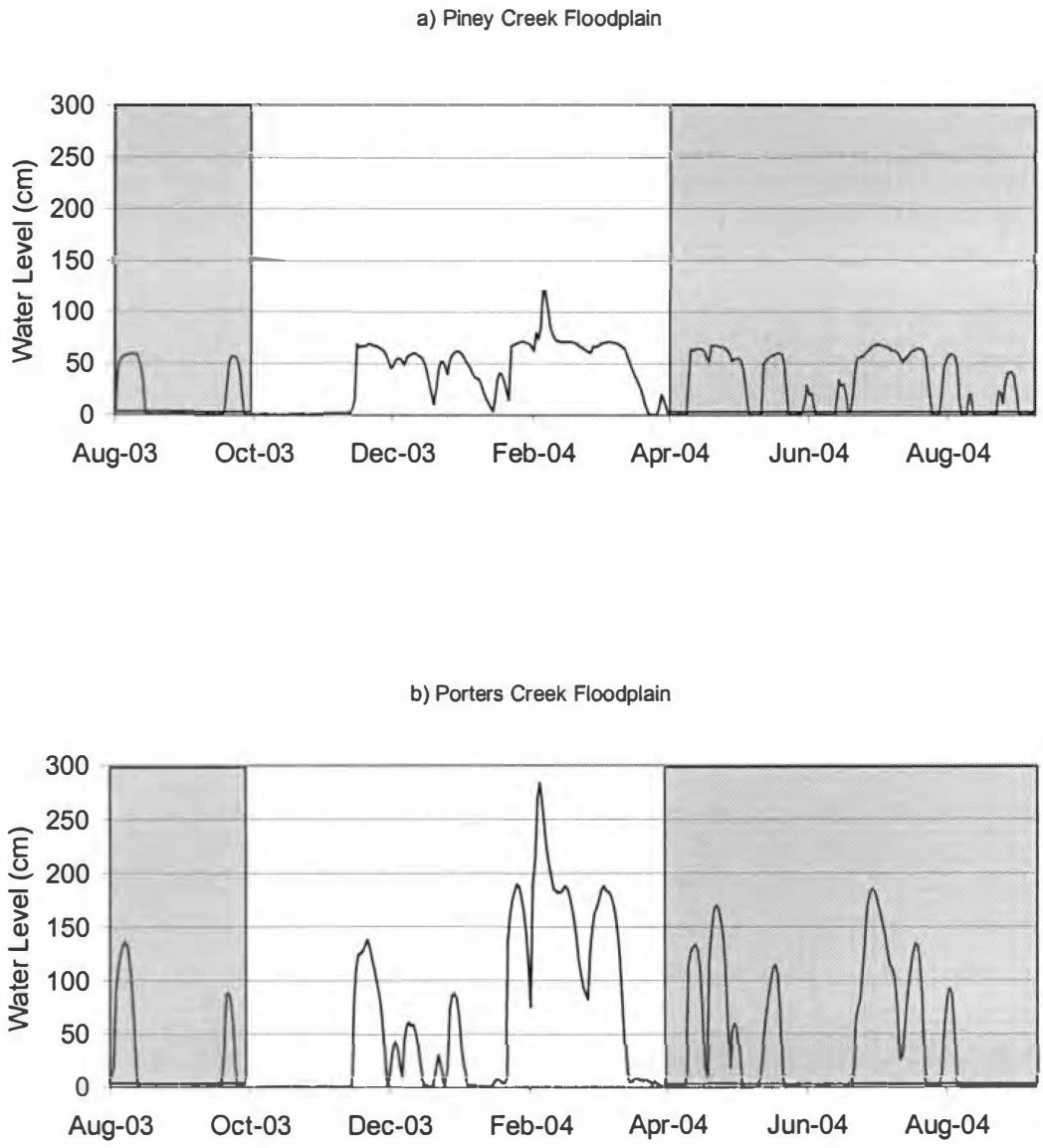
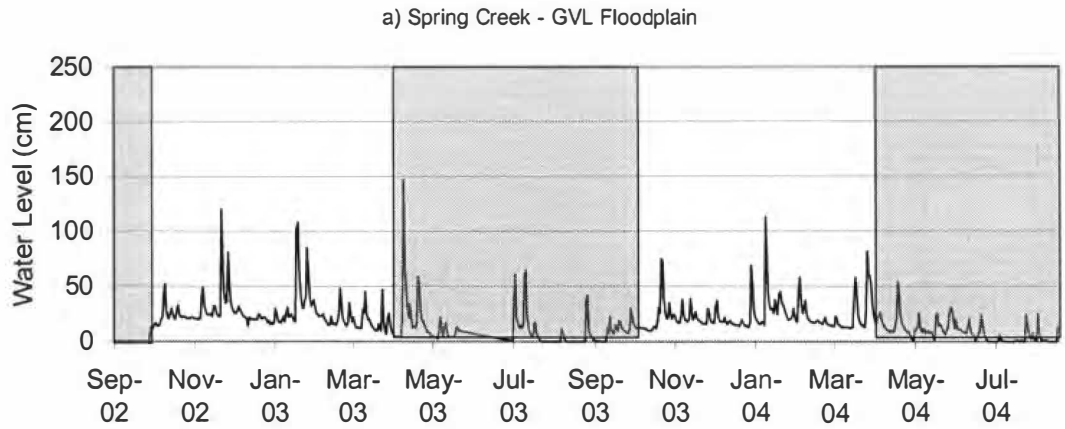


Figure 3-3. Floodplain inundation graphs at the shoal site floodplains from 2003 to 2004: (a) Piney Creek and (b) Porters Creek. Shaded areas indicate time during the growing season. Note that y-axis scale for shoal sites is different from that used for valley plug and unchannelized sites.



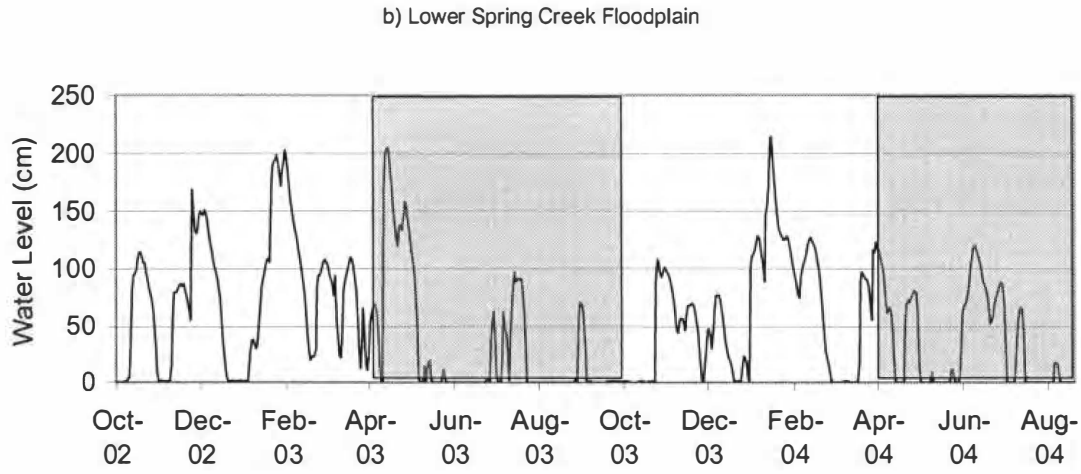


Figure 3-4. Floodplain inundation graphs at the unchannelized site floodplains from 2002 to 2004: (a) Spring Creek –GVL, (b) Lower Spring Creek, and (c) Spring Creek –Sain. Shaded areas indicate time during the growing season. Note that y-axis scale for unchannelized sites is different from that used for valley plug and shoal sites.

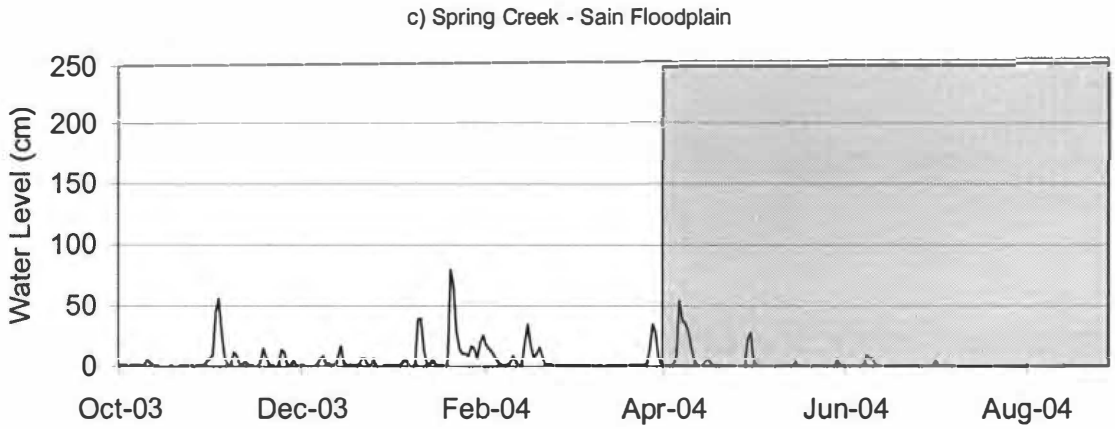


Figure 3-4. Continued

A) Valley plug sites

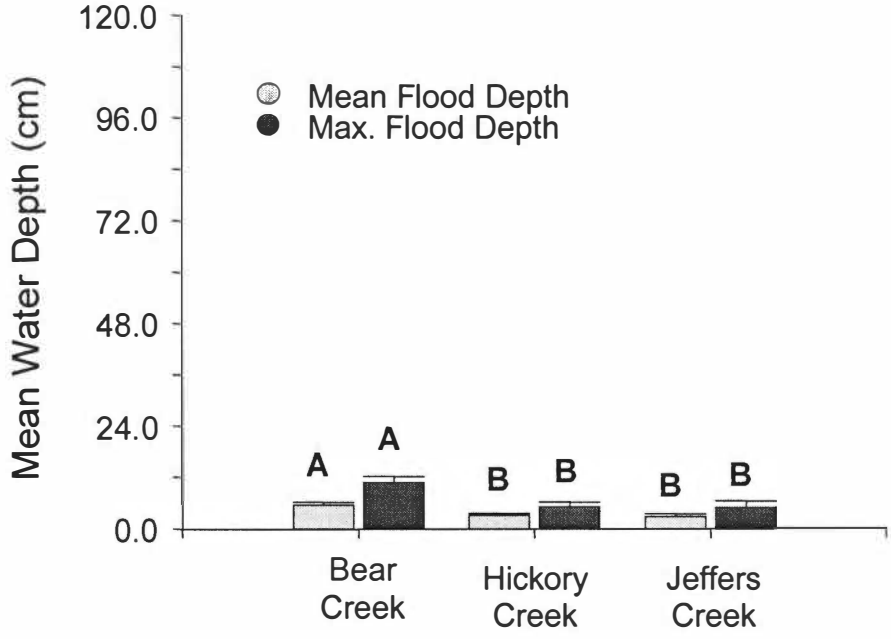


Figure 3-5. Mean flood depth (cm) (+ 1 standard error) and mean maximum flood depth (cm) (+ 1 standard error) by sites of the same type from 2002 to 2004: (a) valley plug sites, (b) shoal sites, and (c) unchannelized sites. Bars within each series that have unlike letters are different ($P < 0.05$).

B) Shoal sites

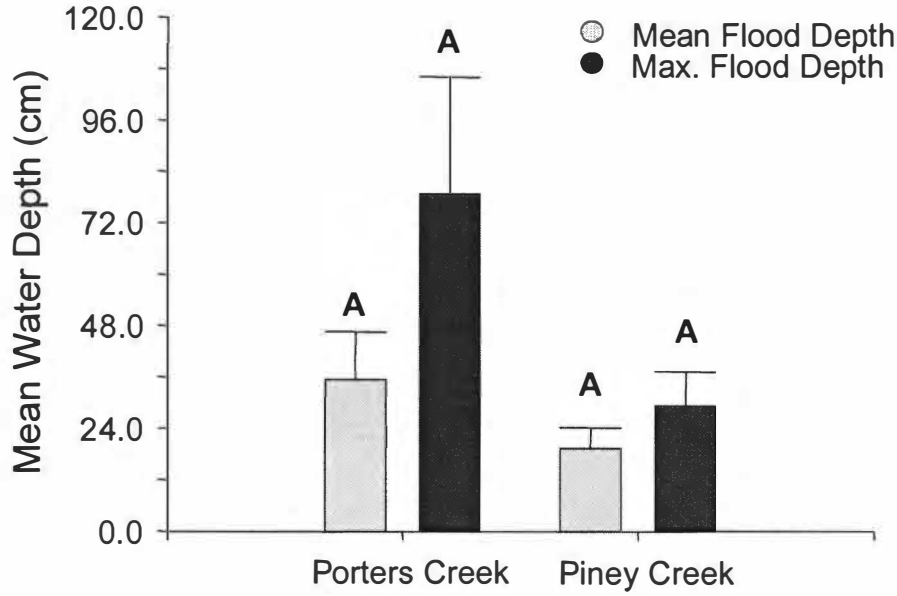


Figure 3-5. Continued.

C) Unchannelized sites

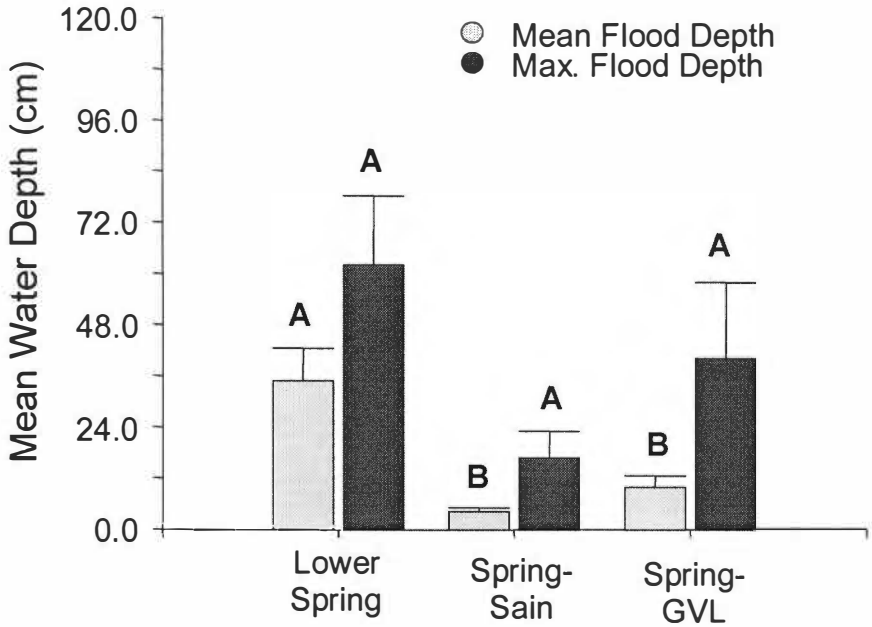


Figure 3-5. Continued.

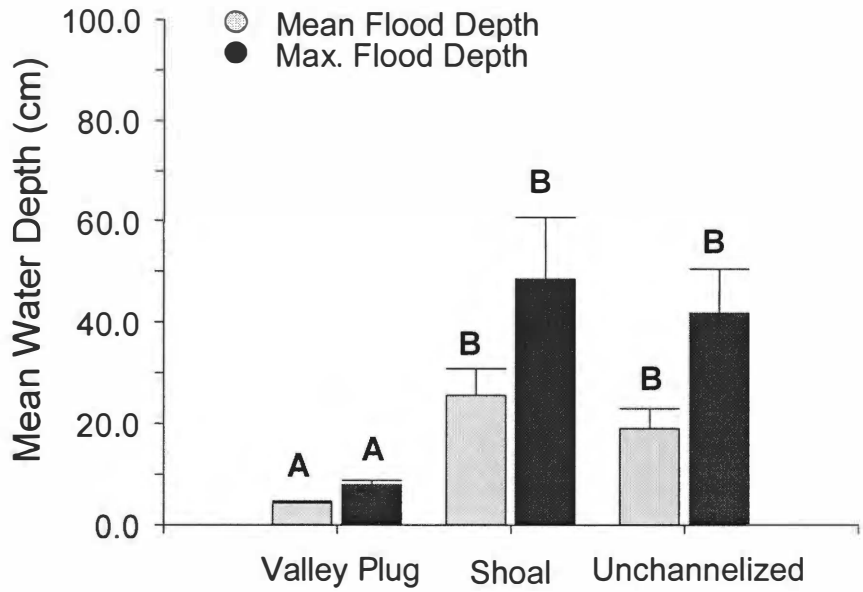


Figure 3-6. Mean flood depth (cm) (+1 standard error) and mean maximum flood depth (cm) (+1 standard error) by site type from 2002 to 2004. Bars for each variable that have unlike letters are different ($P < 0.05$).

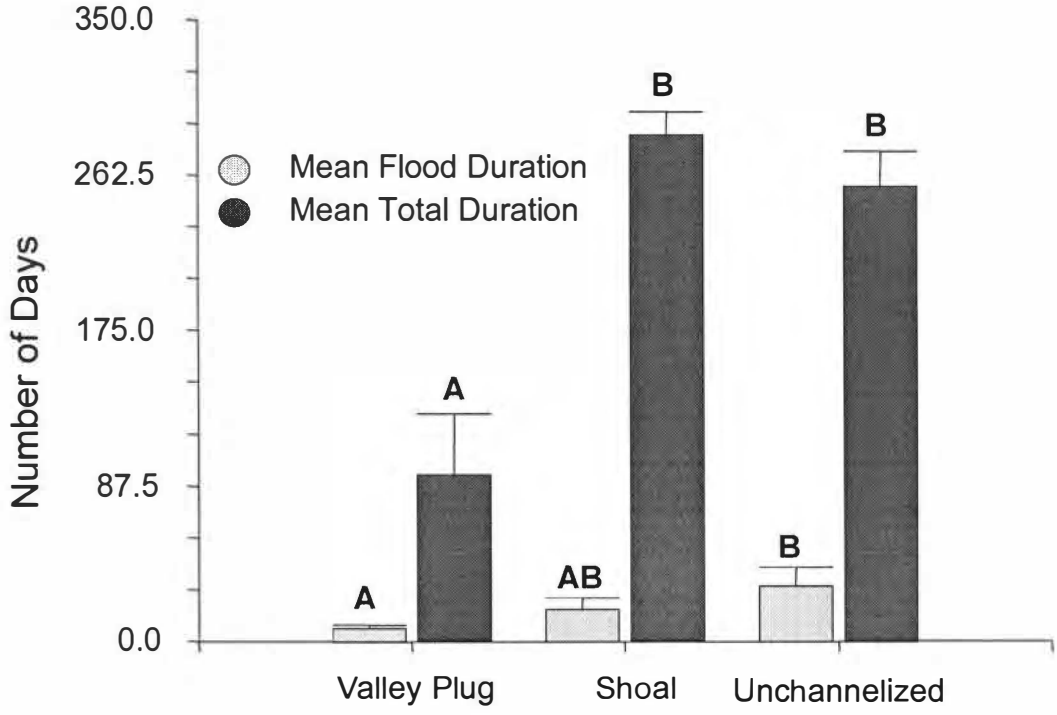


Figure 3-7. Mean total number of days flooded per year (+1 standard error) and mean duration of each flood event (days) (+1 standard error) by site type. Bars within each series that have unlike letters are different ($P < 0.05$).

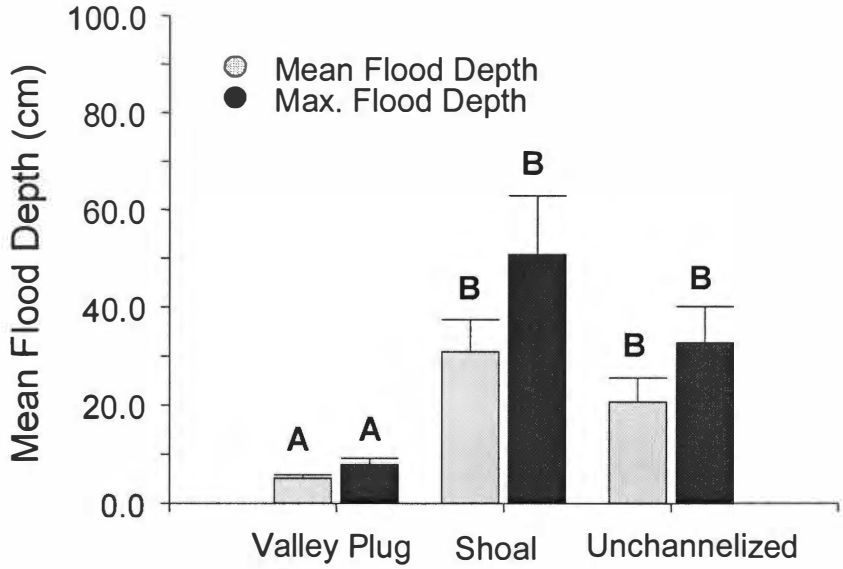


Figure 3-8. Mean flood depth (cm) (+1 standard error) and mean maximum flood depth (cm) (+1 standard error) during the growing season (April to October) by site type. Bars for each variable that have unlike letters are different ($P < 0.05$).

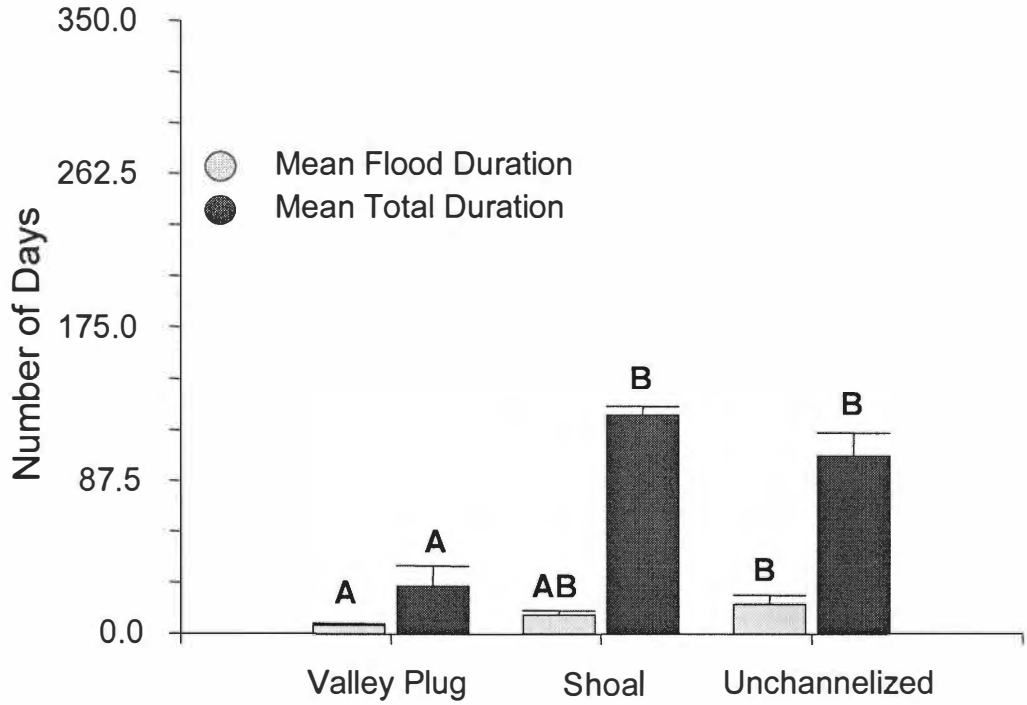
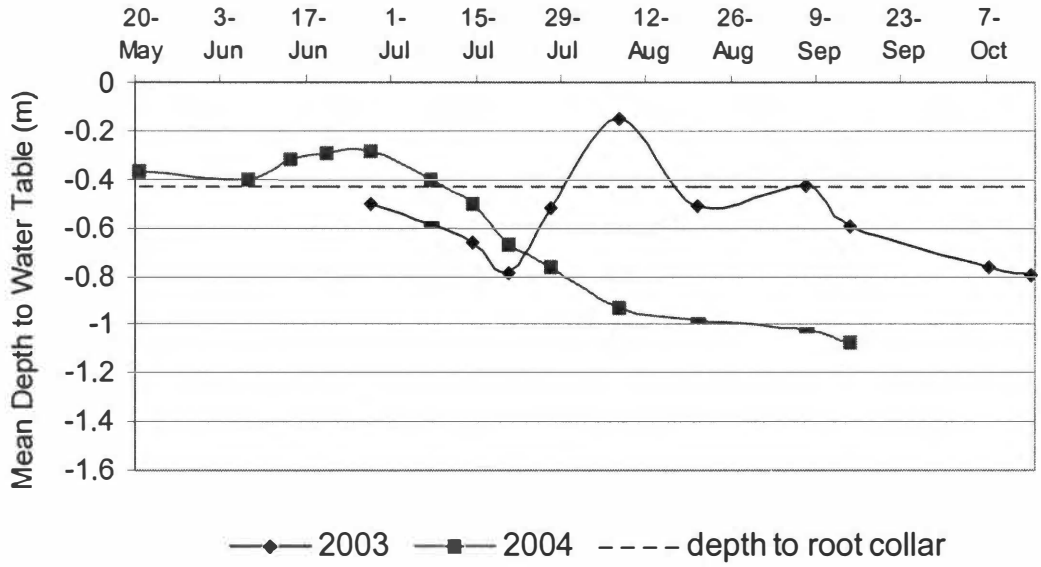


Figure 3-9. Mean total number of days flooded per year (+1 standard error) and mean duration of each flood event (days) (+1 standard error) during the growing season (April to October) by site type. Bars within each series that have unlike letters are different ($P < 0.05$).

