United States
Department of
Agriculture

Forest Service

**Southern
Research Station**

General Technical
Report SRS–50

Proceedings of a Conference on Sustainability of Wetlands and Water Resources:

How Well Can Riverine Wetlands Continue to Support Society into the 21st Century?

**Oxford, Mississippi
May 23–36, 2000**

Cover: Aerial photograph of experimental ponds and mesocosms at The University of Mississippi Field Station (western portion).

Photo by Kevin R. Pigott

DISCLAIMER

PESTICIDE PRECAUTIONARY STATEMENT

This publication reports research involving pesticides. It does not contain recommendations for their use, nor does it imply that the uses discussed here have been registered. All uses of pesticides must be registered by appropriate State and/or Federal agencies before they can be recommended.

CAUTION: Pesticides can be injurious to humans, domestic animals, desirable plants, and fish or other wildlife if they are not handled or applied properly. Use all herbicides selectively and carefully. Follow recommended practices for the disposal of surplus pesticides and their containers.

Papers published in these proceedings were submitted by authors in electronic media. Editing was done to ensure a consistent format. Authors are responsible for content and accuracy of their individual papers.

March 2002

Southern Research Station
P.O. Box 2680
Asheville, NC 28802

Proceedings of a Conference on Sustainability of Wetlands and Water Resources:

How Well Can Riverine Wetlands Continue to Support Society into the 21st Century?

Edited by

Marjorie M. Holland, Melvin L. Warren, and John A. Stanturf

Oxford, Mississippi
May 23–25, 2000

Hosted by

The University of Mississippi

Sponsored by

The University of Mississippi
USDA Forest Service, Southern Research Station

Table of Contents

ACKNOWLEDGMENTS

We would like to thank all of the organizations that were cosponsors of the conference: The Croft Institute for International Studies at The University of Mississippi; The Fertilizer Institute; The Henry L. & Grace Doherty Charitable Foundation, USDA-Agriculture Research Service, National Sedimentation Laboratory; USDA Forest Service, Department of Bottomland Hardwood Research; The Phil Hardin Foundation; The National Center for Natural Products Research; and The National Science Foundation.

We would like to thank all of the organizations that made contributions to the conference: Department of Biology, The University of Mississippi; Fisher Scientific; Sigma Xi, The University of Mississippi Chapter; Society of Wetland Scientists (SWS), National Board; Society of Wetland Scientists, South Central Chapter; USDA Natural Resources Conservation Service, Wildlife Habitat Management Institute; USDA Natural Resources Conservation Service, Wetland Science Institute.

We would like to thank all of the organizations that endorsed the conference: International Association of Theoretical and Applied Limnology; The Ecological Society of America; The Lower Mississippi River Conservation Committee; The Mississippi River Foundation; American Fisheries Society; Society of Environmental Toxicology and Chemistry (SETAC).

We would like to thank the Conference Organizing Committee: Dr. David Arnold, Mississippi Chemical Corporation; Dr. Elizabeth Blood, Jones Ecological Research Center; Dr. W. Franklin Harris, Oak Ridge National Laboratory; Dr. Marjorie M. Holland, The University of Mississippi; Dr. Michael Metcalf, Croft Institute for International Studies; Dr. Gary Miller, The University of Mississippi; Dr. Andy Mullins, The University of Mississippi; Ms. Kathy Sukanek, The University of Mississippi.

We would like to thank the members of the Conference Planning Committee: Dr. Nolan Aughenbaugh, The University of Mississippi; Dr. William Benson, U.S. Environmental Protection Agency; Dr. Alice Clark, NCNPR, The University of Mississippi; Dr. Charles Cooper, USDA-ARS National Sedimentation Laboratory; Dr. Gary Gaston, The University of Mississippi; Dr. Marjorie M. Holland, Chair, The University of Mississippi; Mr. Gerald Inmon, USDA Forest Service (retired); Ms. Janet Jenkins, Bruce High School; Dr. Sammy King, U.S. Geological Survey; Dr. Barbara Kleiss, U.S. Geological Survey; Dr. Al Mikell, The University of Mississippi; Dr. Reza Pezeshki, The University of Memphis; Dr. Catherine Pringle, The University of Georgia; Mr. Paul Rodrigue, USDA-NRCS; Dr. Matt Romkens, USDA-ARS, National Sedimentation Laboratory; Dr. Larry Shaffer, The University of Mississippi; Mr. Duane Shaw, MDWFP; Dr. Jean Shaw, The University of Mississippi; Dr. Stephen Threlkeld, The University of Mississippi; Dr. Melvin Warren, USDA Forest Service; Dr. Robert Wetzel, The University of Alabama; Mr. Darryail Whittington, MS DEQ; Dr. Robert Woolsey, The University of Mississippi; Dr. Joy Zedler, The University of Wisconsin.

Last, but certainly not least, special thanks go to Keith Wright, a student assistant with the UM Field Station, who devoted considerable talent and skill to formatting and reviewing the proceedings manuscripts.

HUMAN ALTERATIONS OF EARTH'S FRESH WATER: SCALE, CONSEQUENCES, AND A CALL TO ACTION

Sandra Postel[1]

Abstract—One of the biggest challenges society faces in this new century is figuring out how to satisfy the water demands of 8 billion people while at the same time protecting the aquatic ecosystems and ecological services that humans and all species depend upon. Since 1950, water demands worldwide have more than tripled, while the scale of our dams and reservoirs, river diversions, and groundwater exploitation have fundamentally altered hydrological systems and the ecological services they perform. On top of all these stresses, we face the added complexity of global climate change. We are going to need a fundamentally different approach if we are going to have any hope of maintaining some degree of ecological health in the face of these rising demographic and economic pressures. This paper addresses some of the goals and key policy levers necessary for wetland ecosystem restoration and protection. This paper is a transcription of Sandra Postel's plenary talk at the May 2000 Conference.

INTRODUCTION

One of the biggest challenges society faces in this new century is figuring out how to satisfy the water demands of eight billion people while at the same time protecting the aquatic ecosystems and ecological services that humans and all species depend upon. This paper explains why I think this is the case, and explores the critical role of science and scientists in helping meet this challenge.

We have an unsettling degree of uncertainty when it comes to just about all aspects of the world's water—from the natural hydrological cycle itself to the amount of water we have stored in reservoirs, to how much groundwater is stored in aquifers, to how much water humanity currently uses. Most of the numbers discussed below are best estimates rather than precise facts. There's no doubt that we need a much better scientific understanding of many aspects of freshwater. But we know enough to say unequivocally that over the last century, human activities have altered the hydrologic cycle and hydrologic regimes in unprecedented ways and on a global scale—and the costs and consequences of these impacts have just begun to come to light.

Since 1950, water demands worldwide have more than tripled. We now remove from the earth's rivers, streams, lakes, and aquifers about 4,000 cubic kilometers of water per year. Irrigated agriculture has been by far and away the biggest driver behind this rise in water use. Irrigated land worldwide has nearly tripled over the last half century, climbing from 100 million hectares to more than 270 million hectares today. Agriculture now accounts for about 70 percent of world water use, industries for about 20 percent, and cities and towns for about 10 percent.

At first glance, it might seem that at current levels of water use we're still in reasonably good shape. Our current use of 4,000 cubic kilometers represents only about 10 percent of the world's total renewable runoff—the water that annually flows back toward the sea via rivers and aquifers. But if we look more carefully at how much of that river and groundwater flow is actually accessible to us—and therefore can serve as a supply for agriculture, industry, and cities—we find that the situation isn't at all comfortable. Only about 31 percent of global runoff can be tapped where and when we can use it. And of this, we already appropriate about half to meet current human needs—which suggests that global limits are closer than we'd previously realized. The only way to increase the accessible runoff is to capture and store more flood water, which typically means constructing more dams. So we face a kind of Faustian bargain with Nature: In order to meet future human needs, it seems, we need to shift an even larger portion of the world's water from serving Nature's purposes to serving humanity's purposes. And I will come back to this dilemma.

But first, a few reality checks. If we are anywhere close to hitting global limits, we should be seeing physical signs of water stress and unsustainable uses in many freshwater systems and across fairly wide geographic regions. So let's look at what is happening.

Rivers

We have seen the number of large dams (those at least 15 meters high) climb from 5,000 in 1950 to more than 40,000 today. We have a less good count of small dams, but they number somewhere in the 800,000 range. The reservoirs behind these dams are capable of storing nearly 20 percent of total annual global runoff, and as is often the case, this global figure masks great variation and extremes among individual river systems. Lake Nassar, behind Egypt's Aswan Dam, is able to fully store two years worth of the Nile's flow, for example.

Recent surveys suggest that nearly 60 percent of the world's largest 237 rivers are moderately to strongly fragmented by

[1] Director, Global Water Policy Project, 107 Larkspur Dr., Amherst, MA 01002-3440

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

dams and diversions. Perhaps more dramatically, we're now seeing that many major rivers are tapped out during the dry season, with virtually no freshwater reaching the sea for months at a time. These include five of Asia's great rivers—the Ganges and Indus Rivers in South Asia, the Amu Darya and Syr Darya Rivers in Central Asia, and the Yellow River in China. The Yellow River first ran dry in 1972. But the situation has worsened greatly over the last 15 years. I remember traveling by train in China in 1988, and awakening early so as not to miss crossing the Yellow River, and being stunned by how little water this great river—the cradle of Chinese civilization-was actually carrying. In 1997 the Yellow ran dry for a record 226 days. Closer to home, the Colorado River doesn't reach the sea at all in an average year, and that's been true more or less since completion of Glen Canyon Dam in the early 1960s.

Groundwater

Groundwater, which forms the base flow of rivers, has come under heavy pressure in the last 50 years as well. Prior to WWII, we just didn't have the technical capabilities of pumping groundwater on a vast scale. But with the advent of powerful pumps and deep drilling technologies, farmers and other water users began to tap groundwater on a historically unprecedented scale. In India, the number of groundwater wells climbed from 4 million in 1951 to 17 million in 1997. In China, the number of irrigation wells has climbed 20-fold over the last three decades. Here in the US, farmers drilled millions of wells into the Ogallala Aquifer—one of the planet's greatest underground water reserves, and which now waters 20 percent of U.S. irrigated land.

The problem is that much of this groundwater use is not sustainable. In just about every area of intensive groundwater-based agriculture, water tables are dropping steadily because pumping exceeds recharge. This is the case in the Punjab of India, the north plain of China, the U.S. Great Plains, California's Central Valley, and much of North Africa and the Middle East. This is a big red flag for future food security, since as much as 5-10 percent of the world's food production may now depend on the overpumping of groundwater. And it's also a great ecological threat as the lowering of water tables dries up springs and wetlands—and, in some cases, turns perennial rivers into seasonal ones by eliminating their base flow.

Together, the scale of our dams and reservoirs, river diversions, and groundwater exploitation have fundamentally altered hydrological systems and the ecological services they perform. It's estimated that half of the world's wetlands have been lost during the 20th century—in part due to direct conversion to agricultural and urban land uses, but also due to the cutting off of rivers from their flood plains and their deltas, and to the overpumping of groundwater.

In many countries, 10-30 percent of freshwater fish are now threatened with extinction. Here in the United States, The Nature Conservancy reports that 38 percent of freshwater fish are at risk, along with 69 percent of freshwater mussels, and 51 percent of crayfish. The U.S. ranks first in the world in the number of known species of freshwater mussels and crayfish and 7th in the number of known freshwater fish species. Freshwater species are proportionately at greater risk than others, and the principal reason for their imperilment is the destruction and degradation of aquatic habitats.

We could, if we had time, run down a long list of aquatic ecosystems on every continent that are in states of rapid decline—from the most dramatic case of the Aral Sea in central Asia, which used to be the world's fourth largest lake, to the Everglades of south Florida, the Danube Delta of Europe, the Ganges and Indus deltas of south Asia, to entire river systems like the Colorado, the Missouri, the Nile, and the Rhine.

On top of all these stresses, we face the added complexity of global climate change. The patterns of river runoff we have measured during the 20th century will almost certainly not be good guides for our planning in the 21st century. Rainfall patterns are expected to shift and droughts and floods to intensify. Many of the world's major rivers are fed by mountain snowpacks, which are fantastic natural reservoirs. They store water through the winter and release it during the spring and summer. As temperatures warm, more of that winter precipitation will fall as rain rather than snow, and the snowpack will melt earlier and faster. In many places, this will mean more intense flooding in the early spring and lower flows during the summer, when water demands are highest.

So we have a large backlog of water problems to confront, even as pressures on water systems continue to increase. Just to supply the food demands of 2025, we could need to find an additional 800 cubic kilometers of irrigation water - a volume nearly equal to the annual flow of ten Nile Rivers.

So I return to our Faustian bargain: We can continue to extract more water from Nature in order to meet rising water demands, but in doing so we place in jeopardy the survival of major pieces of the aquatic world, pieces we depend on for a host of ecological services worth—no one knows how much—but almost certainly in the hundreds of billions if not trillions of dollars annually.

I would submit that there is no winning scenario we can create out of these current trends in water use and management. We are going to need a fundamentally different approach if we are going to have any hope of maintaining some degree of ecological health in the face of these rising demographic and economic pressures.

What are some of the big priorities?

First, I believe we need a multi-disciplinary and cross-professional effort to systematically determine the quantity, quality, and timing of flows needed for freshwater ecosystems to sustain their critical functions. Without an effort such as this, no country or region can answer such basic questions as: how much water is available to meet human needs in a sustainable fashion? Or, how must dams be operated in order to sustain critical ecosystem functions and to protect biodiversity? This initiative would have to be scientifically credible but policy-focussed-that is, produce a strategy that can be implemented. South Africa is the only country I know of that has adopted this kind of goal as a matter of national policy. South Africa's new water policy states that, "The quantity, quality, and reliability of water required to maintain the ecological functions on which humans depend should be reserved so that the human use of water does not individually or cumulatively compromise the long term sustainability of aquatic and associated ecosystems.” Many of the aquatic scientists in South Africa are now involved in the effort to determine these “environmental water reserves” in some way. Of course it remains to be seen how effectively this policy will be carried out.

Another bold step has been taken in Australia’s Murray-Darling river basin, where the Commission has placed a Cap on total water extractions from the basin. The rationale here is simply that the river system has already sustained too much ecological harm, and so no additional water diversions should take place. New demands for water in the region are to be met through conservation and water trading rather than by increasing the supply.

Here in the United States, we have had several calls for major initiatives to generate the knowledge-base needed for sound freshwater management and protection of ecosystems. Bob Naiman, Diane McKnight, Jim Karr and others wrote eloquently of the need for a national initiative of this sort in an article in Science in 1995. These and other researchers have written extensively about the geomorphic, hydrologic, and biological parameters that would form the core of this knowledge base for freshwater ecosystem protection and restoration.

I think the time is right to push harder for an initiative such as this, because much has changed in the last five years. When the Edwards Dam on the Kennebec River in Maine was demolished in the summer of 1999, it was a very public symbol and signal—and Secretary Babbitt said this—that dams are not forever. The idea that we’d be having a serious discussion about breaching major dams in the Columbia River basin—four on the lower Snake—would probably have seemed outlandish even 5-10 years ago.

These debates signal a major shift in public attitudes that provides new opportunities. Many more small dams will likely come down over the next few decades, but probably more important is the possibility of rethinking the operation of dams that will remain standing. To what degree can we mimic a river’s natural hydrograph and still meet a portion of the economic uses for which a given dam was built? What will be the ecological benefits of doing this? Marginal tradeoffs like this will be made, and these collectively may be more important than the big-ticket items like a few big dams coming down.

What are the key policy levers for ecosystem restoration and protection that we’re likely to see greater use of?

- FERC relicensing: of private hydro dams (e.g., Edwards Dam)
- Endangered Species Act—which, if invoked in its full force, would dramatically alter water use and river management nationwide.
- Instream flow requirements—On a case by case basis, these are beginning to get clarified. Judges are making decisions about what “impairment” of a river system means in order to determine what uses and extractions of the river are reasonable and acceptable. This is not just happening in the West, but in the East as well. Very interesting case of the Shepaug River in western Connecticut decided in February 2000.
- Public Trust Doctrine—applied in a historically new and much broader way in the case involving Mono Lake in California. It is not yet clear whether this will be a precedent for broader applications.

Unless we have the knowledge base to assist in these legal, regulatory, and management debates, and are able and willing to translate that knowledge base to policymakers and water managers, we will miss big opportunities for conservation and restoration of aquatic ecosystems.

Second, we need an all-out effort in every sector of the economy to raise water productivity. We simply cannot hope to achieve our ecosystem protection and restoration goals if we do not promote more efficient use of water. I believe that with an all-out effort to promote more efficient and equitable use of water worldwide, we could satisfy year-2025 water demands without extracting much more water from the natural environment than we do today. That’s an exciting prospect-but we’re a long way from getting there.

I once heard it said that “Technology is Nature’s experiment with Man.” We need to get smarter about the technologies we employ and how we employ them—whether it’s how we control floods, how we irrigate our crops, or how we operate our dams.

To do this we need to build new research and management partnerships, especially ones that bridge ecology, engineering, and economics. Some of the best attempts I’ve seen to evaluate ecosystem services have been done through interdisciplinary and/or cross-professional collaborations. We need more efforts not just to think out of our box, but to come out of our box. Even applied science is not getting used in making good policy and management decisions because too few scientists are bridging the divide to the policy world.

