MACROPHYTE RE-ESTABLISHMENT AND DEDUCTIVE GIS MODELING TO IDENTIFY PLANTING LOCATIONS FOR FISH HABITAT ENHANCEMENT PROJECTS

By

Jonathan Paul Fleming

A Thesis
Submitted to the Faculty of
Mississippi State University
in Partial Fulfillment of the Requirements
for the Degree of Master of Science
in Wildlife and Fisheries Science
in the Department of Wildlife, Fisheries, and Aquaculture

Mississippi State, Mississippi

May 2010
MACROPHYTE RE-ESTABLISHMENT AND DEDUCTIVE GIS MODELING TO IDENTIFY PLANTING LOCATIONS FOR FISH HABITAT ENHANCEMENT PROJECTS

By

Jonathan Paul Fleming

Approved:

___________________________________  ____________________________________
Eric D. Dibble  John D. Madsen
Professor of Wildlife, Fisheries and  Associate Extension/Research Professor  
Aquaculture  of Plant and Soil Sciences  
(Co-director of Thesis)  (Co-director of Thesis)

___________________________________  ____________________________________
J. Wesley Neal  Bruce D. Leopold
Assistant Extension Professor of  Professor, Head, and Graduate  
Wildlife, Fisheries and Aquaculture  Coordinator of Wildlife, Fisheries and  
(Committee Member)  Aquaculture

___________________________________
George M. Hopper
Dean of the College of Forest Resources
Aquatic macrophytes are important components in structuring aquatic communities because they provide physical and biological functions that contribute to the stability of the ecosystem. Macrophytes provide the basis for the aquatic food-web and also provide habitat and refugia for aquatic fauna. In systems that lack macrophytes, anthropogenic re-establishment may be a feasible management approach to improve aquatic ecosystems. Understanding environmental factors that regulate colonization, dispersal, and persistence of aquatic macrophytes is pertinent to re-establishment efforts. The purpose of this study is to test hypotheses regarding success of macrophyte re-establishment efforts in Little Bear Creek Reservoir, Alabama using different macrophyte species, water depths, plant patch size and protection against herbivores at planting sites. In addition, a deductive GIS model is used to predict suitable areas to focus re-establishment efforts. Knowledge generated from hypothesis testing and application of GIS modeling provides novel information and tools for managing aquatic ecosystems.
DEDICATION

I dedicate this to my parents, Gary Don and Cindy Fleming. Their unwavering support means more than I could ever adequately express.

"There is no doubt that it is around the family and the home that all the greatest virtues, the most dominating virtues of human society, are created, strengthened and maintained."

--Winston Churchill
ACKNOWLEDGEMENTS

To sufficiently thank everyone that has helped me get to this point I would have to include another chapter and it still probably would not be enough. Funding was provided by the Alabama Department of Conservation and Natural Resources and the Little Bear Creek Millennium Group for which I am greatly appreciative. I would like to thank the Bear Creek Development Authority and Alabama Wildlife and Freshwater Fisheries personnel for assistance and cooperation. Many thanks are owed to the Department of Wildlife, Fisheries, and Aquaculture, the Forest and Wildlife Research Center, and the Geosystems Research Institute at Mississippi State University for the use of facilities and resources. Propagation and field assistance was provided at various times by Joshua Cheshier, Matthew Spickard, Thad Huenemann, Tabitha Kelly, Bonnie Earlywine, B.J. Horton, Amanda Fernandez, Matt Gower, Elizabeth Gower, Jimmy Peebles, and Austin Sharp. Rafael Gonzalez and Ryan Wersal assisted with statistical analysis and had a great deal of patience. My former colleagues at ESRI, Inc., Elizabeth Flannary and Eric Whitton, were always helpful with technical advice and fresh ideas for the GIS work. Dr. Greg Gaston at the University of North Alabama never failed me with his encouragement, wisdom, and guidance. I am indebted to him for his inspiration. I would like to thank Dr. Bruce Leopold for giving me opportunities and responsibilities that have made my graduate education unique and fulfilling. Dr. Wes Neal provided new
perspectives as a committee member and always had an open door. I certainly cannot thank my co-advisors enough for their help and support. Dr. John Madsen initially provided me with this opportunity and Dr. Eric Dibble agreed to serve as a co-advisor and mentor. When I started I was inexperienced, ignorant, and naïve and they both took a chance on me and helped me grow. I have a great deal of respect for them and both have helped to shape who I am professionally. My parents, Gary Don and Cindy Fleming, are my foundation. My love of nature and the outdoors results from the experiences they provided to me from the time I could wear a life jacket. Both of them served, at various times during my research, the role of a field technician and I could not have found anyone better if I had tried. They were always there to help, always supportive, and never complacent.

To every one of these people and the ones that I undoubtedly left out I give my greatest thanks.
TABLE OF CONTENTS

DEDICATION .................................................................................................................... ii

ACKNOWLEDGEMENTS ............................................................................................... iii

LIST OF TABLES ............................................................................................................ vii

LIST OF FIGURES ......................................................................................................... ix

CHAPTER

I. INTRODUCTION ................................................................................................... 1
   Objectives .............................................................................................................. 10
   Study Area Description ...................................................................................... 10
   Scope and Limitations ......................................................................................... 12
   Summary and Plan of Presentation .................................................................... 13
   BIBLIOGRAPHY .................................................................................................. 14

II. MACROPHYTE RE-ESTABLISHMENT IN LITTLE BEAR CREEK RESERVOIR ....... 20
   Introduction ............................................................................................................ 20
   Background on Aquatic Plant Restoration ......................................................... 23
   Objectives .............................................................................................................. 27
   Study Area Description ...................................................................................... 28
   Materials and Methods ....................................................................................... 29
       Statistical Analysis .......................................................................................... 31
   Results .................................................................................................................... 33
       2008 .................................................................................................................. 33
           Species Comparisons .................................................................................. 33
           Depth Comparisons .................................................................................. 34
               *P. nodosus* .......................................................................................... 34
               *V. americana* .................................................................................. 34
       2009 .................................................................................................................. 34
           Species Comparisons .................................................................................. 34
           Depth Comparisons .................................................................................. 35
III. DEDUCTIVE GIS MODEL FOR SUITABLE RE-VEGETATION SITE SELECTION

Introduction............................................................................................................57
Objectives ..............................................................................................................63
Methods..................................................................................................................63
  Parameter Selection, data acquisition, and calculations .........................63
    Light .......................................................................................................64
    Bathymetry .............................................................................................66
    Slope ......................................................................................................69
    Fetch .......................................................................................................71
    Sediment Characteristics ........................................................................72
    Texture ..............................................................................................73
    Organic Matter Content ..................................................................76
    GIS Model Development .......................................................................78
Results and Discussion ..........................................................................................79
BIBLIOGRAPHY ..................................................................................................83

IV. CONCLUSIONS, RECOMMENDATIONS, AND SYNERGY .....................107

BIBLIOGRAPHY ................................................................................................112
LIST OF TABLES

TABLE

2.1 Statistical comparisons of the survival of three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008. Bold Type III Test results represent significant differences in survival among the three species on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date..........................................................................................51

2.2 Statistical comparisons of the presence/absence survival survey results for *P. nodosus* planted in 2008 at three different depths in Little Bear Creek Reservoir, Alabama. Bold Type III Test results represent significant differences in survival among the three depths on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date...........................................52

2.3 Statistical comparisons of the presence/absence survival survey results for *V. americana* planted in 2008 at three different depths in Little Bear Creek Reservoir, Alabama. Bold Type III Test results represent significant differences in survival among the three depths on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date.........................53

2.4 Statistical comparisons of the 2009 survival of three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008. Bold Type III Test results represent significant differences in survival among the three species on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date. ......................................................54

2.5 Statistical comparisons of the 2009 presence/absence survival survey results for *P. nodosus* planted in 2008 at three different depths in Little Bear Creek Reservoir, Alabama. Bold Type III Test results represent significant differences in survival among the
three depths on each date and asterisks (*) indicate significant
differences for each pairwise comparison on each date......................55

2.6 Statistical comparisons of the 2009 presence/absence survival survey
results for *V. americana* planted in 2008 at three different
depths in Little Bear Creek Reservoir, Alabama. Bold Type III
Test results represent significant differences in survival among
the three depths on each date and asterisks (*) indicate
significant differences for each pairwise comparison on each
date ...........................................................................................................56

3.1 Mean light extinction coefficients ($K_d$), maximum colonization depth
($Z_c$), and secchi disk depth ($Z_{sd}$) for each sampling date in
Little Bear Creek Reservoir, Alabama in 2008 and 2009. $K_d$
and $Z_c$ were calculated using equations 3.2 and 3.4,
respectively. .............................................................................................88

3.2 Results of macrophyte growth GIS suitability model for Little Bear
Creek Reservoir, Alabama from data collected in 2008. .........................106
LIST OF FIGURES

FIGURE

2.1  Macrophyte re-establishment areas in Little Bear Creek Reservoir, Alabama from 2008-2009. ...............................................................46

2.2  Water levels in Little Bear Creek Reservoir, Alabama for 2007-2008. Water levels were dropped rapidly in mid-April 2008 and did not reach full pool in 2007 or 2008.................................................................47

2.3  Diagram of macrophyte protective exclosure sites stratified by depth for re-establishment efforts in Little Bear Creek Reservoir, Alabama in 2008. The exclosures were assessed for survival in 2008 and 2009.................................................................48

2.4  Diagram of the experimental design for exclosures planted with Potamogeton nodosus in Little Bear Creek Reservoir, Alabama in 2009. Half of the exclosures should have been removed to assess whether the patch size had an effect on survival without herbivore protection. However, water levels were dropped in 2009 and this experiment was not completed..............................................49

2.5  Water levels in Little Bear Creek Reservoir, Alabama for 2008-2009. Water levels did not reach full pool in 2008 or 2009 and were dropped rapidly in April 2008 and mid-September 2009.......................50

2.6  Percentage of exclosures containing living propagules of one of the three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008 for each surveying date..............................51

2.7  Percentage of exclosures containing living propagules of P. nodosus planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each surveying date.................................52

2.8  Percentage of exclosures containing living propagules of V. americana planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each surveying date.........................53
2.9 Percentage of exclosures in 2009 containing living propagules of one of the three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008 for each surveying date. ..........................54

2.10 Percentage of exclosures containing living propagules of *P. nodosus* planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each 2009 surveying date. ..........................55

2.11 Percentage of exclosures containing living propagules of *V. americana* planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each 2009 surveying date. .................56

3.1 This map of Little Bear Creek Reservoir, Alabama contains a grid of points spaced 100m-1 apart. This grid was the basis for the original depth survey.................................................................89

3.2 This map of Little Bear Creek Reservoir, Alabama is an interpolated raster surface of elevation above mean sea level. The raster surface was derived using the *Topo to Raster* geoprocessing tool with ArcGIS Spatial Analyst..................................................90

3.3 Areas in Little Bear Creek Reservoir, Alabama with light penetration suitable for rooted macrophyte growth estimated from data collected in 2008 and 2009. Suitable depths were determined using equation 3.4. The raster surface was derived using geoprocessing tools in ArcGIS..................................................91

3.4 Percentage slope in Little Bear Creek Reservoir, Alabama derived from bathymetry data collected in 2008. Slope was calculated using geoprocessing tools in ArcGIS...............................................................................92

3.5 Areas with suitable slope standardized to a value between 0 and 1 in Little Bear Creek Reservoir, Alabama. Suitable slope was determined using the lesser of a two class geometrical interval classification applied to the slope surface (Figure 3.4). Suitable slope values were standardized using equation 3.5. The raster surface was derived using geoprocessing tools in ArcGIS.................................................................93

3.6 Effective fetch distances standardized to a value between 0 and 1 in Little Bear Creek Reservoir, Alabama. Effective fetch was calculated using a USGS model based on equation 3.6. Values were standardized using equation 3.5 using the effective fetch

x
distance value. The raster surface was derived using geoprocessing tools in ArcGIS. ...........................................................94

3.7 Sediment sampling locations in the littoral zone of Little Bear Creek Reservoir, Alabama in 2008. ...............................................................95

3.8 Areas with suitable sediment texture (> 50% Silt + Clay) for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008...............................................................96

3.9 Areas with suitable percentage of organic matter content (< 5%) for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008. ...............................................................97

3.10 Final index of suitable areas for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008 calculated using equation 3.7.................................................................98

3.11 Final binary map of suitable areas for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008. .................................................................99

3.12 Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008. The model was created using ArcGIS Model Builder. Magnified colored sections are presented in figures 3.12a-e. ................................................100

3.12a Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the bathymetry surface...............................................101

3.12b Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the slope and organic matter surfaces and weighted linear combination.........................................................102

3.12c Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the silt, sand, and clay surfaces. .................................................103

3.12d Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the normalized sediment particle size surface and suitable sediment particle size surface.................................................104
3.12e Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of suitable and standardized fetch surfaces and the combination of the depth constraint with weighted linear combination results to create a final index of macrophyte suitability surface.
CHAPTER I
INTRODUCTION

Aquatic macrophytes are important components in structuring aquatic communities (Savino and Stein 1982, Carpenter and Lodge 1986). Aquatic macrophytes serve a number of physical and biological functions that are pertinent to the stability of aquatic ecosystems. At the most basic level, macrophytes are primary producers that form the basis of the aquatic food web (Carpenter and Lodge 1986). Not only is macrophyte production important to aquatic fauna, it also leads to the development of other primary producers (e.g., epiphytes and periphytes; Campeau et al. 1994, Kornijow et al. 1995, de Szalay and Resh 2000) which may be the most important contributor to the productivity of a system (Cattaneo and Kalff 1980, Cattaneo et al. 1998). Both levels of production have a bottom-up cascading influence on productivity of other trophic levels such as invertebrate epiplanktivores, planktivores and zooplanktivores, and vertebrate fish, reptiles and birds (Lodge et al. 1998). Native macrophyte assemblages also provide scale dependent habitat heterogeneity and complexity in the three-dimensional aquatic environment (Dibble et al. 2006) which leads to greater richness and density of aquatic animal taxa (Killgore et al. 1989, Diehl 1992, Cheruvelil et al. 2002, Cottenie and De Meester 2004, Thomaz et al. 2008). Likewise, increased complexity within habitats is correlated with greater diversity and generally supports a greater number of invertebrates
(Crowder and Cooper 1982). It is also postulated that areas of high heterogeneity and complexity stabilize predator/prey interactions (Diehl 1992) and also may provide greater ecosystem resilience especially from disturbances (Dodds 2009) such as cultural eutrophication that might otherwise have significant negative impacts (Barko and James 1998, Sondergaard et al. 2007).