A) Hickory Creek



B) Jeffers Creek

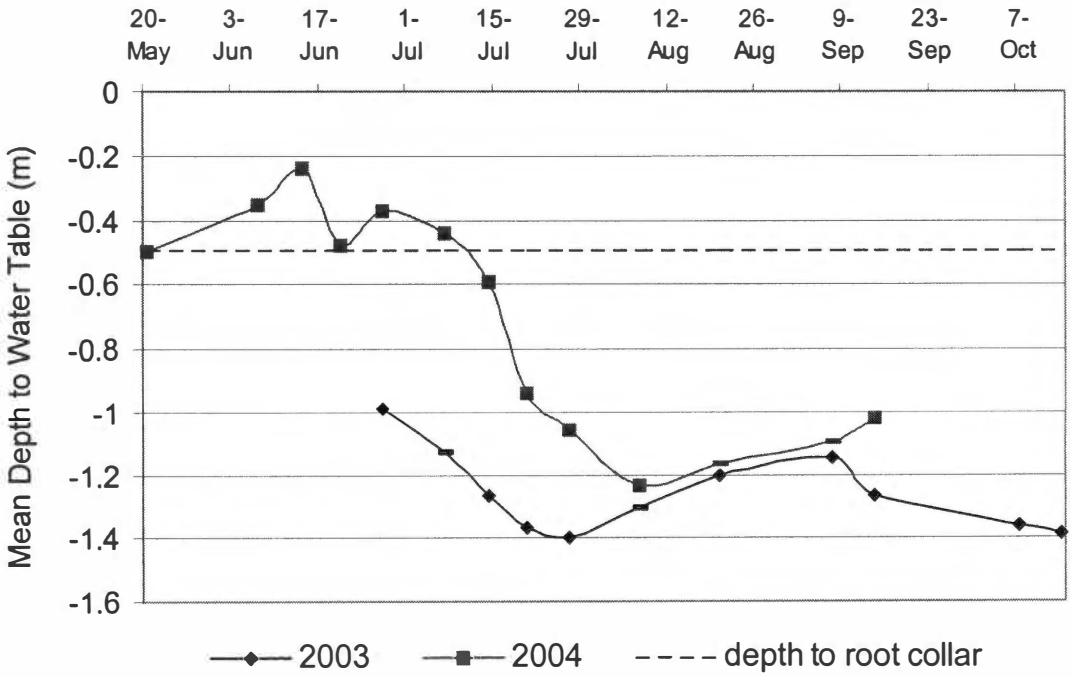
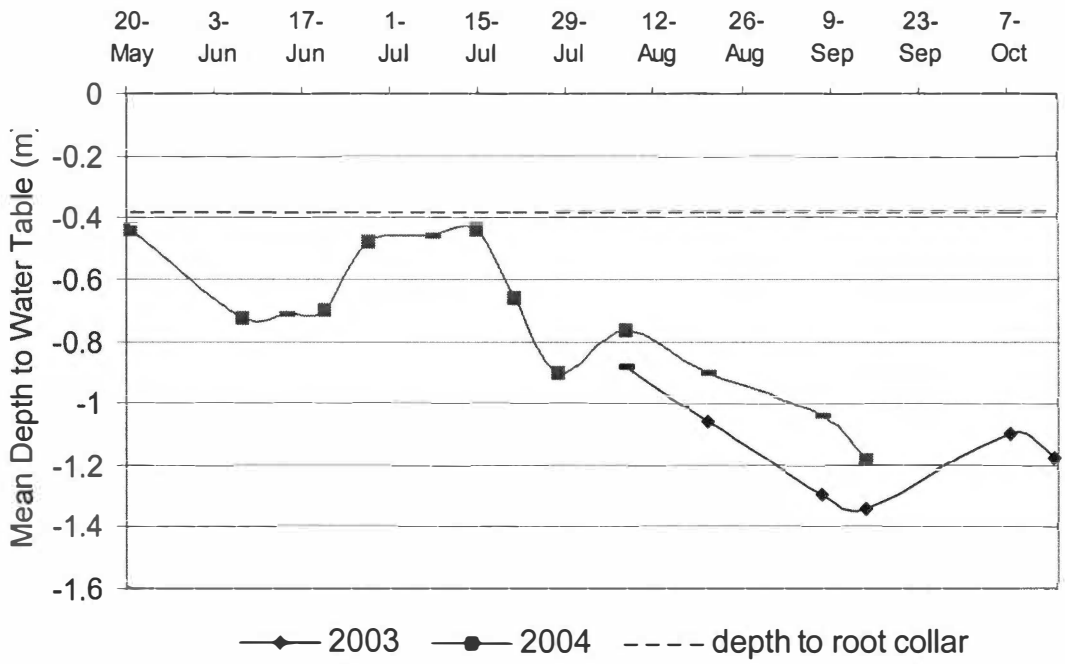


Figure 3-10. Mean water table depths (m) at valley plug sites in 2003 and 2004: (a) Hickory Creek and (b) Jeffers Creek. Average root collar depth is also indicated.

A) Piney Creek



B) Porters Creek

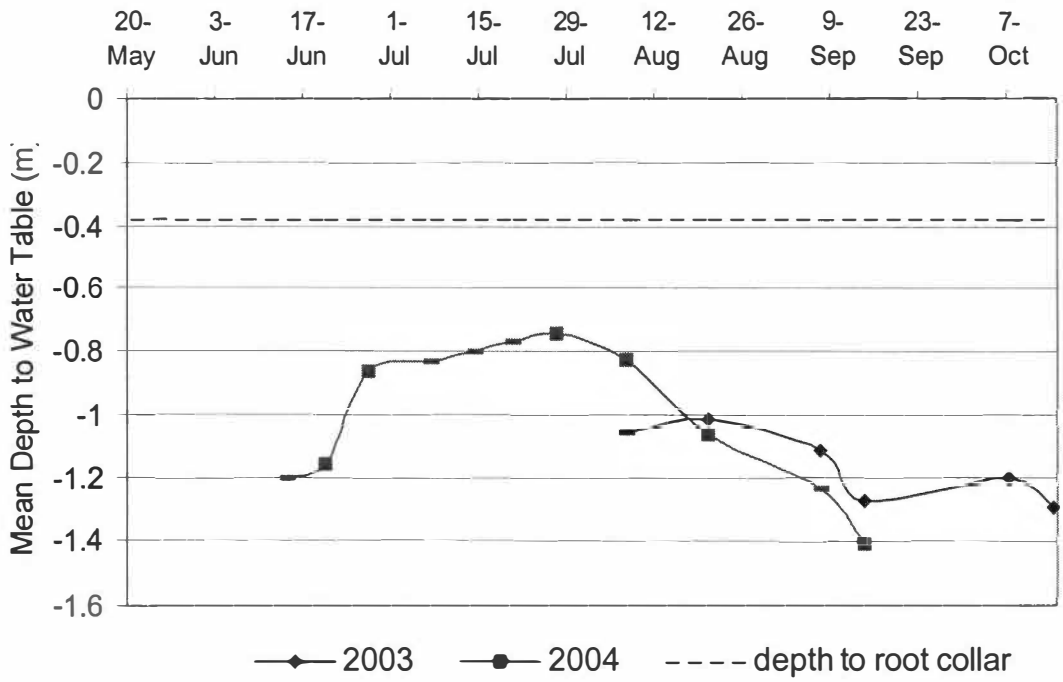
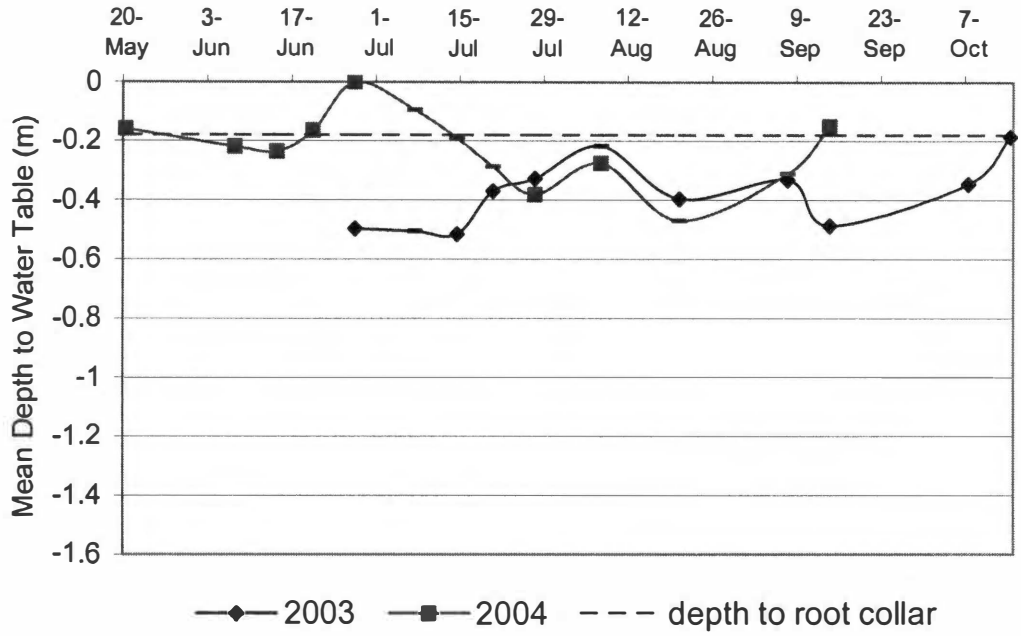


Figure 3-11. Mean water table depths (m) at shoal sites in 2003 and 2004: (a) Piney Creek and (b) Porters Creek. Average root collar depth is also indicated.

A) Spring Creek –GVL



B) Lower Spring Creek

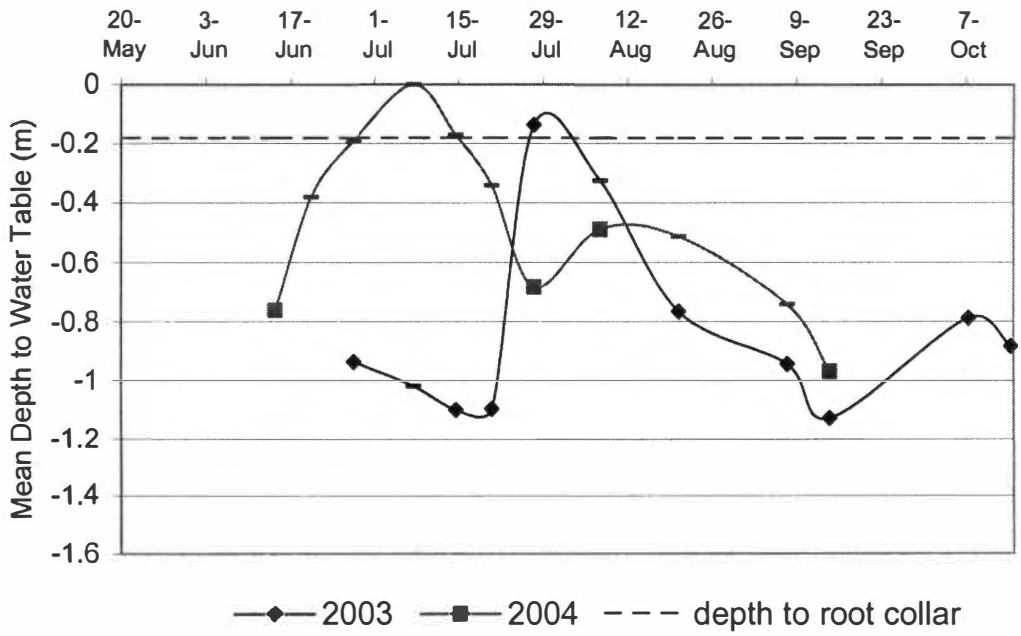


Figure 3-12. Mean water table depths (m) at unchannelized sites in 2003 and 2004: (a) Spring Creek – GVL and (b) Lower Spring Creek. Average root collar depth is also indicated.

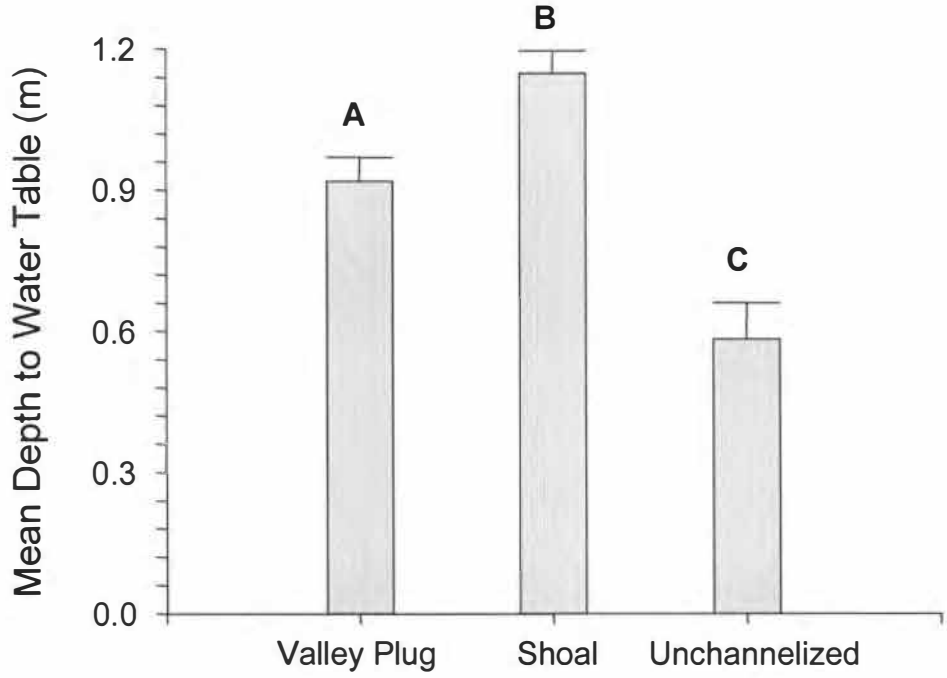


Figure 3-13. Mean depth to water table (m) (+1 standard error) by site for each site type for 2003 and 2004. Bars that have unlike letters are different ($P < 0.05$).

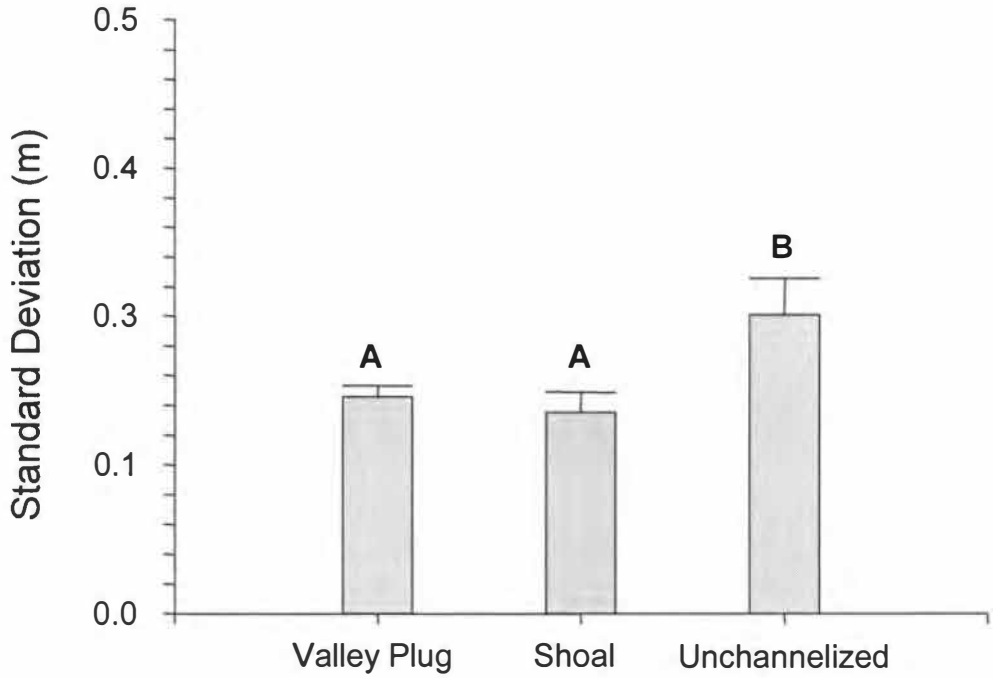


Figure 3-14. Mean standard deviation of water table depth (m) (+1 standard error) by site for each site type for 2003 and 2004. Bars that have unlike letters are different ($P < 0.05$).

A) Spring Creek – GVL (unchannelized site)

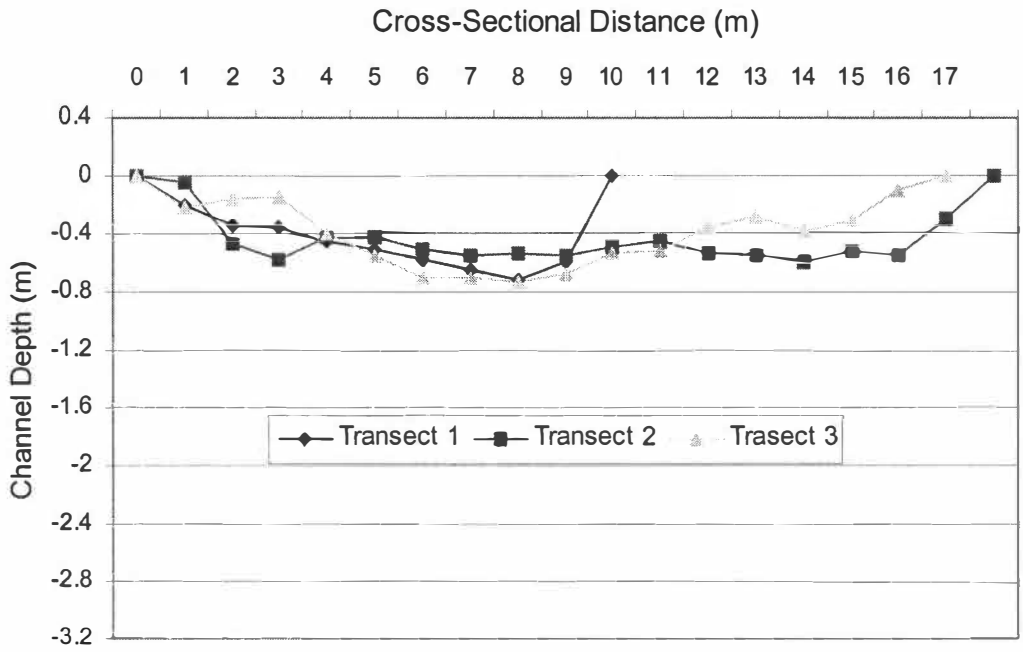


Figure 3-15. Examples of the cross-sectional channel profiles at (a) unchannelized site (Spring Creek-GVL), (b) a shoal site (Porters Creek), and (c) a valley plug site (Hickory Creek) in 2003. The legend identifies the transect profile and distance upstream of the shoal and valley plug of each profile series, with vp indicating the profile at the valley plug.

B) Porters Creek (shoal site)

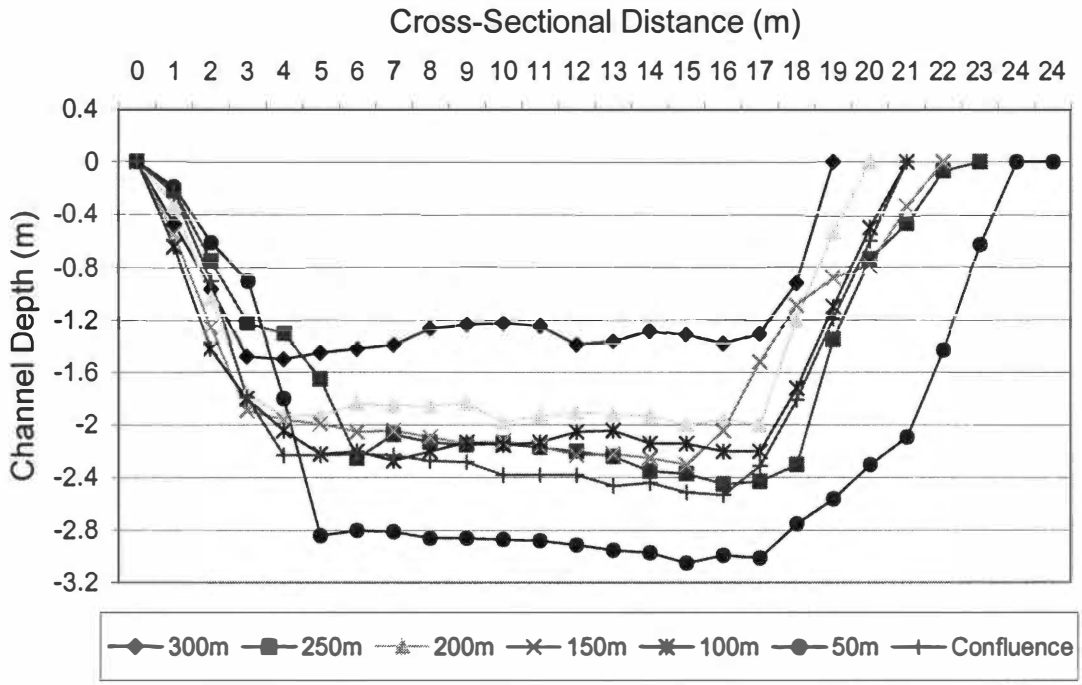


Figure 3-15. Continued

C) Hickory Creek (valley plug site)

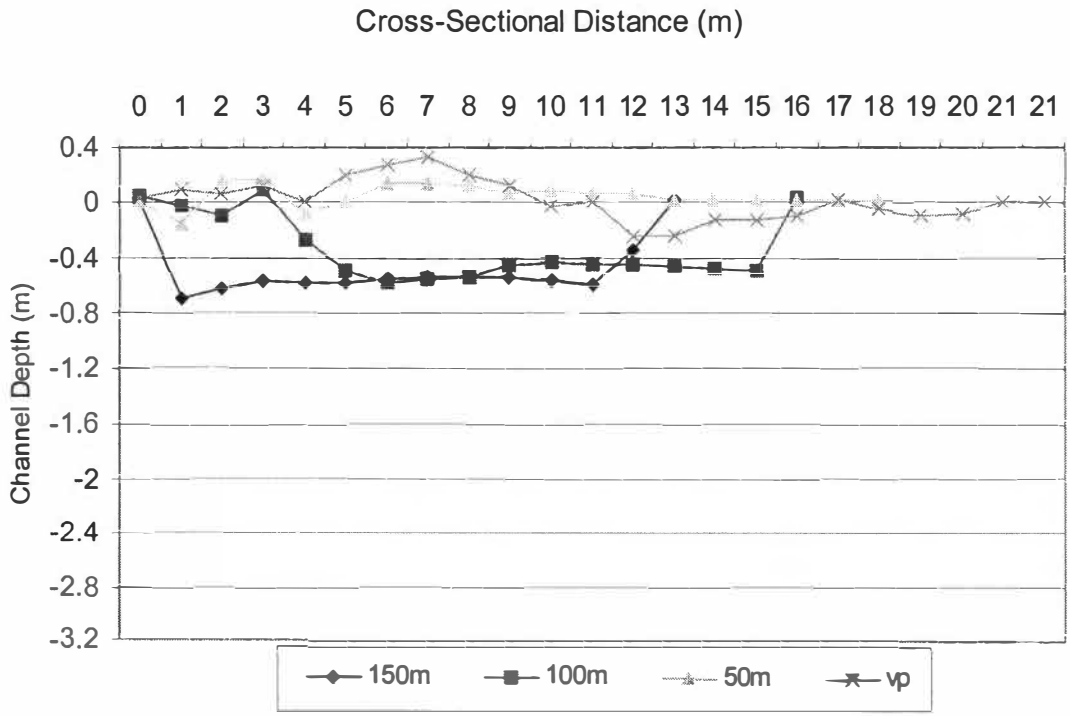


Figure 3-15. Continued

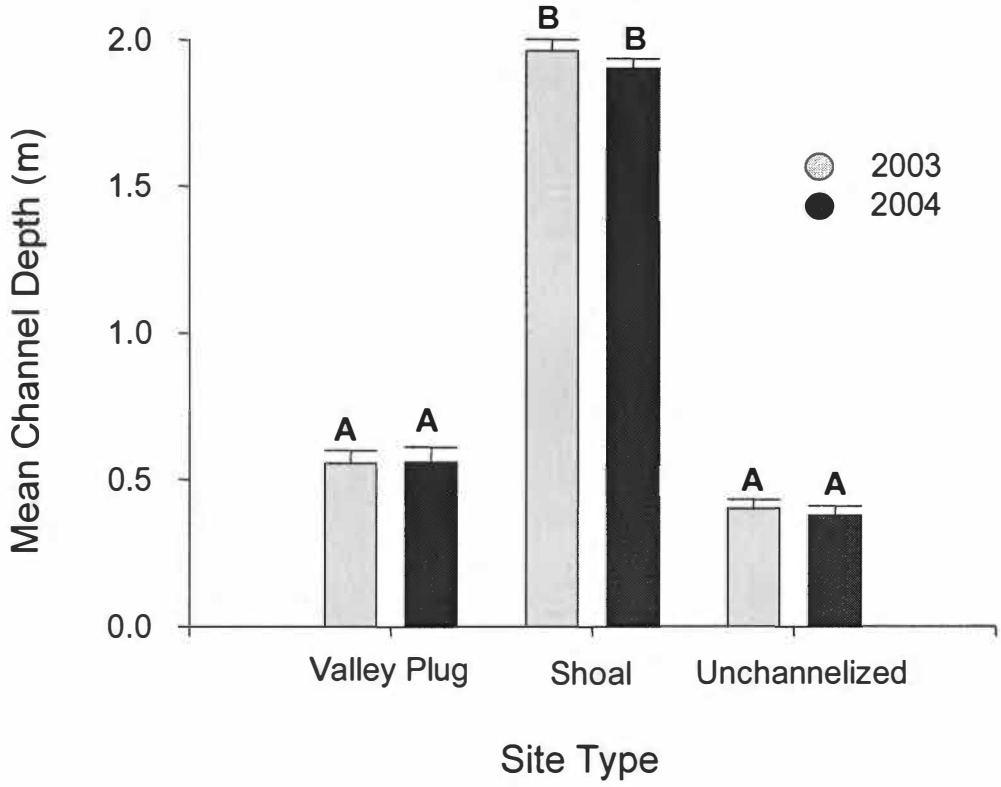


Figure 3-16. Mean channel profile depth (m) (+1 standard error) by site type for 2003 and 2004. Bars that have unlike letters are different ($P < 0.05$).