Finally, I also believe that, as scientists and citizens, we can contribute more to the resolution of our water problems by advocating more forcefully for the adoption of a guiding water ethic in society. The fact that water is the basis of life lends an ethical dimension to every decision that’s made about how to use and manage it. Especially in the face of scientific uncertainty and potentially irreversible change, appealing to an ethic of protection and preservation of freshwater ecosystems can be both justified and compelling.

Eleanor Roosevelt once said, “We should constantly be reminded of what we owe in return for what we have.” I would submit that what we have is more knowledge about the health of our aquatic ecosystems and the threats to them than the vast majority of people on the planet, and that what we owe to society in return is the expression of our best judgement—not perfect judgement, but best judgement—of how to preserve and protect these critical ecological assets.

INSTITUTIONAL CHANGE AND CAMPUS GREENING AT TULANE UNIVERSITY

Aaron S. Allen[1]

Abstract—A case study of Tulane University that examines the institutional change process is presented in this paper. Agents of change can use the examples and conclusions as a basis for making changes at any institution. The inability for Tulane to make the campus environmentally sustainable in terms of operations and education was due to the lack of an institutionalized internal lobbyist and leader dedicated to environmental issues. That argument is supported with a model for institutional change, a historical analysis of nonenvironmental and environmental change initiatives at Tulane, a review of campus greening programs in institutions of higher education, and a series of interviews with Tulane students and employees. In the summer of 1999, as a result of an earlier version of this study,[2] an Office of Environmental Affairs (OEA) was created according to the "Blueprint for a Green Tulane," which outlined the steps necessary for institutional environmental change to occur. The central component of that change is leadership from the OEA's Environmental Coordinator and from students who will, in turn, carry their leadership beyond the campus to create a more sustainable world.

INTRODUCTION

"Greening the campus" means increasing environmental awareness or action or both on campus—in the operational facilities and processes of the campus as well as in the human communities of the campus and surrounding areas. Greening the campus involves working towards some or all of the goals set forth in the Blueprint for a Green Campus.[3] Although the fundamental theme of greening is education, this study focuses on campus operations, the greening of which is pedagogical process itself.

The economics of campus environmental initiatives in higher education are well documented: greening the campus saves money. Twenty-three conservation initiatives at fifteen U.S. institutions of higher education each saved between $1,000 and $9 million, with total annual savings at $16.8 million (Eagan and Keniry 1998).[4] Investing in campus greening is therefore an economic, educational, and environmental investment with handsome returns—both financial and social.

In addition to saving money, campus greening allows students to learn how to infuse environmental sustainability into the larger society. Students must be able to practice (and see the university practice) the lessons of environmental sustainability, which they are taught in the classroom. Tulane has committed to environmental studies along with three other areas of interdisciplinary interest: urban studies, international studies, and information technology. Together, the four are conducive to environmental responsibility and stewardship.

Tulane is located in New Orleans, LA, the southernmost port on the 2,552-mile-long Mississippi River. The Mississippi River Basin drains 41 percent of the landmass of the continental United States. The river is the dominant feature of New Orleans, and the university has designed research agendas and teaching curricula around it. Tulane was established in 1834 as the Medical University of Louisiana to study and treat "the peculiar diseases which prevail in this part of the Union" (Tulane University 1997). The university is now comprised of 11 academic divisions with approximately 6,500 undergraduates, 4,800 graduate students, and 8,000 employees, of which approximately 1,750 are full- or part-time faculty. Tulane is responsible for approximately 24,000 jobs in Louisiana and an annual injection of nearly $1.5 billion into the local economy (Strecker 1998). With its location on the Mississippi River, traditional focus on health, and impact on the local economy, Tulane has a formidable presence in the Southern United States.

INSTITUTIONAL CHANGE

A Model for Institutional Change

Figure 1 is the model of institutional change. It is derived from a literature review of institutional change in higher education.[5] Additionally, case studies in nonenvironmental

---

[1] Environmental Studies Program, 201 Alcee Fortier Hall, Tulane University, New Orleans, LA 70118.

[2] This article is excerpted from a larger study by the author, Greening the Campus: Institutional Environmental Change at Tulane University (1999). It is available on the Internet at www.tulane.edu/~env_stud/greening.htm .

[3] The Blueprint (1995) mentions ten items: integrating environmental knowledge into all relevant disciplines; improving environmental course offerings; providing opportunities to study campus and local environmental issues; conducting a campus environmental audit; purchasing environmentally responsible products; reducing campus waste; maximizing energy efficiency; making environmental sustainability a top priority in campus land-use, transportation, and building planning; establishing a student environmental center; and supporting students who seek environmentally responsible careers.

[4] The summary of financial data from Eagan and Keniry (1998) is available at www.nwf.org/nwf/campus/tools/publications/gigr/cost.html .

[5] The complete literature review is in the Greening the Campus study.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

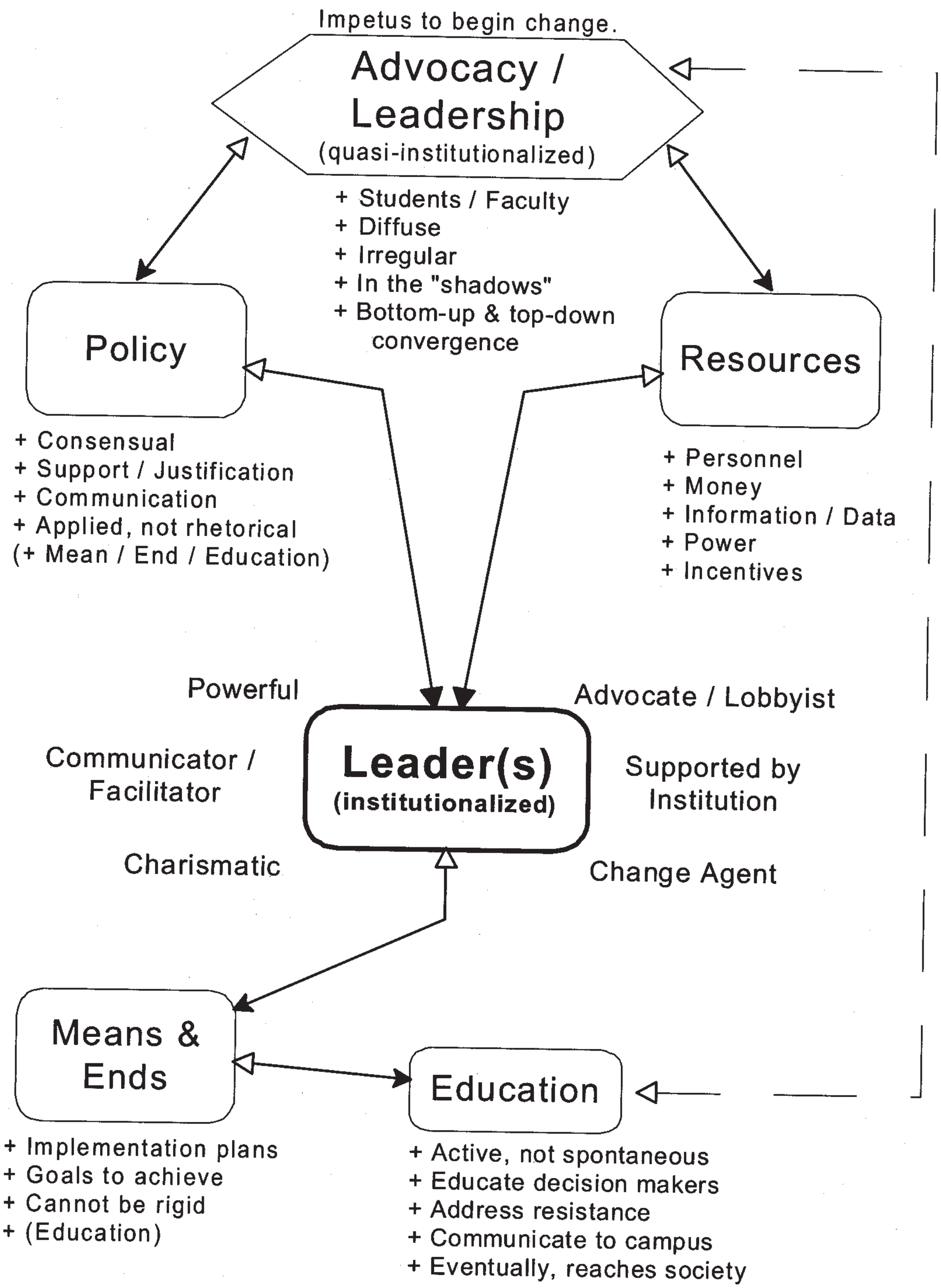


Figure 1—Schematic of the model for institutional change.

and environmental change at Tulane and in academia support the model, which itself is not restricted to environmental change in any way. The key element is a leader who is an administrator or faculty member but not a student because students lack power and connections and are temporary. (Students, however, do play absolutely integral roles in the change process, as discussed in a later section.) In addition to the leader, leadership from the administration is necessary to support the change agenda.

The model is a conceptual framework for understanding and implementing change. It is dynamic: the dark arrows represent normal “flow,” whereas open arrows represent feedback. The model is dynamic not only in itself but also between applications; different circumstances result in different paths. For example, education (the “end”) may result in further advocacy for new changes (thus the dotted line, effectively making the model cyclical); also, procuring policy may return the advocates to the advocacy stage before getting resources. The model is not rigid; for example, policy may be skipped entirely, but the results of the change may not be permanent. Dividing the change process into the segments of the model is artificial but necessary; institutional change is not spontaneous, and greater understanding of the process will increase the likelihood of success for change movements.

Advocacy is the impetus to begin change. It is the product of diffuse, irregular efforts of (primarily) students and faculty found in the “shadow” of the university—the area outside of the “mainstream” of campus life and separate from the traditional governing structures of the institution (Bowers 1997, Mansfield 1998). Advocacy is usually a grassroots or bottom-up effort, but top-down advocacy is just as important: the two converge in the middle to create the integrated advocacy required for institutional change.

Advocacy results in policy. Development of specific and general policies should be consensual with the input of all appropriate parties. Policies should be applicable, enforceable, and nonrhetorical in order to support, justify, and communicate the change goals. Additionally, policy development and having policies in place are forms of education (a means and end) about the change agenda (Altbach 1974, Cerych and Sabatier 1986, Creighton 1998, Fantini 1981, Gitell 1981, Hamburg and Ask 1992, Keniry 1995, Lane 1990, MacTaggart 1996, Smith 1993, Strauss 1996).

Advocacy and policy procure resources. Roughly prioritized, the primary resources are personnel (a leader, support staff, an office), financial resources, information and data, power (or direct access to power), and the ability to offer opportunities and incentives for improvement and positive change. Resource allocations should be in line with the missions of the institution, and a continual supply of necessary resources will maintain the desired changes (Altbach 1974, Cerych and Sabatier 1986, Creighton 1998, Dominick 1990, Fantini 1981, Gitell 1981, Hamburg and Ask 1992, Keniry 1995, Lane 1990, MacTaggart 1996, Smith 1993, Strauss 1996).

Leadership is the key and defining element of the model for institutional change. Advocacy procures the leader, who is supported with policy and resources. The leader is in an institutionalized position dedicated to the change agenda. He or she is the change agent: the communicator and facilitator of the change process, the advocate and lobbyist for the change agenda. The leader needs power or direct access to power. The institution—especially the administration, which should also act as leaders for change—must support the leader. Finally, the leader should be charismatic. Important character traits include communication, interpersonal and listening skills, visionary planning, and the capability to accomplish meaningful projects (Berry and Gordon 1993, Creighton 1998) (see also Altbach 1974; Cerych and Sabatier 1986; Creighton 1998; Dolence and Norris 1995; Dominick 1990; Fantini 1981; Farmer 1990; Gitell 1981; Hamburg and Ask 1992; Keniry 1995; Lane 1990; MacTaggart 1996; Orr 1990, 1992, 1994, 1995, 1996; Rainsford 1990; Smith 1993; Strauss 1996; Wood 1990).

While the leader is the key element to the model, it is also the place for the tragic flaw: how one person can do everything. Some solutions include having other leaders and/ or support staff, or, as discussed below, having a guiding committee and involving students in the change process.

The leader develops well-defined means to achieve agreed-upon ends. Neither the means nor the ends can be rigid. Means are the implementation plans; they are many and specific, and they address education and process re-engineering (physical and administrative). Ends are goals; they are few and broad in scope. Examples of ends might be ecological literacy of graduates and an environmentally sustainable campus (Alinsky 1971) (see also Altbach 1974, Dolence and Norris 1995, Eagan and Orr 1992, Farmer 1990, Keniry 1995, Lane 1990, Smith 1993, Wood 1990).

Education is the primary means and end. Campus decision-makers must be educated on the mechanics of the means and ends of the change agenda. The same issues should be communicated to the entire campus since education about the change agenda is not spontaneous, e.g., the campus community must be educated on the mechanics of a recycling program or the larger goals of environmental sustainability. Eventually, the education reaches society, and such is the ultimate goal for environmental change in higher education (Ackerman 1997; Altbach 1974; Brown and Duguid 1996; De Young 1986; Dolence and Norris 1995; Gitell 1981; Keniry 1995; MacTaggart 1996; Orr 1992, 1994; Smith 1993).

Some theory ties together the model for institutional change in higher education. Change does not happen spontaneously (Ackerman 1997, Bowers 1997, De Young 1986, Williams 1991). The changes pursued must be realistic. They will take time to achieve and will never be 100 percent complete (Cerych and Sabatier 1986, MacTaggart 1996, Steeples 1990). Operational changes affect some people significantly, whereas most are affected only minimally; transformation, not revolution, is needed. A two-dimensional framework of change is appropriate for Tulane: depth is the degree to which a change requires a departure from existing values and practices, and breadth is the number of areas within the institution that a change is expected to introduce modifications. Wide/deep changes result in opposition, and narrow/shallow changes do not take hold. Changes are most likely to succeed when they are moderate in depth and breadth of change (Cerych and Sabatier 1986). Institutional environmental change with regard to campus operations is moderate change (Hamburg and Ask 1992).

Nonenvironmental and Environmental Institutional Change at Tulane

Six case studies of change initiatives at Tulane show that moderate and profound changes are possible—given an empowered leader (or leaders) with resources and policy who introduces means and ends to implement change.

Multicultural Affairs, Bisexual, Gay, and Lesbian Affairs, and Tulane College Programming show that it is necessary to establish offices responsible for oversight and implementation of changes. Advocacy began the establishment of all three, and all established policies and procured resources; then institutionalized leaders implemented educational programs (means) to achieve broad goals (ends). Two other reforms were more ambitious in their scope: Tulane 2000 sought to stabilize the university's budget (and subsequently focus the institution's academic priorities) with cutbacks and resource reallocations; the University Transformation Program sought to improve the quality of staff services and classrooms, create an extracurricular program for first-year students, institute an information technology helpdesk, and establish an international studies office. Both initiatives had a leader (the president and the provost, respectively) and resources to develop and implement policy to affect change. People did not immediately embrace these issues (they were not spontaneous); advocates and leaders convinced the campus that they were meaningful changes. For example, Tulane's management takeover of the Housing Authority of New Orleans was not spontaneous—the leader who initiated the project believed that Tulane's involvement was appropriate and in the best interests of HANO, Tulane, and the citizens of New Orleans.

The necessary elements of achieving change characterize these preceding examples, and most fit into the strategic goals of the university (urban studies, international studies, information technology, and environmental studies). Missing from these six initiatives, however, is a concerted effort to make Tulane more environmentally responsible. Whereas environmental research and education have improved (largely due to grant funding), the third and critical element of a green institution of higher education—operations— has not been greened. The three divisions of the university are research, education, and operations, and, at Tulane, each has been greened to some extent.

Environmental research has been the most successful division. It is a popular area because of the income associated with research grants and the opportunities for publishing. Also, quasi-policy (the environmental studies focus) and resources (multimillion dollar grants) led the development of extensive environmental research programs. The leadership of Dr. John McLachlan of the Center for Bioenvironmental Research at Tulane and Xavier Universities (CBR) has developed, coordinated, and maintained environmental research program opportunities. The research division received a subjective grade[6] of "A-" in the spirit of the Green Gradecard for the Green Wave environmental audit (discussed in a later section).

Tulane's Environmental Studies Program (ENST) has a history that epitomizes how institutional change occurs. In the early 1970s, students lobbied for the creation of the ENST, but the resulting coordinate major program (in which students major in another field in addition to environmental studies) stagnated until the early 1990s because the program was not allocated a budget and had only the devotion of one professor, who was not compensated for his involvement. As a result of the then new environmental studies focus of the university, the program progressed: new faculty became involved and established an environmental education committee, and grant monies provided the resources to offer course development grants, purchase equipment, hold training seminars,[7] and hire a part-time program coordinator. As a result, the program prospered, and enrollment increased dramatically. But the faculty leading the program could not dedicate enough of their professional time to the program; they treated it as if it were a University Senate committee. The student environmental organization, the Green Club, worked cooperatively with the ENST on numerous projects, including the Environmental Forum Newsletter, campus environmental email lists, and the design and publication of the Enviro Counter Culture Catalog, a guide to environmental classes at Tulane, which has received wide acclaim from within and outside of the university.[8] In 1998, the grants ended and the university did not provide a budget for the ENST and its more than 50 students. The CBR stepped in to fund the program, but that funding is also from grants. The future of the environmental education program at Tulane is in question because of the lack of institutional support (a budget). The program is still directed by faculty members who receive no compensation or official credit for their time. Whereas the ENST has potential to be a top program at Tulane and in the Southern United States, the lack of support and the absence of a full-time dedicated leader are hindering such success. The education division received the subjective grade of "B-".