Macrophytes influence the aquatic community through a number of processes that cross multiple scales from small (individual zooplankton) to large (lake ecosystems). They are important in the productivity of an aquatic system through their role as primary producers and the architecture they create (Cattaneo and Kalff 1980, Carpenter and Lodge 1986). Macrophytes facilitate the colonization of other photosynthetic organisms such as epiphyton and periphyton by providing suitable substrate (Cattaneo et al. 1998, de Szalay and Resh 2000). Macrophytes and their epiphytic colonizers serve as a food source for aquatic macroinvertebrates as well as phytophillic zooplankton (Kornijow et al. 1995, Taniguchi et al. 2003). In addition, macrophytes provide refugia for invertebrates to escape predation (Stansfield et al. 1997, Burks et al. 2001, Burks et al. 2002, Warfe and Barmuta 2004). Zooplankton studies in macrophyte beds have been documented with a great deal of work dedicated to understanding the influence macrophytes have on these organisms. Studies have indicated that plants are often vital to the life cycles of macroinvertebrates and zooplankton (Gerking 1957), influencing their diel horizontal and vertical migrations (Burks et al. 2000, Burks et al. 2001, Burks et al. 2002, Marklund et al. 2001). Macroinvertebrate community composition, distribution, and abundance in benthic sediment are significantly greater in vegetated areas (Sagova-
Mareckova and Kvet 2002). Overall, macrophytes are strongly correlated with invertebrate density and diversity which may be important in structuring higher trophic levels (Gilinsky 1984, Schramm et al. 1987, Schramm and Jerka 1989, Waters and San Giovanni 2002).

Not only are macrophytes important as a food source and as refugia for invertebrates, they also play a significant role in the life cycles of some fish species (Dibble et al. 1996). Macrophytes in the littoral zone serve as a prime area for fish spawning due to increased cover and reduced predation (Carpenter and Lodge 1986). Young of the year centrarchids use macrophytes as refugia, especially in the presence of predator species (Savino and Stein 1982, Werner et al. 1983, Werner and Hall 1988, Savino and Stein 1989a, Savino and Stein 1989b). Macrophytes also have the potential to alter foraging strategies and prey consumption (Dibble and Harrel 1997, Theel and Dibble 2008). Age-0 largemouth bass *Micropterus salmoides* often use aquatic plants as a refuge from predators and as a foraging area until they reach the size in which they shift their feeding habits to piscivory (Olsen et al. 1995). Largemouth bass are ecologically important piscivorous species and also are a popular sportfish (Ludsin and DeVries 1997). Therefore, the benefits provided by macrophytes to this species for local economies cannot be overstated.

Largemouth bass do not represent the only species benefiting from the refugia and foraging areas provided by macrophytes. Juvenile bluegills start their life in or near the littoral zone and then move to the energetically rich pelagic zone to forage (Spotte 2007). After this time they return to vegetated areas where they seek refuge from predators until
their size permits a return to the pelagic zone for more advantageous foraging (Persson and Crowder 1998). Both juvenile largemouth bass and bluegill feed on small invertebrates. Competition between these two species as juveniles contributes significantly to the ultimate success of cohorts. Once largemouth bass reach a particular size, they shift from foraging primarily on invertebrates to piscivory and bluegills become a major component of their diet. Thus the interaction between largemouth bass and bluegills change from direct competition to predation (Olsen et al. 1995). In the absence of plants, bluegill may still inhabit the shallow littoral zone, even if the pelagic zone is more energetically profitable (Werner et al. 1983, Werner and Hall 1988, DeVries 1990). Past behavioral studies indicate that when variability in energetic returns and predation exists among different habitats, fish are expected to choose areas with least mortality to growth rate ratio (Turner and Mittelbach 1990, Persson and Crowder 1998).

Macrophytes also are important to invertebrates, fish, and humans by stabilizing the physical and chemical quality of the aquatic environment. Most macrophytes are rooted in the sediment (except for free-floating species such as *Ceratophyllum demersum, Lemna minor, Spirodela polyrhiza, Eichhornia crassipes, Pistia stratiotes, Utricularia* spp. and others). The root systems and other belowground structures of macrophytes stabilize hydric sediments. Macrophytes also slow water velocity leading to increased sedimentation from the water column to the benthos and the prevention or lessening of sediment re-suspension (Barko and James 1998, Sand-Jensen 1998, Madsen et al. 2001). This has numerous effects on the aquatic environment. First and foremost, suspended sediments result in greater turbidity and therefore less water clarity and light availability.
which may alter foraging ability in visual predator species. An increase in suspended sediment and turbidity is often caused by surface runoff at the aquatic-terrestrial interface. In some areas, this runoff may contain high nutrient loads that increase phytoplankton production and may lead to further reductions in water clarity. Plants mitigate these events physically through interactions with water movement and suspended sediment loads (Madsen et al. 2001). Once the nutrients enter the water column they may be taken up by macrophytes or precipitate to the benthos (Barko and James 1998). In some cases, nutrients such as phosphorus can be bound to iron or aluminum oxides and hydroxides in sediments oxygenated by radial oxygen loss from macrophyte roots (Jaynes and Carpenter 1986) given that the pH of the surrounding water is not too high. Under anoxic sediment conditions, phosphorous is released and can be used by phytoplankton leading to potentially harmful algal blooms (Barko and James 1998).

Macrophytes also alter other chemical factors in the water. Plants not only oxygenate benthic sediment, they also can alter oxygen levels in the water column (Sand-Jensen et al. 1982, Honnell et al. 1993). Additionally, aquatic plants that photosynthesize during the day can remove a significant amount of inorganic carbon from the water column and release a significant amount at night or in periods of low light during respiration. This alters the pH of the system in a cycle that increases from a relatively lower pH in the morning, after respiration, to a higher pH by the end of the day, after photosynthesis (Barko and James 1998).
Shallow lakes often demonstrate equilibrium between two states. The alternative stable state hypothesis indicates that shallow lakes exhibit one of two conditions: a clear water state dominated by macrophytes or a turbid, eutrophic state dominated by phytoplankton (Scheffer 1998). The proximate mechanism driving this change is light availability. Competition for light resources may decrease the macrophytes ability to photosynthesize. In systems dominated by phytoplankton, light attenuation may increase to the point that macrophyte growth cannot occur (Jones and Sayer 2003). Alternatively, in clear water states macrophytes can thrive and positive feedbacks contribute to the overall stability of the system. The mechanism for shifts between these states may have different origins and can be caused by high nutrient loading (Sondergaard and Moss 1998) or changes in fish community structure and a loss of top-down phytoplankton controls (Bronmark and Weisner 1992). Neither of these mechanisms directly explains why increased clarity and macrophyte growth constitute the alternative stable state. The answer to this question lies in whether macrophytes themselves limit phytoplankton biomass increases through shading, nutrient reduction, allelopathy or planktivore refugia, or if macrophytes themselves are only present under conditions of high water clarity and serve as a positive feedback for maintaining that clarity (Scheffer 1998, Scheffer 1999). Regardless, the interaction between macrophytes and phytoplankton is significant in determining the overall productivity of an aquatic ecosystem and in structuring the communities of shallow lakes. Aquatic systems lacking littoral vegetation may result in reduced diversity and reduced production of desirable fish species (Savino and Stein 1989a). Declining production and recruitment of fish could lead to cascading trophic
effects resulting in a decrease in overall ecosystem productivity (Carpenter and Lodge 1986).

Due to the biogeographical and hydrological limitations of some aquatic ecosystems (e.g., artificial reservoirs), natural establishment of plant communities may be restricted (Doyle and Smart 1993, Smart et al. 1996, Smart et al. 2005). In areas that lack native aquatic plant communities or in areas that have recently experienced a form of disturbance that resulted in damage to the native community, re-vegetation may be a potential option for restoring the aquatic community (Smiley and Dibble 2006). Re-establishment of aquatic plant communities is an ecosystem approach to aquatic plant and fisheries management that considers the importance of aquatic macrophytes in structuring the system as well as the ability of plants to be a self-renewing habitat (Madsen 2000). This seems like an obvious approach given the importance of aquatic plants as the basis of the aquatic food web and their structural importance in the aquatic environment. However, this method has only recently been tested and work is still underway to assess its potential for widespread use and effectiveness (Smart et al. 1998, Smart et al. 2005, Smiley and Dibble 2006). Smiley and Dibble (2006) found that it was feasible to attempt establishment of some species although they are the only authors that have presented statistical results to advance the science of aquatic plant re-vegetation.

A literature search on macrophyte restoration will reveal that much of the knowledge on restoring aquatic communities regarding submerged aquatic macrophytes has been advanced through studying manipulations of higher trophic levels to control phytoplankton biomass, thus increasing water clarity and encouraging macrophyte
growth (Jeppesen et al. 1990, Perrow et al. 1999, Sondergaard et al. 2007).  This approach relies heavily on the theory of trophic cascades as well as the alternative stable states hypothesis and ultimately depends on the system either having a viable seed bank or being within a suitable distance from a source population of suitable macrophytes.  In some cases this is simply not available and anthropogenic propagation is the best method for re-establishment.  For example, in reservoirs created for flood control, plant establishment may be limited by distance from source populations, age of the reservoir, or both.  In this case, true restoration to a previous state cannot be achieved because the system is technically a human created one.  These systems often have other uses such as recreational fishing and aquatic plants may provide a much needed component in the effective management of the fishery (Smart et al. 1996).

The approach of macrophyte re-establishment was first taken in the early 1990’s in Guntersville Lake Reservoir, Alabama (Doyle and Smart 1995).  Guntersville Lake was infested with nuisance invasive aquatic plant species and the goals were to encourage the growth of native species in hopes of inducing competition for resources and ultimately re-establishing a native plant community.  Trials in Guntersville Lake attempted to demonstrate the interaction between native and exotic species but were eventually unsuccessful in the establishment attempts.  From this foundation, the approach of aquatic plant re-establishment developed into a method for creating fish habitat in areas lacking native plant communities and potentially discouraging exotic invasive species from colonization (Smart et al. 1996, Smart et al. 1998).  The question for managers wishing to apply this technique then becomes one of location (e.g., Where
should plant re-establishment efforts focus to achieve maximum likelihood of success?). The first step in answering the question is to select species that are desirable for re-establishment and plant in areas where conditions are conducive to aquatic macrophyte growth. However, this requires knowledge of what factors affect colonization, growth, and persistence of aquatic plant assemblages (Best et al. 2008) and knowledge of the local aquatic system.

Recognizing the relevant importance of native aquatic macrophytes to invertebrates and fish and the interactions within this community, it is important that managers understand the potential impact they may have on a particular system, especially to justify ecosystem restoration efforts. One approach for assessing impacts of plants on different trophic levels (invertebrates to fish) is to assess the potential trophic impacts of aquatic plants via behavioral shifts (Crowder and Cooper 1982, Savino and Stein 1982, Savino and Stein 1989a, Savino and Stein 1989b, Theel and Dibble 2008) or actual caloric densities (Richardson et al. 1998). Fish growth is correlated with presence of macrophytes, but only general predictions have been made and vary with plant and fish species (Crowder and Cooper 1982, Richardson et al. 1998). Likewise, macroinvertebrate colonization and community structure in macrophyte assemblages have been studied and quantified (e.g., Cyr and Downing 1988, Beckett et al. 1992, Jeffries 1993) in search of mechanisms that lead to differences among species (e.g., biomass, surface area, fractal dimension, respectively with the previously listed citations). However, predictions of the caloric value of invertebrates inhabiting vegetated areas to fish have not been assessed in field experiments (but see Richardson et al. 1998). The
following objectives which directly relate to the proceeding chapters of this thesis outline my attempt to address certain components of this problem.

**Objectives**

The first objective was to identify and successfully cultivate submersed or floating leaved aquatic macrophytes in Little Bear Creek Reservoir, Alabama. This objective consisted of two phases, the first of which was to identify species that would be feasible for the re-establishment effort. The next phase represented an extension of the first phase but with more focused efforts on the successful species. This phase also included testing hypotheses related to plant survival based on species, manipulated planting depths, and removal of herbivore protection.

The second objective was to create a macrophyte colonization suitability model using ArcGIS for Little Bear Creek Reservoir, Alabama. The purpose of this model would be to identify potential zones for focused re-establishment efforts as well as to provide insight to the maximum potential amounts and distributions of macrophytes within the reservoir. This model also can be used to predict impacts of macrophyte re-establishment on potential areal coverage, volume, plant surface area, and biomass. The model also may have a use in making quantitative predictions regarding the impacts of plant re-establishment on invertebrates and fish if appropriate data are available.

**Study Area Description**

Little Bear Creek Reservoir (LBCR) is located in Franklin County in Northwest Alabama. It is one of four lakes that are part of the Bear Creek Development Authority
(BCDA) Lakes and is within the Pickwick watershed of the Tennessee River system. LBCR was impounded in 1975 for flood control and has since become an important driving force for the local economy by providing opportunities for fishing, camping, boating, and other recreational activities. LBCR has a total surface area (at full pool) of approximately 650 ha (1600 ac) and extends 13 km (8 mi) upstream from the dam. It has a fluctuating water level of approximately 3.6m (12ft) each year with full pool occurring from mid-April until late October (TVA 2009).

The remnant flooded timber that once served as habitat for the fishery is in decline and needs to be replaced with a self-renewing habitat for forage fishes and young-of-the-year bass (Cheshier et al. 2008). The best replacement for the flooded timber would be a diverse community of native aquatic plants, such as American pondweed *Potamogeton nodosus* Poir. and water celery *Vallisneria americana* Michx. (Smart et al. 1996).