PART IV

THE EFFECTS OF FLOODING AND SEDIMENTATION ON SEED GERMINATION OF THREE BOTTOMLAND HARDWOOD TREE SPECIES

Introduction

Riparian forests support recognizably distinct assemblages of plants that are associated with particular landforms, soils, and hydrologic regimes (Hodges and Switzer 1979, Mitsch and Gosselink 2000). Plant succession on floodplain sites is directed by both autogenic and allogenic processes (Hodges 1997). Although hydroperiod and light availability are primary determinants of plant species composition (Hall and Harcombe 1998), the type and rate of sediment deposition can influence the composition and successional patterns of floodplain forests (Hodges 1997).

Site quality, germination, and survival of some tree species can change over time due to the influence of small changes in elevation and sediment deposition (Jones et al. 1994, Hodges 1997). Bottomland hardwood (BLH) community level changes, described by Hodges (1997), are a result of differential germination rates and survival of seedlings and overstory trees under the interrelated processes of hydrology and sedimentation.

In the United States, especially in the Lower Mississippi River Alluvial Valley (LMAV), human alteration of hydrology and past and present land use activities have degraded and destroyed wetlands including BLH forests (Pringle 2000). Channelization has occurred extensively throughout the southeastern United States (Shankman 1993). Channelization can reduce flooding in the upper reaches of a system while increasing the peak stage of floods and flood frequency in the downstream reaches (Shankman and Pugh 1992). Channelization can also change the sediment dynamics of the system; however

impacts vary depending on the stretch of river being examined. Typically, channelization causes erosion in the upper reaches and accelerated deposition in the lower reaches of the system (Schumm et al. 1984).

Previous studies have suggested that the influence of environmental factors, especially flooding and sedimentation, on species composition begins with the germination process (Briscoe 1961, Outcalt 2002). Spatial and temporal variation in tree species composition have been found to be influenced mainly by a species' ability to reproduce rapidly during periods of low stress and its germination abilities under the influence of stress (Streng et al. 1989).

The hydrologic regime is a major factor affecting seedling establishment, growth, and survival (Huffman 1980, Harms et al. 1980, Streng et al. 1989, Johnson 1994, Jones et al. 1994, Johnson 2000). However, the germination potential of BLH tree species under different flooding regimes has received little study. Past germination studies (Hosner 1957, Briscoe 1961, Larsen 1963, Outcalt 2002) have shown that flood duration can affect germination rates of some BLH species, yet few species have been studied.

Sediment deposition rates may also influence seedling germination and establishment, but I have found no studies that address these relationships. Stress caused by burial of already established individuals has been studied to a limited extent. Burial of several freshwater lowland plant species to a depth of 5 cm, 10 cm, and 15 cm produced an average reduction in shoot density ranging from 10 to 56% (van der Valk et al. 1983). Rate and texture of sediment deposition may be important factors in determining the response of the

vegetation (van der Valk et al. 1983). In other riparian systems, tree recruitment has been found to be controlled mainly by stream flow pulses and river bed restructuring that cause high seedling mortality by eroding or burying newly germinated individuals (Johnson 2000).

Past studies have shown the range of deposition rates in floodplains of the southeast United States varies from 0.02 to 2.6 cm/year (Hupp and Morris 1990, McIntyre and Naney 1991,Hupp and Brazemore 1993, Kleiss 1996, Heimann and Roell 2000). However, floodplains experiencing excessive sedimentation in western Tennessee have received up to 79 cm of sediment in one year (Chapter 2). Many floodplains experiencing excessive deposition are often buried by infertile sand, which covers the productive silt-clay deposits. Happ et al. (1940) suggest that, when sand deposits reach a depth of 15 cm, the productive potential of the affected area decreases significantly. Excessive sedimentation could reduce the germination potential of typical BLH tree species by burying seed sources under infertile sand.

In this study, I use greenhouse experiments to determine the germination response of three BLH tree species to different hydrologic and sedimentation treatments. Specifically, I determine the effect of hydroperiod and amount and texture of sediment on the germination potential of three BLH tree species after a 10-week period and evaluate differences in above-ground seedling height. Based on results of previous studies (Larsen 1963) and the high tolerance of overcup oak to flooded conditions (Theriot 1993), I expected that the flood duration treatment would have little effect on germination or seedling growth of

overcup oak. The lower tolerance of both swamp chestnut oak and red maple (Theriot 1993) to flooding led me to hypothesize that their germination rates would decrease as flood duration increased. I also tested the hypothesis that flooding before germination retards seedling growth after germination. I predicted that the germination potential of overcup oak and swamp chestnut oak would be reduced by an 8 cm sand treatment but that burial by the 2 cm of topsoil or 2 cm of sand would not have a negative impact on germination because overcup and swamp chestnut oak are commonly found on sandy and loamy soils (Burns and Honkala 1990). The germination potential of red maple was not expected to differ among sediment treatments because of its prevalence at disturbed sites (Oswalt 2003). However, I expected that seedling height would be reduced for all species in the 8 cm sand treatment due to the physical barrier of the treatment.

Methods

Species used in this experiment included red maple (*Acer rubrum*), swamp chestnut oak (*Quercus michauxii*), and overcup oak (*Q. lyrata*). Red maple was chosen because it is common in heavily disturbed sites in western Tennessee. Swamp chestnut oak and overcup oak were chosen because they are typical bottomland hardwood species and are important to both wildlife and the timber industry (Burns and Honkala 1990).

Germination of red maple seeds occurs in April, immediately after seed fall (Burns and Honkala 1990). Overcup oak acorns are dormant during the winter and will germinate in the spring after floodwaters recede (Burns and Honkala

1990). Swamp chestnut oak acorns require no dormancy period, and if suitable habitat is available, they can germinate soon after seed fall in December (Burns and Honkala 1990). No stratification is needed for any of these species (U.S. Forest Service 1974).

All seeds were purchased approximately two weeks before initiation of the experiment in March of 2004 and were stored at 4° C. Red maple seeds were purchased from Lovelace Seeds, Inc. in Missouri. Acorns of both oak species were obtained from western Tennessee collections made by the Tennessee Wildlife Resource Agency. All acorns were float tested and hand sorted to determine seed viability and all nonviable seeds were removed (Lotti 1959, U.S. Forest Service 1974). A total of 50 seeds were planted, at equal spacing, in plastic flats with a single species in each flat. Two flats per species were planted for each sediment and hydroperiod treatment except for swamp chestnut oak. Because of a shortage of swamp chestnut oak acorns, some treatments only had one 50-seed flat.

The experimental design was a split-split plot design with the main plot treatment being a 30-day pre-germination hydroperiod or flood treatment. The hydroperiod treatments were: (1) non-flooded (control), (2) flooded to 10 cm above the soil surface for 15 days, and (3) flooded to 10 cm above the soil surface for 30 days. Germination flats that completed the hydroperiod treatment before the 30-day period was complete were drained and stored at 4° C (Outcalt 2002). After the flood treatment, flats were watered periodically to maintain moist soil. The subplot treatments involved varying amounts and texture of sediment.

The sediment treatments were developed to represent realistic sediment deposition events in western Tennessee floodplains (Chapter 2). The subplot treatments included: (1) seeds buried 2 cm deep in topsoil, (2) seeds buried 2 cm deep in sand, and (3) seeds buried 8 cm deep in sand. All sand used in the experiment was collected in western Tennessee and the topsoil was purchased at Lowe's Home Improvement Warehouse.

To account for confounding variables, dissolved oxygen was measured three times during the first 15 days of flooding in each flat of the 15-day and 30-day hydroperiod treatments. In addition, Hobo temperature data loggers (Onset Computer Corp., Bourne, Ma) were used to measure temperature on an hourly basis during the last four weeks of the germination experiment. Linear regression was used to determine if percent germination was correlated with either of these variables.

After the 10-week germination period, I counted the total number of individuals that germinated and measured the above-ground seedling height. All non-germinated seeds were examined for decay and abnormal germination (seed split or root growth but no shoot development). All percent germination rates were based on seeds that germinated and emerged above the soil-surface.

Analysis of variance was used to determine differences in germination rates among sediment and flooding treatments by species. However, because some treatments could not be replicated for swamp chestnut oak, I used logistic analysis (Sokal and Rohlf 1995) to test for differences (Alpha = 0.05) in proportion of seeds that germinated and that did not germinate among sediment

and flooding treatments. An interaction term was included in the overall model to test for nonadditivity of main effects. Analyses were separated by main effect treatments if a significant interaction existed. When the overall chi-square test for treatment differences was significant, Z-tests, on two proportions that were Bonferroni corrected (Alpha = 0.016) were performed to determine post-hoc pairwise differences (Sokal and Rohlf 1995). ANOVA also was used to determine the effects of sediment and hydroperiod treatments on above-ground seedling height by species. Kruskal-Wallis tests were used in cases when ANOVA assumptions were not met and Tukey-Kramer multiple comparison tests were used to distinguish differences among sediment and flooding treatments (Alpha = 0.05) (Sokal and Rohlf 1995). Statistical analyses were conducted with SAS Version 9.1 (SAS Institute Inc. 2004) and Minitab Release 14 (Minitab Inc. 2003).

Results

When all treatments were combined, overcup oak had the greatest germination at, 72.5%. Of the 900 overcup acorns planted, 653 seeds germinated and emerged above the soil surface. Swamp chestnut oak also had a high percent germination. Of the 650 swamp chestnut acorns planted, 432 germinated (66.5%) and emerged above the surface. Germination of red maple seeds was not as successful as the oak species. Of the 900 red maple seeds planted, only 7 seeds germinated, for a total germination of 0.77%.

Because of the poor germination of red maple in the experiment, a seed viability test was conducted on 100 seeds from the same batch of seeds used in

the experiment. Testing protocol was used from the AOSA Tetrazolium Testing Handbook (2000). The seed viability test determined that only 7% of the seeds were viable. Because of the low viability of the red maple seeds used in this experiment, red maple was excluded from the rest of the analysis.

Linear regression indicated that there was no correlation between percent germination and mean dissolved oxygen (N = 21, df = 1, P = 0.11) or between percent germination and mean temperature (N = 15, df = 1, P = 0.30). Among the treatment combinations the mean dissolved oxygen ranged from 0.26 mg/L to 3.34 mg/L and the mean temperature ranged from 21.72° C to 23.88° C.

Germination Potential

Overcup oak germination differed among flooding treatments (N=18, df =2, F = 7.94, P = 0.004) and flooding and sediment treatment combinations (N = 18, df = 4, F = 4.79, P = 0.02) (Figure 4-1). The percent germination of overcup oak was highest in the 30-day flood and topsoil treatment combination ($\bar{x}$ = 91 ± 3 %), but did not differ in percent germination from any of the other 30-day flood treatments, 15-day flood treatments, or the 0-day and topsoil treatment. The 0-day flood and 8 cm sand treatment combination ($\bar{x}$ = 29 ± 3 %) had the lowest percent germination but did not differ from the 0-day flood and 2 cm sand treatment combination ($\bar{x}$ = 56 ± 6 %) (Figure 4-1).

There were differences in percent germination of swamp chestnut oak among flooding treatments (N = 650, df = 2, X^2 = 47.01, P < 0.001) (Figure 4-2). However, there were no differences among sediment treatments (N = 650, df = 2,

$X^2 = 3.53$, P = 0.17) (Figure 4-3) or flooding and sediment treatment combinations (N = 650, df = 4, P 0.06). The 0-day flooding treatment (77.5%) and the 15-day flood treatment (72.8%) did not differ but both had greater germination rates than the 30-day flood treatment (47.5%) (Figure 4-2).

Height of Seedlings

There were differences in above-ground height of overcup oak seedlings among the flooding treatments (N = 653, df = 2, F = 55.19, P < 0.001), sediment treatments (N = 653, df = 2, F = 130.91, P < 0.001), and flooding and sediment treatment combinations (N = 653, df = 4, F = 5.21, P < 0.001) (Figure 4-4). Above-ground height of the overcup oak seedlings in the 30-day flood and 2 cm sand treatment ($\bar{x}$ = 18.28 ± 0.53 cm) and the 15-day flood and 2 cm sand treatment ($\bar{x}$ = 16.29 ± 0.53 cm) did not differ but were greater in above-ground height than all other treatment combinations. The 0-day flood and 8 cm sand treatment ($\bar{x}$ = 6.40 ± 0.90 cm), 15-day flood and 8 cm sand treatment ($\bar{x}$ = 7.99 ± 0.57 cm), and 30-day flood and 8 cm sand treatment ($\bar{x}$ = 9.22 ± 0.53 cm) did not differ but were lower in above-ground height than all other treatment combinations (Figure 4-4).

Mean above-ground height of swamp chestnut oak seedlings was also different among flooding treatments (N = 432, df = 2, F = 3.20, P = 0.04), sediment treatments (N = 432, df = 2, F = 46.74, P < 0.001), and flooding and sediment treatment combinations (N = 432, df = 4, F = 4.05, P = 0.003) (Figure 4-5). The shortest above-ground seedling height occurred in the 15-day flood

and 8 cm sand ($\bar{x}$ = 8.49 ± 0.68 cm), 30-day flood and 8 cm sand ($\bar{x}$ = 8.70 ± 0.84 cm), and the 0-day flood and 8 cm sand treatment combinations ($\bar{x}$ = 10.45 ± 0.63 cm).

Discussion

This study demonstrates that both hydrology and sedimentation can influence the germination and growth of overcup oak and swamp chestnut oak. The influence of these factors is variable and the response of each species seems to depend on its life history characteristics. For overcup oak and swamp chestnut oak, hydrology seems to have a greater effect than deposition rates on germination potential. However, high deposition rates clearly reduce the above-ground height of seedlings and may reduce a seedling's competitive ability to acquire important resources such as light (Streng et al. 1989, Jones et al. 1994) and to tolerate stresses like flooding (Hosner 1960).

Germination Potential

Contrary to my prediction and to previous work (Larsen 1963) my study showed that germination of overcup oak acorns was influenced by both the hydroperiod treatments and the sediment treatments. The results (Figure 4-1) indicate an increase in mean percent germination with increased duration of flooding. Flooding overcup oak acorns before germination may help seeds imbibe water and activate biochemical processes needed for germination (Kozlowski and Pallardy 1997). The high water-holding capacity of topsoil may have a similar effect in the absence of flooding. Prolonged flooding may still be detrimental to overcup oak germination due to reduced oxygen supply, however,

30 days of flooding does not seem to be long enough for this threshold to be reached. In addition, germination of overcup oak required more time in the 0-day flooded treatment ($\bar{x}$ = 8 ± 0 weeks) than either the 15-day ($\bar{x}$ = 5.3 ± 0.67 weeks) or 30-day flood treatments ($\bar{x}$ = 4.7 ± 0.67 weeks). These results suggest that a flooding period may increase the germination of overcup oak acorns as well as shorten the time needed for germination to occur.

Another negative impact from having no flood treatment for overcup oak is the increased number of acorns that germinated but did not emerge before the end of the study period. The 0-day flood treatment had 24.3% of its seeds germinate but not emerge, compared to the 15-day flood and 30-day flood treatments with 9% and 5.7%, respectively. These results suggest that the lack of a pre-germination flooding period will reduce the germination potential of overcup oak and may also reduce the number of individuals that actually emerge. However, this result may just be an extension of the extended germination time needed by acorns of the 0-day flood treatment. If the 10-week germination period had been extended, the non-emerged individuals might have emerged. In any case, the higher mean percent of non-emerged seeds in the 0-day flood treatment supports the interpretation that timing of germination was delayed by the lack of flooding. This result has important implications on BLH forest succession, as early emergence has been shown to improve survival rates of several BLH tree species (Streng et al. 1989, Jones et al. 1994).

Mean germination rates of overcup oak only differed by sediment treatment in the 0-day flood treatment (Figure 4-1). This suggests that sediment

texture and amount or rates of sedimentation can have a negative influence on the germination of overcup oak when combined with a lack of pre-germination hydroperiod. High sediment deposition rates may present a physical barrier that limits the germination and emergence of overcup oak, or other unknown mechanisms may inhibit the successful germination and emergence of overcup oak. However, germination rates did not differ among sediment treatments for either the 15-day or 30-day flood treatments. This result may explain why overcup oak is commonly found on both poorly drained clay soils and on sites with soil textures with good drainage such as sandy soils (Burns and Honkala 1990). Moreover, the results suggest that the effects of sedimentation on germination may be secondary to the effects of flooding, as sedimentation only had a negative impact on germination in the 0-day flood treatment. Thus, even though sedimentation may be a limiting factor to the germination of overcup oak, the lack of a hydroperiod seems to be more limiting.

The results support my prediction that the germination potential of swamp chestnut oak would be reduced as the pre-germination hydroperiod increased (Figure 4-2). The weakly flood tolerant rating (Theriot 1993) of mature swamp chestnut oak trees seems to apply to swamp chestnut oak acorns as well. The results suggest that the flood tolerance threshold of swamp chestnut oak acorns is reached somewhere between 15 and 30 days, while pre-germination hydroperiods lasting 15 days or less had no significant impact on the germination of swamp chestnut oak. The 30-day flood treatment also had more acorns that germinated but did not emerge ($\bar{x}$ = 5.75 ± 0.25 acorns) than the 15-day ($\bar{x}$ = 1.4

± 1.17 acorns) and the 0-day ($\bar{x}$ = 3 ± 0.71 acorns) flood treatments. This suggests that long pre-germination hydroperiods may also have a negative effect on the emergence of swamp chestnut oak. One explanation for the decreased germination and emergence of swamp chestnut oak might be that flooding for more than 15 days may fill seed pores with water, preventing the uptake of oxygen by the seed and resulting in reduced seed viability (Kozlowski and Pallardy 1997).

Surprisingly, sediment treatments had no significant effects on the germination of swamp chestnut oak. I expected that germination would be the same for the topsoil and 2 cm sand treatments due to the occurrence of swamp chestnut oak on both silty- clay and sandy soils (Burns and Honkala 1990). However, it was unexpected that swamp chestnut oak would be able to overcome any physical barriers imposed by the 8 cm sand treatment. The tolerance of swamp chestnut oak germination to high deposition rates may explain its common occurrence along first-bottom ridges within the floodplain (Burns and Honkala 1990), that are periodically subject to high rates of sediment deposition.

Seedling Growth

This study examined the above-ground height of seedlings to determine if hydroperiod and sediment treatments influenced growth of seedlings and if a competitive advantage might be produced in terms of height and thus access to resources such as light. The results suggest that a pre-germination hydroperiod is beneficial to the growth of overcup oak seedlings. Total height of overcup oak

seedlings in the 2 cm sand treatment was greater than in the other two sediment treatments in both the 15-day and 30-day flood treatments (Figure 4-4), suggesting that sandy soils may enhance the growth of overcup oak. Above-ground height of seedlings in the 8 cm sand treatment was significantly reduced across all flood treatments, suggesting that high deposition rates may adversely affect the growth of overcup oak seedlings and reduce the competitive height advantage of overcup oak seedlings.

Swamp chestnut oak seedlings grew best in the topsoil and 2 cm sand treatments of all three flood treatments, with the 15-day flood treatment producing seedlings with the greatest above-ground height. Swamp chestnut oak is classified as weakly tolerant of flooding (Theriot 1993) and its adaptation to flooded conditions may explain why a short pre-germination hydroperiod produced the tallest seedlings.

Although the topsoil and 2 cm sand treatments did not seem to influence above-ground height of swamp chestnut oak seedlings among any of the flood treatments, the 8 cm sand treatment produced the lowest seedling heights in each flood treatment. Thus, swamp chestnut oak seedlings affected by high levels of deposition do seem to be at a competitive disadvantage in terms of above-ground height.

Conclusion

Plant succession involves changes in species composition that result from differences in life history characteristics that control dispersal ability, establishment requirements, tolerances to various environmental stresses, and

competition with other individuals (Platt and Connell 2003). Establishment of an individual depends on its ability for dispersal to the site and its ability to germinate and survive at the site. This study focused on the effects of flooding and sedimentation on the probabilities of germination and survival of three BLH tree species. Unfortunately, the low viability of red maple seeds used in this experiment prevented the reliable testing of this species. However, the results do suggest that flood duration prior to germination can have both negative and positive effects on germination rates. The direction of the effect depends on the life history characteristics of the plant species. Sediment texture and rate seemed to also have an effect on germination rate, but was secondary to flood duration. The main effect of the sediment treatment was on above-ground height. High deposition rates reduce the above-ground height of individuals and may reduce their competitive abilities to survive in a forest.

Although this study only focused on germination, hydrology and sedimentation may also influence other vegetation processes of establishment, growth, seed production, and survival. Many of these processes have been investigated in relation to hydrology (Hosner 1957, Johnson and Bell 1976, Huffman 1980, Harms et al. 1980, Reily and Johnson 1982, Streng et al. 1989, Johnson 1994, Jones et al. 1994, Keeland and Sharitz 1997, Johnson 2000), however there is a lack of information on the influence of sedimentation. Further research is needed to investigate these relationships and to examine the tolerances of other BLH tree species to varying hydroperiods and sedimentation rates.

The results from this study should be used as a basis for developing and testing hypotheses on the germination potential of other BLH tree species to varying hydroperiods and sedimentation rates. For example, species that typically colonize areas near the stream channel, like cottonwood, boxelder, and black willow, are adapted to frequent flooding and rapid deposition (Hodges 1997), thus would be expected to be less affected by longer hydroperiods and higher rates of deposition than species like sweetgum and cherrybark oak. Testing hypotheses like this will contribute to our overall knowledge of BLH forest succession and benefit management and reforestation efforts.

APPENDIX 4

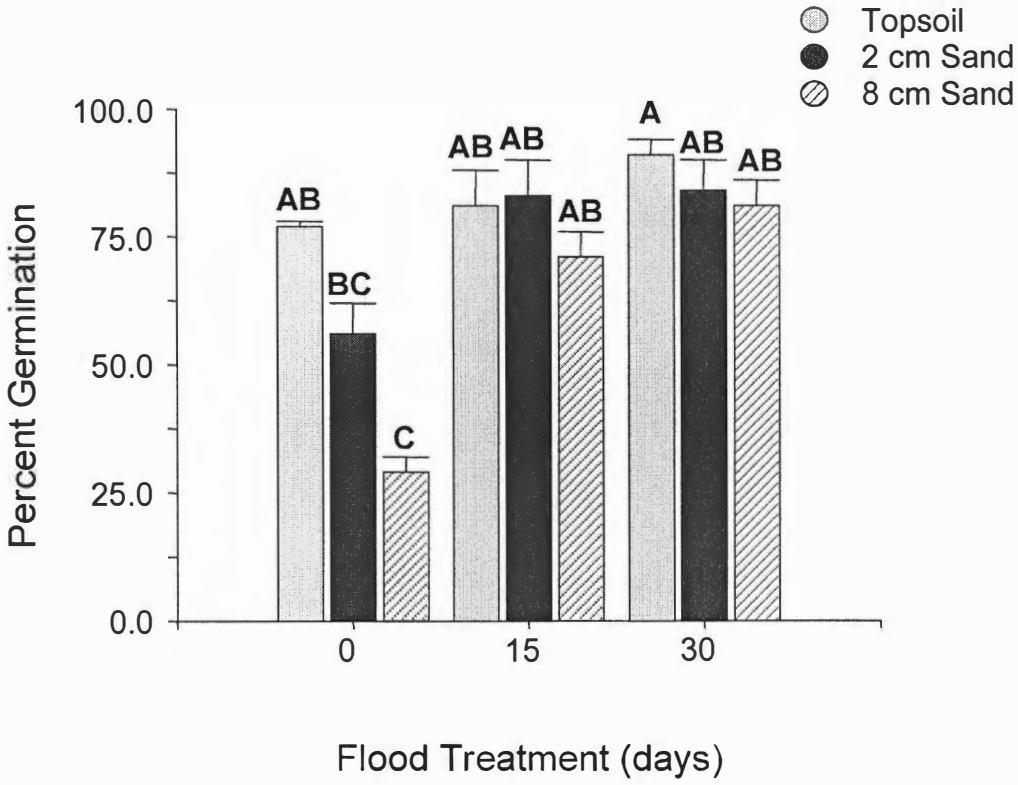


Figure 4-1. Percent germination of overcup oak among flood and sediment treatment combinations after 10-weeks. Bars with unlike letters are different ($P < 0.05$).