The Green Club and the Tulane Environmental Project (TEP) have been significantly involved in the greening of one operational aspect of Tulane: recycling. Recycling at Tulane began in the 1970s as a volunteer effort. In the late 1980s, the Green Club formed to address more institutionalized recycling. In the early 1990s, Green Club leadership petitioned the university to establish a committee to green the campus. Tulane's president at the time, Dr. Eamon Kelly, established the TEP and appointed the chair, Oliver Houck, Professor of Environmental Law. The TEP was active for 2 years. In the first year, the members of the TEP researched and implemented a recycling program, hiring a full-time coordinator and receiving a minimal university budget. (In their second year, they began a recycled procurement program to "close the loop," but that initiative was limited to a few paper products.) Peaks and troughs in student leadership and activism (advocacy), the coming and going of numerous recycling coordinators over the years (leadership), and variable administrative support (resources) have led to peaks and troughs in the success of recycling operations. The Green Club has attempted other operational greening programs, e.g., a "Green Dining" initiative in Tulane dining areas, with minimal success. The administration took on an economics-based lighting retrofit, which did not include any education initiatives for saving energy and had no explicit environmental motives, but no other significant environmental initiatives have been

[6] The Green Gradecard did not use any standardized grading procedure; the approximately 45 students who conducted the audit relied on collective, subjective judgement to grade each area. The same subjectivity was used in this study, although the research behind the decisions was more extensive.

[7] Information on the faculty enrichment seminars is at www.tulane.edu/~efes/ .

[8] The Catalog is available at www.tulane.edu/~greenclb/catalog/ .

institutionalized. The operations division received a subjective grade of “D-”/“D”.

The history of greening at Tulane supports the model and reaffirms the need for a leader. Research has had a supported leader, and that division has been successful. As for education, the Environmental Studies Program should be a university-supported program with a leader. Recycling and procurement programs are in need of improvement; each needs policy, resources, leaders, and comprehensive means to achieve those ends. Additionally, other campus greening programs for operations need to be established for Tulane to live up to its reputation as an environmental (research and education) university.

THE GREENING PHENOMENON IN HIGHER EDUCATION

The Green Gradecard for the Green Wave environmental audit highlights many areas that are in need of improvement at Tulane, especially when compared with other institutions of higher education.[9] Experiences in academia offer caveats, lessons learned, and examples on which Tulane can build and even exceed. The greening initiatives in academia support the model for change, and they show the sound economic, social, and environmental implications of such programs (Blueprint 1995, Creighton 1998, Eagan and Keniry 1998, Keniry 1995, Smith 1993).

Environmental audits are powerful tools for gathering information about the environmental quality of the campus. They are the starting point for environmental change, and they provide information to educate the campus, the community, and especially those involved in the audit. Tulane’s audit, the Green Gradecard for the Green Wave, which an Environmental Sociology class conducted in the spring of 1997, issued letter grades with respect to various areas of environmental performance. Environmental Studies, an energy-saving lighting program, and hazardous waste policies received “A-” grades, whereas recycling, investment practices, and procurement of chemicals and pesticides received failing grades. Overall, the audit graded 22 areas, and Tulane’s “Green GPA” came out to a 1.9/4.0, or a “C” average. The audit concluded that the university should make an “institutional commitment to incorporate environmental decision making into all facets of [campus] operation . . . [and] establish a standing University Committee for Environmental Affairs.” The Gradecard supports the model for change in that it advocates institutional policy and resources that would allow for administrative (leadership) efforts to implement environmental change.

Programs at other institutions concerned with environmental curricula and campus environmental consciousness illustrate the essential role of leadership to provide education. Their success is reflected in campus environmental cognizance. Progressive environmental building, land use, and transportation (parking) policies have social, administrative, and economic benefits. Energy and water conservation programs are financially sound and serve as education about the importance of conserving natural resources. The greening of food service operations has health, environmental, and economic benefits for the campus and local community. Waste issues (recycling, hazardous waste, and medical waste) are visible to many in and out of the campus community; greening them is fiscally responsible, is educational, has positive impacts for the environment, and improves the image of the institution. Green procurement provides market stimulation to keep recycling and waste reduction initiatives available and economical. Finally, environmental research and socially responsible business and investment practices have impacts that can be felt around the world. Case studies from progressive and innovative institutions in the above areas provide examples of what and how Tulane can green (Creighton 1998, Eagan and Orr 1992, Keniry 1995, Smith 1993). Many of the case studies support the model for change.

HEARING FROM THE TULANE COMMUNITY

Interviews with Tulane students, staff, faculty, and administrators further support the model. Five of the six questions support the thesis of the study that a leader is needed to institutionalize and carry out greening efforts.

The four main institutional change barriers, as determined from the interviews, are institutional/organizational (lack of communication, lack of advocacy, and the lack of a leader), financial (lack of allocation of resources), cultural (lack of education),[10] and educational (lack of a modus for education). Interviewees thought that greening programs should relate to operations (administrative and physical) and education (individual and community learning, both in and out of the classroom). The results of the interviews clarify roles of each tier of the university community: students as learners, educators, and advocates; staff as learners and empowered “doers”; faculty as advocates and educators (who should practice environmental sustainability, especially if they teach it); and administrators as leaders in all aspects of the greening process. The responses for the roles of administrators reiterated every element of the model and focused on the need for an environmental coordinator to lobby the administration on environmental issues. Finally, interviewees affirmed that it is possible and appropriate to green Tulane. It is now possible to formulate a proposal for greening Tulane.

THE “BLUEPRINT FOR A GREEN TULANE”

The “Blueprint,” which is based on the model for change, is the plan for implementing institutional environmental change at Tulane. Included in it is the proposal to establish an Office of Environmental Affairs (OEA) and create an Environmental Coordinator position, both of which were accomplished in the summer of 1999. The “Blueprint” is presented here as outlined in the spring of 1999; changes in actual implementation are presented in the Conclusion.

---

[9] Tulane’s mascot is the “Green Wave.” The Green Gradecard for the Green Wave is available at www.tulane.edu/~greenclb/audit/audit.html .

[10] The “cultural barrier” is complex, and more research is necessary to determine specific aspects that could be the target of educational programs.

Advocacy

Re-establish/reinvigorate the Tulane Environmental Project (TEP) as the Tulane Environmental Committee (TEC). Ideally, President Cowen should initiate the new TEC; he should also confirm all appointments to the TEC and appoint the chair who would act as the presidential liaison. The TEC would be charged with approving an annual agenda for campus greening and reviewing OEA projects. A working group from the TEC could develop such an agenda and then continually work with the OEA. The Environmental Coordinator of the OEA would report to the TEC, (which would, in turn, answer to the president). The TEC would meet once or twice each academic year with representatives from the students (Associated Student Body and Green Club), the staff (Staff Advisory Council), the faculty (University Senate, CBR, and ENST), and the administration (President's Executive Working Group). Such representation involves the research, education, and operations divisions. The members of the TEC should be the key players on campus with regards to environmental change. The TEC would be the convergence of grassroots advocacy, which has been displayed for years, and top-down advocacy, which has yet to be shown, while simultaneously holding the power to make environmental change. The TEC, the OEA, and the Environmental Coordinator are interdisciplinary, interdivisional entities pivotal for coordinating comprehensive institutional greening.

Policy

Publish a statement that Tulane will be a leader in environmental research, environmental education, and environmental stewardship. The statement should outline the core values of environmental responsibility that Tulane will espouse. With such a proclamation, the TEC could gather input from the university community via "town meetings" and could draft a university environmental policy statement. The president and the various legislative bodies of the university could then ratify the policy. Additionally, it would be necessary for the university to sign on to national and international environmental platforms, e.g., the Talloires Declaration and the Valdez Principles; such involvement brings national and international attention as well as assistance in implementing sustainability on campus. Finally, project-specific policies, such as for recycling and procurement, should be developed.

Resources

Seek funding for institutionalizing the OEA from internal and external sources. Internal funds could first come from a cooperative funding procedure, whereby each of the academic deans, along with the vice presidents who would be primary representatives on the TEC, would contribute $3,000 to $5,000 for the job search and first year's salary of the Environmental Coordinator. With a job search estimated at $3,500 to $4,000 and with salary and benefits estimated at $36,000 to $36,500 (for a senior program coordinator position), a total of approximately $40,000 is needed; with eight academic deans and three vice presidents, the cooperative funding program could work. (In the spring of 1999, no one was reluctant to contribute to such a cooperative funding measure; they did mention that they would be more willing to participate once they knew that the president supported the OEA proposal.) This literal buy-in into the OEA is important for developing cooperation.

External funds could come from alumni gifts and endowments for programs, such as scholarships and speaker series, and grants for projects and operating expenses. An endowment of $1 million would secure the OEA in perpetuity; the Office of Development could assist in such fundraising. Some grants pending in the ENST include such monies in anticipation of the OEA; the ENST has found, however, that granting agencies will not pay for employee salaries but are more likely to provide funds for students, programs, and operating expenses. A study sponsored by the Nathan Cummings Foundation suggests that granting agencies and foundations fund specific campus projects that have the potential for success and could serve as a model for other institutions to use. Additionally, the report recommends "seed money" funding for projects that will eventually sustain themselves (Strauss 1996).

Other potential funding mechanisms include a university budget, internal "loans" repayable with savings from cost avoidance programs, and a student environmental fee. The more innovative the design of the OEA, the more marketable it is; as such, the OEA could easily raise outside funding—especially from alumni and foundations. Other necessary resources include personnel, information and data, and an office. The issue of personnel is addressed later. Initial sources of information and data already exist (the larger study from which this paper is extracted and the environmental audit mentioned above), but an annual report of the OEA submitted to the TEC, e.g., the "State of the Tulane Environment", could continue to chronicle important information and data. Finally, the OEA has been allocated office space in the new Environmental Science Building complex where it will be in close proximity to most of Tulane's environmental research and education programs. The CBR, Green Club, and Environmental Studies Program can provide necessary office supplies, including a computer, until full funding is raised.

Leadership

Empower the OEA to make a positive impact on campus. The Environmental Coordinator should work closely with various campus entities and constituents to develop and implement greening initiatives.

Means and Ends

Educate the campus on environmental issues. This education could be via the following possibilities: large- and small-scale seminars and programs for students, staff, faculty, and administrators; continued research into and implementation of greening initiatives; a comprehensive measurement/reporting system; the development of an environmental management plan; and classroom and curriculum initiatives. The TEC should initially prioritize projects, and, after the first year, the TEC could approve annual plans and review past performance. The "ends" should be outlined in general and specific policies. The Environmental Coordinator might also teach environmental classes such as "The Campus and the Biosphere" or "Ecological Design."

THE OFFICE OF ENVIRONMENTAL AFFAIRS

Leadership

The OEA will house the leadership that will make environmental change at Tulane: the Environmental Coordinator. As the director of the OEA, the Environmental Coordinator should report to the TEC. Dr. John McLachlan and the CBR would essentially provide a “home” and some day-to-day operational oversight for the OEA, whereas the TEC would provide the approval and guidance for long-range operations. Dr. McLachlan might also chair the TEC. Such an establishment is necessary because of the access to the varied power and resources of TEC members in addition to the valuable experience with successful environmental change initiatives of the CBR and its director. The TEC would involve the people who guide the university in its daily and long-range operations and would ensure that environmental concerns are heard. The TEC could appoint a working group (with ample student involvement) to cooperate with the OEA throughout the year on projects and programs.

The OEA should be “bootstrapped” to each division and tier of the university: research, education, and operations; and students, staff, faculty, and administrators. Bootstrapping involves creating official and unofficial connections that prevent atrophy or abolishment and that foster collaboration and cooperation between all areas of the university. Such connections include research programs with the CBR; educational and service programs with the Green Club, the ENST, and various deans; and operational programs (the ones that will receive much of the focus) with the appropriate vice presidents and facilities administrators. Many other connections are possible with Janitorial Services, Student Programs, Orientation, Admissions, Housing and Residence Life, Athletics, and campus institutes. These connections bootstrap the OEA to the core of the university and provide ways to affect change.

Having an environmental coordinator—the leader—is absolutely critical to the institutional environmental change movement. The leader should be a full-time employee with appropriate experience and degrees; the leader cannot be a student, although students are the second key to success.

Students

Students from the Green Club, ENST, student organizations, and the general campus population are pivotal to the feasibility and success of the OEA. Not only could students carry out office duties in the OEA, they could also participate in and benefit from the myriad programs. To maintain their involvement, ENST and OEA fundraising endeavors could provide work-study funds for student workers, scholarships for leadership and academic excellence, and research assistanceships for student projects. Such opportunities would also be excellent recruiting tools.

As “customers,” students are effective advocates for change; they could advocate and stand up for issues in student milieus by representing the OEA on various campus committees. Through the OEA, students would have an organized outlet for environmental activism, volunteerism, and research opportunities as soon as they arrive on campus. They would provide a constant source of enthusiasm and ideas for the program, continually clarifying the raison d’etre of the OEA. Students would be involved in an active learning and service community, and they could gain valuable leadership and job skills by, for example, taking part in efforts to educate campus denizens through various programs such as greening seminars, the Internet, and publications.

As well as contributing to the success of the programs of the OEA, the students will also be active participants in their own education. In addition to classes, other educational venues include service learning in the community and campus environmental research, effectively using the campus as a laboratory for environmental problem solving and for learning how to make positive environmental change.

Programs of the OEA could also help create connections for students, especially between students and place, i.e., Tulane and New Orleans. The connections they make at Tulane through the OEA—with outside agencies, community members, with professors and, most importantly, with each other— would ensure the lasting success of the OEA because of the broad and dedicated alumni support network that could develop. And the innovative programs of the OEA and ENST could certainly attract talented new students.

The OEA will depend integrally on students; it will also empower, support, and educate them. The relationship will be one of symbiotic, collective leadership, and learning. Campus sustainability programs are an extraordinary boon for the students, the entire university community, and, ultimately, modern civilization. The students will carry their lessons and skills with them, disseminating environmental sustainability wherever they live.

Programs

Potential programs of the OEA range from large-scale projects (conferences with national or international organizations) to smaller scale projects (office recycling education in a department; they would encompass the four divisions and four tiers of the institution, the areas of Tulane’s strategic interest, and all appropriate environmental parameters. Through the TEC, presidential invitations could be sent to key faculty and administrators to strongly encourage them to participate. In doing so, the OEA would educate campus decision-makers and crystallize their involvement with campus stewardship programs, all of which would strive for ecological literacy.

The OEA would not necessarily run all the programs, but it would help coordinate efforts, provide information and experience, and advocate for new programs. Students are an integral part of the programming function of the OEA, and they comprise the crucial links between the OEA and the myriad departments, programs, and organizations on campus and in the community. The successful projects of the OEA should be chronicled in campus, local, and national media. Projects would likely begin focused on campus; once the OEA builds momentum and accomplishes some campus tasks, programming could move into the local community. The program possibilities are virtually endless.[11]

CONCLUSIONS

In the early fall of 1998, it was estimated that, with fundraising throughout the year and with hiring in the spring, the OEA could be in place by the summer in order to prepare for the next academic year. In 1998 when an earlier version of this study was circulated to raise support and funding, eight deans, the CBR director, the vice president for administration and planning, and the provost all said that they were in support of the initiative in principle. Funding was solicited from these senior administrators via a one-page proposal abstracted from the "Blueprint". It cited the research findings (presented earlier) and recent events (outlined later) as sources of campus support and asked for their financial contribution. Follow-up meetings ensued with most of them. Eventually, all were in support but were hesitant to commit resources before they knew the opinions of the president. After convincing them of presidential support, each donated amounts ranging from < $1000 to over $5,000. Then the hiring process began, and the OEA was established.

In 1998–99, Tulane was in a time of profound change—a presidential transition. Tulane's new president, Dr. Scott Cowen, saw that year as a "renaissance of thought and action" to redesign Tulane for the future. That time of strategic planning was an opportune time for institutionalizing the greening process, and the grassroots advocacy pulled out all stops in order to convince President Cowen that the OEA was a good and worthwhile venture. He was not immediately convinced of the validity of the project, but after 1 year of advocacy, he gave his verbal support. At least seven actions were fundamental to the advocacy of 1998–99. All were student initiated and led, thus exhibiting the unique role and power of students in implementing change. The president was their ultimate target, but the advocacy helped crystallize the involvement of the administration and the campus in general.