After impoundment, LBCR was reported to have contained submersed (e.g., *Potamogeton* spp.) and emergent (e.g., water-willow *Justicia americana* (L.) Vahl) aquatic plant species (Phillip Cooper and Gary Don Fleming, personal communication). However, in recent years, there has been no sign of submersed aquatic plant species, with the exception of muskgrass (*Chara* spp.) a non-vascular macro-alga. Additionally, there are local reports of a reduction in emergent plant communities. In 1999, a group of local anglers and conservationists formed the Bear Creek Millennium Group. This group has worked in conjunction with local and state officials to improve the quality of the
fishery at Little Bear Creek Lake and have shown a great interest in establishing aquatic plant communities for fish habitat.

**Scope and Limitations**

This research represents scientific inquiry and prediction that crosses multiple scales. The aim of the research is to present a framework that can potentially be used by aquatic resource managers to make better management decisions. Previous research has been conducted in greater detail at each scale presented here. However, due to the nature of the breadth of this project, only minimal *in situ* measurements have been made. Higher precision is gained with smaller scale detailed work but greater application is available at larger scales (Dodds 2009). Data used for the suitability model were collected along a 100m grid but in its entirety it covers a landscape scale across the entire reservoir. Thus, with the breadth of the project, the precision of estimates may not be applicable for all uses. Suitability is assumed to indicate likelihood of plant establishment but only under optimal conditions and after a period of time. A number of GIS models have been used to predict aquatic plant distributions and it appears that issues related to spatial scale such as size, complexity, and even latitude may be important drivers for accurate prediction in any given location. Therefore, the model presented hereafter should be regarded as a process and not a solution. For model application in other areas, careful consideration of potentially important parameters in those areas should be assessed.
Summary and Plan of Presentation

Native aquatic macrophytes serve a number of purposes that structure and stabilize the aquatic community. Invertebrates and fish are directly and indirectly impacted by macrophytes through physical or biological pathways. Invertebrates feed on epiphytes that colonize the surface of macrophytes and also use the physical structure of macrophyte patches as a refuge from predators. Likewise, juvenile centrarchids such as largemouth bass and bluegill use macrophyte patches as areas to locate their invertebrate prey and as a refuge from predators. Thus aquatic macrophytes are important mediators between predator and prey at multiple scales in aquatic ecosystems. Not only are they directly important for fish and invertebrates, they have direct impacts on water quality. Aquatic plants stabilize shorelines, prevent sediment resuspension, and alter the redox potential of the sediment which may control nutrient cycling.

Assessment of the relationship between aquatic plant presence and invertebrates and/or fish has received a good deal of attention but the prediction of plant growth and the potential impacts has not been reported. The proceeding chapters are presented as a framework for assessing a lentic aquatic system for the suitability of macrophyte growth and the potential impacts of re-vegetation on the aquatic ecosystem.


DeVries, D. R. 1990. Habitat use by bluegill in laboratory pools: Where is the refuge when macrophytes are sparse and alternative prey are present? Environmental Biology of Fishes 29: 27-34.


Madsen, J.D. 2000. Advantages and disadvantages of aquatic plant management techniques. ERDC/EL MP-00-1, U.S. Army Engineer Research and Development Center, Vicksburg, MS.


Scheffer, M. 1999. The effect of aquatic vegetation on turbidity; how important are the filter feeders? Hydrobiologia 408/409: 307-316.


CHAPTER II
MACROPHYTE RE-ESTABLISHMENT IN LITTLE BEAR CREEK RESERVOIR

Introduction

Aquatic macrophytes are important ecological components of freshwater ecosystems (Savino and Stein 1982, Carpenter and Lodge 1986). Macrophytes serve a number of purposes that mediate physical and biological processes. Biologically, aquatic macrophytes provide habitat and food for invertebrates, all ages of fish, and other aquatic organisms, forming the basis of the aquatic food chain (Carpenter and Lodge 1986). Physical and physicochemical contributions by macrophytes include decreasing turbidity levels via removal of suspended particles and nutrients from the water column, and sediment stabilization (Madsen et al. 2001, Bouldin et al. 2004). Native macrophytes may increase dissolved oxygen levels during photosynthesis and by cooling water due to shading of solar radiation via floating leaves (Honnell et al. 1993). Overall, macrophytes are important in structuring aquatic communities (Jeppesen et al. 1998) and provide essential fish habitat (Dibble et al. 1996).

Due to the biogeographical and hydrological limitations of some aquatic ecosystems, natural establishment of plant communities in man-made reservoirs may be restricted (Doyle and Smart 1993, Smart et al. 1996, Smart et al. 2005). Therefore, native plant communities may need to be cultured for colonization of a reservoir to begin (Smart
et al. 1998). In areas where aquatic macrophytes are native but not currently present, restoration of these communities is potentially a tool that could be used by ecosystem managers (Doyle and Smart 1993, Smart et al. 1996, Smiley and Dibble 2006). Methods to establish and restore native aquatic macrophytes have been developed by Smart and others (1996, 1998, and 2005). Additionally, trials on establishment of aquatic plant communities have been performed in Lake Guntersville, Alabama (Doyle and Smart 1993, Doyle and Smart 1995) as well as Oklahoma (Dick et al. 2004a), and Texas (Dick et al. 2004b), although results were mixed and statistical evaluations on the success of these trials were not reported.

Although plants are important components of aquatic ecosystems, some also may become a nuisance or degrade the system. This is especially true of exotic invasive species (e.g., *Hydrilla verticillata* (L. f.) Royle, *Myriophyllum spicatum* L., *Eichhornia crassipes* (Mart.) Solms). Some plants exhibit prolific growth when introduced into a new area or if the environment changes in a way that alters normal growth regulation (Madsen 2004). For example, an increased nutrient load into a small pond may cause an otherwise healthy aquatic plant community to become problematic due to an increase in plant growth. Canopy forming or floating macrophytes can significantly increase shading in the water column which prevents photosynthesis and oxygen release from phytoplankton, reducing dissolved oxygen concentrations. In addition, high plant densities can prevent water column mixing which reduces atmospheric oxygen diffusing into surface waters from reaching lower into the water column. With high plant densities also comes increased organic material and decomposition in the benthos which uses the
oxygen that may be available. The net result is habitat displacement of fish and other fauna that depend on dissolved oxygen for survival (Madsen et al. 1991, Madsen 2005). An increase in plant density beyond the healthy threshold can reduce the foraging ability of several fish species (Theel and Dibble 2008) which may indirectly lead to growth restrictions (Olsen et al. 1995) and stunted populations. Prolific growth also may be aesthetically unpleasing, sometimes leading to incorrect control measures and ultimately harm to the aquatic environment. Pimentel and others (2000) reported that the costs of nuisance aquatic plants totaled over $100 million in the U.S. annually in the early 1990’s. Therefore, the economic impact of these species also is an important consideration. Finally, dense, unhealthy macrophyte growths provide breeding grounds for disease vectors such as mosquitoes that may become a significant threat to human health. Although some macrophyte species can cause significant problems, most native species provide benefits that far outweigh the consequences of their absence. Some have even proposed that native plants can deter exotic invasive species from colonizing and reproducing (Doyle and Smart 1995), although little research has been done and there is currently no consensus.

Numerous research projects have been performed that focus on the interactions between macrophytes, invertebrates and fish (e.g., Theel and Dibble 2008). These studies have tested a variety of hypotheses regarding fish growth rate, density, and behavior, and invertebrate density, species richness and diversity (see Diehl and Kornijow 1998 and Persson and Crowder 1998). The regulating mechanism varies somewhat within these studies but in most cases, a significant explanatory variable is
macrophyte stem density. Studying effects of macrophyte density is important, especially in the context of invasive species which can occur in densities far greater than native species. The results of the research have almost always found that intermediate densities of aquatic plants are preferable to fish because they provide the balance needed to mediate predator/prey interactions in the littoral zone. Likewise, comparisons of native macrophytes with intermediate stem densities to exotic invasive macrophytes with dense stem densities indicate that intermediate stem densities allow more efficient fish foraging, which may result in an increase in overall fitness (Dibble et al. 1996).

Background on Aquatic Plant Restoration

Prior to the early 1990’s there had been little or no research regarding the active propagation of aquatic macrophytes as a management tool for ecosystem restoration. Regarding native aquatic macrophyte re-establishment efforts, the word restoration itself may be a misnomer. The National Research Council (1992) defined restoration as “returning a system to a close approximation of its condition prior to disturbance” and accomplishing restoration as “ensuring that ecosystem structure and function are recreated or repaired, and that natural dynamic ecosystem processes are operating effectively again.” In the case of man-made reservoirs, this definition would imply that the system should return to its pre-human altered state which is unrealistic when dealing with reservoirs constructed for purposes such as flood control. In practice, restoration may involve temporal, multi-phase goals from restoring desired species or species richness (providing community structure), to monitoring development of these
communities, and finally verifying that the desired structure and function have been linked and established (Palmer et al. 1997).

The history of macrophyte restoration is relatively short, indicating the recent emergence of this ecological application. The results of a literature search for aquatic macrophyte restoration often contain documents describing the re-growth or pioneer colonization of native aquatic plant communities as a result of increased light availability and water clarity conducive to plant growth following some environmental manipulation (e.g., Sondergaard et al. 2007). This certainly qualifies as restoration but not in the context of active propagule placement through anthropogenic means. For the purposes of this document, the use and interchange of the terms restoration, re-establishment or re-vegetation will refer to anthropogenically propagated specimens of native aquatic plants to restore native community structure and function and increase fish habitat in an aquatic ecosystem.

Several papers of interest are directly related to native aquatic plant restoration. Starting in the early 1990’s, the U.S. Army Corps of Engineers (USACE) began working on projects related to submersed aquatic plant restoration. The main focus for this work was originally to identify effective biological techniques to help control nuisance invasive species. This was based on the idea that a native plant community could potentially have a competitive advantage over exotic invaders and therefore could help to stabilize systems that were at risk from invasion. The first trial reported took place in Guntersville Reservoir, Alabama from 1991 – 1992. There were two objectives in this original research: a) evaluate methods for promoting establishment and persistence of
populations of native aquatic plant species, and b) evaluate ability of established native plant populations to resist reinvasion by nuisance species (Doyle and Smart 1993, Doyle and Smart 1995). These reports describe interactions between several native species and non-native invasive species but results were inconclusive.

A justification for native submersed aquatic plant species re-establishment can be made based on the importance they serve in structuring aquatic communities, in particular the importance of aquatic vegetation to fish. Smart and others (1996) made this case and further promoted the idea of aquatic plant re-vegetation as part of an American Fisheries Society Symposium dealing specifically with freshwater aquatic vegetation. Developing techniques for establishing native aquatic plant species is an important step in the advancement of restoration in the aquatic environment (Smart et al. 1998). This includes all aspects of restoration from species and site selection to project implementation (Smart et al. 1996). Smart and others (1998) further established the need and justification of this research and made an effort to promote techniques for establishing native aquatic plants to the aquatic plant management field. In addition to this previous work the USACE published a report in 1999 specifically addressing aquatic plant propagation for restoration projects (Smart and Dick 1999). This report was updated in 2005 to include more candidate species and updated techniques for restoration efforts (Smart et al. 2005).

Several specific examples of submerged aquatic plant re-establishment projects have been documented. As mentioned previously, Guntersville Reservoir, Alabama is the first example known (Doyle and Smart 1993, Doyle and Smart 1995). The USACE
also has performed test trials in El Dorado Lake, Kansas from 1996-1998 (Dick and Smart 2004), Arcadia Lake, Oklahoma from 1997-1998 (Dick et al. 2004a), and Cooper Lake, Texas from 1998-2000 (Dick et al. 2004b). The number of species used and sites tested was quite large and results of success were mixed. However, these examples provided an important first step in submersed aquatic plant re-establishment efforts that can be attempted in additional areas.

One of the only examples of experimental hypothesis testing regarding native submersed aquatic plant establishment was performed in Lake Charlie Capps in west-central Mississippi (Smiley and Dibble 2006). The results from this experiment indicated that it is feasible to attempt aquatic plant re-establishment in southern reservoirs, even in systems suffering from cultural eutrophication and high turbidity levels (Smiley and Dibble 2006). Their results, however, indicated that the species tested differed in percentage cover, stem density, and probability of extinction. This indicates that not all species may be successful in re-vegetation projects given a particular set of environmental conditions.

Some of the most recently published documents regarding macrophyte restoration are related to site selection and environmental factors needed for native aquatic plant re-establishment. This includes simulation models combining hydrological, geological, and ecological information to predict the success of planting efforts (i.e., Best et al. 2008, Grodowitz et al. 2009).
Objectives

The first objective of this project was to identify and successfully cultivate submersed or floating leaved aquatic macrophytes in Little Bear Creek Reservoir, Alabama. This objective consisted of two phases, the first of which was to identify species that would be feasible for the re-establishment effort. The first evaluation of species was conducted in 2007 (Cheshier et al. 2008) but discussion here is limited to 2008-2009.

The next phase (phase II: 2008) represented an extension of phase I but with more focused efforts on the successful species (*P. nodosus*, *V. americana*, and *S. pectinata*). The null hypothesis that survival would be different for the three species because species may tolerate environmental conditions in Little Bear Creek Reservoir differently was tested. Also tested was the null hypothesis that survival of each species (considered independently) would be different when initially planted at different depths because of potential environmental variations such as temperature, light availability, and wave action. In addition, trials were performed measuring the survival of specimens planted without herbivore protection.

In 2009, the first objective was to continue monitoring the survival of plants propagated in the 2008 experiment. Also attempted was testing the hypothesis that different patch sizes of *P. nodosus* would respond differently when herbivore protection was removed because a critical rate in the production of plant biomass in larger patches is greater than in smaller patches and should therefore be more resistant to extinction by herbivory (after the hypothesis proposed by Scheffer et al. 2008).
**Study Area Description**

Little Bear Creek Reservoir (LBCR) is located in Franklin County in Northwest Alabama. It is one of four reservoirs that are part of the Bear Creek Development Authority (BCDA) lakes and is within the Pickwick watershed of the Tennessee River system. LBCR was impounded in 1975 for flood control and has since become an important driving force for the local economy by providing opportunities for fishing, camping, boating, and other recreational activities. LBCR has a total surface area (at full pool) of approximately 650 ha (1600 ac) and extends 13 km (8 mi) upstream from the dam (Figure 2.1). It has a fluctuating water level of approximately 3.6m (12 ft) each year with full pool occurring from mid-April until late October (Figure 2.2; TVA 2009).