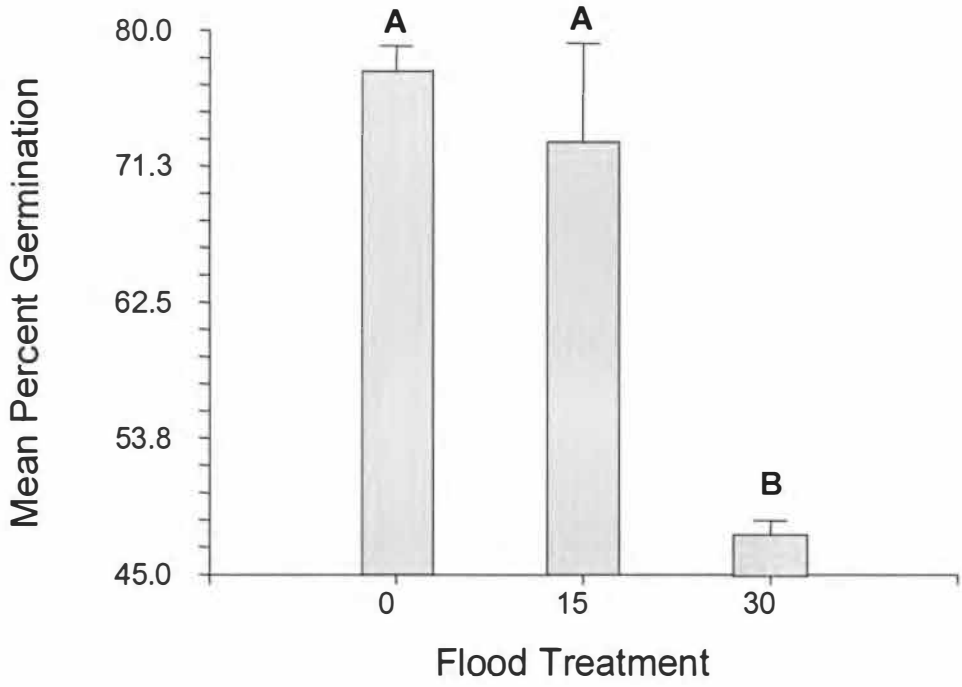


Figure 4-2. Swamp chestnut oak mean percent germination (+ 1 standard error) by flood treatment after 10-weeks. Bars with unlike letters are different ($P < 0.05$).

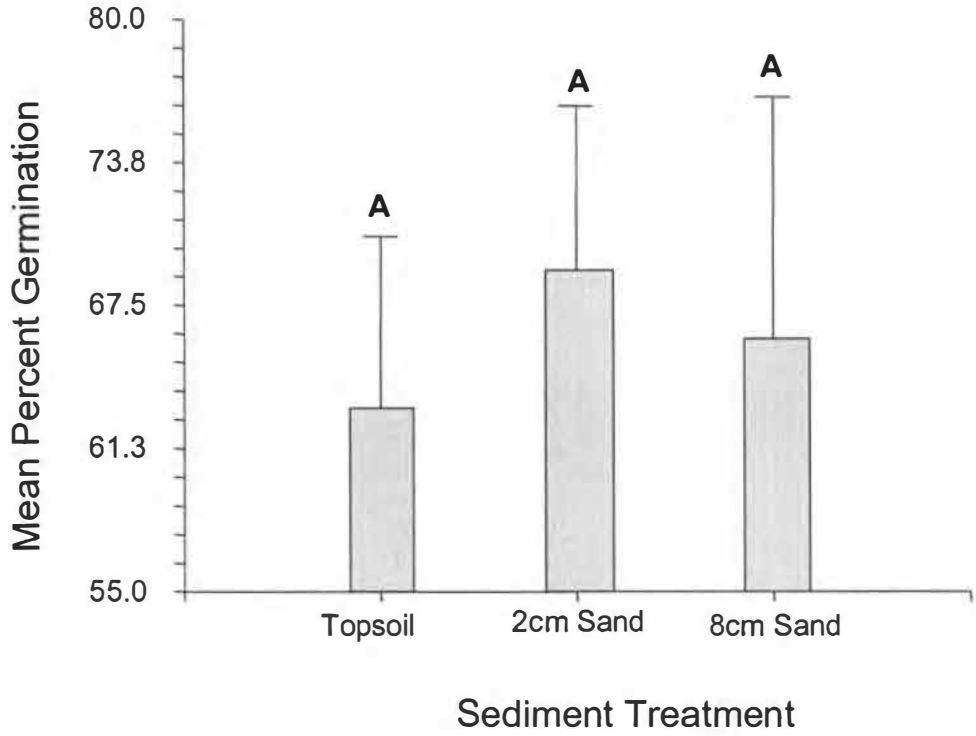


Figure 4-3. Swamp chestnut oak mean percent germination (+1 standard error) by sediment treatment after 10-weeks. Bars with unlike letters are different ($P < 0.05$).

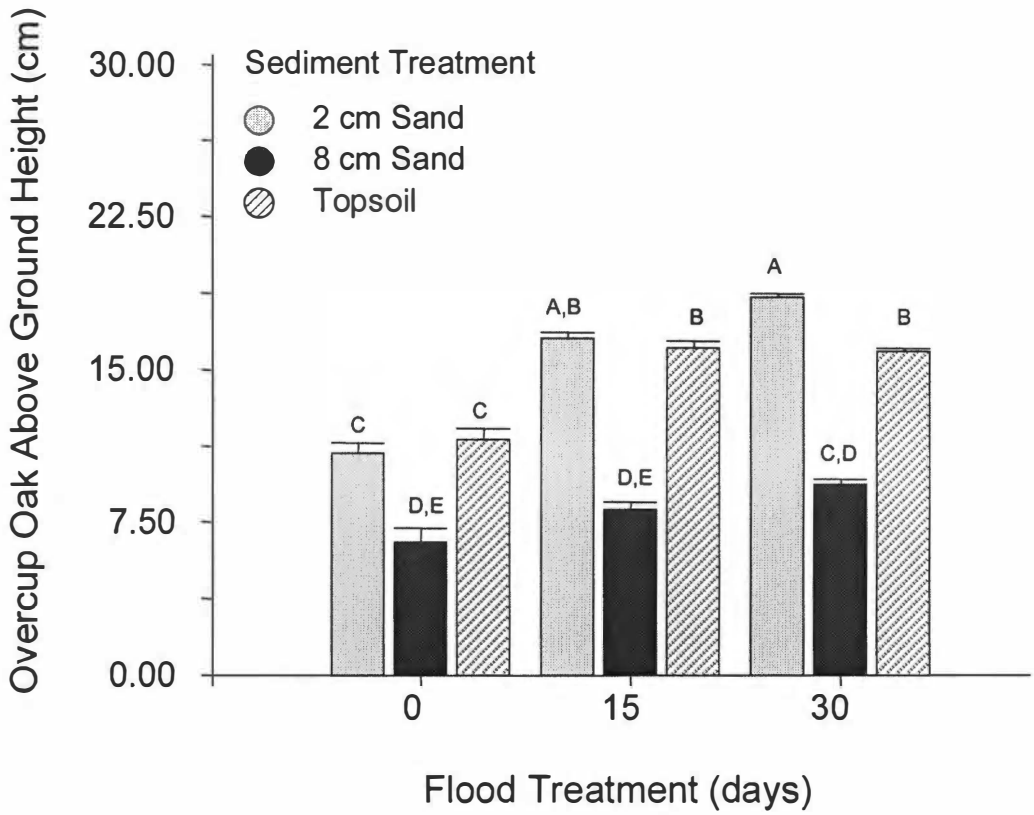


Figure 4-4. Overcup oak seedling mean above-ground height (cm) and standard error by treatment combination. Bars with unlike letters are different ($P < 0.05$).

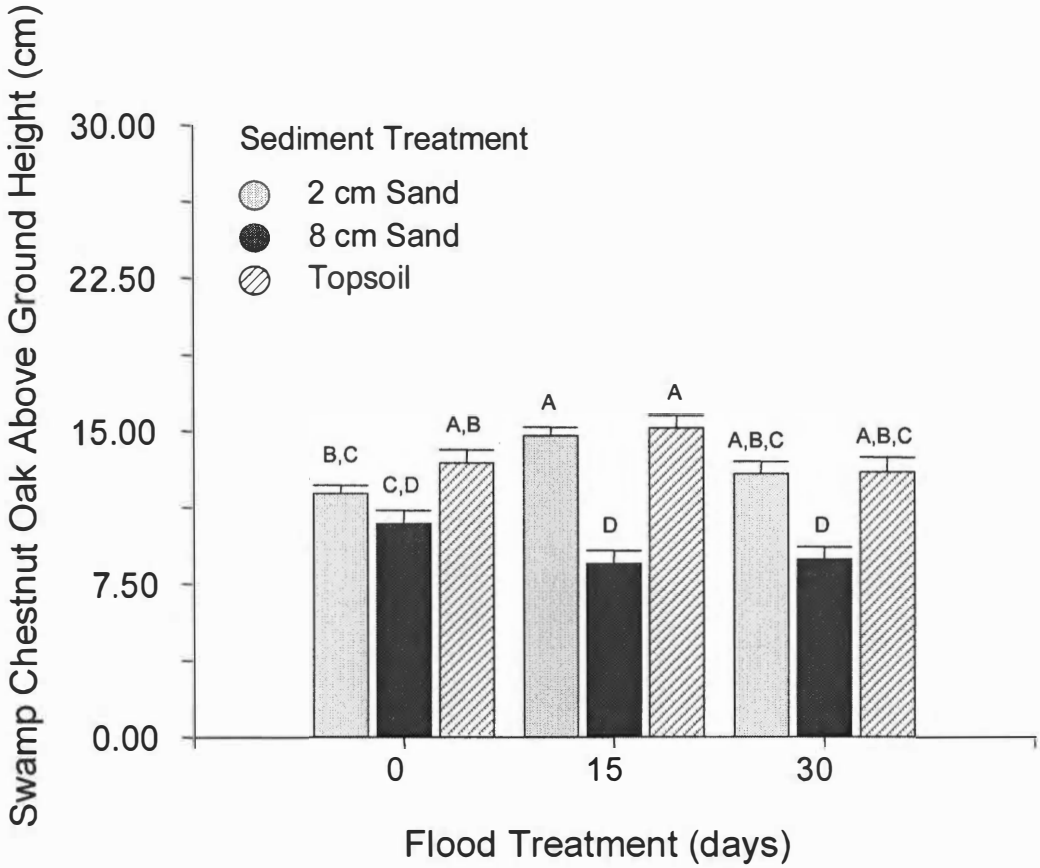


Figure 4-5. Swamp chestnut oak seedling mean above-ground height (cm) and standard error by treatment combination. Bars with unlike letters are different ($P < 0.05$).

PART V

THE EFFECTS OF ALTERED HYDROLOGIC AND SEDIMENTATION PROCESSES ON BOTTOMLAND HARDWOOD FOREST COMMUNITIES

Introduction

Bottomland hardwoods (BLH) occur in floodplains of rivers in the southeastern United States including regions of the Piedmont, Gulf Coastal Plain, and Lower Mississippi River Alluvial Valley (LMAV) (Sharitz and Mitsch 1993). BLH wetlands provide numerous valuable functions to both society and nature, including: water quality enhancement, flood control, erosion control, timber production, and wildlife habitat. BLH forests maintain biologically diverse and remarkably productive ecosystems that are adapted to fluctuating water levels (Odum 1969, Brinson 1990).

BLH forests support recognizably distinct assemblages of plants that are associated with particular landforms, soils, and hydrologic regimes (Wharton et al. 1982, Brinson 1990, Sharitz and Mitsch 1993). Although floodplains have little topographic relief, the continuous erosion and deposition of sediment create diverse micro-topography and geomorphic features that define distinct ecological zones within floodplains. Small differences in elevation can result in unique site conditions that affect the stand composition (Hodges and Switzer 1979).

Flood frequency, depth, timing and duration are primary determinants of floodplain plant species composition (Bedinger 1978, Huffman 1980, Hodges 1997). Sedimentation is also a main determinant of BLH plant species composition because of its interrelatedness with hydrology and its direct impact on vegetation and soil properties (Happ et al. 1940, Wharton et al. 1982, Jones et al. 1994, Hodges 1997, Stanturf and Schoenholtz 1998, Johnson 2000). To a lesser extent, plant species composition is also influenced by an array of factors

within the floodplain including: groundwater levels, soils, micro-topography, light intensity, and human activities (Clark and Benforado 1981).

Channelization has been used extensively throughout the southeastern coastal plain and LMAV to reduce flooding and has a wide range of impacts on floodplain ecosystems (Shankman 1993). Increased channel capacity and transport efficiency of channelized reaches causes channels to be disconnected from the adjacent floodplain. Channelization has been shown to reduce flooding in upstream reaches of a system, while causing lower reaches to experience increased peak flood stage and flood frequency (Shankman and Pugh 1992). Channel degradation occurs in response to channelization (Robbins and Simon 1983, Simon and Hupp 1987, Darby and Simon 1999) and can result in the lowering of water tables in the adjacent floodplain (Tucci and Hileman 1992, Shankman 1996). Channelization can also cause other geomorphic adjustments within the channel, resulting in the accumulation of sediment in lower reaches of the channelized system (Robbins and Simon 1983, Simon and Hupp 1987, Shankman and Samson 1991, Simon 1994).

In western Tennessee, and other areas in the loess region of the LMAV and Gulf Coastal Plain, the effects of channelization have been exacerbated by the geology of the region and past land-use practices. The combination of highly erodible soils, poor land-use practices, and increased transport capacity of channelized streams has resulted in the formation of valley plugs and shoals in the depositional zone of these altered systems. The formation of valley plugs and shoals have a significant influence on hydrologic and overbank

sedimentation processes within the adjacent floodplain (Happ et al. 1940, Chapters 2 and 3). The influence of valley plugs and shoals on floodplain conditions may extend further due to the interrelatedness of hydrologic processes and sedimentation and their effects on other environmental factors such as elevation and soil properties.

Previous research has focused on the distribution of distinct forest communities within the floodplain with respect to geomorphic and hydrologic attributes (Shelford 1954, Marks and Harcombe 1975, Hupp and Osterkamp 1985, Muzika et al. 1987, Hupp 1992). Few studies have investigated additional environmental factors that can influence BLH forest communities such as micro-topography and soil properties (Robertson et al. 1978, Smith 1996, Burke et al. 2003). Currently, in western Tennessee and other areas in the southeastern United States, there is considerable interest in the conservation and restoration of BLH forests that have been affected by channelization. However, there are few data available on the response of BLH forest to channelization (Simon and Hupp 1987, Hupp 1992, Simon and Hupp 1992, Shankman 1993) or to the formation of valley plugs and shoals (Miller 1990, Oswalt and King *In press*.).

Hodges (1997) has summarized past research on BLH forests to develop species-site relationships and patterns of BLH forest community succession. Hodges (1997) outlined three basic pathways of BLH forest succession, all driven by hydrologic events and rates and types of sediment deposition and topographic position within the floodplain. The BLH community level changes, described by Hodges (1997), are a result of differential survival of overstory trees, seedlings,

and germination rates of seeds under the prevailing hydrologic and sediment deposition conditions.

In western Tennessee, Simon and Hupp (1987, 1992) and Hupp (1992) developed a model of channel recovery from channelization and associated different riparian plant communities with different stages of channel recovery. Processes involved in their six stage model of channel recovery included channel bed aggradation, recolonization of streambanks by woody vegetation, and bank accretion (Hupp 1992). Their research determined that pioneer tree species like river birch (*Betula nigra*), black willow (*Salix nigra*), boxelder (*Acer negundo*), and silver maple (*Acer saccharinum*) were adapted to fluvial disturbances because they are fast growing and tolerant of slope instability and sediment deposition. Although their research is useful for understanding channel recovery and riparian forest responses to fluvial-geomorphic disturbances, their study did not include valley plugs and was restricted to within 50 m of the stream channel. My work (Chapter 2 and 3) has shown that valley plugs and shoals can have significant effects on fluvial-geomorphic processes extending out into the floodplain further than 50 m.

Reduced lateral channel migration, associated with channelized streams, can also affect floodplain plant species composition (Shankman 1993). Shankman (1993) noted that the development of baldcypress-water tupelo (*Taxodium distichum-Nyssa aquatica*) swamps is the result of lateral channel migrations through the floodplain over long time periods. As a result of channelization, stabilized channels preclude lateral migration and the formation

of baldcypress-tupelo swamps, thus limiting the occurrence of this forest community in floodplains of channelized streams.

Miller (1990) observed that some floodplains adjacent to valley plugs in the Obion-Forked Deer River watershed in western Tennessee have been permanently flooded causing reduced microhabitat and plant species diversity. Oswalt and King (*In Press*) also investigated the effects of valley plug formation on BLH communities in the Obion-Forked Deer River watershed. Their research suggests that valley plugs have influenced floodplain processes such that recovery of BLH forest communities has been hindered. Typical BLH species such as sweetgum (*Liquidambar styriciflua*) and oak species (*Quercus* spp.) were no longer important in the floodplain and were replaced by the disturbance-tolerant species of red maple (*Acer rubrum*). However, as shown previously (Chapter 2 and 3), the formation of valley plugs and shoals can result in a wide range of responses in fluvial-geomorphic processes, thus our understanding of BLH forest community response is also undeveloped.

Previous research of BLH forest communities and the environmental factors important in structuring them, provide a framework of understanding to develop hypotheses on the response of BLH forest communities to the formation of valley plugs and shoals. In this study, I use the knowledge gained from previous work in both altered (Simon and Hupp 1987, Miller 1990, Hupp 1992, Oswalt and King *In Press*) and unaltered (Hodges 1997) BLH systems and information I obtained on fluvial-geomorphic responses to valley plugs and

shoals (Chapter 2 and 3) to test predictions of BLH forest communities in response to valley plug and shoal formation.

The first objective of this study was to evaluate and determine differences in BLH forests associated with valley plugs, shoals, and unchannelized (natural meandering stream) sites at both the stand level and community level. I tested the hypothesis that altered hydrologic and sedimentation processes associated with channelization and valley plugs are directing widespread responses in BLH tree species composition, structure, and community associations. If the hypothesis is correct, then the following predictions should be supported by the results:

1) Important BLH species associated with valley plug sites should consist of pioneer or disturbance-tolerant species such as river birch, black willow, boxelder, and maple species.

2) Because of the greater rates of deposition occurring over most of the floodplains adjacent to valley plugs, there should be a decreased importance of oak species compared to unchannelized sites.

3) Furthermore, the lack of lateral channel migration associated with channelized streams (Shankman 1993) should reduce the importance of baldcypress and water tupelo. Thus, associations typical of BLH forests including oak species and baldcypress-tupelo swamps should be scarce in floodplains associated with valley plugs.

4) The forest communities at shoal sites, which did not experience greater rates of deposition except in isolated areas corresponding to crevasse splays (Chapter 2), should be more representative of unchannelized sites, containing disturbance-tolerant associations related to streambanks and the natural levee, baldcypress-tupelo swamps in low elevation areas, and oaks, sweetgum, hickories and others associated with flats and ridges.

I also expected that unchannelized and shoal sites would support greater tree species diversity and a greater variety of forest communities associated with micro-topographic features within the floodplain (Hodges 1997). Additionally, the lack of lateral channel migration that has been implicated in the reduced occurrence of baldcypress-tupelo swamps may not be apparent at shoal sites as a result of two factors: (1) greater mean flood depths (Chapter 3) and (2) the proximity of shoal sites to the Hatchie River which is not restricted in its lateral movement.

The second objective of this study was to determine the primary environmental factors important in structuring the identified forest communities. I expected that plot elevation (a surrogate for flooding intensity) and deposition rate and texture would be the most important environmental factors affecting forest communities, based on past studies (Hodges 1997, Burke et al. 2003, Oswalt and King *In Press*) and their influence on soil properties. I also investigated differences in environmental factors among the geomorphic features

to determine if factors important in structuring BLH communities varied among the three geomorphic features.

Methods

Study Reach

The Hatchie River study reach, located in Haywood, Madison, and Hardeman Counties in Tennessee, stretches south from the Hatchie River National Wildlife Refuge in Brownsville to Hickory Valley. Study sites are located along six tributaries of the Hatchie River. The tributaries consist of one unchannelized stream and six channelized streams (Table 5-1).

This study focused on three types of study sites: unchannelized sites (meandering channel), valley plug sites, and shoal sites. Three unchannelized sites were located along Spring Creek being at the confluence to the Hatchie River and spaced at least 2 km apart from each site. Spring Creek is a natural meandering tributary of the Hatchie River and contains extensive BLH forests. It is one of only three unaltered major tributaries in the Hatchie River basin (USDA 1986). Valley plug sites have been identified on several tributaries of the Hatchie River including three tributaries chosen for this study: Bear Creek, Jeffers Creek, and Hickory Creek (Tim Diehl, personal communication). Two shoal sites were also included in this study: one on Porters Creek and one on Piney Creek. These streams each contain a shoal at their confluence to the Hatchie River (Tim Diehl, personal communication).

Vegetation Sampling

Vegetation sampling was conducted at all study sites (Table 5-1) during May through August in 2002, 2003, and 2004. A systematic sampling design was used over the hydrologic gradient of floodplains perpendicular to the stream channels (Wharton et al. 1982, Brinson 1990, Kozlowski 2002). Sampling at unchannelized sites (Figure 5-1a) occurred at plots spaced 50 m apart along transects perpendicular to the stream and spaced 200 m apart. Sampling at shoal and valley plug sites occurred at plots spaced 50 m apart along transects perpendicular to the stream and spaced 50 m apart (Figure 5-1). A total of 80 plots were established at unchannelized sites, 201 plots at valley plug sites, and 86 plots at shoal sites.

The species and diameter at breast height (DBH) of all trees > 10 cm diameter at breast height (DBH) were recorded within a 0.04 ha circular plot from plot center. Similarly, I recorded the species (when possible) and DBH of all snags > 10 cm DBH and greater than 1.8 m tall. All seedlings (>1 m tall) were counted and species identified within a 0.004 ha circular plot (radius = 5.6 m) from plot center.

Environmental Variables

Elevations of each plot center were taken at every sampling plot at all study sites. Reference points were determined from benchmarks established by the U.S. Army Corps of Engineers and the U.S. Geological Survey. Plot elevations were measured using a Topcon Total Station GTS-229. Elevations of the stream channel water surface were also measured during low flow. Plot

elevations were adjusted to reflect elevations relative to the river channel and to eliminate the effects of downstream slope (Smith 1996)

Soil samples were taken at plots spaced 100 m apart starting at the stream's edge at every study site (Figure 5-1). Soil samples were taken to a depth of 20 cm using a 2.5 cm diameter soil core (Dunn and Stearns 1987). Samples were analyzed by A&L Analytical Laboratories, Inc. for texture, organic matter, phosphorous, potassium, calcium, magnesium, pH, and NO3-N.

Analysis

Site-level analysis included summary statistics for each site type (unchannelized, shoal, and valley plug) for all vegetation attributes including: species richness, relative frequency, stems per hectare, basal area (m^2 per hactare), diversity, species' importance values, and relative seedling density. Tree species' importance values were calculated (IV 200, relative density + relative dominance) (Krebs 1994) but only species with greater than 5% frequency were reported (Grace et al. 2000). Simpson's diversity index was calculated to compare tree species diversity among site types (Krebs 1994). ANOVA tests were used to determine differences in vegetation attributes among site types (i.e. stems per ha, basal area). Kruskal-Wallis tests were used in cases where ANOVA assumptions were not valid and Tukey-Kramer multiple comparison tests were used to distinguish differences among groups (Alpha = 0.05) (Sokal and Rohlf 1995). Statistical analyses were conducted with SAS Version 9.1 (SAS Institute Inc. 2004) and NCSS (Hintze 2001).

Community level analysis was performed for all plots of each site type (valley plug, shoal and unchannelized) combined together. Plots that were not sampled for environmental factors were excluded from the community level analysis (Table 5-1). Rare species (occurring less than 5%) were also eliminated from this analysis (Grace et al. 2000). Importance values were calculated for each species by plot.

For the community level analysis, hierarchical cluster analysis was performed using PC-ORD (M.J.M. Software Design 1999) on importance values (Burke et al. 2003) using Euclidean distance measures and Ward's minimum-variance method (McCune and Mefford 1999, McCune and Grace 2002, Burke et al. 2003). The dendrogram grouped plots into clusters by an average distance of 0.9 (Burke et al. 2003). The validity of the identified clusters was evaluated using multi-response permutation procedure (MRPP), that tests for differences within and between clusters (Grace et al. 2000, Burke et al. 2003).

Indicator species analysis (Dufrene and Legendre 1997) was then performed using PC-ORD (McCune and Mefford 1999) to determine statistical significance ($P < 0.05$) of indicator species for each cluster using a Monte Carlo technique. This procedure uses both the concentration of species abundance within clusters and faithfulness of occurrence with clusters to determine the appropriate indicator species for each cluster or community (Burke et al. 2003).

The ordination of study plots was conducted using non-metric multidimensional scaling (NMS) in PC-ORD (M.J.M. Software Design 1999). The NMS ordination was performed using Sorenson distance measures and random

starting points (Grace et al. 2000). Stress (a measure of lack of fit) was used to determine dimensionality of the solution using the autopilot mode of PC-ORD; stress was evaluated using a Monte Carlo test (McCune and Mefford 1999, Grace et al. 2000).