First was the establishment of the Associated Student Body's Committee to Green Tulane, which created a proposal asking the administration, and the president specifically, to implement steps that would make the university more environmentally responsible, including establishment of the OEA. By the spring, the committee presented its recommendations to the student body assembly, which passed the resolution; a subsequent campus newspaper article on the resolution included a quote from President Cowen supporting the recommendations in principle. Second was a year-long series of letters to the editor regarding campus environmental initiatives, all of which were in response to contemporary campus news issues, such as parking and security lighting, which were not originally presented as environmental in nature. The third item involved the collective results of numerous programs by the Green Club, including public events on Earth Day and America Recycles Day as well as numerous smaller events that garnered campus media attention. Fourth was a continual series of meetings on the proposal with the president and other administrators. Fifth was the attention provided to Tulane when the present author, then a graduating senior, was selected for a national scholarship, as well as top campus honors, for work greening Tulane; those events involved local television as well as local and national newspaper coverage.

The penultimate item was of great importance to the eventual success of the advocacy. Using the Greening the Campus study and some of the core texts that informed it (Creighton 1998, Keniry 1995), the present author and Dr. Charles Reith, a visiting professor of business and environmental studies, teamed up to create a class entitled "Ecological Design." The class performed an in-depth audit of Gibson Hall, Tulane's main administration building that houses the offices of the president, provost, senior vice presidents, and other upper administration. The results of the audit were presented in poster form at a campus-wide Earth Day celebration in Alcee Fortier Hall, one of the two new environmental science buildings on campus. President Cowen, many administrators, and hundreds from the campus community were present at the event. A report with suggestions for specific improvements, one of which involved the establishment of the OEA, was prepared and submitted to President Cowen; the campus faculty-staff newsletter featured the Gibson Hall audit, and the report and posters were placed in Gibson Hall itself as well as on the Internet.[12] In addition, earlier in the semester members of the class attended and presented at a national conference on ecological campus design. The eight Tulane delegates—two staff, one faculty, three students, and the two professors—eventually drafted a letter to the administration supporting efforts promoted at the conference, and they formed a working group that continued to meet and follow up on ecological design issues. The class also sponsored two campus-wide events: a public lecture by renowned campus environmental activist David Orr of Oberlin College, which was well attended (especially by administrators) and was pivotal to the campus greening dialogue; and a round-table discussion on the merits of the International Standards Organization's environmental management scheme, ISO 14001.

The ISO issue became the final and pivotal item in the advocacy effort. President Cowen, a professor of business, understood the language of ISO 14001 and eventually lent his support to a concerted effort to decide if it was best for Tulane. A 1-day conference on the feasibility of ISO at Tulane was held in May of 1999; it involved many faculty and senior administrators, and even included an important appearance by President Cowen. The proposal for the OEA (the "Blueprint") now included language that called for pursuing ISO certification, which Tulane saw as a program that could eventually go beyond campus boundaries and connect with

[11] For examples of potential programs, see Greening the Campus (footnote 1), especially Appendix F (also Table 4 of the Executive Summary).

[12] The class report and syllabus are at www.tulane.edu/~greenclb/enst481/ .

local institutions that have environmental impacts (especially industry).

With presidential support and 2 years of funding procured, a national job search began in early June. In early August, a committee hired Elizabeth Davey, who had experience in campus greening initiatives and the added bonus of having a Ph.D. in English, providing her with a familiarity with the academic environment and a certain legitimacy among the faculty. Unfortunately, the group who formed the search committee and who should have eventually become the Tulane Environmental Committee, did not form the TEC, and thus starting institutional change and campus greening moved more slowly than it would have if the TEC were initially in place. Whereas the TEC has yet to be formed, the ISO initiative may result in its creation because it is an important element for ISO certification. Its absence has been noticeable.

It was a controversial decision to place the OEA in the CBR because of the center's isolation from many aspects of the university (especially the operations). However, the most appropriate home for the OEA, the office of the vice president for administration and planning, agreed to consider relocation of the OEA after it becomes established because placing it in the CBR for administrative start-up purposes seemed the best decision despite the political ramifications. The office space in one of the two new environmental studies buildings on campus guaranteed a close proximity to other environmental programs such as the Environmental Studies, the CBR, and the Green Club, whose Student Environmental Center is located there. After a year of operation, the future placement of the OEA is still uncertain.

All the elements of model for institutional change are evident in the events that unfolded at Tulane. Presenting the model to the administration also legitimized and supported proposed decisions since they saw the logic, data, and support of the literature (and academics resonate with academically supported arguments). The advocacy stage involved the collective efforts of many to convince the administration to go forward with the proposal; the TEC as a formal entity was not the instrumental advocacy group as in the "Blueprint." Resources were pooled collectively, and plans are already in place for alumni fundraising and grant writing. Policy exhibited the dynamism provided for in the model. Only informal policy (the early versions of the "Blueprint" and commitments to funding support, for example) was in place before the hiring of Dr. Davey. Eventually, formal policies will be established as in Tulane's Strategic Plan in which, as a result of Davey's work and the advocacy that led to her appointment, President Cowen has recently included environmental issues. The OEA is in place, and its leader, Dr. Davey, is working closely with the various groups, departments, and schools and is developing a variety of means to educate the campus on environmental issues with the eventual ends of environmental responsibility. Although much planning remains to be done, she has already started campus greening projects such as: instituting a grant program for student campus environmental research and education projects, creating with alumni the Tulane Environmental Network, improving recycling and composting efforts, creating an "ecological design" chapter of the campus master plan, developing a preorientation program, and developing an environmental Web page.[13]

The model has provided a helpful framework for understanding and activating institutional change. The limitation of the "Blueprint" (and of the model) has been recognized: it limits to one person the responsibility for making Tulane an environmentally responsible institution, rather than sharing the responsibility and initiative across schools and departments. Some of that limitation could have been alleviated if the TEC had been successfully established before or at the time of Davey's hiring. The second element to remove some of that limitation, however, is being actualized. With Davey's leadership, students are playing key roles in greening Tulane; they are being supported with work-study funding, semester research grants, and extracurricular activities. In fact, it cannot be stressed enough that the students—from the Green Club, the Environmental Studies Program, the Associated Student Body and the general campus population—were absolutely central to the success of the entire initiative to institutionalize campus greening. Moreover, they are key to maintaining the effort, and, after their campus tenure, these students and their peers will take sustainability beyond the campus to create a more sustainable world.

ACKNOWLEDGMENTS

The Tulane Environmental Studies Program, the Center for Bioenvironmental Research at Tulane and Xavier Universities, and the Ecological Society of America provided research funding and administrative support. I would like to thank Timmons Roberts, Michael Zimmerman, Charles Reith, John McLachlan, Teresa Soufas, Yvette Jones, and Christine Murphey for their support and thoughtful suggestions. Heartfelt thanks also go to Julian Keniry and David Orr for inspiration and stimulating conversations.

REFERENCES

Ackerman, F. 1997. Why do we recycle? Markets, values, and public policy. Washington, DC: Island Press. 210 p.

Alinsky, S.D. 1971. Of means and ends. In: Rules for radicals. New York: Vintage Books: 24–27.

Altbach, P.G. 1974. University reform. Cambridge, MA: Schenkman Publishing. 211 p.

Berry, J.K.; Gordon, J.C., eds. 1993. Environmental leadership: developing effective skills and styles. Washington, DC: Island Press. 286 p.

Blueprint for a green campus: the campus earth summit initiatives for higher education. 1995. The Heinz Family Foundation. 63 p.

Bowers, C.A. 1997. The culture of denial: why the environmental movement needs a strategy for reforming universities and public schools. Albany, NY: State University of New York Press. 277 p.

Brown, J.S.; Duguid, P. 1996. Universities in the digital age. Change. July/ August. 28/4: 10–19.

[13] See www.tulane.edu/~eaffairs/ .

Cerych, L.; Sabatier, P., eds. 1986. Great expectations and mixed performance: the implementation of higher education reforms in Europe. Trentham, Stoke-on-Trent, United Kingdom: Trentham Books Ltd. 276 p.

Creighton, S.H. 1998. Greening the ivory tower: improving the environmental track record of universities, colleges, and other institutions. Cambridge, MA: MIT Press. 337 p.

De Young, R. 1986. Encouraging environmentally appropriate behavior: the role of intrinsic motivation. Journal of Environmental Systems. 15: 281–292.

Dolence, M.G.; Norris, D.M. 1995. Transforming higher education: a vision for learning in the 21st century. Ann Arbor, MI: Society for College and University Planning. 100 p.

Dominick, C.A. 1990. Revising the institutional mission. In: Steeples, D.W., ed. Managing change in higher education. San Francisco: Jossey-Bass: 29–36.

Eagan, D.J.; Orr, D.W., eds. 1992. The campus and environmental responsibility. San Francisco: Jossey-Bass. 133 p.

Eagan, D.; Keniry, J. 1998. Green investment, green return: how practical conservation projects save millions on America's campuses. Washington, DC: National Wildlife Federation. 77 p.

Fantini, M.D. 1981. On effecting change in educational bureaucracies. Education and Urban Society. 13: 389–416.

Farmer, D.W. 1990. Strategies for change. In: Steeples, D.W., ed. Managing change in higher education. San Francisco: Jossey-Bass: 7–18.

Gittell, M. 1981. Introduction. Education and Urban Society. 13: 389–398.

Hamburg, S.P.; Ask, S.I. 1992. The environmental ombudsman at the University of Kansas. In: Eagan, D.J.; Orr, D.W., eds. The campus and environmental responsibility. San Francisco: Jossey-Bass: 55–63.

Keniry, J. 1995. Ecodemia: campus environmental stewardship at the turn of the 21st century. Washington, DC: National Wildlife Federation. 222 p.

Lane, J.E. 1990. Institutional reform: a public policy perspective. Brookfield, VT: Dartmouth Publishing Company Limited. 259 p.

MacTaggart, T.J.; Christ, C.L. [and others]. 1996. Restructuring higher education: what works and what doesn't in reorganizing governing systems. San Francisco: Jossey-Bass Publishers. 260 p.

Mansfield, W.H. 1998. Taking the university to task. World Watch. May/June: 24–30.

Orr, D.W. 1990. The liberal arts, the campus, and the biosphere. Harvard Educational Review. 60/2: 205–216.

Orr, D.W. 1992. Ecological literacy: education and the transition to a postmodern world. Albany, NY: State University of New York Press. 210 p.

Orr, D.W. 1994. Earth in mind: on education, environment, and the human prospect. Washington, DC: Island Press. 213 p.

Orr, D.W. 1995. Educating for the environment: higher education's challenge of the next century. Change. 27/3: 43–46.

Orr, D.W. 1996. Reinventing higher education. In: Collett, J.; Karakashian, S., eds. Greening the college curriculum: a guide to environmental teaching in the liberal arts. Washington, DC: Island Press: 8–23.

Rainsford, G.N. 1990. The demographic imperative: changing to serve America's expanding minority population. In: Steeples, D.W., ed. Managing change in higher education. San Francisco: Jossey-Bass: 91–100.

Smith, A.; The Student Environmental Action Coalition. 1993. Campus ecology: a guide to assessing environmental quality and creating strategies for change. Los Angeles: Living Planet Press. 129 p.

Steeples, D.W. 1990. Concluding observations. In: Steeples, D.W., ed. Managing change in higher education. San Francisco: Jossey-Bass: 101–107.

Strauss, B.H. 1996. The class of 2000 report: environmental education, practices and activism on campus. New York: Nathan Cummings Foundation. 125 p.

Strecker, M. 1998. Tulane makes a billion dollar difference. New Orleans: Tulane University Public Relations. [December 3 press release].

Tulane University. 1997. Undergraduate catalog. New Orleans: Tulane University. 4 p.

Williams, E. 1991. College students and recycling: their attitudes and behaviors. Journal of College Student Development. 32: 86–88.

Wood, R.J. 1990. Changing the educational program. In: Steeples, D.W., ed. Managing change in higher education. San Francisco: Jossey-Bass: 51–58.

THE COLLABORATION BETWEEN THE UNIVERSITY OF MISSISSIPPI AND BELIZE ENABLES OPPORTUNITIES FOR SCIENCE EDUCATION

Richard A. Belisle[1]

Abstract—The University of Mississippi and the American Universities International Program (AUIP) enjoy vast educational opportunities in Belize. Bounded by Mexico on the north, Guatemala on the west and south, and the Caribbean Sea on the east, Belize's 22,960 km² of topography range from sea level to 3,688 ft. This variation in altitude and the tropical climate support a great diversity of habitats. The relative underdevelopment and pristine ecosystems of rain forest, pine savannahs, and mangrove swamps are complemented by the longest barrier reef in the Western Hemisphere. Since 1995, the primary purpose of the AUIP study program in Belize has been to provide experiential learning courses for students, professors, and professionals from the United States and Belize. The main emphasis has been on marine biology, ecology, natural resource management, anthropology, ecotourism, and land use.

INTRODUCTION

The University of Mississippi, along with the American Universities International Program, developed an intense study-abroad program in Belize since 1995. This collaboration has seen outstanding success during the past 5 years and involved many university professors, students, professionals, and other officials from both the United States and Belize.

PROGRAM OVERVIEW

The Belize field-study program is a unique multicultural program where participants live and work together with other students from across the United States and other parts of the world. They travel throughout Belize with supervisors, instructors, guides, and local staff to various sites, which provide for affordable living opportunities to study within the natural settings and cultures of the local people. Students are given proper orientation in compliance procedures and regulations to ensure protection of ecosystems being studied while maintaining strict North American safety standards.

The actual classrooms and study sites include spectacular and pristine ecosystems, which range from the largest barrier reef in the Western World, lush rain forests, mangrove swamps, high forested mountains and plateaus, freshwater lagoons, scenic and wild rivers, extensive underground caves, Mayan ruins, and much more. The student is given a truly interactive and hands-on experience that can definitely put the fun and excitement back into science education in a most pragmatic and affordable manner. Such an opportunity can take learning well beyond the greatest expectations, broaden one's scope, and perhaps change one's overall life perspective in a most positive manner. Consequently, almost daily, students are in awe of what they experience and learn.

The current success of the Belize field-study program demonstrates the tremendous opportunities for science education and research that American universities can utilize in collaboration with the many and varied Belizean institutions. Reliable and dependable cooperation with local institutions can be readily established that will provide good accommodations, warm atmosphere, fine food, and a complex mix of pristine subtropical ecosystems—all at a super value and within strict safety standards and regulations.

COUNTRY OVERVIEW

Belize, formerly known as British Honduras, gained independence in 1981 and remained a parliamentary democracy and member of the British Commonwealth. It continues to be a peaceful country, well insulated from the problem areas of Central America.

Belize is on the eastern coast of Central America on the Caribbean Sea. It is located between 15^{o}53' and 18^{o}30' N. and is bounded on the north by Mexico (the State of Quintana Roo and in the extreme northwest, Compeche) to the west and south by Guatemala (the department of Peten and in the extreme south Izabal) and to the east by the Caribbean Sea. In form, the country is roughly rectangular, extending 280 km from north to south and 109 km from east to west. The maximum extension is 180 km, including the territorial sea. Total land area, including some 1,200 cays, is 22,960 km², which is divided into 6 districts, 9 municipalities, and more than 240 villages. The territorial sea adds to a total area of 46,620 km².

In spite of its small size, Belize is composed of a diversity of landscapes. Inland, the Mayan Mountain/Mountain Pine Ridge Massif is the dominant physical feature and rises to 1,124 m at its highest point. It is surrounded by rugged karst limestone hills. Beyond that, most of the north and the entire coastal area, including the south, consists of low-lying plains. Nine land regions, each comprising a particular combination of topography, soils, and vegetation, and, thus, a distinctive landscape, have been distinguished. The

[1] Vice President, Beaus, Ltd., Box 300, Belmopan, Belize, Central America.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

upland of the Mayan Mountain land system constitutes the dominant topographical feature. The geology of this area comprises Carboniferous and Permian metasediments (quartzites, shales, slates) with granite intrusions. Most of this land system is characterized by steep slopes and shallow soils, which are leached, acidic, infertile, and fragile.

There are five upland regions, which are characterized by hilly terrain made up of a thick section of indurated Cretaceous limestone that has eroded into an extremely rugged scenery known as karst. These regions are: the central foothills, located on the northern flank of the Mayan Mountains Massif; the western foothills, located in the western and midwestern areas of the country that include the most extensive cave system in Central America; the eastern foothills, overlooking the coastal plains and hosting the oldest rocks in Central America; the southern foothills, located in the northwestern parts of the southernmost district; and the Bravo Hills, located in the northwest area of Belize.

The other three land systems are comprised of the coastal plains. The southern coastal plain lies in the most southerly district and is further divided into the Machaca (marginal agricultural suitability) and Temash (poorly drained acid and infertile) plains; and the northern and central coastal plains, both underlain by Tertiary limestone, marl, and calcareous clays.

The preliminary classification scheme for the coastal zone, as developed under the current Protected Areas System Plan, further distinguishes four regions that reflect differences in sediments, bathymetry, topology, and hydrology, and represent marine equivalents of the terrestrial land systems. These four coastal regions, running from north to south, are the Ambergris region, the Belize region, the Stann Creek region, and the Punta Gorda region.

The climate in Belize is subtropical with rainfall varying from < 1300 mm per year in the north (with a 4-month dry season) to over 4500 mm in the south (with a shorter dry season). This remarkable variation in rainfall over a small area (280 km in length) is incredible but true.