The remnant flooded timber that once served as habitat for the fishery is in decline and needs to be replaced with a self-renewing habitat for forage fishes and young-of-the-year bass (Cheshier et al. 2008). The best replacement for the flooded timber would be a diverse community of native aquatic plants, such as American pondweed (*Potamogeton nodosus* Poir.) and water celery (*Vallisneria americana* Michx.; Smart et al. 1996).

After impoundment, LBCR was reported to have contained submersed (e.g., *Potamogeton* spp.) and emergent (e.g., *Justicia americana* (L.) Vahl) aquatic plant species (Phillip Cooper and Gary Don Fleming, personal communication). However, in recent years, there has been no sign of submersed aquatic plant species, except for *Chara* spp. a non-vascular macroalgae. Additionally, there are local reports of a reduction in emergent plant communities. In 1999, a group of local anglers and conservationists...
formed the Bear Creek Millennium Group. This group has worked in conjunction with local and state officials to improve the quality of the fishery at Little Bear Creek Reservoir and have shown interest in establishing aquatic plant communities for fish habitat. The Tennessee Valley Authority (TVA) has granted the Millennium Group and Alabama Department of Conservation and Natural Resources three areas in LBCR for aquatic macrophyte re-establishment trials (Figure 2.1).

Materials and Methods

In 2008, three native submersed aquatic plant species (P. nodosus, V. americana (northern ecotype), and S. pectinata) were grown and transplanted into Little Bear Creek Reservoir and their survival was evaluated based on species, depth, and presence of herbivore protection. These species were approved for re-vegetation trials by TVA and are relatively easy to obtain. In addition, several studies involving these species have indicated their use by largemouth bass and bluegill for foraging habitat (e.g., Dibble and Harrel 1997, Richardson et al. 1998). Methods for growing and planting generally follow the recommendations of Smart and others (2005). The plants were first propagated from tubers or fragments in April 2008 and raised in a greenhouse and mesocosm at the R.R. Foil Plant Science Research Center, Mississippi State University, and then transplanted into LBCR. Potamogeton nodosus was collected from a local source in Russellville, AL. Stuckenia pectinata and V. americana were ordered from Kester’s Wild Game Food Nurseries, Inc., Omro, Wisconsin.

Stuckenia pectinata and V. americana were planted in 3-in peat pots and allowed to grow at Mississippi State University facilities for approximately six weeks. Osmocote
fertilizer (19-12-6) was used in each pot to boost production and improve plant survival during initial growth periods. Two specimens of a given species were planted in each pot. Plants were transplanted (early July) when sufficient growth had occurred and water levels in Little Bear Creek Reservoir were stable and acceptable. American pondweed (approximately 45cm stem length) was collected and directly transplanted with bare roots at the time of plantings.

Specimens were transplanted among three sites approved by the Tennessee Valley Authority (TVA) in Little Bear Creek Reservoir. Two sites are located in Trace Branch and one site in Cooper’s Branch (Figure 2.1). Plants were transplanted inside 1m diameter enclosures made of PVC coated wire mesh and re-bar. *Stuckenia pectinata* and *V. americana* were left in biodegradable peat pots and placed in a small excavated hole in the sediment. *Potamogeton nodosus* rhizomes were planted directly into the sediment. Four pots of the same species were planted inside an exclosure (each exclosure only contained one species). Approximately fifteen stems of *P. nodosus* were used as an alternative to four pots. Each planting location contained nine exclosures of each species totaling 27 (three plant species * nine exclosures per species at each site). The treatments of each species were planted along three contour intervals (0.3 meters, 0.6 meters, and 1.0 m) to assess survival and growth of each species based on depth (Figure 2.3). Three additional “patches” of each species at each location were planted without exclosures at a depth of 60 cm. Three exclosures per location in which no pots are planted were used as a reference (one exclosure at each depth). Each exclosure was evaluated individually and the data were analyzed based on species, depth, and enclosed/not enclosed. Water levels
did not reach full pool but were relatively stable throughout the study period (Figure 2.2). Species and depth data from different locations were analyzed together and location was not considered as an experimental variable in statistical tests.

Based on observations of successful species for establishment in 2007 and 2008, \textit{P. nodosus} was the only species involved in the 2009 planting trials. Exclosure size served as the variable of interest for this experiment in 2009 and removal of the exclosures was supposed to serve as the treatment. Three sized exclosures (1m$^2$, 2m$^2$, and 3m$^2$) were placed at each location. These sizes were used because they represented a gradient of patch size change, were manageable for construction, and could be deployed within the limited approved planting areas. The treatments were replicated six times at random locations in the littoral zone of approved planting sites (Figure 2.4). Water levels in Little Bear Creek Reservoir were dropped before the end of the growing season (Figure 2.5). The experimental design for this project required \textit{P. nodosus} to expand from its original number of propagules until it reached approximately 100 percent coverage inside the exclosures. Before this occurred, the water levels were dropped far below the exclosures which did not allow the completion of the experiment. Presence and visual percentage cover data were collected prior to the lowered water levels.

**Statistical Analysis**

Exclosures were initially evaluated based on two factors: 1) presence/absence of plants planted in each location (exclosure or patch) to assess plant survival, and 2) visual estimation of percent coverage inside each exclosure to assess growth. These factors were used because they represent the most basic responses of planting efforts and
indicate the most important information (i.e., which species survive and how much they increase). The analysis and results provided are only the survival of plant species, measured from presence or absence. Visual estimations of percent coverage were confounded by differences in water clarity at each depth and sampling period. In some cases, plant presence was only detectable by reaching into the exclosure to feel for the presence of plants. In addition, differences in the morphologies of the different species do not allow a comparison of percent coverage between species. The results include presence or absence from each sampling period (date) of the 2008 and 2009 season.

Presence/absence data were grouped by sampling period (date) and analyzed using a generalized linear model in SAS software v. 9.2 and SAS Enterprise Guide 4.2 (SAS Institute 2008). Presence/absence data have a binomial distribution and do not fit assumptions of normality for parametric statistical analysis. Therefore the generalized linear model analysis procedure (Proc Genmod) was set to analyze a binomial distribution using a link (logit) function. Hypothesis testing for significant differences was conducted with type III tests of fixed effects and if differences were found, pairwise comparisons to detect means separation of all treatment class levels were performed using a least square means (lsmeans) analysis. All tests were performed with a significance level of 0.05.

Percent coverage was estimated for all exclosures planted in 2009 and analyzed with SAS software (SAS Institute 2008) based on size and location using a one-way analysis of variance (ANOVA). A Bonferroni test was used to detect mean separation with a significance level of 0.05.
Results

2008

Species Comparisons

The propagules of all species planted outside of protective exclosures were absent two days after planting indicating intense herbivory. Therefore, no analysis was performed for the herbivore protection trial.

Of the three species planted, *P. nodosus* had a significantly greater survival rate each sampling period (*P* < 0.05; Table 2.1, Figure 2.6). *V. americana* survival in both Trace Branch sites was minimal. In Cooper’s Branch, *V. americana* grew and expanded on the substrate but did not appear to substantially grow toward the surface. In September, after an initial decline during August, water celery began re-growing in some exclosures. This was likely the result of cooling water temperatures. Approximately 63% of the exclosures planted with *V. americana* had surviving plants in September 2008 (Figure 2.6). This was significantly greater than *S. pectinata* (*P* < 0.001) but less than *P. nodosus* (*P* < 0.001).

Initially, 20% of the *S. pectinata* propagules survived. However, they never expanded beyond the initial propagules inside exclosures and were absent in most exclosures after only a few weeks. In September at the last sampling period, no living plants could be detected inside any exclosures (Figure 2.6). There was no survival of *S. pectinata* at the last sampling period in 2008 and therefore it is excluded from the remaining analysis results.
Depth Comparisons

*P. nodosus*

Planting depth comparisons did not indicate any significant differences in survival rate of *P. nodosus* at any of the three planting depths during any sampling period (P < 0.05; Table 2.2, Figure 2.7). This result is encouraging because several depths had 100 percent survival rate indicating there was no significant negative effect of depth on *P. nodosus* survival during 2008.

*V. americana*

Planting depth comparisons did not indicate any significant differences in survival rate of *V. americana* at any of the three planting depths during 2008 sampling periods (P < 0.05; Table 2.3, Figure 2.8).

2009

Species Comparisons

Survival from exclosures planted in 2008 was evaluated at the beginning of the 2009 growing season and just before water levels were dropped in 2009. *P. nodosus* had a significantly greater survival (P < 0.05) than other species during all sampling periods except for *V. americana* at the June 29 sampling period (P = 0.059; Table 2.4, Figure 2.9). All other pairwise comparisons differed significantly (P < 0.05). *S. pectinata* was absent during every sampling period. *V. americana* was present during every sampling period but did not expand beyond exclosures.
Depth Comparisons

*P. nodosus*

Planting depth comparisons did not indicate any significant differences in survival rate of *P. nodosus* at the May 17, June 1 or September 7 sampling period (P < 0.05; Table 2.5, Figure 2.10). However, there was a significant difference in survival between exclosures planted at 0.60 m and 1.0 m at the June 15 (P = 0.012) and June 29 (P = 0.012) sampling period. This may be attributable to a short rise in water levels that possibly inundated the exclosures planted in 1.0 m water, leaving them exposed to herbivory (Figure 2.5).

*V. americana*

Planting depth pairwise comparisons indicated a significant difference between survival rate of exclosures planted at 1.0m and 0.60m (P < 0.001) and 1.0m and 0.30m (P < 0.001) but did not indicate a difference between exclosures planted at 0.60m and 0.30m (P = 0.638) at the June 1 sampling period. All other sampling periods indicated no significant differences in survival rate of water celery (P < 0.05; Table 2.6, Figure 2.11).

Patch Size Coverage

Water levels were again a problem in 2009, which did not allow the completion of the critical patch size experiment. However, exclosure sizes 1x1, 2x2, and 3x3 had mean percent coverage of 66.9, 73.3, and 69.7, respectively, and did not differ significantly (P < 0.05). Analysis of exclosures by location indicated mean percent coverage of 23.3 (Cooper’s Branch), 87.6 (Trace Branch site 2), and 99.0 (Trace Branch site 1). The two
Trace branch sites did not differ significantly (P < 0.05) but both of these sites differed significantly from the Cooper’s Branch site.

2008 – 2009 Comparisons

Analysis of survival rate of each species at the last sampling period of both years (September 7 for both years) did not indicate significant differences from 2008 to 2009 (P < 0.05; i.e., P. nodosus (2008) did not differ from P. nodosus (2009) and V. americana (2008) did not differ from V. americana (2009)). By using data collected during the last evaluation survey, survival estimates could be made without the confounding effects of intermittent senescent periods that were observed during summer. This finding is particularly interesting to note because it indicates that exclosures containing surviving plants at the end of the growing season in 2008 successfully perennated and re-established inside the exclosures in 2009.

Discussion

Re-establishment of native aquatic plants is potentially a technique used to restore aquatic habitats in the southeastern US (Smiley and Dibble 2006). Aquatic plant community restoration efforts have been attempted in Lake Guntersville, Alabama (Doyle and Smart 1993, Doyle and Smart 1995) as well as in Oklahoma (Dick et al 2004a), Texas (Dick et al. 2004b), and Mississippi (Smiley and Dibble 2006). Establishment of aquatic plant communities can have positive effects on water quality as well as provide essential habitat and sanctuary for fish fauna (Dibble et al. 1996). Successfully cultivating submersed aquatic vegetation in Little Bear Creek was
problematic with difficulties possibly attributable to fluctuating water levels, high water temperatures, and herbivory. In 2008, water levels never reached full pool which complicated restoration efforts. In 2009 water levels were dropped early, further complicating experimental hypothesis testing.

Results from trials performed in Little Bear Creek Reservoir in 2007 indicated that *P. nodosus* was the best candidate for restoration when compared to other species tested (Cheshier et al. 2008). In 2008, *S. pectinata* and *V. americana* were planted along with *P. nodosus* to assess their survival potential for another year in case environmental conditions had changed from past trials. These species all produce some form of subterranean overwintering structures such as tubers or winter-buds. Tubers and winter-buds provide new plants with the necessary carbohydrates needed to initiate growth in subsequent growing seasons (Cronk and Fennessy 2001). They also may serve as a mechanism to aid in plant survival during adverse environmental conditions. Given the nature of water level fluctuations in LBCR (Figures 2.3 and 2.5), focused re-vegetation efforts are only feasible for species adapted to survival when environmental conditions do not meet the necessary requirements for year-round growth.

*Potamogeton nodosus* was the most successful species planted in 2008. *Stuckenia pectinata* never appeared to adjust to lake conditions and disappeared soon after planting. This result is consistent with findings by Doyle and Smart (1995) regarding *S. pectinata*. Water depth may play a key role in the success of submersed macrophytes (Rea et al. 1998). To evaluate this in LBCR planting was stratified at initially controlled water levels (depths). Water temperature in shallow areas can become a significant factor
impacting plant growth (Barko et al. 1982, Pilon and Santamaria 2002). The hypothesis was that water depth would facilitate temperature differences and thus cause differential survival and growth of plant species. However, results indicated that water depths of 0.30, 0.60, and 1.0 m rarely had any significant effect on survival percentage of *P. nodosus* or *V. americana*.

In spring 2008, water levels appeared to be rising in concordance with the operating guide for LBCR; however, structural complications with LBCR dam required TVA to drop the water levels to implement repairs. Therefore, in 2008 water levels still did not reach full pool but were relatively stable throughout the study period. The unpredictable nature of water levels from year to year must be considered when planning additional re-vegetation efforts in LBCR or when considering re-vegetation in other reservoirs. Providing water levels respond in concordance with the operating guide for LBCR, future projects may be started sooner in the year to allow for a longer growing season.