Correlations among ordination axis scores and environmental variables were determined using Pearson and Kendall correlations in PC-ORD. Important variables identified from the Pearson and Kendall correlations where then compared among clusters using ANOVA tests. I also tested for differences in important environmental variables among site types. Multivariate analysis of variance (MANOVA) was performed to determine if differences existed among site types (unchannelized, shoal, and valley plug). If differences were detected, one-way ANOVA tests were used to determine the environmental variables causing the differences and Tukey-Kramer multiple comparison tests were used to distinguish differences among site types.

Results

Across all sites, a total of 6,564 individual trees were sampled and 49 species identified. Each site type had similar tree species richness, with 38 species found at unchannelized sites, 32 species at shoal sites, and 35 species at valley plug sites. Tree species diversity was similar at unchannelized and shoal sites. At unchannelized sites, the Inverse Simpson's Diversity Index was 12.02 or 30.1% of the possible maximum diversity index of 40, while shoal sites had an Inverse Simpson's Index of 10.46 or 30.8% of the possible maximum

diversity index of 34. Tree species diversity was lowest at valley plug sites, with a diversity index of 8.10 or 22.5% of the maximum diversity index of 36.

The number of stems per hectare differed among site types (N = 385, df = 2, F = 15.45, P < 0.001) (Figure 5-2). Shoal sites had more stems per hectare ($\bar{x}$ = 501.74 ± 14.28) than either unchannelized sites ($\bar{x}$ = 373.13 ± 17.81) or valley plug sites ($\bar{x}$ = 415.41 ± 10.90). Basal area also differed among site types (N = 385, df = 2, X^2 = 47.75, P < 0.001) (Figure 5-3). Valley plug sites ($\bar{x}$ = 18.7 ± 0.65 m^2/ha) had significantly lower basal area than either unchannelized ($\bar{x}$ = 25.99 ± 1.53 m^2/ha) or shoal sites ($\bar{x}$ = 27.4 ± 1.44 m^2/ha). The number of snags per plot also differed among site types (N = 385, df = 2, X^2 = 11.73, P = 0.003). Valley plug sites ($\bar{x}$ = 2.83 ± 0.17 had a greater number of snags per plot than shoal sites ($\bar{x}$ = 1.97 ± 0.19), but neither valley plug nor shoal sites differed from unchannelized sites ($\bar{x}$ = 2.19 ± 0.30).

Twenty-five species occurred in at least 5% of all unchannelized site plots (Table 5-2). Baldcypress (*Taxodium distichum*) had the greatest importance value (37.25), followed by silver maple (*Acer saccharinum*) with a value of 33.15. Species that were found in greater than a third of all plots at unchannelized sites included: baldcypress, silver maple, American hornbeam (*Carpinus caroliniana*), American elm (*Ulmus americana*), and green ash (*Fraxinus pennsylvanica*) (Table 5-2). Species occurring in at least 5% of the unchannelized plots included five oak species: overcup oak (*Quercus lyrata*), water oak (*Q. nigra*), cherrybark oak (*Q. falcate*), swamp chestnut oak (*Q. michauxii*), and willow oak (*Q. phellos*).

Twenty-two species occurred in at least 5% of shoal site plots (Table 5-3). Silver maple (36.58) had the greatest importance value of all species at shoal sites followed by green ash with 27.72. Six species were found in at least a third of all shoal plots: silver maple, green ash, American hornbeam, sweetgum (*Liquidambar styriciflua*), hackberry (*Celtis occidentalis*), and American elm. There were also four oak species that occurred in at least 5% of all shoal site plots including: swamp chestnut oak, cherrybark oak, overcup oak, and water oak.

Fifteen species occurred in at least 5% of all valley plug plots (Table 5-4). Boxelder (*Acer negundo*) had the greatest importance value at 38.41, followed by black willow (*Salix nigra*) with a value of 30.34. Eight species occurred in at least a third of all valley plug plots: boxelder, black willow, green ash, red maple (*Acer rubrum*), sweetgum, sycamore (*Platanus occidentalis*), river birch (*Betula nigra*), and American elm. Only two oak species, swamp chestnut oak and willow oak, occurred in at least 5% of all valley plug plots.

At unchannelized sites, species with the greatest density of seedlings were American hornbeam, red maple, green ash, and oak species (Table 5-5). The density of baldcypress and tupelo species (*Nyssa spp.)* were also high at 5.59 and 5.45, respectively. At shoal sites, boxelder and American hornbeam, and pawpaw (*Asimina triloba*) had the greatest densities. The species with greatest density of seedlings at valley plug sites were boxelder, green ash, river birch, and elm species.

The community-level analysis identified eight different vegetation clusters or groups (Figure 5-4). Indicator species analysis (Table 5-6) identified (1) an overcup oak and silver maple association, (2) a sycamore association that also included American elm and river birch, (3) a mixed association that included swamp tupelo, cherrybark oak, hackberry, mockernut hickory, and swamp chestnut oak, among others, (4) a green ash association, (5) a baldcypress and water tupelo association, (6) a sweetgum and willow oak association, (7) a boxelder association, also including cottonwood, and (8) a black willow and red maple association. The multi-response permutation procedure (MRPP) produced a t-test statistic of –78.46, and a chance-corrected within group agreement of 0.33 ($P < 0.001$), demonstrating between-group heterogeneity and within-group homogeneity.

The proportion of sample plots grouped by the cluster analysis and indicator species analysis varied among unchannelized, shoal, and valley plug sites (Table 5-7). The overcup oak/silver maple association and the baldcypress/tupelo association were only present at unchannelized and shoal sites. Conversely, the black willow/red maple association only occurred at the valley plug sites and over 94% of the plots grouped into the boxelder association also occurred at valley plug sites (Table 5-7).

The ordination analysis resulted in a three-dimensional model that explained a total of 67.4% of the variation in the data (Figure 5-5). Axis one was positively correlated with geomorphic feature ($r = 0.383$) and several chemical properties of the soil, including potassium, magnesium, percent organic matter

and cation exchange capacity (CEC) (Table 5-8). Axis two was also correlated with several chemical soil properties including percent organic matter and CEC, among others and geomorphic feature; however, it was also negatively correlated with percent sand ($r = -0.317$). Axis three was most correlated with relative elevation ($r = 0.246$).

Environmental variables that were highly correlated with the axis scores of the ordination model were compared across vegetation groups determined from the cluster and indicator species analysis (Table 5-9). Site characteristics including relative elevation ($N = 175$, $df = 7$, $F = 9.48$, $P < 0.001$) and deposition rate ($N = 175$, $df = 7$, $X^2 = 35.16$, $P < 0.001$) differed among forest communities (Table 5-9). The green ash association ($\bar{x} = 179.93 \pm 25.80$ cm) had a greater relative elevation than all other communities except for the mixed association ($\bar{x} = 101.5 \pm 15.99$ cm). Deposition rate was greatest among the boxelder, black willow/red maple, sycamore, and sweetgum/oak associations (Table 5-9).

The soil textural property of percent sand differed among forest communities ($N = 175$, $df = 5.73$, $P < 0.001$). The black willow/red maple, boxelder, and sweetgum/oak associations had greater percent sand content than the baldcypress/tupelo, green ash, mixed and overcup oak/silver maples associations (Table 5-9). Chemical properties of the soil also differed among the forest associations (Table 5-9). Both percent organic matter ($N = 175$, $df = 7$, $F = 12.49$, $P < 0.001$) and CEC ($N = 175$, $df = 7$, $F = 12.11$, $P < 0.001$) differed among the forest communities. Macronutrient levels of phosphorus, potassium,

and magnesium also varied significantly among the forest communities (Table 5-9).

Environmental factors also varied among the three geomorphic features. Soil textural properties, including percent sand ($N = 176$, $df = 2$, $X^2 = 28.35$, $P < 0.001$), percent silt ($N = 176$, $df = 2$, $X^2 = 23.02$, $P < 0.001$), and percent clay ($N = 176$, $df = 2$, $X^2 = 19.25$, $P < 0.001$), all differed among valley plug, shoal, and unchannelized sites (Figure 5-6). Valley plug sites had the greatest percent sand ($\bar{x} = 67.93 \pm 2.40\%$) and the least percent silt ($\bar{x} = 23.26 \pm 1.96\%$) and clay ($\bar{x} = 8.82 \pm 0.57\%$). Shoal and unchannelized sites did not differ in soil textural properties (Figure 5-6).

Levels of phosphorus ($N = 176$, $df = 2$, $X^2 = 116.23$, $P < 0.001$), calcium ($N = 176$, $df = 2$, $F = 9.56$, $P < 0.001$), potassium ($N = 176$, $df = 2$, $X^2 = 26.99$, $P < 0.001$), and magnesium ($N = 176$, $df = 2$, $F = 41.98$, $P < 0.001$) also varied among valley plug, shoal, and unchannelized sites (Figure 5-7). The percent organic matter ($N = 176$, $df = 2$, $X^2 = 56.81$, $P < 0.001$) and cation exchange ($N = 176$, $df = 2$, $F = 46.29$, $P < 0.001$) also differed among valley plug, shoal, and unchannelized sites (Figure 5-8). Percent organic matter differed among all three site types with valley plug sites ($\bar{x} = 0.90 \pm 0.06\%$) having the least percent organic matter and unchannelized sites ($\bar{x} = 2.03 \pm 0.08\%$) having the greatest. Cation exchange differed similarly to percent organic matter among types with valley plug sites ($\bar{x} = 4.0 \pm 0.20$ meg/100g) having the lowest and unchannelized sites ($\bar{x} = 8.45 \pm 0.44$ meg/100g) having the greatest.

Discussion

The results of this study (including Chapters 2 and 3) indicate that surface and subsurface hydrology, sediment deposition rates, types of sediment deposited, and macronutrient concentrations are affected by valley plug formation. These alterations have created environmental gradients that are strongly affecting species composition and stand structure of associated BLH forests. Floodplain forests associated with valley plugs had lower tree species diversity and no longer contained the typical associations of oak species and baldcypress/tupelo that existed at both shoal and unchannelized sites. Forest associations consisting of disturbance-tolerant tree species dominated floodplains adjacent to valley plugs; seedling densities suggest that these associations will continue at least in the near future.

Chapters 2 and 3 provided detailed information on the surface and sub-surface hydrologic and sedimentation responses to valley plug and shoal formation. In general, surface flooding was reduced and water tables lowered at valley plug sites while shoal sites experienced more surface flooding but reduced water table levels. Valley plug sites also experienced greater rates of sediment deposition consisting mostly of coarse sands. Overall, sedimentation rates were similar among shoal sites and unchannelized sites, although distinct areas of greater sedimentation occurred along crevasse splays at shoal sites.

Sediment texture, particularly percent sand, was found to be an important control of the structure of forest communities. At valley plug sites, most of the floodplain surfaces experienced greater rates of deposition, mostly of coarse

sand (67% sand), while sand content at shoal and unchannelized sites was significantly less. In the case of soil texture, the high percentage of sand identified in the soil at valley plug sites is a result of sedimentation processes associated with valley plugs.

Likewise, many of the other patterns of soil characteristics are related to hydrologic conditions and soil texture. For example, the textural properties of soil can determine the percent organic matter and cation exchange capacity (CEC). Fine-textured soils consisting of silt and clay contain greater amounts of organic matter than sandy soils because decomposition rates are slower and clay particles form electrochemical bonds that hold organic compounds. In turn, CEC, which is the total exchangeable cations that a soil can hold, increases as a function of percent organic matter (Mitsch and Gooselink 2000). This relationship is also demonstrated in my results of percent organic matter and CEC among valley plug, shoal, and unchannelized sites (Figure 5-8).

Abundance of essential plant macronutrients such as phosphorus, potassium, and magnesium, is also related to hydrologic and sedimentation processes (Wharton et al. 1982, Hopkins 1995, Lockaby and Walbridge 1998, Stanturf and Schoenholtz 1998). For example, magnesium, which is an essential constituent of the chlorophyll molecule needed for photosynthesis, is typically deficient in sandy soils because of poor CEC (Hopkins 1995). Availability of essential macronutrients is also influenced by hydrologic processes that provide nutrient inputs, influence decomposition rates, and mobilize nutrients for plant uptake (Lockaby and Walbridge 1998). Thus, although soil characteristics

formed important environmental gradients affecting the distribution of forest communities, hydrologic and sedimentation processes related to geomorphic features were the main determinant of these gradients.

In this study, relative floodplain elevation was not as important as expected based on research in relatively unaltered systems (Dollar et al. 1992, Robertson et al. 2001, Burke et al. 2003). In unaltered systems, relative floodplain elevation is an effective surrogate of flooding conditions, but in this study it was not because the greater deposition rates associated with valley plugs cause dramatic changes in local floodplain elevation over short time periods. Similarly, in a study of the channelized Middle Fork Forked Deer River, (Oswalt and King *In Press*) demonstrated that baldcypress/tupelo associations, which are typically found in the lowest elevation areas, were located at higher elevations than most other communities and that red maple was replacing these species at the lower elevation sites. In this study, lower elevation communities of baldcypress/tupelo and overcup/silver maple were absent from valley plug sites. The lowest elevation sites at valley plug sites were dominated by the disturbance-tolerant black willow/red maple association.

In addition to relative floodplain elevation, sediment deposition rate and sand content differed among the forest communities. The forest communities of sycamore, sweetgum/oak, boxelder, and black willow/red maple, all had the greatest deposition rates and greatest percent sand. In addition, the majority of sample plots (> 75%) that were grouped within these forest communities occurred at valley plug sites. The accelerated deposition of coarse sand that is

occurring at these sites (Chapter 2) as a result of valley plug formation is responsible for the current site conditions and dominance of these disturbance-tolerant forest associations at valley plug sites.

Overall, the stand-level analysis revealed that tree species diversity and basal area were lower at valley plug sites and snag densities were greater than those of shoal sites. The lower tree species diversity is consistent with results of swamped floodplains of the Middle Fork Forked Deer River as a result of valley plug formation (Miller 1990). The lower basal area at valley plug sites may reflect the new establishment of pioneer species in response to channelization or reduced growth rates in response to fluvial-geomorphic disturbances. Valley plug sites also contained a greater number of snags than shoal sites, thus, increased tree mortality is also contributing to the lower basal area.

Stand-level analysis supported my prediction that important tree species at valley plug sites would consist mainly of pioneer species that have a high tolerance of disturbance. Species with greater importance values at valley plug sites included boxelder, black willow, red maple, sycamore, river birch, and cottonwood. These species are characteristic of stream bank and natural levee plant communities that are subject to the most fluvial-geomorphic disturbance in unaltered systems (Hodges 1997), early colonization of streams recovering from channelization (Hupp 1992), and floodplains influenced by valley plugs (Oswalt and King *In Press*). The community-level analysis also indicated that the most common forest communities associated with valley plug sites represented pioneer species that are tolerant of fluvial-geomorphic disturabance (Hupp 1992).

Other typically common BLH species including oaks, baldcypress, and water tupelo were either missing entirely from valley plug sites or reduced to very low importance. Seedling densities also support the results of the stand-level analysis. At valley plug sites, the species with the greatest density of seedlings were boxelder, green ash, and river birch and there was almost no regeneration of oak, baldcypress, or tupelo species.

There were exceptions to the dominance of pioneer species at valley plug sites, however, as green ash and sweetgum had high importance values and were common at these sites. Although the high importance of green ash was not expected, it does commonly occur on sandy soils and has a fast growth rate (Burns and Honkala 1990). Green ash is also typically associated with boxelder and has been used to re-vegetate spoil-banks resulting from strip-mining (Wright 1965). These characteristics suggest that green ash may be able to tolerate the fluvial-geomorphic disturbances occurring at valley plug sites and remain a remnant species of the pre-channelization and pre-valley plug formation forest. Results from my dendrogeomorphic study (Chapter 2) also support the contention that green ash has remained as a remnant species. The mean age of green ash trees ($\bar{x}$ = 37.56 ± 2.29 years) sampled in the dendrogeomorphic study was older than all other species (for N > 10), except for sweetgum ($\bar{x}$ = 43.42 ± 3.37 years).

Sweetgum is typically found on higher elevation ridges and flats (Hodges 1997) that would be least affected by fluvial-geomophic disturbances, thus enabling sweetgum to also remain an important remnant species at valley plug

sites. There were nine valley plug plots determined to be sweetgum/oak associations, including seven plots at the Hickory Creek valley plug site. Six of the seven sweetgum/oak plots at the Hickory Creek valley plug site were located upstream of the valley plug. Deposition at these plots mainly occurred during November 2003 to March 2004, when the valley plug at Hickory Creek expanded approximately 80 m upstream, resulting in failure of a spoil bank along the channel and subsequent floodplain deposition in excess of 30 cm. Vegetation sampling at the upstream plots at this site occurred before plots experienced deposition rates greater than 1 cm/yr with the exception of one plot with 27.15 cm of sediment deposition. After vegetation sampling, the mean deposition rate at these plots was 10.04 cm/yr, mainly as a result of valley plug expansion. Thus, the timing of my vegetation sampling prevented any detection of negative impacts on sweetgum trees in response to increased deposition. The large diameter size of the remaining sweetgum trees ($\bar{x}$ = 23.34 ± 0.45 cm), lack of regeneration (relative seedling density = 1.23 seedlings), and the mean age of sweetgum ($\bar{x}$ = 43.42 ± 3.37 years) indicate that sweetgum is a remnant species at valley plug sites and will not successfully reproduce except in higher elevation areas outside of the deposition zone. Based on plant community composition downstream of the plug, however, in time these sweetgum/oak associations will likely revert to disturbance-tolerant vegetation communities. Plots located downstream of the valley plug were formerly subjected to greater deposition rates and are now dominated by boxelder and black willow/red maple associations.

In contrast, both shoal and unchannelized sites contained several species of oaks with relatively high importance values and had greater importance values of baldcypress and tupelo, with baldcypress being the most important species at unchannelized sites. Seedling densities at unchannelized and shoal sites suggest that these species are also regenerating relatively well. The community-level analysis indicated the presence of baldcypress/tupelo associations that are commonly found along sloughs and in backswamps and other associations typical of BLH forests that are distributed over different elevations along flats and ridges within the floodplain (Hodges 1997). Previous research (Shankman 1993) suggests that baldcypress/tupelo associates would not be present in floodplains along channelized streams, due to reduced lateral migration; however, this was not the case in this study. The presence of baldcypress/tupelo swamps at shoal sites may be a result of hydrologic influences linked to the proximity of shoal sites to the confluence of the Hatchie River.

The disturbance-tolerant species that dominated valley plug sites were also found at shoal and unchannelized sites but had low importance values, with the exception of silver maple. The high importance value of silver maple at shoal and unchannelized sites is probably a result of forest management practices instead of a response to fluvial-geomorphic disturbances. Several of the unchannelized sites and one of the shoal sites are either owned by a lumber company or logged by the private landowner. Timber harvests, mainly high-grading, have impacted both shoal and unchannelized sites. The lower economic value of silver maple compared to other BLH tree species and its fast

growth rate in response to increased light (Burns and Honkala 1990) are probably the reason why silver maple had such a high importance value at these sites.

Interestingly, seedling density of boxelder was the second highest of all species at shoal sites. Furthermore, shoal sites also contained the only two plots (5.6%) that were grouped in the boxelder association, outside of valley plug sites. These boxelder plots were located along crevasse splays that experienced the greatest deposition rates (4 cm/yr and 17.8 cm/yr; Chapter 2) of all shoal sample plots. Additionally, sample plots located along the identified crevasse splays account for 63% of all boxelder seedlings sampled at shoal sites. These findings indicate that although forest stands and forest communities appear similar between unchannelized and shoal sites, the altered abiotic processes associated with channelization and shoal formations are influencing areas within the floodplain, specifically corresponding to crevasse splays.

Conclusion

The results of this study indicate that the hydrologic and sedimentation conditions associated with channelized streams and valley plug formation are the main processes influencing site conditions, including soil characteristics, resulting in changes to the floodplain forest communities. Typical BLH forest associations of oak species and baldcypress/tupelo are being reduced along channelized streams and replaced by several disturbance-tolerant species. Seedling densities suggest that these patterns may continue at least in the near future. However, this study and Chapters 2 and 3 have demonstrated that there is

considerable temporal and spatial variability in hydrologic and sedimentation processes associated with valley plugs creating temporal and spatial variability in the observed forest changes. The lack of predictability of abiotic processes associated with valley plugs, particularly with plug expansion, makes the future of these forests, especially in upstream sections, uncertain. Hupp (1992) suggests that 65 years may be required for streams to recover from channelization, but currently no data exist on the time period needed for floodplain forests to recover from valley plug formation. Further research is needed to determine if and when BLH forests recover from valley plugs and the processes involved in their recovery. The floodplain recovery process may be dependent on channel recovery but may also be complicated by anthropogenic disturbances and limitations on seed availability and dispersal.

APPENDIX 5

Table 5-1 Study tributaries with identification of geomorphic feature studied (Diehl 2000). "No. plots*" refers to plots used in the community analysis.

Site - Tributary	Feature	No. Plots	No. Plots*
Spring Creek - GVL	Unchannelized	28	10
Spring Creek - Lower	Unchannelized	36	11
Spring Creek – Sain	Unchannelized	16	5
Bear Creek	Valley Plug	77	34
Jeffers Creek	Valley Plug	55	39
Hickory Creek	Valley Plug	69	34
Piney Creek	Shoal	44	21
Porters Creek	Shoal	42	21

Table 5-2. Importance values (relative density + relative dominance) and frequency of overstory species occurring at unchannelized sites. Frequency is defined as the percentage of plots in which a given species occurred. Species occurring in less than 5% of the plots are not included.

Species	Common Name	Imp. Value	Frequency
Taxodium distichum	Baldcypress	37.25	42.50
Acer saccharinum	Silver Maple	33.15	43.75
Carpinus caroliniana	American Hornbeam	15.08	50.00
Liquidambar styriciflua	Sweetgum	13.58	28.75
Ulmus americana	American Elm	11.33	46.25
Fraxinus pennsylvanica	Green Ash	10.84	35.00
Betula nigra	River Birch	10.70	27.50
Nyssa aquatica	Water Tupelo	7.14	23.75
Quercus lyrata	Overcup Oak	6.95	25.00
Quercus nigra	Water Oak	6.41	28.75
Acer rubrum	Red Maple	6.09	22.50
Celtis occidentalis	Hackberry	5.79	23.75
Nyssa sylvatica	Swamp Tupelo	4.49	21.25
Quercus falcata	Cherrybark Oak	3.64	7.50
Liriodendron tulipifera	Yellow Poplar	3.16	8.75
Salix nigra	Black Willow	2.72	6.25
Quercus michauxii	Swamp Chestnut Oak	2.46	8.75
Carya ovata	Shagbark Hickory	2.22	5.00
Ostrya virginiana	Eastern Hornbeam	2.10	11.25
Quercus phellos	Willow Oak	1.67	7.50
Ulmus alata	Winged Elm	1.39	11.25
Carya tomentosa	Mockernut Hickory	1.17	6.25
Ulmus rubra	Slippery Elm	1.10	8.75
Acer negundo	Boxelder	1.06	5.00
Platanus occidentalis	Sycamore	1.06	6.25

Table 5-3. Importance values (relative density + relative dominance) and frequency of overstory species occurring at shoal sites. Frequency is defined as the percentage of plots in which a given species occurred. Species occurring in less than 5% of the plots are not included.

Species	Common Name	Imp. Value	Frequency
Acer saccharinum	Silver Maple	36.58	67.44
Fraxinus pennsylvanica	Green Ash	27.72	65.12
Carpinus caroliniana	American Hornbeam	22.52	67.44
Liquidambar styriciflua	Sweetgum	13.10	44.19
Nyssa aquatica	Water Tupelo	10.86	23.26
Taxodium distichum	Baldcypress	9.92	15.12
Celtis occidentalis	Hackberry	9.77	39.53
Planera aquatica	Water Elm	8.45	17.44
Quercus michauxii	Swamp Chestnut Oak	8.21	23.26
Ilex opaca	American Holly	6.83	27.91
Ulmus americana	American Elm	5.99	44.19
Quercus falcata	Cherrybark Oak	5.88	27.91
Betula nigra	River Birch	5.38	25.58
Quercus lyrata	Overcup Oak	4.62	13.95
Nyssa sylvatica	Swamp Tupelo	4.58	29.07
Platanus occidentalis	Sycamore	3.20	13.95
Carya tomentosa	Mockernut Hickory	3.07	24.42
Carya aquatica	Water Hickory	2.80	15.12
Acer rubrum	Red Maple	2.56	19.77
Salix nigra	Black Willow	2.21	9.30
Quercus nigra	Water Oak	2.05	6.98
Acer negundo	Boxelder	1.31	6.98

Table 5-4. Importance values (relative density + relative dominance) and frequency of overstory species occurring at valley plug sites. Frequency is defined as the percentage of plots in which a given species occurred. Species occurring in less than 5% of the plots are not included.