The climate, along with the diversity of geology, topography, and edaphic factors, maintains pristine ecosystems of rain forest, pine Savannahs, jungle mountains, rivers, deep caves, lagoons, estuaries, huge waterfalls and more. Likewise, the natural vegetation shows astonishing variation and diversity with some 49 distinct types already recognized. Together, the terrestrial ecosystems of Belize display as much habitat diversity at low altitude as any other Mesoamerican country. The exceptional inland ecosystems of Belize are home to over 500 species of birds, 107 species of reptiles, more than 120 species of mammals, 26 amphibian species, and over 700 plant species.

Seven principal habitats make up the Belizean coastal zone: coral reefs, sea grass beds, mangrove forests, littoral forests, watersheds, wetlands, and estuaries. The shallow crystal clear waters of the coastal zone of Belize are host to the largest barrier reef of the Northern Hemisphere. This is one of the most spectacular, complex, and biologically diverse ecosystems in the world. The barrier reef complex runs from north to south and extends some 230 km along the coastline from the border with Mexico to the threshold of the border with Guatemala. Three offshore coral atolls occur east of the barrier reef and are associated with multiple sand and mangrove islands enclosing a lagoon with sparse coral formation. The southernmost atoll, however, has frequent and well-developed structures.

Some 65 coral reef species have been identified with suggestions that as many as 113 coral-associated species may occur. Indications are that apart from corals, as many as 343 species of other invertebrates may occur. In December 1996, seven locations on the barrier reef were designated World Heritage Sites under the World Heritage Convention; and in 1998, one inland wetland site was designated under the RAMSAR Convention.

Over 4,000 years ago, the Mayan Civilization took root in what is now Belize. Some 2 million Mayas may have lived there during its height. They built sophisticated cities and temples, and the record of their lives in over 900 archeological finds has survived the years. Although many temples have been excavated from the earth, many more, it is certain, remain hidden in the dense jungle. The reign of the Mayas has vanished, but direct blood ancestors still live in Belize in harmony with an unlikely mix of newcomers. They include descendants of English settlers, European pirates, African slaves, Atlantic Coast Indians, East Indians, Mennonites, Chinese, Lebanese, and some veterans of the U.S. Civil War. More recently, an influx of Central American immigrants has increased the population of Belize to some 230,000 strong.

Although multilingualism prevails in Belize, communication is not a problem because almost everyone speaks English. However, the different ethnic groups maintain their cultural traditions and dialects, thereby adding to the natural charms of Belize.

Getting to Belize from the United States is facilitated by direct flights out of the cities of Houston, New Orleans, Dallas, and Miami, each with a short duration of 2 hours. Central standard time is observed year round in Belize, and, similarly, the currency exchange is stable at $2.00 BZE to $1.00 U.S. Hotels, restaurants, and stores accept U.S. and Belizean currencies interchangeably.

Belize is still an outpost on the edge of the world, where all things wild live undisturbed. If the country is famous for anything, it would be for its genuinely friendly people, touring, diving along its outer cays, and ecotourism inland. The richness of the traditional medicine that preserves the value of many species of plants and animals in the unspoiled ecosystems is likewise appealing to tourists, students, professors, and researchers alike. Furthermore, the availability of modern and highly efficient telecommunications and Internet facilities throughout Belize adds irresistible appeal to the high-tech generation of today and provides exceptional delight to both students and professors.

The primary purpose of the current University of Mississippi collaborative program in Belize is to provide an experiential learning adventure whereby undergraduate students, along with their professors, can develop an understanding of living, working, and learning in a developing country where pristine ecosystems prevail. It is designed to entail field study for persons with backgrounds and interests in natural resources, biology, geology, ecology, tourism, and anthropology. It is for anyone who enjoys working and learning in remote environments and cares about further conservation of global resources, particularly in a diverse and challenging natural situation. Special emphasis is placed on natural and social sciences and how land use and ecological decisions are developed within existing socioeconomic constraints.

Some recent direct spin-off activities include: baseline environmental studies, evaluation and monitoring of environmental factors, natural product research, research and development of bioremediation systems and prototypes, and current and tide data generation within a U.S. Navy hydrographic project. Also, possibilities exist through collaboration with the Mississippi Research Consortium in the completion of a diagnostic for the World Bank’s Regional Mesoamerican Reef Project. Subsequently, several new avenues for additional marine biology research activities and operations have evolved. Clearly, Belize is now being recognized as a country that affords ideal hands-on situations for field studies that can certainly provide tremendous opportunities for science education within a regional and global context.

FEASIBILITY OF USING ORNAMENTAL PLANTS IN SUBSURFACE FLOW WETLANDS FOR DOMESTIC WASTEWATER TREATMENT

Marco A. Belmont and Chris D. Metcalfe[1]

Abstract—Constructed wetlands are possible low-cost solutions for treating domestic and industrial wastewater in developing countries such as Mexico. However, treatment of wastewater is not a priority in most developing countries unless communities can derive economic benefit from the water resources that are created by the treatment process. As part of our studies directed at improving the quality of water in the Rio Texcoco in central Mexico, we are determining the feasibility of using ornamental flowers for treatment of domestic wastewater. In a laboratory-scale study, we determined that subsurface flow wetlands planted with calla lilies could reduce levels of ammonia and nitrate in simulated domestic sewage. Results are presented on the optimal conditions for treatment of domestic wastewater in these systems. Floriculture activities in constructed wetlands could provide the economic benefits necessary to encourage communities in developing countries to maintain wastewater treatment systems.

INTRODUCTION

Constructed wetlands are a feasible solution to treat domestic wastewater in small rural communities. These systems are especially valuable for onsite wastewater treatment in developing countries because they are low tech, and the costs of construction and operation are low (Denny 1997).

The gains in vegetative biomass in constructed wetlands can provide economic returns to communities when harvested. These can be realized through biogas production, animal feed, fiber for papermaking, and compost (Lakshman 1987). There have been no reports of the use of ornamental plants, such as commercial flowers, in constructed treatment wetlands even though these plants can be harvested and sold.

Most of the research on constructed wetlands has been conducted in northern countries. Therefore, the plants most studied for treatment purposes are cattail (*Typha* spp.), bulrush (*Scirpus lacustris*), and reeds (*Phragmatis australis*) as these plants are native to northern countries (Denny 1997). It is known that ornamental plants like canna lily (*Canna flaccida)*, calla lily (*Zantedeschia aethiopica*), elephant ear (*Colocasia esculenta*), ginger lily (*Hedychium coronarium*), and yellow iris (*Iris pseudacorus*) can be used in rock/plant filters to treat septic tank effluents (Wolverton 1990). However, there is not enough information about the efficacy of these “warm weather” ornamental plants for use in treatment wetlands. Because most developing countries are located in tropical and subtropical areas, the use of ornamental plants for treatment in wetlands should be explored. Floriculture opportunities would also provide economic benefits to the communities in addition to the benefit of the wastewater treatment.

This study is focused on the use of calla lily in subsurface flow wetlands to treat domestic wastewater.

OBJECTIVES

The objectives of this study are to set up a lab-scale subsurface flow wetland to study the feasibility of using the calla lily to treat simulated wastewater and to assess the efficacy of the system in removing nitrogen from the wastewater.

METHODS

A lab-scale subsurface flow wetland was installed in a greenhouse at Trent University. The system consists of six cells of 38 cm by 240 cm by 30 cm (width by length by depth) filled with crushed rock of 3 to 5 cm in diameter. Two cells were unplanted and four were planted with calla lilies. The cells were provided with the same simulated domestic wastewater, stored in an elevated tank, at the same flow. The output water was collected and pumped back into the tank. Each cell was planted with 16 plants at a distance of 15 cm between them.

The plants were 8 cm tall when planted. The system was maintained at a temperature range of 14 to 20 ^{o}C and 16 hours of light per day. After plantation, nutrients were added to the water in the same composition as is used for hydroponics to allow the plants to root and grow. After 120 days, once the plants were well established, lawn fertilizer and tannic acid were added to increase the nitrogen and carbon levels to those typically found in domestic wastewater (table 1). The flow was adjusted to operate the system at 1 day of hydraulic retention time.

Ion-selective electrodes were used to measure N-NH_3, N-NO_3, N-NO_2; phosphate, sulfate, and chloride were measured by ion chromatography. Total organic carbon (TOC) was measured using a TOC analyzer, Shimadzu model TOC-5000; pH and Eh were also measured. The monitoring of the water was performed according to techniques described in the Standard Methods (American

[1] Belmont, Graduate Student, and Metcalfe, Professor, Environmental Resources Studies, Trent University, Ontario, Canada.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

Table 1—Typical water-quality parameters in domestic wastewater

Parameter	Sewage in North America medium-strong	Experimental wetland[a] April-May 2000
	Mg/L	
TSS	720 – 1200	6.5 – 14.5
TOC	160 – 290	110 – 155
$N\text{-}NH_3$	25 – 50	70 – 180
$N\text{-}NO_3$	0 – 0	60 – 170
$N\text{-}NO_2$	0 – 0	1.2 – 11.16
pH	6 – 8	6.9 – 8.1
Phosphate	5 – 10	25 – 75
Sulfate	30 – 50	600 – 620
Chloride	50 – 100	43 – 57

[a] Metcalf and Eddy 1991.

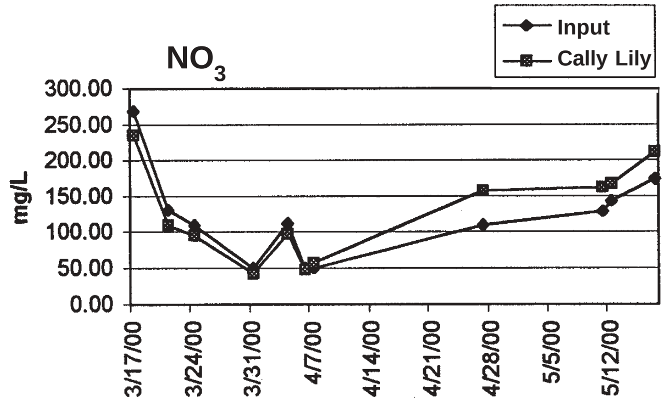


Figure 2—Nitrate concentration in mg/L.

Public Health Association and others 1998). The parameters were measured in input and output water of the cells.

The simulated wastewater was prepared to meet the usual characteristics of sewage as shown in table 1 (Metcalf and Eddy 1991).

RESULTS AND DISCUSSION

The experiment shows that the calla lily can grow very well on crushed rock beds at warm temperatures. The growth of the plants was faster than expected, and the plants started flowering after 6 weeks of plantation.

Ammonia and nitrate measurements showed that the wetland removed ammonia (fig. 1), but nitrate reduction occurred only at the beginning of the experiment before the addition of organic nitrogen (fig. 2). In fact, once fertilizer was added, the nitrate concentration was higher in the output than in the input water. This probably means that the production of nitrate by the conversion of organic nitrogen into ammonia followed by conversion of the ammonia to nitrate exceeded the nitrate removal capability of the system. In other words, the concentration of nitrate increases because of the lower capability of the system for denitrification than for mineralization and nitrification. It is necessary to quantify organic nitrogen to support this hypothesis.

Nitrite was found in very low concentrations compared with nitrate and ammonia; therefore, nitrate and ammonia are more descriptive of the nitrogen transformation in the artificial wetland.

Measurements of pH showed that the output water from the wetland had a quite constant pH, slightly higher than 7, even when the input water pH changed with time for almost 2 units (fig. 3). The redox potential (Eh) in the input and output water was very similar through the sampling period (fig. 4). This parameter showed a tendency of decrease with time throughout the duration of the experiment. This is probably due to the addition of organic nitrogen, ammonia, and organic carbon to the input water.

CONCLUSIONS

Calla lily plants can live and grow very well in subsurface flow beds of crushed rock. In warm places, the calla lily can be used in subsurface flow wetlands to treat wastewater and produce cut flowers. This adds a financial benefit to the environmental benefit of wastewater treatment. Under proper

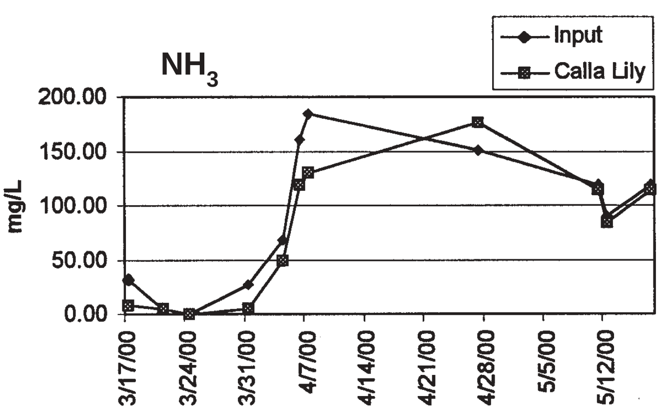


Figure 1—Ammonia concentration in mg/L.

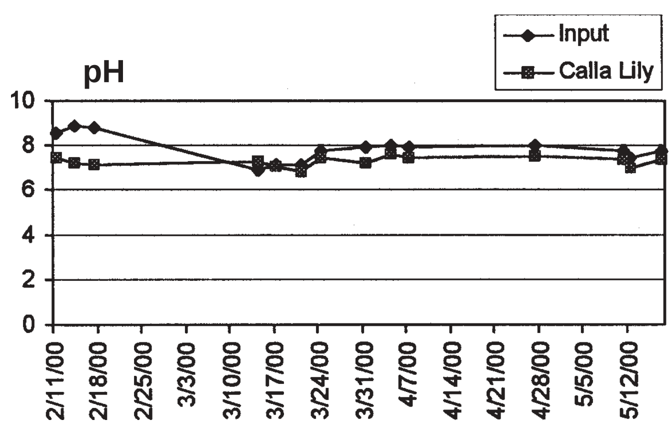


Figure 3—Measured pH.

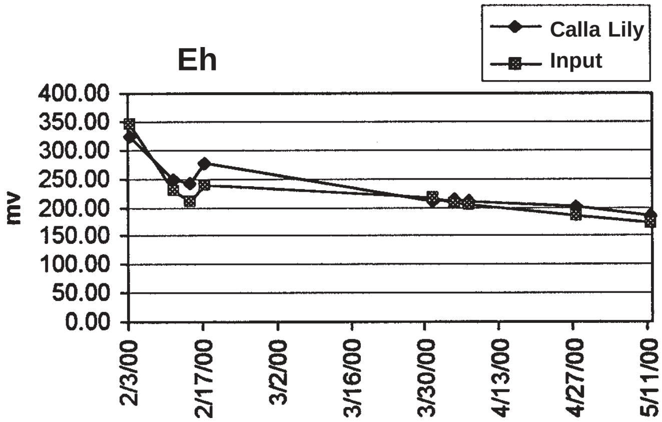


Figure 4—Redox potential (Eh) in mV.

conditions of hydraulic retention time, calla lilies can be used in subsurface flow wetlands to reduce levels of nitrogen and to convert ammonia to nitrate.

For the next phase of the experiment, it is necessary to quantify total organic nitrogen. The artificial wetland regulates the water pH buffering changes in the input water pH.

ACKNOWLEDGMENTS

We thank the Canadian International Development Agency for the financial support received through the INSTRUCT project and Consejo Nacional de Ciencia y Tecnologia (CONACyT, Mexico).

REFERENCES

American Public Health Association; American Water Works Association; Water Environmental Federation. 1998. Standard methods for the examination of water and wastewater. 20th ed. Washington, DC.

Denny, P. 1997. Implementation of constructed wetlands in developing countries. Water Science Technology. 35(5): 27–34.

Lakshman, G. 1987. Ecotechnological opportunities for aquatic plants—a survey of utilization options. In: Reddy, K.R.; Smith, W.H., eds. Aquatic plants for water treatment and resource recovery. Orlando, FL: Magnolia Publishing Inc.: 49–68.

Metcalf and Eddy Inc. 1991. Wastewater engineering. Treatment, disposal and reuse. 3rd ed. New York: MacGraw Hill Inc.: 109–110.

Wolverton, B.C. 1990. Aquatic plant/microbial filters for treating septic tank effluent. In: Hammer, D.A., ed. Constructed wetlands for wastewater treatment. New York: Lewis Publishers: 173–185.

REHABILITATION OF COASTAL WETLAND FORESTS DEGRADED THROUGH THEIR CONVERSION TO SHRIMP FARMS

Peter R. Burbridge and Daniel C. Hellin[1]

Abstract—International demand for shrimp has stimulated large-scale conversion of mangrove and other coastal wetlands into brackish water aquaculture ponds. Poor site selection, coupled with poor management and over-intensive development of individual sites, has led to nonsustainable production and often, wholesale abandonment of ponds. This has been followed by further conversion of wetlands in an attempt to maintain aquaculture production, incomes, and employment. This has also often proved nonsustainable. The net result is that extensive areas of formerly biologically rich and productive wetland forest are lying idle. In limited cases, natural regeneration of wetlands is taking place, and there are sporadic attempts to stimulate regeneration. However, the drive to convert further wetlands is far greater than efforts at rehabilitation. The development of alternative, sustainable uses of former wetland forests is examined as a means of reducing the pressures to convert further areas of wetland forest.