After two years of trials, it appeared that efforts should focus on *P. nodosus* as the primary species to maximize re-vegetation success. Other studies also have indicated this species as a good candidate for re-establishment (Doyle and Smart 1995, Smart et al. 2005, Smiley and Dibble 2006). In 2009, a continuation of planting *P. nodosus* was planned with exclosure size instead of species or planting depth as the variable of interest. *Potamogeton nodosus* has shown promising results in the ability to grow and expand within exclosures. However, observations of plant growth outside of the exclosures are rare and have not been sustained. This is likely due to some form of
herbivory from fauna that cannot penetrate the exclosures, but can consume or otherwise desiccate plant material that grows beyond the 1m diameter protection of the exclosure. This has been observed in past re-vegetation trials (Doyle and Smart 1995, Dick et al. 2004a). Presence of aquatic turtles and common carp were observed in LBCR and on occasion were found inside the plant exclosures. In the 2009 study, efforts focused on building larger protected areas to allow for greater expansion of plant assemblages. The initial experimental design planned for the removal of exclosures from well established patches to assess whether patch size had an effect on the ability of the plant patch to reproduce and continue to expand even with herbivory. Due to sudden water level changes, this experiment was not completed. However, an assessment of percent coverage within each exclosure indicated that *P. nodosus* can spread into areas larger than had previously been tested (> 1m diameter), as long as protection from herbivores is still present. Because the infrastructure is still in place, this experiment can potentially be completed in the future.

Future hypothesis testing and monitoring may be the best way to continue advancing the art and science of native aquatic plant re-establishment. Future work should focus on mitigating for negative abiotic (e.g., water level fluctuations) and biotic (e.g., herbivory) factors. Future studies might also focus on establishing emergent species such as American water-willow *Justicia americana*. Recently, quantitative studies regarding its establishment have been advanced (Strakosh et al. 2005, Collingsworth et al. 2009). Research also has suggested that waterwillow can have a significant impact on largemouth bass density with little or no negative effect on growth.
rate (Strakosh et al. 2009). Using this species as a candidate for re-establishment has been suggested (Smart et al. 2005) but was not performed in LBCR due to lack of time and resources.
BIBLIOGRAPHY


Figure 2.1  Macrophyte re-establishment areas in Little Bear Creek Reservoir, Alabama from 2008-2009.
Figure 2.2  Water levels in Little Bear Creek Reservoir, Alabama for 2007-2008. Water levels were dropped rapidly in mid-April 2008 and did not reach full pool in 2007 or 2008.
Figure 2.3  Diagram of macrophyte protective exclosure sites stratified by depth for re-establishment efforts in Little Bear Creek Reservoir, Alabama in 2008. The exclosures were assessed for survival in 2008 and 2009.
Figure 2.4 Diagram of the experimental design for exclosures planted with *Potamogeton nodosus* in Little Bear Creek Reservoir, Alabama in 2009. Half of the exclosures should have been removed to assess whether the patch size had an effect on survival without herbivore protection. However, water levels were dropped in 2009 and this experiment was not completed.
Figure 2.5  Water levels in Little Bear Creek Reservoir, Alabama for 2008-2009. Water levels did not reach full pool in 2008 or 2009 and were dropped rapidly in April 2008 and mid-September 2009.
Table 2.1 Statistical comparisons of the survival of three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008. Bold Type III Test results represent significant differences in survival among the three species on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date.

<table>
<thead>
<tr>
<th>Date (2008)</th>
<th>7/12</th>
<th>7/19</th>
<th>8/6</th>
<th>8/24</th>
<th>9/7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Type III Test</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td><em>P. nodosus vs. V. americana</em></td>
<td>0.001*</td>
<td>0.004*</td>
<td>0.017*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
</tr>
<tr>
<td><em>P. nodosus vs. S. pectinata</em></td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
</tr>
<tr>
<td><em>V. americana vs. S. pectinata</em></td>
<td>0.088</td>
<td>0.054</td>
<td>&lt;0.001*</td>
<td>0.008*</td>
<td>&lt;0.001*</td>
</tr>
</tbody>
</table>

Figure 2.6 Percentage of exclosures containing living propagules of one of the three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008 for each surveying date.
Table 2.2  Statistical comparisons of the presence/absence survival survey results for *P. nodosus* planted in 2008 at three different depths in Little Bear Creek Reservoir, Alabama. Bold Type III Test results represent significant differences in survival among the three depths on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date.

<table>
<thead>
<tr>
<th>Potamogeton nodosus</th>
<th>Date (2008)</th>
<th>7/12</th>
<th>7/19</th>
<th>8/6</th>
<th>8/24</th>
<th>9/7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth Pairwise Comparison</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Type III Test</td>
<td>&gt;0.999</td>
<td>&gt;0.999</td>
<td>0.094</td>
<td>&gt;0.999</td>
<td>&gt;0.999</td>
<td></td>
</tr>
<tr>
<td>0.30 m vs. 0.60 m</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
</tr>
<tr>
<td>0.30 m vs. 1.0 m</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
</tr>
<tr>
<td>0.60 vs. 1.0 m</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
</tr>
</tbody>
</table>

Figure 2.7  Percentage of exclosures containing living propugles of *P. nodosus* planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each surveying date.
Table 2.3  Statistical comparisons of the presence/absence survival survey results for *V. americana* planted in 2008 at three different depths in Little Bear Creek Reservoir, Alabama. Bold Type III Test results represent significant differences in survival among the three depths on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date.

<table>
<thead>
<tr>
<th>Vallisneria americana Depth Pairwise Comparison</th>
<th>Date (2008)</th>
<th>7/12</th>
<th>7/19</th>
<th>8/6</th>
<th>8/24</th>
<th>9/7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Type III Test</td>
<td></td>
<td>0.635</td>
<td>0.862</td>
<td>0.079</td>
<td>0.856</td>
<td>0.348</td>
</tr>
<tr>
<td>0.30 m vs. 0.60 m</td>
<td></td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
</tr>
<tr>
<td>0.30 m vs. 1.0 m</td>
<td></td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
</tr>
<tr>
<td>0.60 vs. 1.0 m</td>
<td></td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td>...</td>
</tr>
</tbody>
</table>

Figure 2.8  Percentage of exclosures containing living propagules of *V. americana* planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each surveying date.
Table 2.4  Statistical comparisons of the 2009 survival of three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008. Bold Type III Test results represent significant differences in survival among the three species on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date.

<table>
<thead>
<tr>
<th>Date (2009)</th>
<th>5/17</th>
<th>6/1</th>
<th>6/15</th>
<th>6/29</th>
<th>9/7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Type III Test</td>
<td>0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td><em>P. nodosus vs. V. americana</em></td>
<td>0.002*</td>
<td>0.004*</td>
<td>0.003*</td>
<td>0.059</td>
<td>&lt;0.001*</td>
</tr>
<tr>
<td><em>P. nodosus vs. S. pectinata</em></td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
</tr>
<tr>
<td><em>V. americana vs. S. pectinata</em></td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
<td>&lt;0.001*</td>
</tr>
</tbody>
</table>

Figure 2.9  Percentage of exclosures in 2009 containing living propagules of one of the three macrophyte species planted in Little Bear Creek Reservoir, Alabama in 2008 for each surveying date.
Table 2.5 Statistical comparisons of the 2009 presence/absence survival survey results for *P. nodosus* planted in 2008 at three different depths in Little Bear Creek Reservoir, Alabama. Bold Type III Test results represent significant differences in survival among the three depths on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth Pairwise Comparison</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Type III Test</td>
<td></td>
<td>0.118</td>
<td>0.424</td>
<td><strong>0.010</strong></td>
<td><strong>0.009</strong></td>
<td>&gt;0.999</td>
</tr>
<tr>
<td>0.30 m vs. 0.60 m</td>
<td></td>
<td>...</td>
<td>...</td>
<td>0.067</td>
<td>0.277</td>
<td>...</td>
</tr>
<tr>
<td>0.30 m vs. 1.0 m</td>
<td></td>
<td>...</td>
<td>...</td>
<td>0.325</td>
<td>0.069</td>
<td>...</td>
</tr>
<tr>
<td>0.60 vs. 1.0 m</td>
<td></td>
<td>...</td>
<td>...</td>
<td>0.012*</td>
<td>0.012*</td>
<td>...</td>
</tr>
</tbody>
</table>

Figure 2.10 Percentage of exclosures containing living propagules of *P. nodosus* planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each 2009 surveying date.
Table 2.6  Statistical comparisons of the 2009 presence/absence survival survey results for *V. americana* planted in 2008 at three different depths in Little Bear Creek Reservoir, Alabama. Bold Type III Test results represent significant differences in survival among the three depths on each date and asterisks (*) indicate significant differences for each pairwise comparison on each date.

<table>
<thead>
<tr>
<th>Vallisneria Americana</th>
<th>Date (2009)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth Pairwise Comparison</td>
<td>5/17</td>
</tr>
<tr>
<td>Type III Test</td>
<td>0.514</td>
</tr>
<tr>
<td>0.30 m vs. 0.60 m</td>
<td>...</td>
</tr>
<tr>
<td>0.30 m vs. 1.0 m</td>
<td>...</td>
</tr>
<tr>
<td>0.60 vs. 1.0 m</td>
<td>...</td>
</tr>
</tbody>
</table>

Figure 2.11  Percentage of exclosures containing living propagules of *V. americana* planted at three different depths in Little Bear Creek Reservoir in 2008 recorded for each 2009 surveying date.
CHAPTER III

DEDUCTIVE GIS MODEL FOR SUITABLE RE-VEGETATION SITE SELECTION

Introduction

Diverse native aquatic macrophyte communities serve a number of physical and biological functions in the aquatic environment (Carpenter and Lodge 1986). It is postulated that shallow freshwater ecosystems occur as eutrophic, phytoplankton dominated or clear macrophyte dominated systems (Scheffer 1998). However, in some cases native macrophyte communities are absent even when trophic status and light availability are conducive to rooted submersed macrophyte growth, especially in young systems that may not have a macrophyte seed bank or are too distant for dispersal of native species (Smart et al. 1996, Smart et al. 2005).

Native aquatic macrophyte re-establishment is a tool that may be used to restore littoral freshwater communities (Smiley and Dibble 2006). Reestablishing plants has numerous of advantages such as improving water quality (Scheffer 1998), providing essential habitat and relatively safe foraging areas for young of the year fishes (Dibble et al. 1996) and refugia for macroinvertebrate and zooplankton phytoplanktivores (Diehl and Kornijow 1998). Research on the propagation and establishment of native submersed aquatic plant communities is limited. Most research regarding the restoration of submersed aquatic plant communities is related to the natural recolonization of
macrophytes following some environmental manipulation that creates conditions that are conducive to plant growth (Sondergaard et al. 2007). However, there have been a number of trials dealing specifically with native plant re-establishment, and techniques for propagation and establishment have been advanced in recent years (Smart et al. 1998, Smart et al. 2005). One important consideration that is currently a hindrance to re-establishment is the selection of suitable sites to focus the restoration efforts (Grodowitz et al. 2009). The first step to hurdle this obstacle is to identify the factors that are important for macrophyte colonization, growth, and persistence.

Factors affecting macrophyte colonization and growth have received considerable attention in the literature. Many studies both in the field and laboratory have been performed in an attempt to understand the factors that contribute to the habitat requirements of macrophytes. It is commonly noted that the most limiting factor for freshwater macrophytes is light availability which directly impacts their maximum colonization depth (Van Duin et al. 2001, Case and Madsen 2004, Wersal et al. 2006). Light is important to all aquatic organisms because it is the energy source that fuels primary production and consequently all trophic levels in aquatic ecosystems (Scheffer 1998). Sediment texture also is important because it directly relates to benthic stability as well as biogeochemical variables important to macrophytes such as nutrient availability and presence of organic materials (Barko and Smart 1986). Temperature is important in life-cycles of macrophytes, but vertical and seasonal variability as well as latitude may be of greater importance than its horizontal distribution in lentic systems (Barko et al. 1986, Madsen and Adams 1989). Other physical factors that may significantly contribute to
presence of macrophytes are fetch, wave action, suspended sediment and littoral slope (Duarte and Kalff 1986, Doyle 2001, Koch 2001, Bini and Thomaz 2005). Suspended sediment is a limiting factor only because of its affect on light attenuation and therefore, if there is suitable light availability data then suspended sediment data is likely redundant.

Although the environmental conditions that affect macrophyte growth are often continuous and spatially explicit (e.g., depth), few approaches have considered scale dependent or lake wide geographic distribution of these environmental metrics within the aquatic system (Cheruvelil and Soranno 2008). Likewise, species composition or only biomass has been considered in most predictive studies without regard to geographic distribution (e.g., Narumaiani et al. 1997, Best and Boyd 2008, Best et al. 2008). Van Nes and others (2003) proposed a spatially explicit model named Charisma to simulate the growth of single or multiple macrophyte species. Interestingly enough, their spatially explicit model contains no graphical output capabilities (maps), no coordinates linking to Earth’s surface, and is likely far too complex for realistic management applicability. They also noted that their model was developed as a method for generating hypotheses and not as a prediction tool.

In an effort to solve the site selection conundrum, Grodowitz and others (2009) developed a multi-attribute utility analysis model as a deductive method for selecting submersed macrophyte restoration sites. This modeling approach is more “application-friendly” than other complex models. However, there are some particular weaknesses that make its scientific applicability questionable. First, model performance is based on expert ratings which are subject to bias and may be influenced more by opinion than
scientific information. Second, it relies on pre-selected sites for areas to be analyzed. This defeats the purpose of selecting optimal locations in a reservoir because pre-model site selection is arbitrary and introduces bias and uncertainty before the model is even applied. Lastly, the model is not based solely on optimal growth of macrophytes but instead gives high priority to propagation convenience (i.e., the model has parameters that select sites which are easily accessible rather than sites that could be more successful). Similar to the model presented by van Nes (2003), this model is location based but does not have any spatial visualization capabilities. It appears that the multi-attribute utility analysis model is missing an important step in the restoration process that relies on scientifically based site selection before the application of a model that selects locations convenient for human access.