Species	Common Name	Imp. Value	Frequency
Acer negundo	Boxelder	38.41	69.41
Salix nigra	Black Willow	30.34	38.36
Fraxinus pennsylvanica	Green Ash	28.74	55.71
Acer rubrum	Red Maple	17.13	46.12
Liquidambar styriciflua	Sweetgum	15.28	35.16
Platanus occidentalis	Sycamore	15.14	36.07
Betula nigra	River Birch	14.97	45.21
Ulmus americana	American Elm	11.28	52.05
Populus deltoides	Cottonwood	6.43	12.79
Carpinus caroliniana	American Hornbeam	4.31	19.18
Taxodium distichum	Baldcypress	3.91	5.48
Ulmus rubra	Slippery Elm	2.53	15.98
Quercus michauxii	Swamp Chestnut Oak	1.70	9.13
Quercus phellos	Willow Oak	1.55	6.85
Acer saccharinum	Silver Maple	1.52	6.39

Table 5-5. Relative density (number of individuals of species x/ total number of individuals of all species) of the most abundant seedlings at valley plug, shoal, and unchannelized sites. Seedlings were defined as tree species >1 m tall and DBH < 4 cm.

Species	Common Name	Valley Plug	Shoal	Unchannelized
Taxodium distichum	Baldcypress	0	0.93	5.59
Salix nigra	Black Willow	3.93	0	1.77
Acer negundo	Boxelder	37.75	12.25	0.55
Cornus spp.	Dogwood	3.42	0	3.96
Ulmus spp.	Elm	7.40	5.33	3.96
Fraxinus pennsylvanica	Green Ash	15.14	6.79	9.00
Carya spp.	Hickory	0.62	2.80	0.68
Carpinus caroliniana	American Hornbeam	3.87	43.81	18.42
Quercus spp.	Oak	3.08	5.33	7.64
Asimina triloba	Pawpaw	0.73	7.19	6.96
Acer rubrum	Red Maple	3.14	3.73	18.28
Betula nigra	River Birch	8.97	1.46	0.68
Acer saccharinum	Silver Maple	0.62	4.53	4.50
Liquidambar styriciflua	Sweetgum	1.23	1.20	1.64
Nyssa spp.	Tupelo	0	2.53	5.45

Table 5-6. Results of indicator species analysis. Significant indicator values appear in bold text.

	Common Name	P-Value	Vegetation Group							
			1	2	3	4	5	6	7	8
1	**Overcup/Maple**									
	Overcup Oak	0.009	**20**	1	0	0	10	0	0	0
	Silver Maple	0.001	**74**	1	2	1	0	1	1	0
	Water Elm	0.035	13	0	0	0	4	0	0	0
2	**Sycamore**									
	American Elm	0.044	3	19	5	5	12	3	7	1
	River Birch	0.169	0	13	4	2	5	4	5	11
	Sycamore	0.001	2	**35**	0	3	0	1	2	5
3	**Mixed**									
	American Holly	0.001	0	0	**35**	0	0	0	0	0
	Swamp Tupelo	0.001	0	0	**46**	0	0	0	0	0
	Cherrybark Oak	0.002	1	0	**36**	0	0	0	0	0
	Hackberry	0.044	10	0	12	1	0	0	0	0
	Mockernut Hickory	0.002	2	0	**27**	0	0	0	0	0
	American Hornbeam	0.001	2	1	**43**	1	20	1	4	0
	Swamp Chest. Oak	0.009	1	0	**20**	2	0	0	2	0
4	**Green Ash**									
	Green Ash	0.001	13	2	2	**53**	0	1	7	2
5	**Baldcypress/Tupelo**									
	Baldcypress	0.001	1	1	0	0	**85**	0	0	0
	Slippery Elm	0.262	0	4	2	5	8	0	1	1
	Water Oak	0.089	0	0	6	0	10	8	0	0
	Water Tupelo	0.001	3	0	1	0	**57**	0	0	0
6	**Sweetgum/Oak**									
	Sweetgum	0.001	0	1	2	6	0	**71**	1	0
	Willow Oak	0.020	0	0	0	3	0	**16**	0	0
7	**Boxelder**									
	Boxelder	0.001	0	12	0	9	1	0	**51**	6
	Cottonwood	0.095	0	4	0	0	0	0	10	4
8	**Willow/Maple**									
	Black Willow	0.001	0	1	0	0	0	0	8	**72**
	Red Maple	0.001	0	14	0	1	0	12	4	**27**

Table 5-7. Percentage of plots that were grouped into each of the eight forest communities by site type.

Association	% Unchannelized	% Shoal	% Valley Plug
Overcup/S. Maple	25.0	75.0	0.0
Sycamore	13.6	9.1	77.3
Mixed	26.7	56.7	16.7
Green Ash	3.3	13.3	83.3
Baldcypress/Tupelo	87.5	12.5	0.0
Sweetgum/Oak	16.7	8.3	75.0
Boxelder	0.0	5.6	94.4
Willow/R. Maple	0.0	0.0	100.0

Table 5-8. Correlations of environmental variables with ordination axes. Variables with the highest correlations are listed in bold text.

Environmental Variable	**Axis 1**		**Axis 2**		**Axis 3**	
	r	**R-sq**	**r**	**R-sq**	**r**	**R-sq**
% Sand	-0.192	0.037	**-0.317**	**0.101**	0.053	0.003
% Silt	0.162	0.026	0.279	0.078	-0.075	0.006
% Clay	0.198	0.039	0.299	0.089	0.023	0.001
Relative Elevation (cm)	0.215	0.046	0.280	0.078	**0.246**	**0.060**
Deposition (cm/yr)	-0.248	0.062	-0.203	0.041	-0.017	0.000
Soil pH	-0.151	0.023	-0.258	0.066	0.110	0.012
Phosphorus	-0.250	0.063	**-0.358**	**0.128**	0.030	0.001
Potassium	**0.362**	**0.131**	**0.362**	**0.131**	0.046	0.002
Calcium	0.291	0.084	0.295	0.087	-0.017	0.000
Magnesium	**0.354**	**0.125**	**0.306**	**0.094**	-0.073	0.005
% Organic Matter	**0.329**	**0.108**	0.253	0.064	-0.176	0.031
NO3-N	0.104	0.011	0.142	0.020	0.179	0.032
Cation Exchange	**0.353**	**0.125**	**0.356**	**0.126**	-0.024	0.001
K:Mg Ratio	-0.181	0.033	-0.046	0.002	-0.102	0.010
Cation Saturation % K	-0.240	0.057	-0.104	0.011	-0.028	0.001
Cation Saturation % Ca	-0.199	0.040	-0.220	0.049	0.002	0.000
Cation Saturation % Mg	0.107	0.011	-0.047	0.002	0.025	0.001
Cation Saturation % H	0.150	0.022	0.206	0.043	-0.013	0.000
Geomorphic Feature	**0.383**	**0.147**	**0.366**	**0.134**	-0.132	0.018

Table 5-9. Mean (1 s.e.) of important environmental variables grouped by species associations. Means with unlike letters within a row differ (P < 0.05).

Association	Relative Elevation (cm)	Deposition (cm/yr)	% Sand	% Organic
Overcup/S.Maple	47.8(14.35)c	0.87(0.20)bc	47.26(6.0)c	1.51(0.16)abc
Sycamore	50.2(27.62)bc	4.72(1.68)abc	64.96(4.79)abc	1.14(0.12)c
Mixed	101.50(15.99)ab	0.88(0.38)c	48.65(3.53)c	1.83(0.13)ab
Green Ash	179.93(25.80)a	2.83(0.68)bc	52.15(5.14)c	1.07(0.11)c
Cypress/Tupelo	-9.99(17.15)c	0.35(0.12)bc	43.24(2.48)c	2.43(0.19)a
Sweetgum/Oak	40.08(43.50)c	5.44(1.88)abc	71.74(7.81)a	0.82(0.20)cd
Boxelder	66.89(15.35)bc	6.63(1.15)a	70.91(4.0)a	0.94(0.11)d
Willow/R. Maple	-56.22(17.09)d	5.33(1.56)ab	76.85(5.09)a	0.65(0.10)d

Association	CEC (meq/100g)	P (ppm)	K (ppm)	Mg (ppm)
Overcup/S.Maple	6.39(0.77)abc	9.05(0.78)cd	55.05(4.07)ab	102.65(13.9)bc
Sycamore	4.9(0.59)cd	18.59(1.75)ab	54.41(4.14)ab	86.45(12.47)d
Mixed	8.02(0.47)a	8.37(1.20)d	68.37(2.92)a	112.5(6.83)b
Green Ash	5.50(0.48)bc	19.13(0.94)b	61.93(4.12)a	104.7(8.16)bc
Cypress/Tupelo	8.09(0.23)ab	7.13(2.06)d	70.38(3.21)a	178.25(12.03)a
Sweetgum/Oak	3.88(0.84)cd	16.17(2.43)bc	38.58(5.86)b	72.83(18.64)d
Boxelder	3.79(0.31)d	21.11(1.15)ab	44.36(2.82)b	72.86(6.46)cd
Willow/R. Maple	2.79(0.37)d	20.94(1.68)ab	38.35(2.71)b	46.29(9.06)d

A) Unchannelized sites

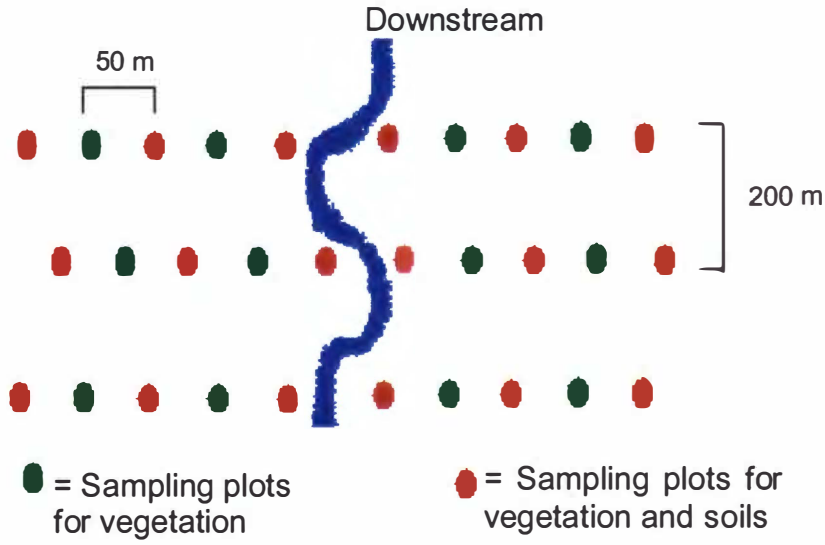


Figure 5-1. Sampling design for vegetation and soil at a) unchannelized sites, b) shoal sites, c) valley plug sites.

B) Shoal sites

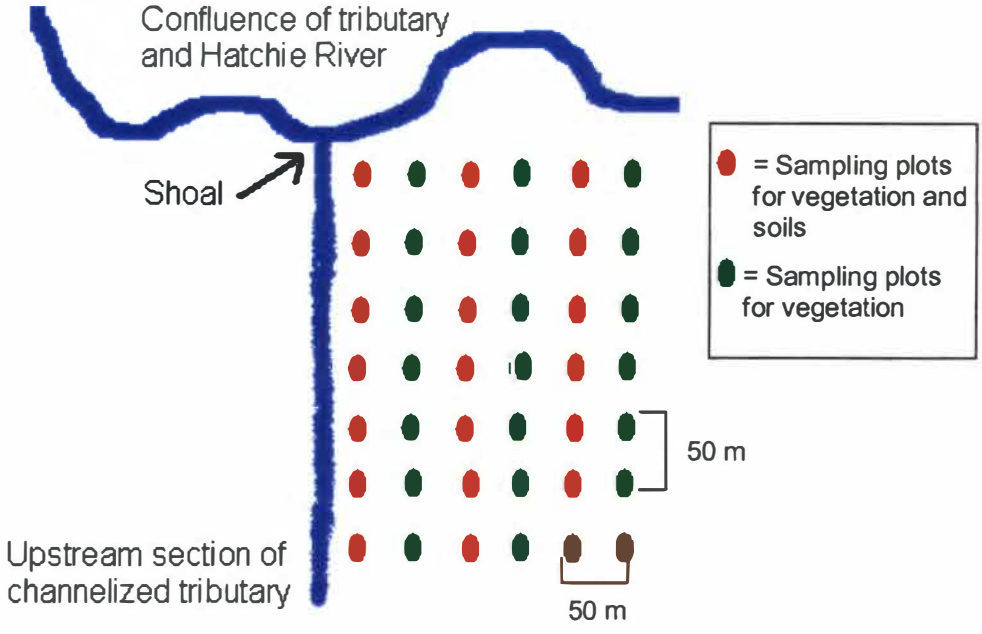


Figure 5-1. Continued.

C) Valley plug sites

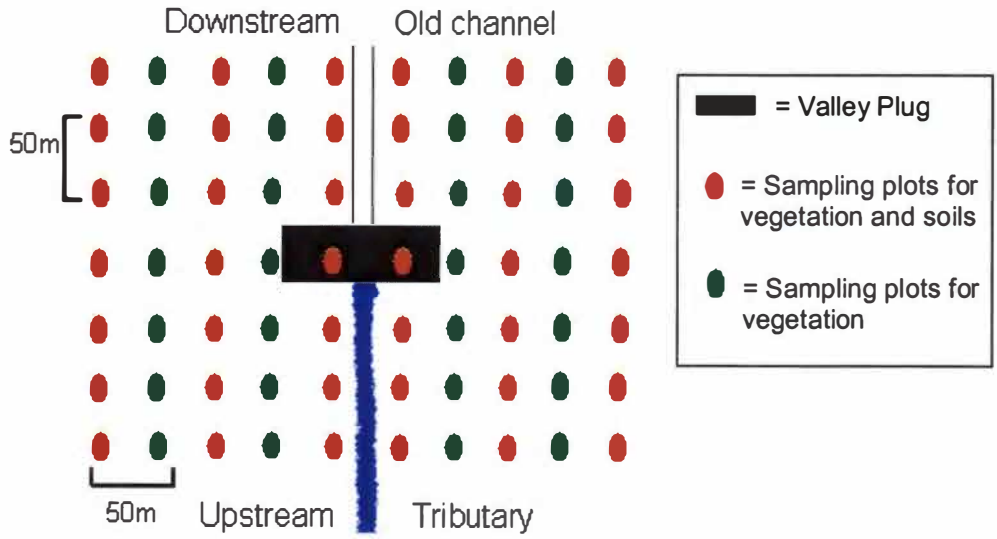


Figure 5-1. Continued.

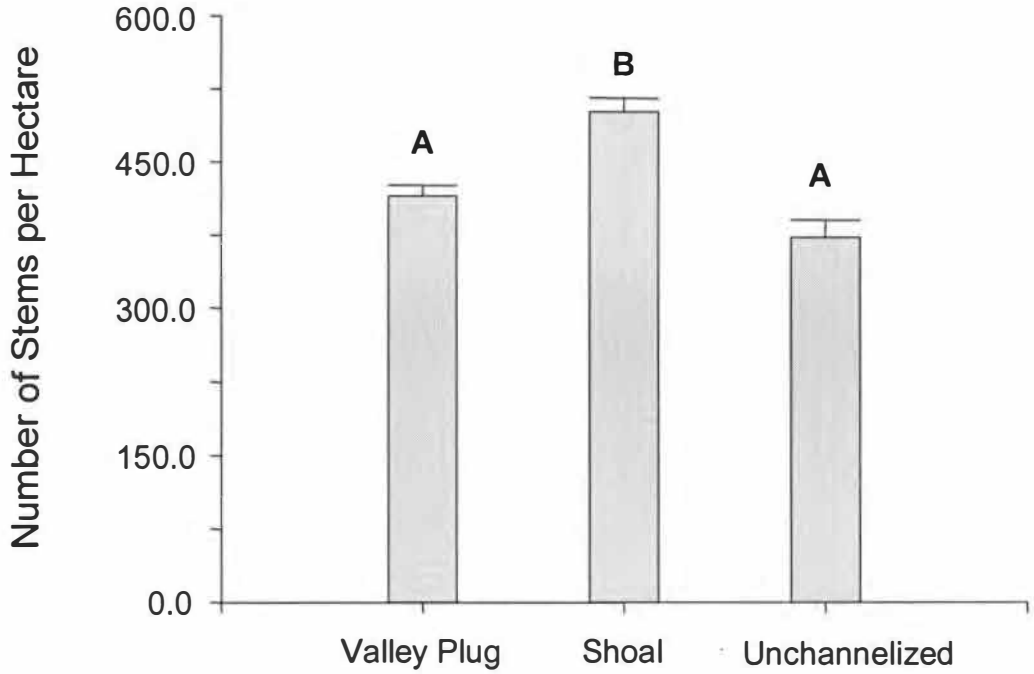


Figure 5-2. Mean number of stems per hectare (+1 standard error) at valley plug, shoal, and unchannelized sites. Bars with unlike letters are different ($P < 0.05$).

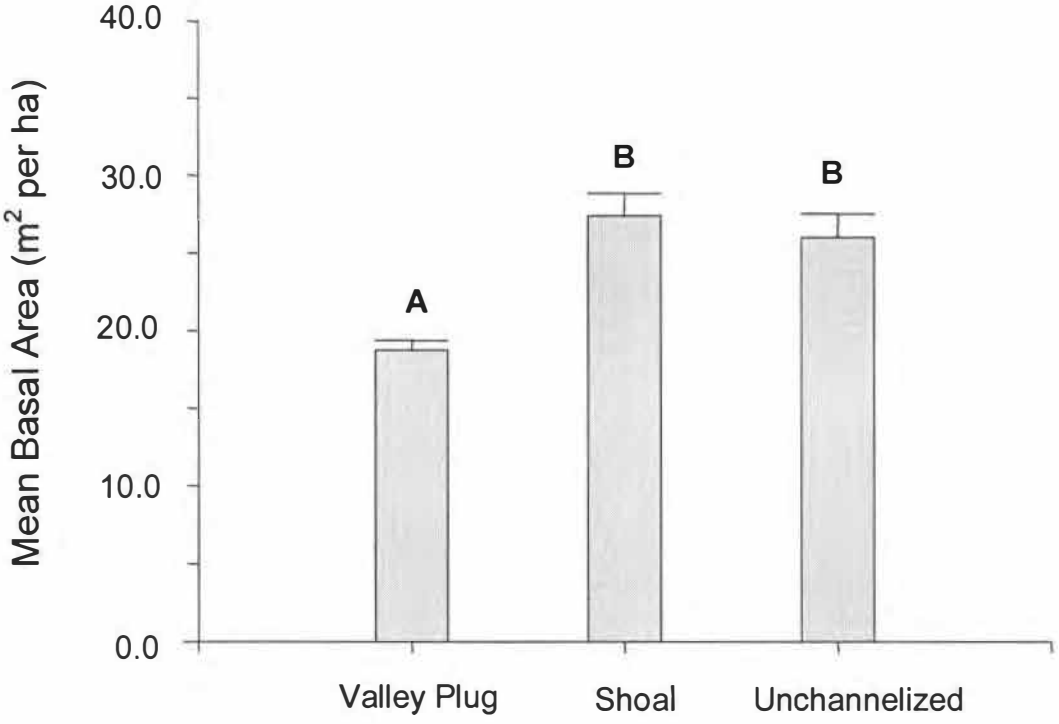


Figure 5-3. Mean basal area (m^2 per ha) (+1 standard error) at valley plug, shoal, and unchannelized sites. Bars with unlike letters are different ($P < 0.05$).

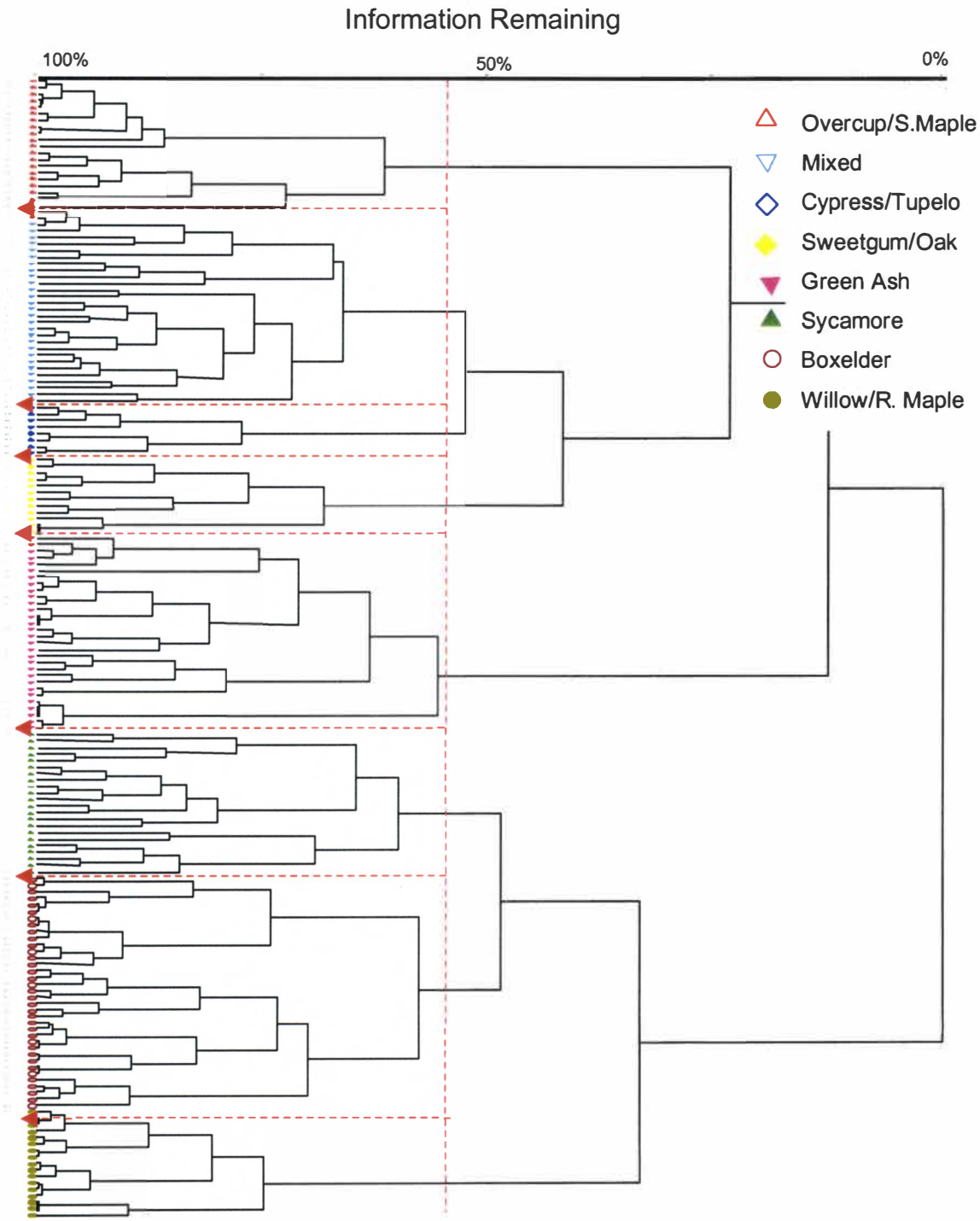


Figure 5-4. Dendrogram of the cluster analysis and indicator species associations. Dashed red lines indicate the decision for group separation.

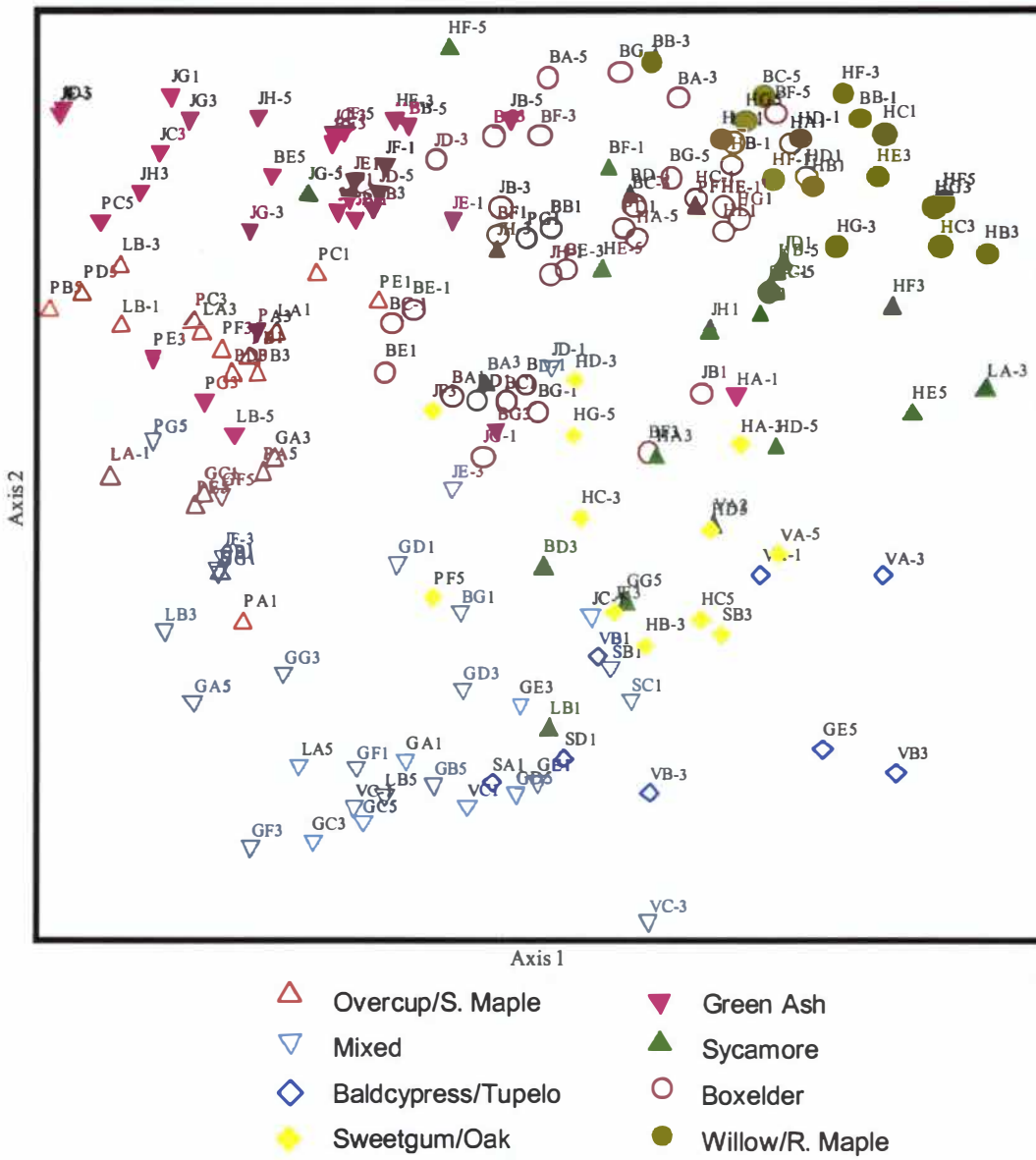


Figure 5-5. Non-metric multidimensional scaling ordination diagram for all plots – Axis 1 vs. Axis 2.