INTRODUCTION

Burgeoning human populations with inherent needs for food and income continue to drive the settlement and exploitation of coastal regions. Some 20 percent of the human population (over 1 billion people) live within 30 km of the coast (Gommes and others 1998). In tropical developing countries, approximately 90 percent of fishery landings come from shallow coastal waters and provide for 40 to 90 percent of national animal protein consumption (Holdgate 1993). In Asia, it has been estimated that one billion people depend exclusively on fish for their protein requirements (Anon. 1985). Population densities in excess of 1,000 people per square kilometer are commonplace in rural areas of developing countries in Asia, the Pacific, Central America, and the Caribbean (Lundin and Linden 1993). Such population densities place adverse pressures on coastal zones, which are used for settlement, transport, waste disposal, agriculture, forestry, aquaculture, and fishing. In many cases, the sustainable levels of exploitation in fisheries, harvest of mangroves, or the use of the assimilative capacity of wetlands to deal with sewage has long been exceeded. Competition for and overuse of renewable resources have been identified as a major problem in many regions. There are also mounting conflicts between different forms of resources development that reduce the effectiveness of investment and threaten the sustainability of resource production. For example, logging and mining activities in upland areas of Southeast Asia have brought short-term and localized economic benefits to the communities involved. At the same time, they have caused major environmental damage and imposed negative economic impacts that damaged capture fisheries, aquaculture, and tourism interests in lowland areas (Chou and others 1991, Hodgeson and Dixon 1988).

We are therefore facing a very serious challenge in most coastal regions where rapid growth in coastal populations, rapid urbanization, competition for land and water resources, and pollution are undermining the potential of coastal zones to sustain social and economic development objectives. One such objective is sustainable protein production through wise fisheries management and the development of aquaculture. However, throughout the world, coastal ecosystems believed to play a significant role in supporting fisheries and aquaculture are being lost or severely degraded. Coastal wetland forests play such a role but are not being effectively conserved. For example, estimates suggest that 70 percent of Thailand's mangrove has been converted to industrial sites, agriculture, and shrimp ponds. A policy to encourage the conversion of mangrove and other coastal ecosystems into shrimp ponds was stimulated by the reduction of open-access fishing areas as a result of the declaration of Exclusive Economic Zones by countries bordering the Gulf of Thailand. The policy aimed to relieve unemployment among fishermen and to increase fisheries production. However, little or no account was taken of the impact of the loss of mangrove in terms of decreasing support of capture fisheries or aquaculture. This lack of attention to such a basic issue may undermine the benefits derived from aquaculture when good planning and management could avoid a conflict of interests between forestry, aquaculture, and capture fisheries.

With the mounting pressures on coastal areas and resources, it is increasingly important to strive for the sustainable use of ecosystems and the renewable natural resources they generate. However, this takes sophisticated development planning and careful management of development activities. In the case of shrimp-pond based aquaculture development, we are facing two distinct sets of issues affecting the sustainability of production. The first is the adverse impact of other forms of development on aquaculture; the second is the poor siting and management of the aquaculture units.

[1] Burbridge, Professor, and Hellin, MSc, Department of Marine Sciences and Coastal Management, University of Newcastle, Newcastle upon Tyne, NE1 7RU, England.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

Aquaculture has a legitimate right of access to and use of coastal resources as well as a right not to be adversely affected by the poor planning and management of other forms of coastal development. These issues have been addressed in a comprehensive manner in recent reports of the Working Group on Environmental Interactions of Mariculture (ICES 1995, 1996, 1997, 1998, 1999). The second set of issues has been addressed by FAO (Barg 1992) and numerous other technical groups and individual experts. For example, as early as 1981, Juliano and others (1981) were able to demonstrate that production from shrimp ponds in the Philippines could be doubled through marginal improvements in the management of ponds using existing knowledge, thus reducing the need to expand the area of mangrove converted to aquaculture.

Agencies such as the World Bank and FAO are addressing basic institutional and technical issues concerning the establishment of coastal management in developing nations. However, no international organization has yet developed proactive approaches or practical guidelines for the integration of aquaculture into coastal planning and management activities in both developing and developed nations. Where integration is being attempted, it is based upon local or regional planning within individual nations (ICES Working Groups on CZM 1995). As a result, opportunities for promoting the sustainable growth and diversification of coastal aquaculture are not being fully realized, and options for future development are being foreclosed.

Despite the feasibility of greatly enhancing the sustainable production of aquaculture through integrated development planning and sound management, there has been a rapid and largely unplanned expansion of shrimp aquaculture in developing nations. As a consequence, there has been large-scale destruction of mangrove and other wetland systems with consequent impacts on human activities that are supported directly, or indirectly, by their ecosystem functions. A second consequence has been the cessation of production in extensive areas of aquaculture sites. Land is frequently lying idle, yet there is continuing pressure to convert more wetlands in an attempt to maintain aquaculture production. This trend cannot be sustained, and it is argued that rehabilitation of disused and unproductive aquaculture sites presents a positive means of bringing these lands back into some form of productive and sustainable use while reducing the pressure to convert remaining productive wetlands. This presents a major challenge to all of us involved in forestry, aquaculture, capture fisheries, and other aspects of human development in coastal regions.

THE ISSUE OF NONSUSTAINABLE CONVERSION OF COASTAL WETLANDS TO AQUACULTURE

The actual scale of conversions of mangrove and other wetlands to aquaculture is very difficult to document. It has been estimated that over 15 million ha of mangrove had been cleared. This accounts for approximately half of the mangrove that had previously existed, and this is thought to be further decreasing at a rate of between 2 and 8 percent per year (Kunstadter and others 1985). In the Philippines, 279,000 ha of mangrove were lost between 1951 and 1988, and conversion to culture ponds accounted for approximately 50 percent of this loss (Primavera 1995).

In addition to areas converted from mangrove to shrimp ponds, other areas have been converted from salt flats, salt marshes, freshwater wetlands, fishponds, rice paddies, or agricultural lands. Frequently, ponds were built in regions already degraded by other practices and already in need of active management.

Aquaculture of marine species has been a well-established feature of coastal development for more than a thousand years in parts of Asia. An example of this is the polyculture of milkfish and shrimp in “Tambaks” (brackish water ponds) in Indonesia. Most of these systems were extensive in nature and many integrated mangrove conservation into the pond management system as both a means of enriching the food supply in the ponds and of protecting ponds from coastal storms. The traditional extensive or semi-intensive pond systems have largely given way to intensive systems, which require very careful site selection and high levels of inputs of feed and other materials to balance water chemistry and maintain the health of the cultured species. The capital investment, technical skills, and knowledge of markets required to sustain production in intensive aquaculture systems often puts them beyond the reach of most rural communities. Where they are attempted without the requisite inputs and protection from adverse external impacts, production cannot be sustained, and they become disused.

The Scale of Disuse of Aquaculture Sites

Accurate estimates of pond disuse (in both mangrove and nonmangrove areas) are difficult to obtain because land tenure records are often unreliable and out of date, and assessments using remote sensing are hampered by the inability to discern between productive and disused ponds (Stevenson and others 1999). Unofficial estimates of pond disuse have suggested that the percentage of ponds left idle after a period in production can be as high as 70 percent (Stevenson 1997). Attempts to quantify the scale of pond disuse have been marred by the belief that an admission of pond abandonment is tantamount to an admission of management failure, and, to date, comprehensive surveys of disused shrimp ponds have not been undertaken (Stevenson and others 1999). In practice, ponds are often converted to other uses; for instance, in Thailand, some ponds have been sold for housing and industrial development, converted to salt farms or fish or crab culture operations, and some shrimp farmers have sold topsoil for construction projects (Stevenson 1997).

In Thailand, Potaros (1995) stated that 19 900 ha of shrimp farms in the five provinces of the Inner Gulf of Thailand were closed in 1990–91. A report produced by the Network of Aquaculture Centres in Asia-Pacific (NACA) indicates that in 1989 about 62 percent of farms were operating “under capacity” and another 22 percent of farms were “abandoned” in Samut Sakhon province (Office of Environmental Policy and Planning - OEPP 1994). This is supported by Briggs and Funge-Smith (1994) who reported that an area of 40 000 to 45 000 ha south of Bangkok became derelict after shrimp production collapsed in 1989–90.

Yap (1997) reports that nearly all of the 54 912 ha of shrimp ponds in the Philippines were abandoned, and another 83 000 ha of brackish water ponds were "idle". Reports from NGOs in the Philippines in late 1997 stating that pond disuse is common in the Philippines have supported this, although pond operators have frequently returned to traditional forms of milkfish [*Chanos chanos* (Forsskal)] culture after shrimp production has ceased (Stevenson and others 1999).

The disuse or abandonment of coastal aquaculture sites has been reported elsewhere but not quantified. Extensive areas of disused shrimp ponds are thought to exist in Bangladesh, China, Malaysia, and Colombia (Stevenson 1997) and, more recently, Mexico.

Causes of Disuse and Abandonment of Ponds

There are numerous reasons for the cessation of production in shrimp ponds. Examples include:

- poor site selection (reported in Sri Lanka by Jayasinghe 1995);
- flooding due to poor catchment management as well as from storm surges where the buffer function of mangrove has been lost due to their removal;
- predation by nontarget species in the ponds, e.g., birds and other animals;
- poor cohesivity of soils, which causes the pond walls to collapse;
- acidification of soils and water as a result of the exposure of potential Acid Sulfate Soils (reported in Vietnam by Tuan 1996, in Cambodia by Sreng 1996);
- contamination of pond water from agricultural wastes (noted in Indonesia as a result of a shortage of fresh water and problems of water quality by Burbridge pers. obs. 1997); and
- diseases resulting from a lack of hygiene, which can be rapidly transmitted among ponds through poor water management, for example, reported in India as a result of white spot disease (Sammut and Mohan 1996), in Sri Lanka (Jayasinghe 1995), in the Philippines (Ogburn and Ogburn 1994), and in Taiwan (Stevenson and others 1999).

Briggs and Funge-Smith (1994) were among the first to highlight the problem of poor hygiene and diseases in a report to the British Overseas Development Agency (now the Department for International Development, DFID). Hambrey (1996) reported that chronic disease and water-quality problems have caused 'significant' pond abandonment. For instance, disease problems have caused abandonment in India, (Sammut and Mohan 1996), the Philippines (Yap 1997), and Thailand (Macintosh 1996). Poor water quality and poor site selection have caused production failure in Sri Lanka (Jayasinghe 1995) and Indonesia, and problems with Acid Sulfate Soils (A.S.S.) have caused abandonment in Vietnam (Tuan 1996) and Cambodia (Sreng 1996). These problems often lead to financial difficulty, causing farmers to either sell or abandon their farms (Fegan 1996). Ponds may also be abandoned due to a drop in profits or yields (Flaherty and Karnjanakesorn 1995), or political intervention, such as the revoking of leases or license agreements (Stevenson and others 1999).

Where the development of coastal aquaculture has not been well planned and managed, it undermines the potential for coastal zones to sustain economic and social development. This raises the question of whether it would be better to rehabilitate these areas and restore the original ecosystem, or to find a means of modifying the aquaculture system to allow it to be more productive and sustainable. There are good arguments for both alternatives. On one hand, the rehabilitation of the original ecosystem may help to rejuvenate coastal capture fisheries stocks and the income of fishermen, improve biodiversity, ecotourism, and reduce salinization of soils and groundwater, which adversely affect agriculture and domestic water supplies. However, it must be realized that rehabilitation will cost money, will take considerable time, and may not be welcomed by local people who may see little benefit for themselves.

On the other hand, developing more productive use of unsuccessful aquaculture sites could allow the original developers to achieve a reasonable return on their investment and could provide opportunities to diversify and expand local employment. Mixed aquaculture systems may help improve the food security of rural communities and reduce organic pollution loads from other forms of development, including more intensive aquaculture. One example might be the change from nonprofitable intensive or semi-intensive shrimp culture to a less intensive polyculture system or an integrated aquaculture-agriculture forestry system.

These and other alternatives will vary from place to place depending on environmental, social, and economic conditions. At this meeting, it would be useful to discuss how to develop a system to identify and evaluate opportunities to put nonproductive and idle aquaculture sites into a more productive use that helps to meet sustainable development objectives at both a local and national level. In the following paragraphs, factors that influence options for rehabilitating disused aquaculture sites are set out.

OPTIONS FOR THE REHABILITATION OF DISUSED OR ABANDONED AQUACULTURE PONDS OR BOTH

The term restoration has been adopted in recent studies to mean any activity that aims to return a system to a pre-existing condition, whether or not this was pristine (sensu Lewis 1990b). Whereas, the term rehabilitation is used to denote any activity that aims to convert a degraded system to a stable alternative use, which is designed to meet a particular management objective (Stevenson and others 1999). The term rehabilitation is used here to describe a continuum of management options for altering the state of the ponds to some alternative condition where human activities can be sustained. This can include the reinstatement of a wetland forest ecosystem such as a mangrove, where uses foreclosed through the conversion of the wetland can be regained and benefit a range of different interests. For example, mangrove forests have been restored to meet commercial purposes such as silviculture (Watson 1928), for restoring fisheries habitat (Aksornkoea 1997, Lewis 1992), for sustainable multiple community use purposes, or for shoreline protection purposes. None of these are mutually exclusive.

There are three basic rehabilitation options. The first is to rehabilitate the pond sites so that they can be put back into sustainable shrimp production. The second is to rehabilitate the pond sites so that they can be put to some alternative, sustainable use. The third option is to restore the environmental conditions within the pond sites and the surrounding area, and to re-establish a productive wetland ecosystem (Stevenson and others 1999). Each of these options is influenced by the causes of production failure and the conditions that remain in the pond after production has ceased (Stevenson 1997). The continuum of rehabilitation options is illustrated in figure 1.

Factors That Influence the Choice of Rehabilitation Options

The basic rationale for attempting rehabilitation is to address factors that alter a wetland or other productive coastal ecosystem to such an extent that it can no longer self-correct or self-renew. Under such conditions, ecosystem homeostasis has been permanently stopped, and the normal processes of secondary succession (Clements 1928, Watson 1928) or natural recovery from damage are inhibited in some way.

Before any restoration is attempted, the goals should be determined through an active dialogue with, and effective participation of, the local stakeholders. The stakeholders comprise both individuals or interest groups that have promoted aquaculture, as well as those who may have been influenced by the conversion of wetlands and other systems. Those with interests in aquaculture may have invested scarce capital, as well as their labor and materials, to develop what they saw as a means of improving their welfare. In the process of developing the aquaculture sites, they may have attained user rights or title to the land. They will want to see some productive outcome from restoration efforts that will help them achieve sustainable return on their investment. On the other hand, people who did not benefit from the aquaculture development may have suffered a loss of access to common property resources such as the crabs that breed and thrive in mangrove forests or fish and crustaceans that depend upon wetlands for part of their life cycle. Balancing the needs and aspirations of the different stakeholders and achieving broad agreement on the goals are critical to the sustainability of rehabilitation initiatives. The importance of this involvement of stakeholders is often overlooked, and, without local support, restoration and rehabilitation programs have little chance for long-term success (Primavera and Agbayani 1996).

It may be possible to restore the functionality of a system even though factors such as soil type and condition may have been altered with consequent effects on the flora and fauna. If the goal is to return an area to pristine condition, then the likelihood of failure is high and must be regarded as unrealistic (Stevenson and others 1999). The restoration of certain ecosystem traits and the replication of some functionality stand more chance of success (Lewis and others 1995).

The reference conditions for restoring the functions of a system also need to be examined carefully because “pre-existing conditions” may not have been pristine due to factors outside those associated with the conversion to aquaculture. Potaros (1995) reports that in Thailand, “the mangrove area that has been used for shrimp farms is often previously degraded forest so it is very difficult to assess the economic damage related directly to shrimp farming”. In such cases, it is not sensible to return the area to a previously degraded condition. The goal may be to achieve a level of functionality that will allow productive alternatives to be developed and sustained. These alternatives could include mangrove timber production or development of new forms of aquaculture.

Additionally, restoration to the original habitat type may not be the best social, economic, or ecological option. For example, rather than restoring the area to a relatively common ecosystem type, it may be preferable to restore not to the original condition, but rather to a scarce type of habitat within the ecosystem (Cairns 1988, Lewis Environmental Services, Inc. and Coastal Environmental, Inc. 1996). This is also known as “out-of-kind” restoration (Stevenson and others 1999).

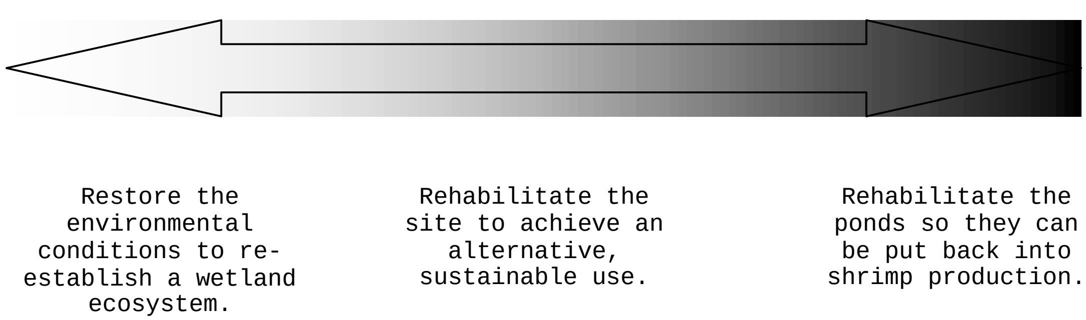


Figure 1—Diagram illustrating the continuum of options that are available for the rehabilitation of disused or abandoned aquaculture ponds.