The increased availability of geographic information system (GIS) technology in academia has provided new opportunities to approach scientific inquiry and is useful in mapping, analyzing, predicting and visualizing macrophyte distributions. GIS provides a coherent set of tools and procedures to relate spatial data with elaborate habitat models (Lehmann et al. 1997). In an exceptional overview of the methodological aspects of GIS in aquatic botany, Caloz and Collet (1997) note that GIS is viewed by some as only a means of cartographic representation. However, they argue that its [GIS] gravitational center is the capacity to give structure to the organization and processing of spatial information. Although GIS has traditionally been used more in the terrestrial realm, it has a great deal of applicability in aquatic systems as well. A 1997 issue of Aquatic Botany was dedicated solely to the application of geographic information systems for the
mapping, analysis and prediction of aquatic vegetation and highlighted some important advantages of GIS to the field of aquatic botany (Lachavanne et al. 1997). In this issue, Lehmann and Lachavanne (1997) highlighted three reasons environmental scientists need to embrace these new tools: 1) to improve understanding of the environment, 2) to enhance communication between different specialists, and 3) to affect decision-makers. Additionally, Muller (1997) advocated that vegetation mapping can drive research, moderate subjectivity of conceptual statements, and validate ecological theories. Now, over a decade later, GIS technology has undoubtedly been adopted by more universities and management agencies and both of these institutional groups now have access to the same powerful geographic technologies. This presents a unique opportunity for researchers to apply the current state of knowledge regarding the fundamental niche of macrophytes to the development of spatially explicit tools that can actually be applied by management personnel to enhance re-establishment efforts.

Most studies related to using GIS in aquatic vegetation science thus far have applied an inductive approach by explaining macrophyte distributions based on empirical data specific to a particular study area (e.g., Remillard and Welch 1993, Narumalani et al. 1997, Lehman 1998, Rea et al. 1998). While this approach is useful for identifying spatially explicit factors that are important to macrophyte distributions, they rarely produce predictive models or processes that are general enough to be applied elsewhere (but see Narumaiani et al. 1997 and Lehman 1998). In addition, variables used for inputs into these models are not always available. This is not a problem if data for a particular variable can be readily collected, but in some cases that is not possible (e.g., aerial
photography of a reservoir prior to impoundment for the delineation of soils as in Narumaiani et al. 1997 and Lehmann 1998 is unlikely to be available). This makes their management application impractical.

Models using GIS functionality have tested the importance of specific variables to macrophyte occurrence or distribution with a variety of methods. Several statistical methods used to compute the significance of different variables in conjunction with GIS for predicting macrophyte occurrence are Chi Square test of independence (Remillard and Welch 1993), logistic multiple regression (Narumaiani et al. 1997), logistical regression (Rea et al. 1998), and Generalized Additive Models (Lehmann 1998). The numerous techniques that have been used in these inductive prediction models are indicative of the emergence of using this technology for studying the distribution of vegetation in the aquatic environment. However, it is interesting to note that similar variables emerge in all of these studies as important for prediction models (e.g., light, depth, and fetch).

Ecological models can be defined as representations or abstractions of reality (Remillard and Welch 1993) and modeling reality is the basis for the scientific method (Caloz and Collet 1997). To maximize effectiveness of aquatic plant re-establishment in areas devoid of vegetation, predictive models of optimal (or just suitable) locations should be investigated. Data regarding the important variables for macrophyte occurrence and distributions should be the basis for these predictive models. Use of Geographic Information Systems provides a framework for organizing spatial data and
modeling ecological processes and has the potential to be a powerful tool for enhancing macrophyte re-establishment efforts.

**Objectives**

The objectives here are to synthesize information related to the prediction of macrophyte occurrence in lentic littoral habitats, collect appropriate data and create a deductive model in GIS based on this information to identify and locate suitable areas for native aquatic plant re-establishment in Little Bear Creek Reservoir. This is accomplished mainly through development of a methodological framework for the model. Although not tested here, the hypothesis is that native aquatic macrophytes will have greater survival rates if areas conducive to plant growth are selected for re-establishment sites which will make efforts more efficient and management more effective.

**Methods**

**Parameter Selection, data acquisition, and calculations**

In developing a model, defining and justifying the necessary variables is an essential first step. Literature regarding factors important for macrophyte occurrence was collected and summarized. The following parameters and methods describe the most commonly implicated factors in determining the occurrence and distribution of aquatic macrophytes, the reasons these factors are important, and the methods used to collect and apply the data to the GIS model. The study area for model development is Little Bear
Creek Reservoir (LBCR) in northwest Alabama. Study area descriptions are provided in the previous chapters.

Light

Availability of light is used to define littoral zones in freshwater lentic systems because it provides the energy source that fuels photosynthesis. For this reason, light has been noted as the most limiting factor to macrophyte occurrence (Barko et al. 1986). Photosynthetically Active Radiation (PAR) was collected using a cosine corrected Li-Cor LI-1000 light meter with surface and underwater quantum light sensors in June 2008 and from June to September 2009. Water clarity also was measured with a standard 20cm black and white Secchi disk during these times. The Li-Cor sensors measure quanta of light within the photosynthetic spectra (400 – 700 µm) as µmol m⁻²s⁻¹. PAR measurements were taken in 50cm increments in the water column at a location in the middle of the lake to avoid shading from trees. Underwater and surface (deck cell) readings were taken at every depth interval until the underwater sensor reached the bottom of the lake or the entire data cord was used (9m), whichever came first. Attempts were made to collect at least three separate profiles each sampling date but due to time constraints some dates only have one or two profiles. Light transmittance (T) was calculated as:

\[ T = \frac{L_{Zn}}{L_{0n}} \times 100 \]  

Equation 3.1

where \( L_{Zn} \) was underwater light intensity at depth \( n \) and \( L_{0n} \) was light intensity of the deck cell that corresponded to the reading for depth \( n \) (modified from Madsen et al.)
Light transmittance was not used in the model but may be useful for future comparisons of light attenuation percentages and responses of macrophytes at various depths. For example, Middelboe and Markager (1997) found means of five and sixteen percent surface irradiance for the colonization depths of caulescent and rosette-type angiosperms respectively.

A commonly used metric for assessing light availability for submersed macrophytes that was used for this model is the light extinction coefficient \( (K_d) \) which is calculated as:

\[
K_d = \left[ \ln(L_{Z1}) - \ln(L_{Z2}) \right] / (Z_2 - Z_1) \tag{ Equation 3.2 }
\]

where \( L_{Zn} \) is the light intensity at depth interval \( n \), and \( Z_n \) is the depth (in meters) of depth interval \( n \) (Madsen et al. 1999). Coefficients are generated for each depth interval except for the final measurement. The light extinction coefficient for a particular day was estimated by calculating the arithmetic average of \( K_d \). Furthermore, the seasonal arithmetic average extinction coefficient \( (K_{ds}) \) was generated by averaging the mean coefficient for each sampling date (Table 3.1). Light extinction coefficients have been used in several models as an explanatory variable for macrophyte distribution or growth. However, the coefficient itself is only useful for suitability selection when incorporated with bathymetry (see below).

The Secchi disk was used to measure transparency (or clarity) in the water column. The disk was lowered into the water until it wasn’t visible, depth recorded to the nearest cm, and then raised until it reappeared, and depth recorded. These two numbers were averaged to calculate the arithmetic mean Secchi disk depth \( (Z_{sd}) \) which is often
used to describe transparency (Table 3.1; Madsen et al. 1999). Units between Secchi disk and PAR meter are not comparable. However, past studies have indicated that the extinction coefficient is correlated negatively with Secchi disk depth. It also has been noted that maximum colonization depth of macrophytes can often be calculated by doubling Secchi disk depth. Secchi disks are relatively simple pieces of equipment that are readily available or can be constructed with ease and therefore for management applicability it may be more feasible for widespread use than a PAR meter to assess light conditions.

Other factors affect light penetration in the water column. Turbidity is often measured and used as a surrogate for water transparency. Turbidity, Secchi disk depth and light extinction coefficients are all correlated. However, the extinction coefficient is the most useful because it takes into account attenuation from particulate and dissolved material whereas turbidity only measures particulate material and Secchi disk is subject to visual errors from the observer (Bini and Thomaz 2005).

**Bathymetry**

PAR measurements are only useful in this model when incorporated with depth. Prior to the start of this study no bathymetric data existed for LBCR. In June 2008 a point-intercept survey (Madsen 1999) was performed with a 100m separated point grid (Figure 3.1). Each point was precisely located using a Panasonic Toughbook equipped with a Trimble AgGPS 106 GPS receiver and Farmworks Site Mate geospatial software. At each point, depth was determined using a Lowrance depth sonar and recorded to the nearest meter. These data were exported as a shapefile and added to ArcMap GIS.
software (ESRI 2009). Depth was converted to a mean sea level elevation ($\text{MSL}_{\text{di}}$) with the equation:

$$\text{MSL}_{\text{di}} = \text{MSL}_{\text{si}} - Z_{\text{di}}$$  \hspace{1cm} \text{Equation 3.3}

where $\text{MSL}_{\text{si}}$ is the mean sea level of the water surface in ft determined with the GPS unit and $Z_{\text{di}}$ is the recorded depth in ft at location $i$. There are a few reasons $\text{MSL}_{\text{di}}$ is a better metric than depth. When the points are interpolated to form a continuous hydrologically correct surface, they should be bound by the perimeter of the lake. The lake perimeter was defined by generating a series of pseudo-points by converting the lake perimeter to 5m raster pixels and then converting those pixels back into vector points (this process may seem counterintuitive but it is necessary to achieve controlled point spacing). A value of MSL in m$^{-1}$ for the lake perimeter was then assigned to each point so that bathymetry would be consistent with the lake boundaries rather than assigning a border value of zero to the border and using the collected depth. Another reason this is important is because in June 2008 LBCR was not at full pool but no imagery was available to digitize the lake boundary for the water surface elevation of the surveying days. Therefore, using depth as the interpolating variable would result in the generation of erroneous slopes if depth data were interpolated without further adjustment. Data adjustment for depth could have been performed but would have resulted in a much more complicated process in the future if there is a desire to run model simulations at different lake surface MSL values.

Once elevation adjustments were made the points were interpolated into a 10m$^{-2}$ continuous bathymetric surface using the ArcGIS Spatial Analyst geoprocessing tool.
Topo To Raster in ArcMap (Figure 3.2). *Topo To Raster* is specifically designed to generate hydrologically correct elevation surfaces and if needed allows finer control of input variables. Maximum colonization depth ($Z_c$) in $\text{m}^{-1}$ was estimated from the mean seasonal light extinction coefficient ($K_{ds}$) using the Lambert-Beer equation:

$$Z_c = -\ln(I_z/I_0)/K_{ds} \quad \text{Equation 3.4}$$

where $I_z/I_0$ is the percentage of light required by the species under consideration or the percentage of light at the maximum depth distribution of the plants. Vant et al. (1986) found that the value of $-\ln(I_z/I_0)$ could be set at a constant coefficient equal to 4.34 to explain most (93 percent) of the variability of maximum colonization in North Island, New Zealand lakes of differing clarity. This location has similar latitude, albeit in the southern hemisphere, as northern Alabama where LBCR is located. Middelboe and Markager (1997) indicated that maximum colonization depth for a species should be constant in lakes at the same latitude if the true maximum colonization depth is determined exclusively by light availability. Therefore, $Z_c$ is estimated by using the constant coefficient (4.34) found by Vant et al (1986).

Mean maximum colonization depth was used as the depth barrier in the GIS model by first subtracting $Z_c$ from MSL of LBCR. Alternatively, for different lake level simulations MSL$_{si}$ could be substituted for MSL. This determines the maximum MSL elevation at which plants should be able to colonize. The GIS layer for $Z_c$ used in the model is binary since plants either can or cannot survive at a given depth based on light availability. Therefore, the bathymetry raster was queried using the *Greater Than Equal* logical math geoprocessing operator. This operator returns a value of 1 for pixels where...
the statement is true and 0 for pixels where the statement is false. The resulting layer (Figure 3.3) indicates areas suitable for macrophyte growth based on light and depth.

**Slope**

Duarte and Kalff (1986) found significant differences between maximum submerged macrophyte biomass at steep and gentle slopes with greater biomass occurring at gentle slopes. This indicates that slope may be a significant factor not only in submersed plant biomass but also for optimal growth. Rea and others (1998) found that slope was a significant factor in some years but not others indicating collinearity with another variable. Others have recommended aquatic macrophyte re-establishment locations should only be performed on gentle slopes (e.g. Smart et al. 2005). Quantitative estimates of suitable slope have ranged from 2.24% to 5.33% (Duarte and Kalff 1986). However, Rea and others (1998) only classified slope into steep and less steep. This presents a conundrum for modeling macrophytes using the slope variable. Typically, slope is judged only qualitatively from site visits or quasi-quantitatively by using existing paper bathymetric maps (Duarte and Kalff 1986). However, using GIS, estimates can be made rather easily from the bathymetric data generated in the previous step. The *Slope* operator was used for calculating percentage slope of the bathymetric data layer (Figure 3.4). To reach some type of comparative data that was study area dependent rather than static, the slope layer was classified into two classes using a geometric interval classification method. The geometric interval algorithm was designed specifically for continuous data. It minimizes variance within classes and works well with data that may be non-normal (ESRI 2009). The slope layer was then queried using the *Greater Than*
Equal logical math geoprocessing operator (as described above). Unlike bathymetry, it was necessary to retain the ability to rank slope values to incorporate into the model because most studies have indicated that less-steep slopes are generally more conducive to plant growth than steeper slopes. Ordinarily, multiplying the binary slope layer by the original slope layer using the *Times* math geoprocessing operator would return valid results. However, because zero percentage slope is considered an acceptable value the *Set Null* conditional operator was used. This creates a raster with the original slope values for each pixel within the suitable range and sets all non-suitable pixels to a null value instead of zero. To incorporate it into the model the data were standardized to a scale of 0.0 to 1.0 using the formula:

\[ V_i = \frac{(S_i - S_{\text{min}})}{(S_{\text{max}} - S_{\text{min}})} \]  \hspace{1cm} \text{Equation 3.5}

where \( V_i \) is the standardized value for the original slope percentage \( S_i \) at a particular pixel, \( S_{\text{min}} \) is the least original slope percentage, and \( S_{\text{max}} \) is the greatest original slope percentage (modified from Chang 2010). This standardized number is greater for greater slope percentage values but when incorporated into the model the lesser value should be considered more advantageous for plant growth. To fix this problem \( V_i \) was subtracted from 1 using the *Minus* math geoprocessing operator so that lesser standardized values (lesser slopes) would get a greater score. The transformation was applied using *Divide* and *Minus* math geoprocessing tools in ArcMap. The results of suitable slope areas with a standardized rank are presented in Figure 3.5.
Fetch