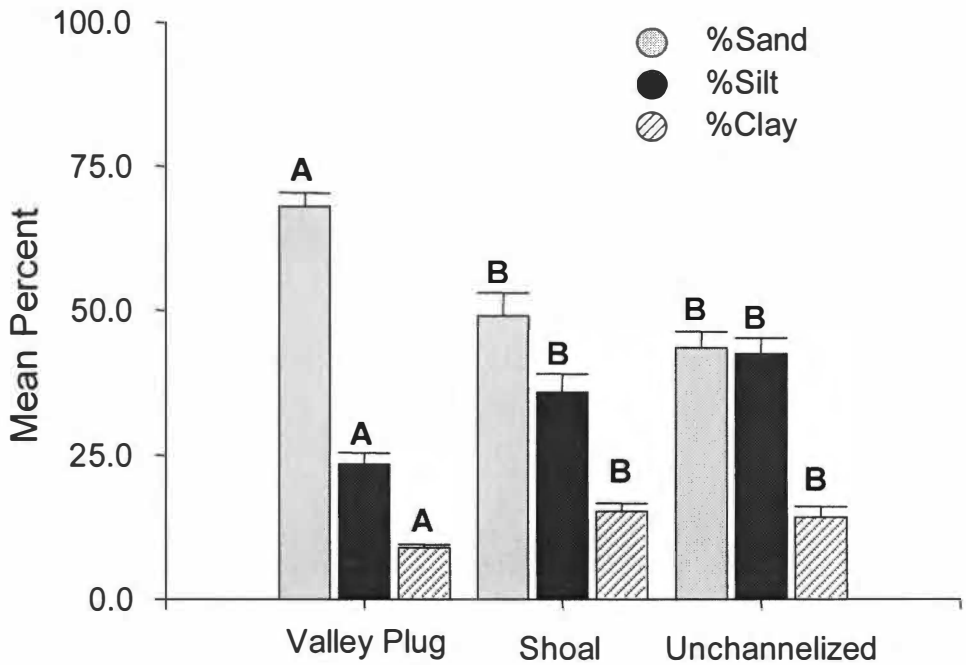


Figure 5-6. Mean (+1 standard error) of the percentage of sand, silt, and clay at valley plug, shoal, and unchannelized sites. Bars for the same variable with unlike letters are different ($P < 0.05$).

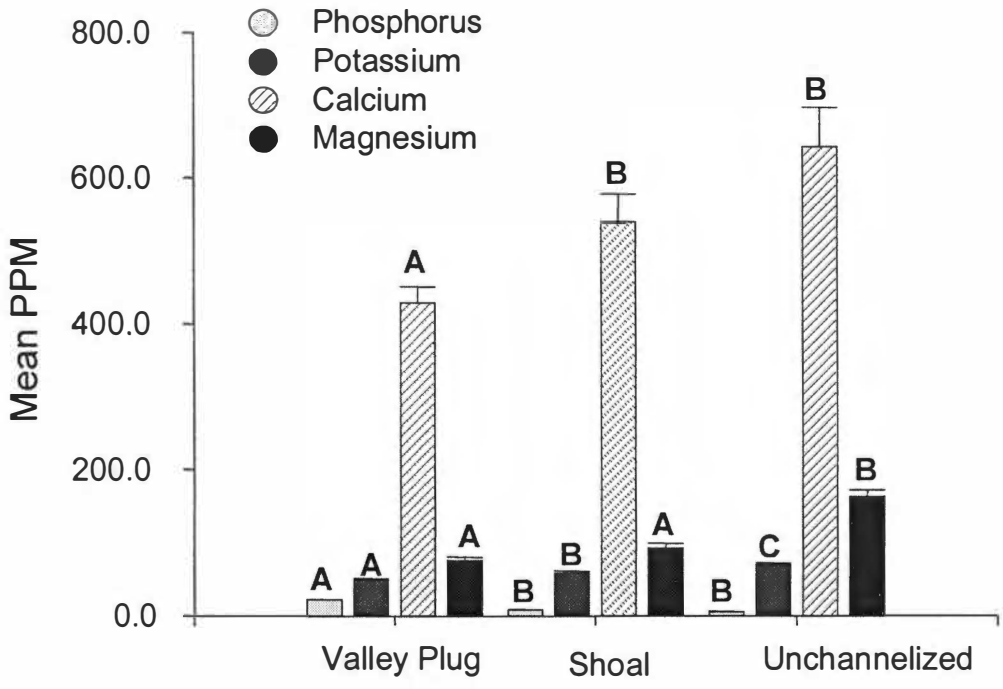


Figure 5-7. Mean (+1 standard error) of phosphorus, calcium, potassium, and magnesium found in the soils of valley plug, shoal, and unchannelized sites. Bars for the same variable with unlike letters are different ($P < 0.05$).

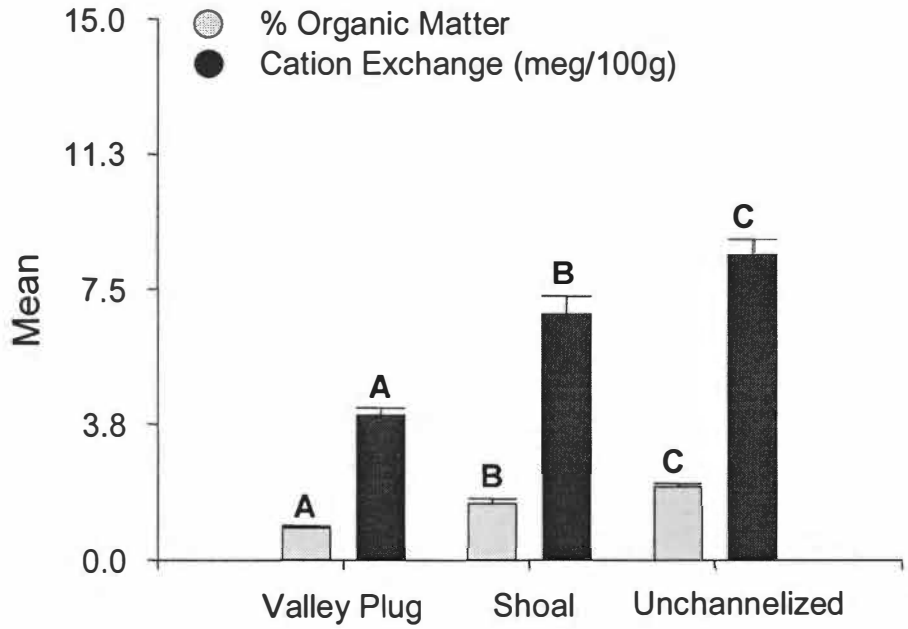


Figure 5-8. Mean (+1 standard error) of the percent organic matter and cation exchange found in the soils of valley plug, shoal, and unchannelized sites. Bars for the same variable with unlike letters are different ($P < 0.05$).

PART VI

CONCLUSIONS

Conclusions

This study was conducted, in part, to aid restoration and conservation efforts by providing a better understanding of the effects that valley plugs and shoals have on sedimentation and hydrologic processes and their resulting impacts on BLH forests. This study determined that the formation of a valley plug can cause greater deposition rates, 10 times greater than at unchannelized sites, over a large extent of the adjacent floodplain. The types of sediment deposited at valley plug sites also differ from sediments deposited in unaltered systems. The greater proportion of coarse sand deposited at valley plug sites suggests that high velocity overbank flows are created as a result of valley plug formation. The geospatial analysis of short-term deposition rates also showed that valley plugs can strongly affect the spatial dynamics of deposition rates by changing the direction of spatial dependence from parallel to stream flow to perpendicular to the stream flow. However, the spatial patterns were variable because of other factors such as channel recovery processes and anthropogenic disturbances. This change in direction of spatial dependence has enabled high deposition rates to impact a large proportion of the floodplain. At valley plug sites, deposition rates had greater temporal variability than in unaltered systems; possibly because of local climatic variability and large amounts of sediment moving through the system when geomorphic thresholds are met.

The dendrogeomorphic analysis has indicated a dramatic increase in sediment deposition rates at valley plug sites since approximately 1970. This result not only corresponds to the time period of channelization of most western

Tennessee streams but also supports previous findings (Hupp and Bazemore 1993) that showed an increase in deposition rates during this time period. The lack of difference in long-term deposition rates by proximity to valley plugs suggests that valley plugs are progressing upstream and can impact new floodplain areas.

Sedimentation in response to shoals was also variable. At one site, the shoal did not seem to influence the sedimentation dynamics, however, the Piney Creek site did have some areas that experienced high deposition rates that correspond to crevasse splays. The crevasse splays also disrupted the spatial continuity of deposition rates across the floodplain and influenced the direction of spatial dependence. Differences in the effects of shoals on deposition rates may be a result of shoal development, but more research is needed to understand the influence of shoals on floodplain sedimentation dynamics during the early stages of formation.

Valley plugs and shoals show altered surface and sub-surface hydrology relative to unchannelized sites; however, some of the differences were unexpected based on our current understanding of valley plugs and shoals. Several authors suggested that flooding would increase around valley plugs as a result of decreased channel storage capacity (Happ et al. 1940, Miller 1990, Shankman and Samson 1991, Diehl 2000). Miller (1990) confirmed these findings as he found open water and marsh communities developing near some valley plugs.

My study, however, indicated that both surface and sub-surface hydrology were affected by channelization and subsequent formation of valley plugs and shoals. Contrary to previous research, surface flooding at valley plug sites in the Hatchie River watershed was less than at unchannelized and shoal sites. This result demonstrates the variability in hydrologic responses to valley plug formation. Water tables were also lower at valley plug and shoal sites, as a result of channel bed lowering during channelization.

Evaluation of environmental variables indicated that surface and sub-surface hydrology, sediment deposition rates, types of sediment deposited, and macronutrient concentrations of floodplain soils are affected by valley plug formation. These alterations have created environmental gradients that are strongly affecting species composition and stand structure of associated BLH forests. Floodplain forests associated with valley plugs had lower tree species diversity and no longer contained the typical associations of oak species and baldcypress/tupelo that existed at both shoal and unchannelized sites. Forest associations consisting of disturbance-tolerant tree species dominated floodplains adjacent to valley plugs; seedling densities suggest that these associations will continue at least in the near future. Nevertheless, the considerable variability associated with abiotic processes influenced by valley plugs and shoals have created temporal and spatial variability in the observed forest changes. The considerable variability of abiotic processes associated with valley plugs, particularly with plug expansion, makes the future of these forests, especially in upstream sections, uncertain.

Plant succession involves changes in species composition that result from differences in life history characteristics that control dispersal ability, establishment requirements, tolerances to various environmental stresses, and competition with other individuals (Platt and Connell 2003). Establishment of an individual depends on its ability for dispersal to the site and its ability to germinate and survive at the site. My greenhouse experiment focused on the effects of flooding and sedimentation on the probabilities of germination and early growth of two BLH tree species. The results suggest that flood duration prior to germination can have both negative and positive effects on germination rates. The direction of the effect depends on the life history characteristics of the plant species. Sediment texture and rate of deposition seemed to also have an effect on germination rate, but was secondary to flood duration. The main effect of the sediment treatment was that greater deposition rates reduce the above ground height of individuals and may reduce their competitive abilities to survive.

In the case of overcup oak, which has a high tolerance to flooding, germination potential and above-ground height were reduced by a shorter hydroperiod and lowest when subjected to sedimentation rates combined with a short hydroperiod. This study has shown that high deposition rates and reduced flooding are occurring at valley plug sites and may explain the low relative density of oak seedlings at valley plug sites. In addition, if other species with a high tolerance of flooding, such as baldcypress and water tupelo, react in a similar way to reduced flooding and high deposition rates as overcup oak, this may also explain the low seedling densities of these species at valley plug sites.

The hydrologic and sedimentation responses to valley plug formation demonstrated in this study may have dramatic impacts on floodplain processes and functions. In floodplain systems, the primary driving process responsible for the existence, productivity, and interactions of the major biota is periodic overbank flooding, also known as the flood pulse (Junk et al. 1989). The fertility of floodplain soils depends on nutrient inputs from the main channel and the quality of deposited sediments from overbank flooding (Wharton et al. 1982, Junk et al. 1989, Stanturf and Schoenholtz 1998). The reduced flood pulse and increased sand deposits (Chapter 2) that are occurring at valley plug sites may be reducing the fertility of the floodplain soils and directly influencing the establishment and growth of BLH tree species. Although the effects of flood reduction on BLH forests have received little study (Bedinger 1978), decreased flooding has been shown to reduce tree growth and seed production (Burgess et al. 1973). Reduced flooding at valley plug sites may also influence seed dispersal of many BLH tree species that have adapted seed production cycles to the timing of flood pulses for dispersal by water and fish (Junk et al. 1989).

There is also some evidence that suggest that even though water tables were lower at valley plug sites, root systems of trees at these sites may be inundated for extended periods of time ($\bar{x}$ = 32.75 ± 12.76 days) during the growing season. Prolonged inundation of root systems can cause stress, low seed production, reduced growth, and mortality of some BLH tree species (Happ et al. 1940, Kozlowski 2002). Thus, prolonged inundation of root systems may

have a strong selective force on BLH forest composition and structure that may lead to a forest dominated by disturbance tolerant species.

Hosner and Boyce (1962) showed that seedlings of some BLH tree species, like cherrybark oak (*Quercus falcate*) and willow oak (*Quercus phellos*), experienced shoot mortality and root dormancy in response to 30 days of soil saturation. However, soil saturation also resulted in increased growth of other species like green ash (*Fraxinus pennsylvanica*), suggesting that soil saturation of 30 days or more results in a disparity of competitive advantages for BLH tree species. On the other hand, groundwater levels at shoal sites never reached the root collar depths of trees during the growing season of 2003 and 2004. The response of groundwater levels to channelization at shoal sites may be causing a drought effect and limiting the access of BLH trees to groundwater during the growing season, thus favoring drought resistant species.

Restoration

Selection of the appropriate restoration approach depends on the specific objectives of the project, however, in the case of the Hatchie River watershed, the objectives have not been clearly defined. This has resulted in confusion and concern among landowners, conservation organizations, state, and federal agencies, over the several restoration options that have been presented and discussed for the tributary systems of the Hatchie River. These restoration options include: (1) "hands off" approach, where valley plugs are left in place to prevent sediment from moving downstream into the Hatchie River, (2) creating artificial valley plugs to prevent tributaries from transporting sediment into the

Hatchie River, (3) restoring the fluvial portion of the tributaries, including restoration of the channels and gully regions, and (4) restoring both the fluvial systems and the floodplain.

The short-term deposition analysis indicated that valley plugs are protecting downstream sections from excessive sedimentation. This result is extremely important, as one of the restoration options under consideration is to leave valley plugs in place or add new ones to prevent large quantities of sediment from reaching the Hatchie River. However, it is essential to note that this effect of downstream protection is only short-term. Field observations and pervious research (Happ et al. 1940) suggest that eventually the plug will be circumvented by the formation of new channels. This may result in formation of other plugs either upstream or downstream of the previous plug. For example, along a three mile stretch of Clover Creek, I encountered four former valley plugs in which the stream had created new channels through and around the valley plugs, resulting in a swamped floodplain similar to those, described by Miller (1990) and Oswalt and King (*In Press*), along the Middle Fork-Forked Deer River. In the Hatchie River watershed, tributary floodplains that have been degraded by channelization and valley plugs have been reduced in economic value, mainly timber value, by \$5,438 ha^{-1} (Wells 2004). Changes in forest composition and structure may also affect wildlife communities.

Another option that has been discussed is hydrologic restoration to reduce the transport capacity of the tributaries. This strategy may be effective if (1) there is also a stabilization of sediment sources, including gully erosion and channel

erosion/bank failure, and (2) the restoration occurs over the entire tributary system with particular attention to the stream gradient and channel depth at the confluence of the Hatchie River to prevent head-cutting. If these factors are included in the restoration project, then sediment input into the system would be stabilized and geomorphic readjustment caused by changes in stream power, flow velocity, and stream gradient may be minimized. However, this would only restore the fluvial system, which may benefit the Hatchie River, but may not ensure restoration of the floodplain system. Although restoration of the fluvial system is a necessary first step in restoring the floodplain system, there are other processes involved that could prevent the establishment of typical BLH tree species in the floodplain. Additional concerns include the loss of microtopography as a result of excessive deposition and seed availability/dispersal. Microtopography has a direct effect on flooding, which can determine the distribution of BLH tree species (Hodges 1997). At valley plug sites, high deposition rates over a large extent of the floodplain may have reduced the microtopography of the floodplain. My greenhouse experiments and previous research (Hosner 1957, Briscoe 1961, Larsen 1963, Outcalt 2002) have shown that BLH tree species respond differently to flood duration depending on life history characteristics.

Seed availability may also be a limiting factor in the restoration of the floodplain. The results from my study indicated that typically common BLH tree species including oaks, baldcypress, and tupelo were absent from valley plug sites. Dispersal of seeds from upstream locations may also be minimal because

of intensive agricultural practices that have reduced most of the upstream BLH forests. Particular attention should also be given to the impact of channel restoration projects on water tables in the floodplain. It has been demonstrated in this study and others (Tucci and Hileman 1992) that channel alterations can have a significant impact on water tables.

Restoration of both the fluvial system and the floodplain would have to involve consideration of the above-mentioned factors as well as others. Other factors included successful collaboration with landowners and overcoming communication difficulties and differences in goals. The cooperation of private landowners will be a critical factor in any restoration effort within the Hatchie River watershed. Organizations involved in restoration activities have to be able to successfully communicate with landowners about the variability within these systems and the variety of restoration options open to them. Successful restoration will depend on their collaborative efforts.

Needs

Although this study was able to further our understanding of BLH systems and the effects of valley plugs and shoals on critical processes of these systems, many questions still remain to be answered. Valley plug expansion seems to be unpredictable as a result of climate variability and geomorphic thresholds. Further study is necessary to understand the rates and processes involved in upstream expansion of valley plugs; this would be useful in prioritizing restoration efforts and predicting potential impacts to upstream forest communities.

Further investigation is also needed to understand the influence that factors, such as channel recovery processes and anthropogenic disturbances, have on overbank sedimentation dynamics in conjunction with valley plugs and shoals. The results of this study clearly show the variability in responses of sedimentation to valley plug and shoal formation and indicate the complexity of these systems. For BLH conservation and restoration efforts to be successful, a clearer understanding of overbank sedimentation associated with valley plugs, shoals, and other potential influential factors is needed.

Recovery processes of channelization also seem to be changing the hydrological conditions at valley plug and shoal sites, but our understanding of these relationships is still rudimentary. The mechanisms involved in the creation of permanently flooded areas and different developmental stages of valley plug formation are still poorly understood. Further research is needed to test hypotheses of surface and sub-surface hydrological response to valley plug formation related to stage of development and specific site conditions in order to understand the factors influencing the variability of hydrological responses. This information will also be useful for understanding the implications of valley plug formation on BLH forests and will enhance management and restoration efforts.

The variability of abiotic processes associated with valley plugs, particularly plug expansion, makes the future of the adjacent forests, especially in upstream sections, uncertain. Hupp (1992) suggested that 65 years may be required for streams to recover from channelization, but currently no data exist on the time period needed for floodplain forests to recover from valley plug

formation. The floodplain recovery process may depend on channel recovery, but may also be complicated by anthropogenic disturbances and limitations of seed availability and dispersal. Further research is needed to determine if and when BLH forests recover from valley plugs and the processes involved in their recovery.

Overbank flooding velocity has received little study in floodplain systems, but flow velocity can have a major influence on plant communities. Johnson (2000) showed that tree recruitment and seedling mortality were mainly influenced by stream flow pulses that either eroded or buried seedlings. The deposition rates measured at valley plug sites (Chapter 2) suggest that flooding velocities are much greater at valley plug sites than at unchannelized sites. This difference may also influence the composition and structure of floodplain plant communities. High flow velocities, sediment loads, and turbidity also reduce primary production affecting the biological processes within the systems (Junk et al. 1989). Additional research is needed to gain a better understanding of the variability in hydrological responses to valley plugs and the influence of channelization recovery processes.

Restoration efforts would also benefit by research that identifies and categorizes the amount of degradation that has occurred within the Hatchie River watershed. Sites could then be prioritized based on level of degradation, landowner desires, and potential to improve the ecological conditions of the site and contribution to the entire system. In addition, if sacrificing some tributary systems to protect the Hatchie River main stem is going to be considered a

viable option, then research should be conducted to determine the ecological costs of losing these systems and potential impacts on the watershed system. My research on the bird communities supported by the BLH forests along the tributary systems suggests that tributary systems may have significant ecological importance as undisturbed sites supported a diverse array of species including the Swainson's warbler (*Limnothlypis swainsonii*) and Cerulean warbler (*Dendroica cerulea*). More research is needed to determine the overall significance of these tributary systems to the entire watershed, including their role in the conservation of floodplain fish and Neotropical migrant songbirds.

LITERATURE CITED

Literature Cited

Allen, J.R.L. 1965. A review of the origin and characteristics of recent alluvial sediments. Sedimentology **5**(2):89-191.

AMS. 2000. American Meteorological Society Glossary, 2nd edition. Boston, MA.

AOSA Tetrazolium Testing Handbook. 2000.

Ashley, G.H. 1910. Drainage Law of Tennessee. State of Tennessee, State Geological Survey, Nashville, TN. Bulletin: Drainage reclamation in Tennessee.

Aurenhammer, F. 1991. Voronoi Diagrams – A survey of a fundamental geometric data structure. ACM Computing Surveys **23**:345-405.

Barbujani, G. 1988. Detecting and comparing the direction of gene-frequency gradients. Journal of Genetics **67**:129-140.

Barnhardt, M.L. 1988. Historical sedimentation in West Tennessee gullies. Southeastern Geographer **28**(1):1-18.

Barnes, B.V., D.R. Zak, S.R. Denton, S.H. Spurr. 1998. Forest Ecology, 4th edition. John Wiley & Sons, Inc. New York. Pages 445-446.

Barstow, C.J. 1971. Impact of channelization on wetland habitat in the Obion-Forked Deer Basin, Tennessee. Pages 362-376 in J.B. Trefethen (eds.) Transactions of the Thirty-Sixth North American Wildlife and Natural Resources Conference. Wildlife Management Institute, Washington, D.C.

Baumann, R., J. Day, and C. Miller. 1984. Mississippi deltaic wetland survival: sedimentation versus coastal submergence. Science **224**:1093-1095.

Bazemore, D., C. Hupp, T.H. Diehl. 1991. Wetland sedimentation and vegetation patterns near selected highway crossings in west Tennessee. Nashville,TN, U.S. Geological Survey Water Resource Division: 1-46.

Bedinger, M.S. 1978. Relation between forest species and flooding. Pages 427-435 in P.E. Greeson, J.R. Clark, and J.E. Clark, editors. Wetland functions and values: the state of our understanding. Proceedings of the national symposium on wetlands, Lake Buena Vista, Florida, USA.

Begon, M., J.L. Harper, and C.R. Townsend. 1990. Ecology: Individuals, populations and communities. Second edition, Blackwell Scientific Publications, Boston , Mass.

Bloom, A.L. 1978. Geomorphology: A systematic analysis of late Cenozoic landforms. Prentice-Hall, Inc. New Jersey.

Boto, K.G. and Patrick, Jr. W.H. 1979. Role of wetlands in the removal of suspended sediments. In P.E. Greeson et al. edited, Wetland functions and values: State of our understanding. American Water Resources Association, Minneapolis, MN, p. 479-489.

Brinson, M. 1990. Riverine forests. Pages 87-141 *in* S. Brown, editor. Forested wetlands. Elsevier Science Publishers, New York.

Briscoe, C.B. 1961. Germination of cherrybark and nuttall oak acorns following flooding. Ecology 42:430-431.

Burke, M.K., S.L. King, D. Gartner, and M.H. Eisenbies. 2003. Vegetation, soil, and flooding relationships in a blackwater floodplain forest. Wetlands **23**:988-1002.

Burgess, R.L., W.C. Johnson, and W.R. Keammerer. 1973. Vegetation of the Missouri River floodplain in North Dakota. North Dakota Water Resources Research Institute Report WI-221-018-73, p. 162.