Some areas may not be considered suitable for restoration. In such cases, the best practical option may be to re-establish shrimp farming in a well-managed and sustainable manner. Similarly, the primary redevelopment goal of coastal managers may be to recommence shrimp farming in disused ponds.

In an area subject to storm activity, the primary goal may be to restore natural coastal protection afforded by the buffer function served by mangrove. Different goals are not necessarily mutually exclusive. Rehabilitation of a degraded mangrove can help to sustain shrimp aquaculture by acting as a buffer to storm surges and reducing flooding of ponds. It can help to provide nutrients to ponds as well as helping to assimilate wastes drained from ponds.

The choice of options can be limited by the information available on key factors that influence the costs and technical difficulties of rehabilitating ponds. For example, ponds sited in areas where Acid Sulfate Soils have been exposed to air pose difficult problems of restoring neutral soil conditions without costly liming or leaching the acids by flushing, which can impose unacceptable impacts on neighboring rivers and estuaries. There are fundamental gaps in the information available on the environmental effects of leachate from Acid Sulfate Soils and the persistence of chemicals, disinfectants, and other materials used during shrimp culture. If new and innovative treatments develop, then a variety of new alternatives may be open to the coastal manager for consideration (Stevenson and others 1999).

FACTORS THAT INFLUENCE OPTIONS FOR THE REHABILITATION OF MANGROVE

Factors influencing the rehabilitation of mangrove were reviewed by Chapman (1976), Field (1997), Hamilton and Snedaker (1985), Lewis (1982, 1990a, 1990b), Lugo (1992), Noakes (1951), Saenger and Siddiqi (1993), Siddiqi and others (1993), and Watson (1928).

Some of the main factors that have a major influence on both the sustainability of aquaculture and the rehabilitation of disused sites include:

- the presence of Potential Acid Sulfate Soils (P.A.S.S.);
- erosion and subsidence of soils and associated increased risks of flooding;
- waterlogging of soils; and
- the presence and longevity of antimicrobials and other chemical compounds used in aquaculture operations.

There is a shortage of studies that quantify the environmental conditions in disused shrimp ponds and their effect on surrounding areas. Disused or abandoned ponds or both may represent heavily degraded environmental conditions that could prove technically difficult, expensive, and time consuming to rehabilitate. In some case, environmental changes brought about by developing ponds in inappropriate situations may be irreversible. For example, it is not possible to reverse the formation of Actual Acid Sulfate Soils (A.A.S.S.) resulting from the oxidation of Potential Acid Sulfate Soils (P.A.S.S.) caused by excavating ponds. However, it may be possible to transform A.A.S.S. to a pH neutral soil that has less acid-producing capacity.

Great care must be taken in choosing methods of dealing with A.A.S.S. Methods commonly recommended by agriculturists involve flushing the acids from the soil with copious amounts of either fresh or brackish water, for example, in Agency for Agricultural Research and Development and the Land and Water Research Group (1990). Although this may reduce the acidity in local areas, it does not address the issue of the impact of acids draining into adjacent streams, rivers, or estuaries. Field observations by one of the authors in the Mekong Delta indicated acid levels as low as pH 2.3 in canals dredged in P.A.S.S. in an attempt to reclaim Melaleuca wetland forests for agriculture. P.A.S.S. can extend over large areas and represents a very large reservoir of potential acid materials. Flushing of acids from pond sites may in the short term appear to be a viable solution. However, unless water levels are maintained above the P.A.S.S. layers, oxidation will continue with the result of continuing leaching of acids. Maintaining water levels in the soil profile can be very difficult where groundwater is being extracted to irrigate crops or to moderate salinity and temperatures in aquaculture sites.

With the clearance of trees and other vegetation to create ponds, surface erosion and subsidence and compaction of the soil profile can take place. This can have a series of knock-on effects, including increased risks of flooding from river systems as well as increased vulnerability from oceanic storm surges. These conditions are very difficult to reverse.

Additionally, it is difficult to counteract the adverse effects on surrounding areas resulting from clearance of mangrove and other wetland forest types. Lahmann and others (1987) discuss the impact of shrimp aquaculture siting in basin mangrove forests and “salitrales” (salt flats) in the southern Gulf of Guayaquil, Ecuador. The basin forests are often characterized as “unproductive” in major shrimp aquaculture countries like Thailand, but Lahmann and others (1987) point out that the local“...declining abundance of shrimp postlarva in Ecuadorian estuaries...” may at least in part be due to “...the disproportionate elimination of sources of dissolved organic matter...” and “...may be the dominant cause of the reduction in wild shrimp postlarva stocks...” (p. 242).

There is evidence that mangrove forests around the world can recover from a degraded or heavily harvested condition if: (a) normal freshwater flows entering the mangrove are not disrupted, (b) the normal tidal hydrology is not disrupted, and (c) the availability of waterborne seeds or seedlings (propagules) from adjacent mangrove stands is maintained. Through afforestation, mangroves can also be established on unvegetated, intertidal flats where they would not normally grow. These areas, however, are limited in extent and often serve other ecological purposes such as feeding areas for wading birds like herons and egrets. They may also support valuable submerged aquatic vegetation like seagrass meadows that are a valuable marine habitat in their own right (Phillips and McRoy 1980).

Based on the ability of mangrove to restore itself, it has been recommended that plans for restoration should first look at the potential existence of stresses such as blocked tidal inundation, which might prevent secondary succession from occurring and to remove those stresses before attempting additional restoration (Hamilton and Snedaker 1985, Lugo 1992). The identification of stresses and evaluation of their effects on natural regeneration are not simple tasks. Studies in Panama (Duke 1996) and Indonesia (Soemodihardjo and others 1996) report successful natural regeneration of mangrove subjected to stresses from oil spills and logging, respectively. In Panama, Duke (1996) observed that "...densities of natural recruits far exceeded both expected and observed densities of planted seedlings in both sheltered and exposed sites..." (p. 228). Soemodihardjo and others (1996) report that only 10 percent of a logged area in Tembilahan, Indonesia, needed replanting because "...the rest of the logged over area...had more than 2,500 natural seedlings per hectare" (p. 109). In cases where natural recovery is not occurring through natural seedling recruitment, assisting natural recovery through planting is one option.

Unfortunately, many mangrove restoration projects move immediately into planting of mangroves without determining if natural recovery is taking place and which stresses may prevent natural regeneration and the likely effectiveness of replanting. This often results in major failures of planting efforts (Stevenson and others 1999). For example, Sanyal (1998) has recently reported that between 1989 and 1995, over 9000 ha of mangroves were planted in West Bengal, India, with only a 1.52-percent success rate. The World Bank funded Central Visayas Regional Project-I, Nearshore Fisheries Component in the Philippines that targeted 1000 ha for mangrove planting between 1984–92. An evaluation of the success of the planting in 1995–96 (Silliman University 1996) indicated only 18.4 percent of 2,927,400 mangroves planted over 492 ha had survived. Another planned 30 000-ha planting effort in the Philippines (Fisheries Sector Program, 1990–95) funded by a $150 million loan from the Asian Development Bank and the Overseas Economic Cooperation Fund of Japan was cut short after only 4792 ha were planted due to similar problems (Ablaza-Baluyut 1995).

Restoration of Disused Aquaculture Ponds to Mangrove Forests

Detailed studies by Lewis and Marshall (1997) of pond restoration at field sites in Central America and the Philippines indicate that the most important factor in successful restoration of ponds to mangroves is the re-establishment of the tidal hydrology to the maximum extent possible. Restoration of mangroves through restoring the natural hydrology has been emphasized before by Hamilton and Snedaker (1984), Lugo (1992), and by Olsen and Arriaga (1989). Turner and Lewis (1996) also give examples of successful restoration of mangroves through restored hydrology alone.

Lewis and Marshall (1997) have suggested five critical steps to achieve successful mangrove restoration in general and pond restoration in particular:

- understand the autecology (single species ecology) of the mangrove species at the site, in particular the patterns of reproduction, propagule distribution, and successful seedling establishment;
- understand the normal hydrologic patterns that control the distribution and successful establishment and growth of targeted mangrove species;
- assess the modifications of the previous mangrove environment that occurred and that currently prevent natural secondary succession;
- design the restoration program to initially restore the appropriate hydrology and utilize natural volunteer mangrove propagule recruitment for plant establishment; and
- only utilize actual planting of propagules, collected seedlings, or cultivated seedlings after determining, through steps 1 through 4, that natural recruitment will not provide the quantity of successfully established seedlings, or rate of stabilization, or rate of growth of saplings, established as goals for the restoration project.

These critical steps are often ignored, and failure in most restoration projects can be traced to proceeding directly to step 5 without considering steps 1 through 4. Lewis and Marshall (1997) refer to this approach as "gardening", where simply planting mangroves is seen as all that is needed.

SELECTING APPROPRIATE REHABILITATION OPTIONS

The selection of the most appropriate options for rehabilitation depends heavily upon local as well as national, economic, social, and environmental priorities. From an economic viewpoint, assessing the value of the flows of resources derived from the former ecosystem provides an important benchmark against which to assess the relative costs and benefits that can be derived from different rehabilitation options. Such benchmarks should incorporate both the tangible economic goods and services as well as the less tangible environmental goods and services provided by the wetland forest or other ecosystems before they were subjected to conversion. These environmental goods and services include flood and storm protection, sediment and toxicant removal, erosion mitigation, and nutrient export.

Of particular importance are the benefits provided by mangrove in coastal protection from typhoons or hurricanes. In areas subject to strong storm activity for West Bengal, Bangladesh, and Mozambique, the removal of mangrove to form shrimp ponds has made coastal activities more vulnerable to storm surges and flooding. In some states in India, e.g., Andrha Pradesh, the buffering capacity and the erosion mitigation provided by mangrove is particularly important because the costs associated with the construction of artificial structures to combat erosion and to protect from storm damage can total more than.$12,000 (U.S.) per meter. In such areas, the case for restoration may be very strong, and the costs involved in restoration may be small when compared with the costs of constructing these artificial structures. The benefits in such cases would be both financial, in terms of costs avoided, and social, in terms of the avoidance or minimization of the loss of human life and property.

There are well-established economic techniques for undertaking such analyses. For example, see section 6 of the Mangrove Area Management Handbook (Hamilton and

Snedaker 1985). Mangroves are regarded as particularly rich in terms of the goods and services they provide (Burbridge 1990, Dixon 1989, Hamilton and Snedaker 1984) and some researchers have made estimates of the economic benefits of mangroves both in terms of fisheries and other benefits. There is less information available on the economic value of other land-use types or other habitats in tropical regions, e.g., salt pans, tidal swamp forests, Melaleuca wetlands, or mud flats.

However, well-constructed economic cases that include a full range of social as well as environmental factors for different rehabilitation options are seldom undertaken. Justifications for rehabilitation such as those by Sidall and others (1985) who state with reference to Ecuador, Panama, and the Philippines that “reclamation of abandoned ponds should be encouraged... and poorly sited or engineered ponds should be breached to promote eventual recolonization” would benefit from rigorous assessment of the benefits of different rehabilitation options. This also applies to justifications for rehabilitating general aquaculture related degradation in mangroves. For example, Ishwaran (1996) argues that given the importance of mangroves, there is a need to rehabilitate degraded areas through planting and the introduction of environmentally friendly aquaculture technology.

The case for rehabilitating disused shrimp ponds should also be viewed objectively. The direct loss of shrimp sales revenue from ponds, which fail due to poor management, can be in the region of.$15,000 to $25,000 (U.S.) per hectare per year. The costs of the system degrading further and posing a risk to neighboring habitats or land-use types should also be taken into consideration. Damaged resources are often unstable and actively deteriorating, and, in general, if deterioration is not arrested, repair may become progressively more expensive and difficult; i.e., redevelopment costs must be balanced against ‘costs avoided’ (Stevenson and others 1999).

There can be added benefits from the rehabilitation of degraded pond sites to an alternative income-generating activity. For example, by redeveloping an area that has become degraded and helping stakeholders to regain productive use of the sites, pressure will be reduced on neighboring areas, which are perceived to be of high value for conversion and are at risk of degradation. The resulting re-establishment of production of useful goods and services in a rehabilitated or restored area may serve to help maintain the flows of economic and environmental goods and services provided by undisturbed ecosystems. This will allow these systems to continue to sustain other forms of human activity, as well as meeting international obligations such as maintaining biological diversity.

Where ponds were converted from other land-use types, such as fish ponds, paddy fields, or other agricultural lands, socio-economic reasons may exist for restoring ponds to their prior uses, particularly where local reliance on subsistence agriculture or aquaculture exists. However, there is little in the literature that supports restoration to prior land-use types. Where restoration to a ‘natural’ habitat (such as mangrove) is considered inappropriate, technically very difficult, or too expensive, then restoration to prior or other land-use types may be considered. This may represent the best option in terms of economic feasibility, environmental acceptability, and maximal sustainable productivity (Stevenson and others 1999).

Where ponds have been constructed in inappropriate locations, such as those frequently hit by typhoons or hurricanes, there is a strong case for not rehabilitating ponds to alternative uses that also will be vulnerable and difficult to sustain. In cases where areas which are not normally hit by hurricanes but experience El Niño years, for instance, along the Pacific Coast of Latin America and the United States, there can be a case for rehabilitation (or even restoration) of ponds if the benefits outweigh the risks. In such cases, management practices can reduce the vulnerability of aquaculture production to periodic events.

Selection of Rehabilitation Options and Their Implementation in Practice

There are limited references to the practical restoration of abandoned shrimp ponds. Given the extensive areas of mangrove converted unsuccessfully to shrimp ponds, there have been surprisingly few reports of attempts to restore aquaculture ponds back to mangroves. Anecdotal reports of up to 13 000 ha of ponds restored in Thailand and several thousand hectares in Vietnam have been noted by Stevenson and others (1999). It is significant to note that Field (1997), in compiling and editing reports on mangrove restoration from 13 countries including Thailand, Malaysia, Vietnam, Indonesia, India, Pakistan, and Bangladesh, does not report a single occasion where pond restoration was attempted. In fact, the only mention of aquaculture ponds in this work is by Aksornkoae (1997), in which mangroves were restored between existing shrimp aquaculture ponds in Pattani Province, Thailand. In several areas in Thailand (and probably elsewhere), mangrove has been planted in bund walls and in areas adjacent to ponds as part of the aquaculture management system in order to stabilize sediments and to improve water quality.

Stevenson and others (1999) observed that pond rehabilitation is an ongoing concern, and there are probably many localized efforts to either reforest ponds in former mangrove areas or to put disused ponds to alternative uses. However, the data that do exist are often of poor quality, or they are poorly disseminated and difficult to obtain or verify. Consequently, it is not possible to draw any substantive conclusions from them. The authors concluded that in most cases, evaluation procedures or assessment of pond condition do not take place prior to the initiation of pond rehabilitation projects. Therefore, the reasons behind either the success or failure of pond rehabilitation projects are not known, and there is no ‘learning curve’ or lessons learned from these endeavors. This is not a rational way to continue (Stevenson and others 1999).

CONCLUSIONS

Aquaculture forms an important activity in many of the world’s coastal regions, and its importance as a source of income, employment, and exports is likely to continue to expand for the foreseeable future. Aquaculture of a generally nonintensive nature has been a part of coastal land and water use for many centuries in Asia, and it has proven sustainable. However, the rapid expansion of poorly planned and managed semi-intensive and intensive shrimp aquaculture has created a number of significant adverse environmental, economic, and social effects. In turn, shrimp aquaculture has often been adversely affected by impacts from other forms of human development.

The combined effects of poor standards of aquaculture development and adverse impacts on aquaculture operations have led to unnecessary destruction of coastal wetland forests, nonsustainable aquaculture production, disuse of pond sites, and even abandonment of the land and the loss of the investment of peoples' labor and capital. Reluctance on the part of the aquaculture industry and government to admit that shrimp aquaculture has often proven nonsustainable has helped to disguise the extent of unproductive shrimp farming. However, extensive areas are believed to be involved in many of the poorer developing nations such as India, the Philippines, and Indonesia.

Developing nations cannot afford to allow the extensive areas of nonsustainable, unproductive shrimp farms to remain idle if they are to meet the development needs of coastal communities, as well as national development objectives and international obligations such as the conservation of biodiversity. Substantial scientific effort is required to analyze the factors that lead to nonsustainable aquaculture and to help the 50 or more shrimp-producing nations find ways to rehabilitate unproductive, disused, and abandoned areas. This will serve both to help people to develop sustainable uses of coastal areas and resources already committed to aquaculture and to reduce pressures on remaining coastal ecosystems and the renewable resources they generate.

Options for the rehabilitation of areas that can no longer sustain shrimp production need to be identified, tested, evaluated, and demonstration projects established as a means of engaging the interest and active support of stakeholders. Efforts to improve the dialogue between donors, researchers, aquaculturists, and governmental bodies will be essential in developing workable solutions. Unfortunately, much of the research into pond rehabilitation carried out to date has been conducted without adequate site assessment, without documentation of the methodologies or approaches used, and often lacks subsequent follow up or evaluation. Those projects, which have not been successful, are rarely documented and information on them is largely anecdotal and hard to obtain. The reasons for success or failure are still largely guesswork, and we are still at the beginning of what may prove to be a steep 'learning curve.'