Effective fetch length is the weighted distance around a wind direction that wind can travel over the water surface without obstruction from land (Lehmann 1998). This is important in aquatic plant models because the longer the length the greater the probability of generating larger waves that can dislodge plants or increase suspended sediments which Doyle (2001) demonstrated as a significant factor limiting total plant mass accumulation. The USGS developed a model (python script) that runs in ArcGIS to calculate effective fetch length (with no correction for wind speed) for a water body. The model requires a binary raster of the water body (water body = 1, land = 0) and an angular degree measurement of wind direction (direction of origin). The existing vector polygon of LBCR was converted to a 10m$^{-2}$ raster grid for use as the input. Wind direction was estimated from weather data collected from a weather station in Belgreen, Alabama (approximately 5 miles from LBCR). Wind direction has the potential to change throughout the year but most wind direction is from due west ($270^\circ$). The effective fetch model calculates distances to land for nine radials spaced every three degrees on each side of the input degree of wind direction. Each measurement ($n = 9$) is then weighted by the cosine of the angle deviation. Effective fetch for each pixel is calculated in the USGS model with the equation:

$$L_f = \frac{\sum x_i \cdot \cos \gamma_i}{\sum \cos \gamma_i} \quad \text{Equation 3.6}$$

where $L_f$ is the effective fetch, $x_i$ is the distance to land for a given angle, and $\gamma_i$ is the deviation angle. A suitable value for effective fetch is rarely reported in the literature even though it is often used as an explanatory factor for macrophyte occurrence. Because
there is no suitable evidence to justify selecting prime areas based on a set range of values, defining this as a binary layer would be subjective. Therefore, to incorporate it into the model, data were standardized to a scale of 0.0 to 1.0 using the same linear transformation used above for slope except that S was replaced by D which is equal to the effective fetch distance calculated with the USGS fetch model. The standardized number for fetch is greater for greater fetch value but when incorporated into the model the lesser value should be more advantageous for plant growth, exactly like the slope values calculated above. Again, to fix this problem $V_i$ was subtracted from 1 so that lesser standardized values would get a greater score. This transformation was applied using multiple math geoprocessing tools in ArcMap and is presented in Figure 3.6.

**Sediment Characteristics**

Several studies have indicated that sediment texture and organic matter content can impact the distribution of submersed aquatic macrophytes (Barko and Smart 1983, Barko and Smart 1986, Barko et al. 1986, Lehmann et al. 1997, Koch 2001, Madsen et al. 2006). General conclusions have been that fine sediment texture increases macrophyte productivity by increasing nutrient availability and cation exchange capacity whereas increasing organic matter decreases occurrence because of unfavorable chemical accumulation from decomposition (Koch 2001). Several studies have found no relationship between sediment texture and macrophyte distribution within reservoirs but have noted differences in growth when different sediment types were analyzed in the laboratory (Madsen et al. 1996). Given its relative importance in the distribution of
macrophytes, sediment cores were collected in LBCR for analysis of texture and organic matter content to be included in the habitat suitability model.

Texture

After surveying depth LBCR with a 100m point grid in 2008, points less than 2 X Secchi disk depth were selected for the sediment survey. This method of selection is sometimes used as a rough estimate of maximum macrophyte colonization depth and to define the littoral zone of a lentic system. Surveying all points of the 100m grid (n = 566) was not feasible and unnecessary considering that light is the main limiting factor on colonization depth (Barko et al. 1986). The points that were left (n = 213) were defined as sample sites in the littoral zone and surveyed in July 2008 (Figure 3.7). Using a modified 2in diameter by 5in length tip biomass sampler made of pvc pipe (after Madsen et al. 2007), each point was located with a HP Ipaq pocketPC equipped with a Holux GPS ultra receiver (GR-271) and Farmworks Site Mate geospatial software and sediment cores were collected and stored in whirl-packs. The exact location of the sample was logged in the GPS and the unique number provided by the GPS for the location was used to label the sediment sample. All samples were immediately put in a cooler and when sampling was completed they were stored in a freezer at Mississippi State University. Samples in deeper water (>4m) could not be collected with the modified sediment corer. Instead, samples were collected by free diving to the bottom with a plastic zip-lock bag and a hand-grabbing. Samples in some areas were not collected because of the substrate type. If sediment could not be collected using the corer, the point was marked and the substrate type (gravel, cobble, or bedrock) was recorded in the GPS.
In the lab the samples were thawed, weighed to approximately 150g in an aluminum weighing pan and dried in an oven at 105°C for 24 hours. Sediments were then ground with mortar and pestle and precisely weighed to 50g back into the aluminum pan with an analytical balance. Sediment also was weighed to 10g in a separate aluminum pan and labeled with the corresponding sample number for use in organic matter content analysis (see below). Some samples did not contain enough sediment to weigh 50g after drying. If this occurred, only 10g samples for the organic matter analysis were weighed.

Texture was measured as the particle size distribution of sand, silt and clay in each sample using the Bouyocous hydrometer method (Bouyocous 1962, Ashworth et al. 2001). Raw data were entered into a spreadsheet and corrected according to the formulas in Ashworth and others (2001). Data were then joined to a point shapefile generated from GPS coordinates of the sample sites in ArcGIS using the sample identification number as the join field (n = 136).

Point interpolation algorithms do not accept multivariate inputs. Therefore, to get accurate sediment surface predictions for the littoral zone of LBCR each particle class was interpolated separately using the ArcGIS Spatial Analyst kriging interpolator. Verification for the use of various interpolation methods for hydric sediments was sought out but materials that were relevant for this model were not found. A likely reason for this is the limited effort that has been put forth in mapping sediment in shallow-water habitats (Demas et al. 1996). However, Valley and others (2005) reviewed interpolation techniques for mapping submersed vegetation and found that kriging provided the best
results. Therefore, it was decided that if macrophyte distribution is correlated with sediment (as is an assumption of this model) then sediment interpolation surfaces should resemble those of macrophyte distribution surfaces. This resulted in three raster layers (sand, silt, and clay) containing the interpolated percentages. By interpolating each surface separately it was apparent that over and under estimations could potentially be made in areas with skewed influences. For example, an area with a relatively low content of one particle size but high contents of other particle sizes were intersected, the intersected value could be above or below 100. To reduce this error the pixels in each layer were normalized to 100 percent. Raster layers were first added together to get a total value (theoretically when each layer is added together the total should be 100). Interestingly, the range of values when all layers were added together was 99.5 to 100.6 percent which was surprisingly accurate. Each texture layer was then normalized by dividing it by the total value layer. The Plus and Divide math geoprocessing operators were used for this process.

Texture preferences for freshwater macrophytes vary in the literature. However, there is a consistent theme that seems to always emerge that finer sediments are more preferable. Koch (2001) used the percentage of silt + clay to define fine sediments for classifying macrophyte preferences. By contemplating the sediment texture triangle the decision was made that, for the GIS model, suitable fine sediments would be defined as such if the silt plus clay content were greater than or equal to 50 percent. The normalized silt layer and the normalized clay layer were combined with the Plus math geoprocessing operator. The Greater Than Equal logical math geoprocessing operator was used to
select pixels with a value greater than or equal to 50. This created a binary raster with preferable sediment texture (Figure 3.8). There were no assumptions made that an increase from 50 to 100 would increase preference so no standardization was performed on suitable fine sediment values.

Points marked as bedrock, gravel or cobble were considered unsuitable for macrophyte colonization. These points were queried in ArcMap and converted to a 100m² binary raster and used as a constraint in the final model.

**Organic Matter Content**

The percentage of organic matter content generally has a negative correlation to macrophyte growth (Barko and Smart 1983, Barko and Smart 1986) although the limiting mechanism is not well understood (Koch 2001). It is likely that this limitation is due to nutrient limitations or the generation of unfavorable chemicals through decomposition (Barko and Smart 1986). However, it is hypothesized that oxygen released in the rhizosphere may buffer these effects on plants but at the same time facilitating the need for greater light availability for the production of this oxygen through photosynthetic processes (Koch 2001).

Organic matter content was analyzed using the same sediment samples that were used for texture analysis. While weighing samples for PSD analysis, 10g were weighed precisely separately with an analytical balance. Organic matter content was determined using the combustion (also called loss on ignition) method (Storer 1984, Jones 2001). First, a 150ml stone crucible was weighed using an analytical balance to the nearest ten-thousandth gram. Sediment weighed to 10g was then emptied into the crucible.
Sediment was often powdery and therefore became suspended when it was poured from one container to the other. To account for this, approximately 20 seconds was allowed to elapse after pouring the sediment in before weighing the filled crucible. As soon as the weight was determined, the crucible was capped with a glass crucible cover to prevent accidental loss. The samples were then put in a muffle furnace with the lids removed and the furnace was set to 375°C. After 1 hour the temperature of the muffle furnace temperature was set to 550°C to fully combust any organic material in the samples. The purpose of the initial warm-up period was to prevent initial volatilization that would result in erroneous final weights as recommended by Jones (2001). The samples were burned for 24 hours and then the furnace was turned off and the samples were allowed to cool for approximately 8 hours inside the furnace. Cooling time could have been significantly lowered if samples were removed from the furnace, but moisture acquired from the air could contaminate the samples and compromise the data. When samples were cool they were removed from the furnace, immediately capped with the glass crucible cap to prevent loss during transport and weighed again to the nearest 0.001g (without cap). Ten samples were completed at a time.

Raw data were entered into a spreadsheet and percentage organic matter was calculated by subtracting the filled crucible weight after combustion from the filled crucible weight before combustion and then dividing that number by the filled crucible weight before combustion. Data were then joined to a point shapefile generated from GPS coordinates of the sample sites in ArcGIS using the sample identification number as
the join field \( n = 139 \). Percentage organic matter was interpolated using the kriging method in ArcGIS Spatial Analyst (same as texture, described above).

Organic matter preferences are reported to be below 5 percent (Barko and Smart 1983). Therefore, the organic matter percent raster was queried using the \( \text{Less Than Equal} \) logical math geoprocessing operator. This created a binary raster with preferable organic matter content (Figure 3.9). There were no assumptions made that a decrease in value from 5 to 0 would increase suitability so no standardization was performed on suitable organic matter content values.

**GIS model development**

Once all suitable layers were generated they were combined for an overall suitability map. If all layers were binary and each had equal value then multiplying the raster datasets together would result in a binary suitability raster dataset where pixels with a value of 1 are suitable and areas with a value of 0 are not suitable. However, in reality some variables are more important than others and there may be more preferable ranges within variables that should be included. Of the layers that were generated; depth, texture, bedrock (consisting of bedrock, gravel or cobble presence) and organic matter content were binary. Fetch and suitable slope were standardized to a value between 0 and 1. As was already mentioned, light is considered the most important factor in determining macrophyte distribution because if light is not suitable then macrophytes simply cannot survive. Of the substrate types, bedrock was considered completely unsuitable. However, among the other variables it would be subjective to rank them in importance based on the data currently available. Therefore, an index of suitability was
calculated using all of the variables except depth and bedrock. The other four layers were weighted and summed using a weighted linear combination method to create an index between 0 and 1 using the formula:

\[ I = \frac{\sum w_i x_i}{\sum w_i} \]  

Equation 3.7

where \( I \) is the index value, \( w_i \) is the weight for layer \( i \), and \( x_i \) is the standardized (or binary) value of criterion \( i \) for all layers \( i = 1 \) to \( n \) (modified for this data from Malczewski 2000). All layers were given the equal weight of 0.25. This index does not include depth or bedrock because even if these layers were given a greater weighted value, all other criteria summed together could still be greater than 0 which in reality should not occur. Therefore, to insure that no depths greater than the suitable range and no areas with bedrock as the substratum were selected the binary raster pixel values of 0 for both layers individually were set to null using the Set Null conditional geoprocessing operator and all pixel values of 1 were replaced with the composite index layer pixel values to yield a final indexed suitability layer (Figure 3.10) as well as a binary suitability layer (Figure 3.11) after a reclassification. A schematic of the final model is presented in figure 3.12.

**Results and Discussion**

The purpose of the discussion here is to focus on the usability, limitations and management applications of the model. The original reason for the creation of this model was to have a tool that used scientifically based information to select suitable sites to focus aquatic macrophyte re-establishment efforts. The specific results from each
process of the model for Little Bear Creek Reservoir are presented in Table 3.2 but could be adjusted to any waterbody where the model is applied.

In the past, only subjective methods have been used in site selection and many have been based on convenience rather than environmental suitability for macrophytes (e.g., Grodowitz et al. 2009). Best and others (2008) proposed a combination of models to maximize submerged macrophyte restoration success. Their approach involved complex hydrodynamic and sediment transport models and predicts macrophyte growth using carbon flow models. In addition, they predicted biomass of two species (*V. americana* and *S. pectinata*) with the primary goal of deciding which species had a greater likelihood of survival given a set of environmental conditions. Although these approaches may be used to generate hypotheses (such as the model proposed by van Nes et al. 2003), they are not practical for management application because they require high amounts of location specific field work and data collection. In addition, rarely do these models use technology that is readily available for management or easily understood without specialized training. Finally, although data input for these models is often spatially explicit (e.g., van Nes et al. 2003), there is rarely an output provided that would allow managers to assess the spatial distribution of the results.

Geographic Information System technology provides a solution to many of these problems. The application of GIS to the field of aquatic botany has been warranted for over a decade (Lachavanne et al. 1997). However, GIS modeling approaches for macrophytes thus far have relied on inductive, site specific applications. Similar to models discussed earlier, these approaches are suitable for generating hypotheses and
determining site specific environmental influences on aquatic plant assemblages but lack the necessary next step for management application (i.e., the creation of a deductive model based on scientific data that can be widely applied by managers with minimal tweaks).

The model presented here is a deductive approach using technology and methodology that can be easily used to help make management decisions. Each process was performed using technology that is readily available to management personnel. Given the flexible nature of GIS software, this model also allows simulations of different environmental conditions with only minimal adjustments or data manipulations. The model proposed by Grodowitz and others (2009) has the potential to be applied by managers but the GIS model presented here should be the first step in deciding the preliminary locations necessary for their model.