Burns, R.M., and B.H. Honkala. 1990. Silvics of North America. Volume 2. Hardwoods. U.S. Forest Service, Washington, D.C.

Burt, T.P., P.D. Bates, M.D. Stewart, A.J. Claxton, M.G. Anderson, and D.A. Price. 2002. Water table fluctuations within the floodplain of the River Severn, England. Journal of Hydrology **262**:1-20.

Clark, J.R. and J. Benforado. 1981. Wetlands of bottomland hardwood forests: Proceedings of a workshop on bottomland hardwood forest wetlands of the southeastern U.S. Elsevier Science Publishing Company, New York, New York, USA.

Darby, S.E. and Simon, A. 1999. Incised River Channels: Processes, Forms, Engineering, and Management. John Wiley and Sons, New York, pp. 442.

DeLaune, R.D., R.J. Buresh, and W.H. Patrick, Jr. 1978. Sedimentation rates determined by 137 Cs dating in a rapidly accreting salt marsh. Nature **275**:532-533.

Diehl, T. 2000. Shoals and valley plugs in the Hatchie River Watershed. 00-4279, USGS, Nashville, TN.

Diehl, T. 2002. U.S. Geological Survey. Personal communication. Nashville, TN.

Dollar, K., S. Pallardy, and H. Garrett. 1992. Composition and environment of floodplain forests of northern Missouri. Canadian Journal of Forest Resources **22**:1343-1350.

DuBarry, A.P. 1963. Germination of bottomland tree seed immersed in water. Journal of Forestry **61**:225-226.

Dufrene, M., and P. Legendre. 1997. Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecological Monographs **67**:345-366.

Dunn, C., and F. Stearns. 1987. A comparison of vegetation and soils in floodplain and basin forested wetlands of southeastern Wisconsin. American Midland Naturalist **118**:375-384.

Emerson, J. 1971. Channelization: A case study. Science **173**(3994):325-326.

EPA. 2005. Hatchie River watershed. http://cfpub.epa.gov/surf/huc.cfm?huc_code08010208.

ESRI. 2004. ArcGIS Version 9.0. Redlands, California.

Flowers, R. L. 1964. Soil survey of Fayette county, Tennessee. U.S. Dept. of Agric. Soil Surv. Service. No. 13. 168 pp.

Gilvear, D., and J.P. Bravard. 1996. Geomorphology of temperate rivers. Pages 69-97 in G.E. Petts and C. Amoros, eds. Fluvial Hydrosystems. Chapman & Hall, London.

Gomez, B. 1991. Bedload Transport. Earth-Science Review, **31**:89-132.

Grace, J. B., L. Allain, and C. Allen. 2000. Vegetation associations in a rare community type - coastal tallgrass prairie. Plant Ecology **147**:105-115.

Gross, K.L., K.S. Pregitzer, and A.J. Burton. 1995. Spatial variation in nitrogen availability in three successional plant communities. Journal of Ecology **83**(3):357-367.

Hall, R.B.W. and P.A. Harcombe. 1998. Flooding alters apparent position of floodplain saplings on a light gradient. Ecology **79**:847-855.

Happ, S.C. 1975. Genetic classification of valley sediment deposits. Pages 286-292 in V.A. Vanoni ed. Sedimentation engineering: New York, American Society of Civil Engineers.

Happ, S., G. Rittenhouse, and G. Dobson. 1940. Some principles of accelerated stream and valley sedimentation. Technical Bulletin 695, U.S. Department of Agriculture.

Harms, W., H. Schreuder, D. Hook, and C. Brown. 1980. The effects of flooding on the Swamp Forest in Lake Ocklawaha, Florida. Ecology **61**:1412-1421.

Harrelson, C.C., Rawlins, C.L., Potyondy, J.P. 1994. Stream channel reference: an illustrated guide to field technique. Gen. Tech. Rep. RM-245. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 61 p.

Hefner, J., and J. Brown. 1985. Wetland trends in the southeastern United States. Wetlands **4**:1-11.

Heimann, D.C. and M.J. Roell. 2000. Sediment loads and accumulation in a small riparian wetland system in Northern Missouri. Wetlands **20**(2):219-231.

Heitmeyer, M., and L. Fredrickson. 1981. Do wetland conditions in the Mississippi Delta hardwoods influence mallard recruitmen? Transactions of the North American Wildlife and Natural Resources Conference **46**:44-57.

Hidinger, L.L. and A.E. Morgan. 1912. Drainage problems of Wolf, Hatchie, and South Fork of Forked Deer Rivers, in West Tennessee. The Resources of Tennessee **2**(6):231-249.

Hintze, J. 2001. NCSS and PASS. Number Cruncher Statistical Systems. Kaysville, Utah.

Hodges, J.D. 1997. Development and ecology of bottomland hardwood sites. Forest Ecology and Management **90**:177-125.

Hodges, J.D. and G.L. Switzer. 1979. Some aspects of the ecology of southern bottomland hardwoods. Pages 360-365 in North America's forests: gateway to opportunity. Proceedings of the Society of American Foresters and the Canadian Institute of Forestry. Society of American Foresters, Washington, D.C.

Hohensinner, S., H. Habersack, M. Jungwirth, and G. Zauner. 2004. Reconstruction of the characteristics of a natural alluvial river-floodplain system and hydromorphological changes following human modifications: The Danube River (1812-1991). River Research and Applications **20**:25-41.

Hoover, J.J, and K.J. Killgore. 1997. Fish communities. Pages 237-260 in M.G. Messina and W.H. Conner, eds. Southern forested wetlands: ecology and management. Lewis Publishers, Boca Raton, FL.

Hopkins, W.G. 1995. Introduction to plant physiology. John Wiley & Sons, Inc. New York.

Hopkinson, C.S. and J.J. Vallino. 1995. The relationships among mans activities in watersheds and estuaries. Estuaries **18**:598-621.

Hosner, J. 1957. Effects of water upon the seed germination of bottomland trees. Forest Science **3**:67-70.

Hosner, J. 1960. Relative tolerance to complete inundation of fourteen bottomland tree species. Forest Science **6**(3):246-251.

Hosner, J., and S. Boyce. 1962. Tolerance to water saturated soil of various bottomland hardwoods. Forest Science **8**:180-186.

Huffman, R.T. 1980. The relation of flood timing and duration to variation in selected bottomland hardwood communities of southern Arkansas: Final report. US Army Corps of Engineers miscellaneous paper EL-80-4. Washington, D.C.

Hughes, F.M.R., N. Barsoum, K.S. Richards, M. Winfield, and A. Hayes. 2000. The response of male and female black poplar (Populus nigra L. subspecies betulifolia (Pursh) W. Wettst.) cuttings to different water table depths and sediment types: implications for flow management and river corridor biodiversity. Hydrological Processes **14**:3075-3098.

Hunter, W., M. Carter, D. Pashley, and K. Barker. 1993. The Partners in Flight Species Prioritization Scheme. General Technical Report RM-227, USDA Forest Service, Fort Collins, CO.

Hupp, C. 1992. Riparian vegetation recovery patterns following stream channelization: a geomorphic perspective. Ecology **73**(4): 1209-1226.

Hupp, C. 2000. Hydrology, geomorphology, and vegetation of Coastal Plain Rivers in the southeastern United States. Hydrological Processess **14**:2991-3010.

Hupp, C., and D. Bazemore. 1993. Temporal and spatial patterns of wetland sedimentation, West Tennessee. Journal of Hydrology **141**:179-196.

Hupp, C. and E. Morris. 1990. A dendrogeomorphic approach to measurement of sedimentation in a forest wetland, Black Swamp, Arkansas. Wetlands **10**:107-124.

Hupp, C., and W. Osterkamp. 1985. Bottomland vegetation distribution along Passage Creek, Virginia, in relation to fluvial landforms. Ecology **66**:670-681.

Hupp, C.R. and A. Simon. 1986. Vegetation and bank-slope development. Interagency Sedimentation Conference, 4th, Las Vegas, Nevada. Proceedings: Interagency Committee on Water Data, Subcommittee on Sedimentation, Vol. 2:5-83 to 5-92.

Hupp, C. and A. Simon. 1991. Bank accretion and the development of vegetated depositional surfaces along modified alluvial channels. Geomorphology **4**:111-124.

Isaaks and Srivastava 1989. An introduction to applied geostatistics. Oxford University Press, New York.

Johnson, F., and D. Bell. 1976. Tree growth and mortality in the streamside forest. Castanea **41**:34-41.

Johnson, W. 1994. Woodland expansions in the Platte River,Nebraska: Patterns and causes. Ecological Monographs **64**:45-84.

Johnson, W. 2000. Tree recruitment and survival in rivers: influence of hydrological processes. Hydrological Processes **14**:3051-3074.

Johnston, C.A., G.D. Bubenzer, G.B. Lee, F.W. Madison, and J.R. McHenry. 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. Journal of Environmental Quality **13**:283-290.

Jones, R., R. Sharitz, P. Dixon, D. Segal, and R. Schneider. 1994. Woody plant regeneration in four floodplain forests. Ecological Monographs **64**:345-367.

Junk, W.J., P.B. Bayley, and R.E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Pages 110-127 in D.P. Dodge, editor. Proceedings of the International Large River Symposium. Can. Spec. Publ. Fish. Aquat. Sci. 106.

Keeland, B.D. and R.R. Sharitz. 1997. The effects of water-level fluctuations on weekly tree growth in a southeastern USA swamp. American Journal of Botany **84** (1):131-139.

King, S. 1995. Effects of flooding regimes on two impounded bottomland hardwood stands. Wetlands **15**:272-284.

Kleiss, B.A. 1996. Sediment retention in a bottomland hardwood wetland in Eastern Arkansas. Wetlands **16**(3):321-333.

Knighton, D. 1998. Fluvial forms and processes, London, Arnold. Pp.383.

Kozlowski, T. and S. Pallardy. 1997. Growth Control in Woody Plants. Academic Press. Page 23.

Kozlowski, T.T. 2002. Physiological-ecological impacts of flooding on riparian forest ecosystem. Wetlands **22**(3):550-561.

Krebs, C.J., 1994. Ecology. HaperCollins College Publishers, New York, NY. pp. 438.

Larsen, H.S. 1963. Effects of soaking in water on acorn germination of four southern oaks. Forest Science 9:236-241.

Leck, M.A., R.L. Simpson, and V.T. Parker. 1989. The seed bank of a freshwater tidal wetland and its relationship to vegetation dynamics. Pages 189-205 in R.R. Sharitz and J.W. Gibbons Eds., Freshwater Wetlands and Wildlife. USDOE Office of Scientific and Technical Information, Oak Ridge, TN.

Le Corre, V., G. Roussel, A. Zanetto, and A. Kremer. 1998. Geographical structure of gene diversity in *Quercus petraea* (Matt.) Liebl. III. Patterns of variation identified by geostatistical analyses. Heredity **80**:464-473.

Leopold, L.B., G.M. Wolman, and J.P. Miller. 1992. Fluvial processes in geomorphology. Dover Publications, Inc., New York, New York, USA. Pp.522.

Lockaby, B.G and M.R. Walbridge. 1998. Biogeochemistry. Pages 149-172 in M.G. Messina and W.H. Conner, editors. Southern Forested Wetlands: ecology and management. Lewis Publishers, Chelsea, MI.

Lotti, T. 1959. Selecting Sound Acorns For Planting Bottomland Hardwood Sites. Journal of Forestry **57**:923.

Marks, P.L. and P.A. Harcombe. 1975.Community diversity of coastal plain forests in southeast Texas. Ecology **56**:1004-1008.

McCune, B. and J.B. Grace. 2002. Analysis of ecological communities. MJM Press, Gleneden Beach, OR.

McCune, B. and M.J. Mefford. 1999. PC-ORD. Multivariate analysis of ecological data, Version 4. MjM Software Design, Gleneden Beach, Oregon.

McIntyre, S.C. and J.W. Naney. 1991. Sediment deposition in a forested inland wetland with a steep-farmed watershed. Journal of Soil and Water Conservation **46**(1):64-66.

McGee, C.E. 1986. Budbreak for twenty-three upland hardwoods compared under forest canopies and in recent clearcuts. Forest Science **32**:924-935.

Meade, R.H. 1985. Wavelike movement of bedload sediment, East Fork River, Wyoming. Environmental Geology and Water Sciences **7**: 215-232.

Microsoft Corporation. 2000. Excel. USA.

Miller, N.A. 1990. Effects of permanent flooding on bottomland hardwoods and implications for water management in the Forked Deer River Floodplain. Castanea **55**:106-112.

Miller, R.A., W.D. Hardeman, and D.S. Fullerton. 1966. Geologic map of Tennessee. Tennessee Division of Geology, TN.

Minitab Inc. 2004. Minitab Release 14 Statistcal Software.

Mitsch, W.J., G.L. Dorge, and J.R. Wiemhoff. 1979. Ecosystem dynamics and a phosphorus budget of an alluvial swamp in southern Illinois. Ecology **60**:1116-1124.

Mitsch, W.J. and J.G. Gosselink. 2000. Wetlands: 3rd edition. John Wiley and Sons, Inc. New York, New York, USA.

M.J.M. Software Design. 1999. PC-ORD. Multivariate analysis of ecological data. MJM Software Design. Gleneden Beach, OR.

Morgan, A.E. and S.H. McCrory. 1910. Preliminary report upon the drainage of the lands overflowed by the North and Middle Forks of the Forked Deer River and the Rutherford Fork of the Obion River in Gibson County, TN. State of Tennessee, State Geological Survey, Nashville, TN.

Muzika, R.M., J.B. Gladden, and J.D. Haddock. 1987. Structural and functional aspects of succession in southeastern floodplain forests following major disturbance. American Midland Naturalist **117**:1-9.

Nakamura, F., M. Jitsu, S. Kameyama, and S. Mizugaki. 2002. Changes in riparian forests in the Kushiro Mire, Japan, associated with stream channelization. River Research and Applications **18**:65-79.

National Oceanic and Atmospheric Administration (NOAA). 2005. Climate Division. http://www.noaa.gov.

Nichols, M. 2002. U.S. Fish and Wildlife Service. Personal communication. Brownsville, TN.

Odum, E.P. 1969. The strategy of ecosystem development. Science **164**: 262-270.

Oswalt, S.N. 2003. Evaluation and description of a floodplain system: the Middle Fork Forked Deer River, West Tennessee. M.S. Thesis. University of Tennessee, Knoxville, TN.

Oswalt, S.N. and S.L. King. *In Press*. Channelization and floodplain forests: impacts of accelerated sedimentation and valley plug formation on floodplain forests of the Middle Fork Forked Deer River, Tennessee, USA. Forest Ecology and Management.

Outcalt, Kenneth W., ed. 2002. Proceedings of the eleventh biennial southern silvicultural research conference.Gen. Tech. Rep. SRS–48. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. Page 55-58.

Pickett, S.T.A., P.S. White. 1985. Natural disturbance and patch dynamics: an introduction. Pages 3-13 in S.T.A. Pickett and P.S. White, eds. The ecology of natural disturbance and patch dynamics. Academic Press, Orlando, FL.

Platt, W.J. and J. Connell. 2003. Natural disturbances and directional replacement of species. Ecological Monographs **73**(4):507-522.

Pringle, C. 2000. Threats to U.S. public lands from cumulative hydrologic alterations outside of their boundaries. Ecological Applications **10**:971-989.

Reily, P., and C. Johnson. 1982. The effects of altered hydrologic regime on tree growth along the Missouri River in North Dakota. Canadian journal of Botany **60**:2410-2423.

Risotto, S., and R. Turner. 1985. Annual fluctuation in abundance of the commercial fisheries of the Mississippi River and tributaries. North American Journal of Fisheries Management **5**:557-574.

Robbins, C.H. and A. Simon. 1983. Man-induced channel adjustment in Tennessee streams. USGS Water Resources Investigations Report 82-4098.

Robertson, P.A., G.T. Weaver, and J.A. Cavanaugh. 1978. Vegetation and tree species patterns near the northern terminus of the southern floodplain forest. Ecological Monographs **48**:249-267.

Robertson, A.L., P. Bacon, and G. Heagney. 2001. The responses of floodplain primary production to flood frequency and timing. Journal of Applied Ecology **38**:126-136.

Rossi, R.E., D.J. Mulla, A.G. Journel, E.H. Franz. 1992. Geostatistical tools for modeling and interpreting ecological spatial dependence. Ecological Monographs **62**:277-314.

SAS Institute Inc. 2004. SAS Version 9.1. Cary, NC, USA.

Saucier, R.T. 1994. Geomorphology and quaternary geologic history of the Lower Mississippi Valley Volume1. US Army Corps of Engineers. Vicksburg, MS.

Schumm, S.A. 1977. The Fluvial System. John Wiley & Sons Inc., New York.

Schumm, S.A., M.D. Harvey, and C.C. Watson. 1984. Incised Channels: Morphology, dynamics, and control. Water Resources Publications, Littleton, Colorado.

Schweitzer, J. 2000a. Forest statistics for Tennessee, 1999. Resource Bulletin SRS-49. Asheville, NC: USDA Forest Service, Southern Research Station.

Schweitzer, J. 2000b. Forest statistics for Tennessee, 1999. Resource Bulletin SRS-51. Asheville, NC: USDA Forest Service, Southern Research Station.

Schweitzer, J. 2000c. Forest statistics for Tennessee, 1999. Resource Bulletin SRS-52. Asheville, NC: USDA Forest Service, Southern Research Station.

Scott, M.L., P.B. Shafroth, and G.T. Auble. 1999. Responses of riparian cottonwoods to alluvial water table declines. Environmental Management **23**(3):347-358.

Scott, M.L., G.C. Lines, and G.T. Auble. 2000. Channel incision and patterns of cottonwood stress and mortality along the Mojave River, California. Journal of Arid Environments **44**:399-414.

Shankman, D. 1993. Channel migration and vegetation patterns in the Southeastern Coastal Plain. Conservation Biology **7**(1):176-183.

Shankman, D. 1996. Stream channelization and changing vegetation patterns in the U.S. coastal plain." Geographical Review **86**(2): 216-232.

Shankman, D. and T.B. Pugh. 1992. Discharge response to channelization for a Coastal Plain stream. Wetlands **12**(3):157-162.

Shankman, D. and S.A. Samson. 1991. Channelization effects on Obion River flooding, western Tennessee. Water Resources Bulleting **27**(2):247-254.

Sharitz, R. and W. Mitsch (1993). Southern floodplain forests. Biodiversity of Southeastern United States/Lowland Terrestrial Communities. W. Martin, S. Boyce and A. Echternacht, John Wiley & Sons Inc.: 311-371.

Shelford, V.E. 1954. Some lower Mississippi valley biotic communities: their age and elevation. Ecology **35**(2):126-142.

Simon, A. 1989. The discharge of sediment in channelized alluvial streams. Water Resource Bulletin **25**:1177-1188.

Simon, A. 1994. Gradation processes and channel evolution in modified West Tennessee streams: process, response, and form. USGS Professional Paper 1470. USA, Washington, D.C.

Simon, A. and C.R. Hupp. 1987. Geomorphic and vegetative recovery processes along modified Tennessee streams: an interdisciplinary approach to disturbed fluvial systems. Pages 251-262 in Forest Hydrology and Watershed Management (Proceedings of the Vancouver Symposium, August 1987. International Association of Scientific Hydrology.

Simon, A. and C.R. Hupp. 1992. Geomorphic and vegetative recovery processes along modified stream channels of West Tennessee. U.S. Geological Survey, Nashville, TN.

Simon, A. and M. Rinaldi. 2000. Channel instability in the loess area of the Midwestern United States. Journal of the American Water Resources Association **36**(1):133-150.

Simon, A. and C.H. Robbins. 1987. Man-induced gradient adjustment of the South Fork Forked Deer River, West Tennessee. Environmental Geology and Water Sciences **9**(2):108-118.

Smith, R.D. 1996. Composition, structure, and distribution of woody vegetation on the Cache River floodplain, Arkansas. Wetlands **16**:264-278.

Soil Conservation Service. 1977. Land treatment plan for erosion control and water quality improvement in the Obion-forked Deer Basin River. USDA, SCS, Nashville, TN.

Sokal, R.R. and F.J. Rohlf. 1995. Biometry: The principles and practice of statistics in biological research. 3rd Edition. W.H. Freeman and Company, New York.

Sparks, R.E. and A. Spink. 1998. Disturbance, succession, and ecosystem processes in rivers and estuaries:effects of extreme hydrologic events. Regulated Rivers:Research & Management **14**:155-159.

Speer, P.R., W.J. Perry, J.A. McCabe, O.G. Lara, and others. 1965. Low-flow characteristics of streams in the Mississippi embayment in Tennessee, Kentucky, and Illinois, with a section of quality of water, by H.G. Jeffery: USGS Professional Paper 448-H.

Stanturf, J.A. and S.H. Schoenholtz. 1998. Soils and landforms. Pages 123-148 in M.G. Messina and W.H. Conner, editors. Southern Forested Wetlands: ecology and management. Lewis Publishers, Chelsea, MI.

Steiger, J., A.M. Gurnell, and J.M. Goodson. 2003. Quantifying and characterizing contemporary riparian sedimentation. River Research and Applications **19**:335-352.

Streng, D., J. Glitzenstein, and P. Harcombe. 1989. Woody seedling dynamics in an east Texas floodplain forest. Ecological Monographs **59**:177-204.

Thein, S.S. 1979. A flow diagram for teaching texture-by-feel analysis. Journal of Agronomic Education **40**:54-55.

Theriot, R.F. 1993. Flood tolerance of plant species in bottomland forests of the southeastern United States. U.S. Army Corps of Engineers Technical Report WRP-DE-6. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Trimble, S.W. and W.P. Carey. 1984. Sediment characteristics of Tennessee streams and reservoirs. U.S. Geological Survey, Open File Report, p. 84-749.

Tucci, P. and G.E. Hileman. 1992. Potential effects of dredging the South Fork Obion River on ground-water levels near Sidonia, Weakley County, Tennessee. U.S. Geological Survey, Water-Resources Investigations Report 90-4041.

Turner, R.E., S.W. Forsythe, and N.J. Craig. 1981. Bottomland hardwood forest land resources of the southeastern U.S. Pages 13-43 in J.R. Clark and J. Benforado, eds. Wetlands of Bottomland Hardwood Forests: Proceedings of a workshop on bottomland hardwood forest wetlands of the southeastern U.S. Elsevier Science Publishing Company, New York, NY.

U.S. Department of Agriculture Soil Conservation Service. 1970. Hatchie River basin survey report, Tennessee and Mississippi: U.S. Department of Agriculuture Soil Conservation Service.

U.S. Department of Agriculture Soil Conservation Service. 1986. Sediment transport analysis report, Hatchie River Basin special study, Tennessee and Mississippi: U.S. Department of Agriculture Soil Conservation Service, p. 17.

U.S. Fish and Wildlife Service. 2001. Hatchie National Wildlife Refuge. Brownsville, Tennessee.

U.S. Forest Service. 1974. Seeds of woody plants in the United States. Agriculture Handbook No. 450. Forest Service, U.S. Department of Agriculture, Washington, D.C.

U.S. Geological Survey. 2003. Real-time data for Tennessee. http://waterdata.usgs.gov/tn/nwis/uv?site_no07029500.

van der Valk, A.G., S.D. Swanson, and R.F. Nuss. 1983. The response of plant species to burial in three types of Alaskan wetlands. Can. J. Bot. **61**:1150-1164.

Walling, D.E. and Q. He. 1998. The spatial variability of overbank sedimentation on river floodplains. Geomorphology **24**:209-223.

Wells, A.R. 2003. Integrating geographic information systems and remote sensing with spatial econometric and mixed logit models for environmental valuation. Doctoral Dissertation. University of Tennessee, Knoxville, TN.

Wharton, C.H., W.M. Kitchens, E.C. Pendleton, and T.W. Sipe. 1982. The ecology of bottomland hardwood swamps of the southeast: a community profile. Publ. No. FWS/OBS-81/37, U.S. Fish and Wildlife Service, Washington, D.C.

Wilder, T.C. 1998. A comparison of mature bottomland hardwood forests in natural and altered settings in West Tennessee. M.S. Thesis. Tennessee Technological University, Cookeville, TN.

Wilen, B., and W. Frayer. 1990. Status and trends of U.S. wetlands and deepwater habitats. Forest Ecology and Management **33/34**:181-192.

Wolfe, W.J. and T.H. Diehl. 1993. Recent sedimentation and surface-water flow patterns on the flood plain of the North Fork Forked Deer River, Dyer County, Tennessee. USGS, in cooperation with TWRA, Nashville, TN. Water Resources Investigations Report 92-4082.

Wright, Jonathan W. 1965. Green ash *(Fraxinus pennsylvanica* Marsh.). In Silvics of forest trees of the United States. H. A. Fowells, comp. p. 185-190. U.S. Department of Agriculture, Agriculture Handbook 271. Washington, D.C.

Vita

Aaron R. Pierce was born in 1976 and grew up in Oneonta, New York. Aaron graduated from Oneonta High School in June 1994, and began his undergraduate work in biology at Hartwick College, Oneonta, N.Y., in August 1994. He received his Bachelor degree in biology in May 1998. Following graduation, he worked as the biology intern in the Department of Biology at Hartwick College for one year before starting his graduate work at Purdue University. Aaron completed his master's degree in ecology at Purdue University in 2001 before starting his doctoral work at the University of Tennessee, Knoxville. He completed his Ph.D. in natural resources through the Department of Forestry, Wildlife, and Fisheries in the summer of 2005.

4535 0545 17
11/09/05 MAB