Future adjustments could be made with the GIS model to simplify data collection and processing even further. One consideration would be to develop a different way for calculating maximum colonization depth. The Secchi disk is sometimes used to define the photic zone of a lentic system. Also, studies have shown a negative correlation between Secchi disk depth and the light extinction coefficient in moderately productive waters. Using this technique would eliminate the need for the use of a PAR meter which may not be available.

Sediment analysis is the most time consuming process of data collection for the model input. One solution for this problem is to identify soils by sight or texture in the field. Data collectors trained in soil analysis might be able to simply take sediment cores
in the field and record the texture. This also might be useful in areas where the within lake sediment has high variability and therefore some sediments may be easily ranked for their macrophyte suitability. Several studies have used aerial photography to digitize soils (e.g., Narumalani et al. 1997, Lehman et al. 1997) but this requires accurate photography of a site before impoundment. Another issue with using historic aerial photography is that soils may change because of erosion, sedimentation or anoxic conditions depending on inundation time (Wetzel 2001) which could compromise information derived from pre-impoundment data. Sediment interpolation also is questionable. Kriging is the most complicated statistical operation in the model. It is likely that different geographic locations may have sediments that are more or less feasible for the use of this interpolation method. However, this is a consideration that needs to be made on a location by location basis and cannot be automated by the model.

As with all models, this one represents a process and not a solution. Validation of the model has not been performed because it is used to predict suitable areas for macrophyte re-establishment efforts based on data obtained in other studies. Furthermore, the study area it has been applied to is lacking in aquatic plant communities thus making validation unattainable. However, this model represents an important step in the management application of spatially explicit predictive models for aquatic systems. At the very least, the model provides a framework for the development of future macrophyte prediction models and can still be used for generating hypotheses.
BIBLIOGRAPHY


Table 3.1  Mean light extinction coefficients ($K_d$), maximum colonization depth ($Z_c$), and secchi disk depth ($Z_{sd}$) for each sampling date in Little Bear Creek Reservoir, Alabama in 2008 and 2009. $K_d$ and $Z_c$ were calculated using equations 3.2 and 3.4, respectively.

<table>
<thead>
<tr>
<th>Date</th>
<th>$K_d$</th>
<th>$Z_c$ (m$^{-1}$)</th>
<th>$Z_{sd}$ (m$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>6/5/2008</td>
<td>*</td>
<td>*</td>
<td>2.18</td>
</tr>
<tr>
<td>6/12/2008</td>
<td>0.935</td>
<td>4.64</td>
<td>2.29</td>
</tr>
<tr>
<td>5/18/2009</td>
<td>0.843</td>
<td>5.15</td>
<td>2.20</td>
</tr>
<tr>
<td>6/2/2009</td>
<td>0.734</td>
<td>5.91</td>
<td>2.07</td>
</tr>
<tr>
<td>6/15/2009</td>
<td>0.931</td>
<td>4.66</td>
<td>1.43</td>
</tr>
<tr>
<td>6/29/2009</td>
<td>0.922</td>
<td>4.71</td>
<td>2.12</td>
</tr>
<tr>
<td>7/27/2009</td>
<td>0.952</td>
<td>4.56</td>
<td>2.25</td>
</tr>
<tr>
<td>8/22/2009</td>
<td>0.580</td>
<td>7.48</td>
<td>2.05</td>
</tr>
<tr>
<td>9/7/2009</td>
<td>0.552</td>
<td>7.86</td>
<td>*</td>
</tr>
</tbody>
</table>

**Mean**  0.806  5.62  2.07

*Data not available*
This map of Little Bear Creek Reservoir, Alabama contains a grid of points spaced 100m-1 apart. This grid was the basis for the original depth survey.
Figure 3.2 This map of Little Bear Creek Reservoir, Alabama is an interpolated raster surface of elevation above mean sea level. The raster surface was derived using the Topo to Raster geoprocessing tool with ArcGIS Spatial Analyst.
Figure 3.3 Areas in Little Bear Creek Reservoir, Alabama with light penetration suitable for rooted macrophyte growth estimated from data collected in 2008 and 2009. Suitable depths were determined using equation 3.4. The raster surface was derived using geoprocessing tools in ArcGIS.
Figure 3.4 Percentage slope in Little Bear Creek Reservoir, Alabama derived from bathymetry data collected in 2008. Slope was calculated using geoprocessing tools in ArcGIS.
Figure 3.5 Areas with suitable slope standardized to a value between 0 and 1 in Little Bear Creek Reservoir, Alabama. Suitable slope was determined using the lesser of a two class geometrical interval classification applied to the slope surface (Figure 3.4). Suitable slope values were standardized using equation 3.5. The raster surface was derived using geoprocessing tools in ArcGIS.
Figure 3.6 Effective fetch distances standardized to a value between 0 and 1 in Little Bear Creek Reservoir, Alabama. Effective fetch was calculated using a USGS model based on equation 3.6. Values were standardized using equation 3.5 using the effective fetch distance value. The raster surface was derived using geoprocessing tools in ArcGIS.
Figure 3.7 Sediment sampling locations in the littoral zone of Little Bear Creek Reservoir, Alabama in 2008.
Figure 3.8  Areas with suitable sediment texture (> 50% Silt + Clay) for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008.
Figure 3.9 Areas with suitable percentage of organic matter content (< 5%) for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008.
Figure 3.10  Final index of suitable areas for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008 calculated using equation 3.7.
Figure 3.11 Final binary map of suitable areas for macrophyte growth in Little Bear Creek Reservoir, Alabama in 2008.
Figure 3.12  Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008. The model was created using ArcGIS Model Builder. Magnified colored sections are presented in figures 3.12a-e.
Figure 3.12a Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the bathymetry surface.
Figure 3.12b  Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the slope and organic matter surfaces and weighted linear combination.
Figure 3.12c  Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the silt, sand, and clay surfaces.
Figure 3.12d  Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of the normalized sediment particle size surface and suitable sediment particle size surface.
Figure 3.12c  Schematic of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama in 2008 showing the generation of suitable and standardized fetch surfaces and the combination of the depth constraint with weighted linear combination results to create a final index of macrophyte suitability surface.
Table 3.2 Results of macrophyte growth GIS suitability model for Little Bear Creek Reservoir, Alabama from data collected in 2008.

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Maximum Colonization Depth (m)</th>
<th>Slope (%)</th>
<th>Fetch (m)</th>
<th>Sediment Particle Size (%)</th>
<th>Organic Matter (%)</th>
<th>Substrate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Suitability Criteria</td>
<td>$&lt; Z_c$ (Mean $Z_c = 5.62$)</td>
<td>Std. (0-1)*</td>
<td>Std. (0-1)</td>
<td>$&gt; 50%$ Clay + Silt</td>
<td>$&lt; 5%$</td>
<td>Bedrock, Gravel, Cobble</td>
</tr>
<tr>
<td>Original Pixel Type</td>
<td>Floating Point</td>
<td>Floating point</td>
<td>Floating Point</td>
<td>Floating Point</td>
<td>Floating Point</td>
<td>String</td>
</tr>
<tr>
<td>New Pixel Type</td>
<td>Integer</td>
<td>Floating point</td>
<td>Floating Point</td>
<td>Floating Point</td>
<td>Integer</td>
<td>Binary (Discrete)</td>
</tr>
<tr>
<td>Layer Type</td>
<td>(within Range)</td>
<td>Continuous</td>
<td>Continuous</td>
<td>(within Range)</td>
<td>(within Range)</td>
<td></td>
</tr>
<tr>
<td>Cell Size (m²)</td>
<td>10</td>
<td>10</td>
<td>10</td>
<td>10</td>
<td>10</td>
<td>100</td>
</tr>
<tr>
<td>Original Range</td>
<td>169 - 189 (MSL)</td>
<td>0 - 98</td>
<td>10 - 1916 m</td>
<td>4 - 94.0 % Clay + Silt</td>
<td>1.3 - 29.1 %</td>
<td>N = 72</td>
</tr>
<tr>
<td>Suitable Range</td>
<td>183 - 189 (MSL)</td>
<td>0 - 16.2</td>
<td>NA</td>
<td>$&gt; 60%$ Clay + Silt</td>
<td>$&lt; 5%$</td>
<td>Absent</td>
</tr>
<tr>
<td>Suitable Area (ha)</td>
<td>292</td>
<td>273</td>
<td>NA</td>
<td>383</td>
<td>466</td>
<td>7.2</td>
</tr>
<tr>
<td>Percent Area</td>
<td>49.7</td>
<td>46.5</td>
<td>NA</td>
<td>65.2</td>
<td>79.3</td>
<td>1.2</td>
</tr>
<tr>
<td>Index Weight</td>
<td>NA</td>
<td>0.25</td>
<td>NA</td>
<td>0.25</td>
<td>0.25</td>
<td>NA</td>
</tr>
</tbody>
</table>

**Total Suitable Area** 124.4

**Percentage Area** 21.2

* First classified into two classes using a geometric interval algorithm. The lesser class was selected and classified.
Aquatic macrophytes are important components of freshwater ecosystems. They serve numerous physical and biological purposes that structure aquatic communities (Carpenter and Lodge 1986). In systems that lack native aquatic macrophyte assemblages, re-establishment may be a management technique used to improve system quality (Smart et al. 1998, Smiley and Dibble 2006). For successful propagation, native plants should be planted in areas suitable for plant growth. Selecting these sites requires knowledge of the system as well as knowledge of the factors that regulate aquatic macrophyte growth.

Little Bear Creek Reservoir is lacking in submersed or floating leaved macrophyte assemblages. This makes it a good candidate for native macrophyte re-establishment efforts. Efforts so far have been successful in establishing *P. nodosus* inside protective exclosures. However, future management efforts should focus on encouraging the expansion of macrophytes beyond protective exclosures. Considerations should also be made for additional species testing.

Predictive ability is a fundamental goal of scientists, particularly in the study of community ecology (Dodds 2009). One of the objectives of this study was to build a spatial model to assist in locating areas that were optimal for plant growth. This was
developed on the basis that suitable areas could be predicted by selecting areas that contained habitats that were conducive to macrophyte growth. However, there are other management uses for the model that may transcend macrophyte re-establishment such as predicting plant biomass, water body surface coverage, or plant surface area within a waterbody. Biomass is generally measured on small scales but extrapolated to large scales to determine waterscape wide predictions (Cheruvelil and Soranno 2008). With this in mind, if species specific biomass is predictable based on data derived from small scale samples, then predictions should be possible on larger scales by using predicted occurrence data.

There also are opportunities to use this model for the prediction of macrophytes that are not desirable. In a waterbody that already has plants the model can be validated or used to predict invasion risk in areas lacking plants within the water body. The model also can be used to develop a landscape scale macrophyte index that can be compared not only within a given lake, but between lakes as well. For example, some have postulated the use of macrophyte surface coverage in a water body for predicting impacts on fish (Killgore et al. 2008). This may be a predictor of the benefits or risks of macrophytes but predicting the impacts of macrophytes on fish is likely scale dependent (Dibble et al. 2006). The use in this case could be to develop an index in which reservoirs of different size, shape, or dimension can be used in comparative analysis.

Additionally, modeling on a macroscale can potentially be used to predict waterscape wide impacts of macrophytes. Not only can the model be used to predict occurrence or biomass, it may have potential as part of the development of equations to
make quantitative predictions for invertebrate and fish communities. Many authors have studied the potential to predict invertebrate abundance, diversity or biomass based on macrophyte biomass, surface area, or complexity (Jeffries 1993, Thomaz et al. 2008, Cheruvelil et al. 2002). Some of these studies have obtained significant results with the measured variables (Beckett et al. 1992) and others have provided methods for ensuring statistically significant results (Cheruvelil et al. 2000). The ability to make accurate prediction also may be macrophyte species specific, suggesting invertebrate colonization may depend on different variables for different species, or that there are additional variables that have not been tested (Cyr and Downing 1988). By assessing this for specific locales with species native to a particular area, it may be possible to make more specific impacts of macrophyte re-establishment, introduction, or current occupation on fish communities.

For example, Beckett and others (1992) performed experiments to test if the surface area of *P. nodosus* was correlated with abundance of invertebrate taxa and found a significant positive relationship. Of the invertebrates they collected, chironomid larvae composed an average of 38.3%. Chironomid larvae are a common and important forage food for juvenile bluegill (Spotte 2007) and therefore an increase in chironomid abundance with increasing plant surface area can be viewed as an increase in prey forage potential for these and other fish species.

Samples of *P. nodosus* plants collected in Little Bear Creek Reservoir using a 15cm$^3$ box sampler and digitized using Sigma Scan Pro 5 had a mean surface area (stems and leaves) of 272.5cm$^2$ (± 145.0cm$^2$; N = 45) for that volume of water (15cm$^3$). Using
the results from Beckett and others (1992) which found an average of 0.62 chironomids per cm\(^2\) plant surface area, this translates to an average potential chironomid abundance of 169.0 individuals/15cm\(^3\) (50,000 individuals/m\(^3\)) of water containing \(P.\ nodosus\) plants in Little Bear Creek Reservoir. Energetically, chironomids provide a value of approximately 5420 Cal/gm dry mass (22,700 joules/gm dry mass; Cummins and Wuychuck 1971). Therefore, if number of chironomids needed to constitute 1gm dry mass is known, a prediction can be made for the potential energy density provided by chironomids living on \(P.\ nodosus\) to bluegill, largemouth bass, or other fish.

Taking this approach one step further, the GIS model that predicts areas suitable for plants to grow can be used to calculate volume of water for these areas by adding water depth values of pixels that intersect suitable plant growth areas. This information enables managers to predict potential plant surface area, chironomid abundance, and caloric value. Incorporating potential caloric energy densities into this spatial framework demonstrates the flexibility, scalability, and topological advantages of using GIS for management predictions.

GIS suitability models should represent a process, not a solution or an answer. Therefore, location specific variables should be assessed. However, for Little Bear Creek Reservoir, locations indicated with the model should be used to indicate areas for additional planting effort and potentially for predicting other impacts to the aquatic environment. Based on the results of the model there is little risk from plant colonization that would be problematic for recreational uses of the water body. However, aquatic
vegetation could provide benefits that improve the quality and production of the overall fishery and aquatic ecosystem.
BIBLIOGRAPHY