To reduce the length of time required to address this learning curve and avoid further unnecessary damage to wetland forests, consideration should be given by the international scientific community and donors to placing greater emphasis upon working with developing nations where these issues are acute and where poverty and lack of effective development assistance will drive people to degrade more areas through repeating mistakes of the past.

The potentially adverse effects of new aquaculture development can generally be avoided through good planning and management. To be fully successful, such plans and management arrangements must recognize that aquaculture should have equal rights of access to and use of natural resources and a good quality environment. It is suggested that Integrated Coastal Zone Management can provide a beneficial framework for the development of aquaculture where due care and attention are given to the maintenance of the functional integrity of coastal ecosystems that sustain aquaculture and other natural resource-dependent activities. It is also suggested that multiple use management of coastal ecosystems will provide a better basis for integrating aquaculture with other activities, which have a common dependence on the functions and resources provided by one or more coastal ecosystems.

ACKNOWLEDGMENTS

This paper draws heavily on papers prepared by the first author in collaboration with N.J. Stevenson, R.R. Lewis, A.J. Edwards, and S. Jennings, and we wish to acknowledge the value of their work in supporting this paper.

REFERENCES

Anon. 1985. The state of food and agriculture 1984. FAO Agric. Ser. 18. Rome: Food and Agriculture Organization of the United Nations. 185 p.

Ablaza-Baluyut, E. 1995. The Philippines fisheries sector program. In: Proceedings of a workshop on coastal and marine environmental management; 1995 March 27–29; Bangkok, Thailand. Bangkok, Thailand: Manila, Philippines: Asian Development Bank: 156–177.

Agency for Agricultural Research and Development and the Land and Water Research Group. 1990. Workshop on acid sulphate soils in the humid tropics; 1990 November 20–22; Bogor, Indonesia. 392 p.

Aksornkoae, S. 1997. Reforestation in mangrove forests in Thailand-a case study in Pattani Province. In: Field, C.D., ed. Restoration of mangrove ecosystems. Okinawa, Japan: International Society for Mangrove Ecosystems: 53–63.

Barg, U.C. 1992. Guidelines for the promotion of environmental management of coastal aquaculture development. FAO Tech. Pap. 328. Rome: Food and Agriculture Organization of the United Nations. 122 p.

Briggs, M.R.P.; Funge-Smith, S. 1994. Unsustainable shrimp culture-causes and potential solutions from experience in Thailand. In: Development of strategies for sustainable shrimp farming. Rep. to the Overseas Development Administration, Res. Pro. R4751. App. 1. Stirling, UK: Institute for Aquaculture, Stirling University. 31 p.

Burbridge, P.R. 1990. Tidal wetland resources in the tropics. Journal of Resource Management and Optimization. 7(1–4): 115–138.

Cairns, J. 1988. Restoration ecology: the new frontier. In: Cairns, J., ed. Restoration of damaged ecosystems. Boca Raton, FL: CRC Press: 1–12. Vol. 1.

Chapman, V.J. 1976. Mangrove vegetation. Vaduz, Germany: J. Cramer. 447 p.

Chou, L.M.; Chua, T.E.; Khoo, H.W. [and others], eds. 1991. Towards an integrated management of tropical coastal resources. In: International center for living aquatic resources management (ICLARM): Conference proceedings 22; 1988 October 28–31; Singapore. Manila, Philippines: International Center For Living Aquatic Resources Management. 455 p.

Clements, F.E. 1928. Plant succession and indicators. New York: H.W. Wilson Co. 453 p.

Dixon, J.A. 1989. Valuation of mangroves. Tropical Coastal Area Management. 4(3): 2–6.

Duke, N. 1996. Mangrove reforestation in Panama. In: Field, C.D., ed. Restoration of mangrove ecosystems. Okinawa, Japan: International Society for Mangrove Ecosystems: 209–232.

Fegan, D.F. 1996. Sustainable shrimp farming in Asia: vision or pipedream. Aquaculture Asia. 1(2): 22–28.

Field, C.D., ed. 1997. Restoration of mangrove ecosystems. Okinawa, Japan: International Society for Mangrove Ecosystems. 250 p.

Flaherty, M.; Karnjanakesorn, C. 1995. Marine shrimp aquaculture and natural resource degradation in Thailand. Environmental Management. 19(1): 27–37.

Gommes, R.; de Guerny, J.; Nachtergaele, F.; Brinkman, R. 1998. Potential impacts of sea-level rise on populations and agriculture. Rome: Food and Agricultural Organization of the United Nations.

Hambrey, J. 1996. Comparative economics in land use options in mangrove. Aquaculture Asia. 1(2): 10–14.

Hamilton, L.S.; Snedaker, S.C., eds. 1984. Restoration and establishment. Sec. 111. In: Handbook of mangrove area management. Honolulu: East West Center, International Union for the Conservation of Nature, United Nations Education Social and Cultural Organization (UNESCO), United Nations Development Program (UNDP): 102–108.

Hodgeson, G.; Dixon, J.A. 1988. Logging versus fisheries and tourism in Palawan. Pap. 7. Honolulu: East West Environment and Policy Institute. 95 p.

Holdgate, M.W. 1993. The sustainable use of tropical coastal resources-a key conservation issue. Ambio. 22: 481–482.

ICES. 1995. Principles and practical measures for the interaction of mariculture and fisheries in coastal area planning and management; Rep.: Working Group on Environmental Interactions of Mariculture. Copenhagen, Denmark: Mariculture Committee; International Council for Exploration of the Sea. 28 p.

ICES. 1996. Report: working group on environmental interactions of mariculture. Copenhagen, Denmark: Mariculture Committee; ICES CM 1996; International Council for Exploration of the Sea.

ICES. 1997. Report: working group on environmental interactions of mariculture. Copenhagen, Denmark: Mariculture Committee; ICES CM 1997; International Council for Exploration of the Sea.

ICES. 1998. Report: working group on environmental interactions of mariculture. Copenhagen, Denmark: Mariculture Committee; ICES CM 1998; International Council for Exploration of the Sea.

ICES. 1999. Report: working group on environmental interactions of mariculture. Copenhagen, Denmark: Mariculture Committee; ICES CM 1999; International Council for Exploration of the Sea.

Ishwaran, N. 1996. Global concepts and local practice: UNESCO experience in implementing the biosphere reserve concept and its relevance to integrated coastal resource management. In: Phan Nguyen Hong, Nataragjan Ishwaran, eds. Proceedings of the ECOTONE V seminar on community participation in conservation, sustainable use and rehabilitation of mangrove in South East Asia; 1996 January 8–12; Ho Chi Minh City, Vietnam. Ho Chi Minh City, Vietnam: Agricultural Publishing House of Vietnam: 221–228.

Jayasinghe, J.M.P.K. 1995. Sri Lanka. In: FAO/NACA report on regional study and workshop on the environmental assessment and management of aquaculture development (TCP/RAS/2253). Network of Aquaculture Centers in Asia and the Pacific (NACA), Environment and Aquaculture Development Series 1. Bangkok, Thailand: Network of Aquaculture Centers in Asia and the Pacific; Food and Agriculture Organization of the United Nations: 357–376.

Juliano, R.; Anderson, J.; Librero, A.R. 1981. Coastal zone management in the Philippines. In: Collier, W.; Chia, L.S.; Soysa, C., eds. Man, land, and sea. Bangkok, Thailand: Agricultural Development Council: 219–240.

Kunstadter, P.; Bird, E.C.F.; Sabhasri, S., eds. 1985. Proceedings of the workshop on man in the mangroves. Tokyo, Japan: United Nations University. 117 p.

Lahmann, E.J.; Snedaker, S.C.; Brown, M.S. 1987. Structural comparisons of mangrove forests near shrimp ponds in southern Ecuador. Interciencia. 12(5): 240–243.

Landesmann, L. 1994. Negative impacts of coastal aquaculture development. World Aquaculture. 25(2): 12–17.

Lewis Environmental Services, Inc. and Coastal Environmental, Inc. 1996. Setting priorities for Tampa Bay habitat protection and restoration: restoring the balance. St. Petersburg, FL: Tampa Bay National Estuary Program. 106 p.

Lewis, R.R. 1982. Mangrove forests. In: Lewis, R.R., ed. Creation and restoration of coastal plant communities. Boca Raton, FL: CRC Press: 153–172.

Lewis, R.R. 1990a. Creation and restoration of Coastal Plain wetlands in Florida. In: Kusler, J.A.; Kentula, M.E., eds. Wetland creation and restoration: the status of the science. Washington, DC: Island Press: 73–101.

Lewis, R.R. 1990b. Wetlands restoration/creation/enhancement terminology: suggestions for standardization. In: Kusler, J.A.; Kentula, M.E., eds. Wetland creation and restoration: the status of the science. Washington, DC: Island Press: 417–422.

Lewis, R.R. 1992. Coastal habitat restoration as a fishery management tool. In: Stroud, R.H., ed. Stemming the tide of coastal fish habitat loss: Proceedings of a symposium on conservation of coastal fish habitat; 1991 March 7–9; Baltimore. Savannah, GA: National Coalition for Marine Conservation, Inc.: 169–173.

Lewis, R.R.; Kusler, J.A.; Erwin, K.L. 1995. Lessons learned from five decades of wetland restoration and creation in North America. In: Montes, C.; Oliver, G.; Molina, F.; Cobos, J., eds. Bases ecologicas para la restauracion de humedales en la Cuenca Mediterranea: Proceedings of a meeting; 1993 June 7–11; University of La Rabida, Spain. Junta de Andaluca, Spain: Andaluca: 107–122.

Lewis, R.R.; Marshall, M.J. 1997. Principles of successful restoration of shrimp aquaculture ponds back to mangrove forests [Abstract]. Programa/resumes de Marcuba '97; 1997 September 15–20; La Habana, Cuba. Palacio de Convenciones: 126.

Lugo, A.E. 1992. Restoration of mangroves. In: Thayer, G.W., ed. Restoring the nation's marine environment. College Park, MD: Maryland Sea Grant Program. 716 p.

Lundin, C.G.; Linden, O. 1993. Coastal ecosystems: attempts to manage a threatened resource. Ambio. 22: 468–473.

Macintosh, D.J. 1996. Mangroves and coastal aquaculture; doing something positive for the environment. Aquaculture Asia. 1(2): 3–8.

Noakes, D.S.P. 1951. Notes on the silviculture of the mangrove forest of Matang, Perak. Malay Forester. 14: 183–196.

Office of Environmental Policy and Planning (OEPP), Ministry of Science and Technology, Thailand. 1994. The environmental management of coastal aquaculture: a study of shrimp culture in Samut Sakhon and Chantaburi Provinces. Bangkok, Thailand: Network of Aquaculture Centers in Asia-Pacific. 195 p.

Ogburn, D.M.; Ogburn, N.J. 1994. Intensive pond culture trials of the green grouper (*Epinephelus malabaricus* Bloch et Schneider) in the Philippines. In: Chou, L.M.; Munro, A.D.; Lam, T.J. [and others]. The third Asian fisheries forum 1; 1992 October 26–30; Singapore. Manila, Philippines: Asian Fisheries Society: 74–77.

Olsen, S.; Arriaga, L. 1989. Establishing a sustainable shrimp mariculture industry in Ecuador. Narragansett, RI: The University of Rhode Island, Coastal Resources Center; Washington, DC: U.S. Agency for International Development. 139 p.

Phillips, R.C.; McRoy, C.P, eds. 1980. Handbook of seagrass biology: ecosystem perspective. New York: Garland STPM Press. 353 p.

Potaros, M. 1995. Country report on Thailand. Annex II–16. In: FAO/NACA 1995 regional study and workshop on the environmental assessment and management of aquaculture development (TCP/RAS/2253). Network of Aquaculture Centers in Asia and the Pacific (NACA), Environment and Aquaculture Development Series 1. Bangkok, Thailand: Network of Aquaculture Centers in Asia and the Pacific: 377–390.

Primavera, J.H. 1995. Mangroves and brackishwater pond culture in the Philippines. Hydrobiologia. 295: 303–309.

Primavera, J.H.; Agbayani, R.F. 1996. Comparative strategies in community based mangrove rehabilitation programs in the Philippines. In: Proceedings of the ECOTONE V regional seminar: community participation in conservation, sustainable use and rehabilitation of mangroves in Southeast Asia; 1996 January 8–12; Ho Chi Minh City, Vietnam. Ho Chi Minh City, Vietnam: Mangrove Ecosystem Research Centre (MERC), Vietnam National University: 34.

Saenger, P.; Siddiqi, N.A. 1993. Land from the sea: the mangrove afforestation programme of Bangladesh. Ocean and Coastal Management. 20: 23–29.

Sammut, J.; Mohan, C.V. 1996. Processes and impacts of estuarine acidification: the Richmond River, NSW and Karnataka, India. In: One-day symposium on issues for eastern Australia and Asia: Acid sulphate soils-causes, management, effects, rehabilitation, prevention; 1996 July 29; Sydney, Australia. Sydney, Australia: University of Sydney, Earth Sciences Foundation, Department of Geology and Geophysics. 4 p.

Sanyal, P. 1998. Rehabilitation of degraded mangrove forests of the Sunderbans of India [Abstract]. Program of the international workshop on the rehabilitation of degraded coastal systems; 1998 January 19–24; Phuket, Thailand. Phuket, Thailand: Phuket Marine Biological Center: 25.

Sidall, S.E.; Atchue, J.A.; Murray, R.L., Jr. 1985. Mariculture development in mangroves: a case study of the Philippines, Ecuador, and Panama. In: Clark, J.R., ed. Coastal resources management: development case studies. Renewable Resources Information Series, Coastal Management Publ. 3. Columbia, SC: Research Planning Institute: 1–53. [Prepared for the U.S. Department of the Interior, National Park Service, and the U.S. Agency for International Development].

Siddiqi, N.A.; Islam, M.R.; Khan, M.A.S.; Shahidullah, M. 1993. Mangrove nurseries in Bangladesh. Occas. Pap. 1. Okinawa, Japan: International Society for Mangrove Ecosystems. 15 p.

Silliman University. 1996. Assessment of the Central Visayas Regional Project-I: nearshore fisheries component. Dumaguete City, Philippines: Silliman University. 126 p. Vol. 1.

Soemodihardojo, S., Wiroatmodjo, P.; Mulia, F.; Harahap, M.K. 1996. Mangroves in Indonesia-a case study of Tembilahan, Sumatra. In: Fields, C.D., ed. Restoration of mangrove ecosystems. Okinawa, Japan: International Society for Mangrove Ecosystems: 96–110.

Sreng, L. 1996. Mangroves in Kampot Province, Cambodia. In: Proceedings of the ECOTONE V regional seminar: community participation in conservation, sustainable use and rehabilitation of mangroves in Southeast Asia; 1996 January 8–12; Ho Chi Minh City, Vietnam. Ho Chi Minh City, Vietnam: Mangrove Ecosystem Research Centre (MERC), Vietnam National University: 13.

Stevenson, N.J. 1997. Disused shrimp ponds: options for redevelopment of mangrove. Coastal Management. 25(4): 423–425.

Stevenson, N.J.; Lewis, R.R.; Burbridge, P.R. 1999. Disused shrimp ponds and mangrove rehabilitation. In: Streever, W.J., ed. An international perspective on wetland rehabilitation. Dordrecht, The Netherlands: Kluwer Academic Publishers: 277–297.

Tuan, M.S. 1996. Building up the strategy for mangrove management in Vietnam. In: Proceedings of the ECOTONE V regional seminar: community participation in conservation, sustainable use and rehabilitation of mangroves in Southeast Asia; 1996 January 8–12; Ho Chi Minh City, Vietnam. Ho Chi Minh City, Vietnam: Mangrove Ecosystem Research Centre (MERC), Vietnam National University. 35 p.

Turner, R.E.; Lewis, R.R. 1997. Hydrologic restoration of coastal wetlands. Wetlands Ecology and Management. 4(2)(Special): 65–72.

Watson, J.G. 1928. Mangrove forests of the Malay Peninsula. Rec. 6. Malayan Forester. 276 p.

Yap, W.G. 1997. Can the Philippines produce enough fish for the multitude? Aquaculture Asia. 2(3): 32–38.

EFFECTS OF HYDROLOGIC CONDITIONS ON BIOGEOCHEMICAL PROCESSES AND ORGANIC POLLUTANT DEGRADATION IN SALT MARSH SEDIMENTS

W. James Catallo[1]

Abstract—This work addressed the influence of tidal vs. static hydrologic conditions on biogeochemical processes and the transformation of pollutant organic chemicals (eight representative N-, O-, and S-heterocycles (NOSHs) from coal chemicals, crude oils, and pyrogenic mixtures) in salt marsh sediments. The goals were to: (1) determine the effects of static (flooded, drained) vs. dynamic (tidal) hydrology on redox potential (Eh) dynamics, trace gas evolution, and pollutant transformation; (2) deploy hydrodynamic microcosms for this purpose that were reproducible, well controlled, and adequately monitored; and, (3) develop analytical approaches for target pollutant chemicals that allowed for detection of small but significant concentration differences between time points and treatments, i.e., isotopic dilution. NOSH-amended sediments were exposed to three hydrologic conditions: static drained (oxidized redox potentials), static flooded (reduced redox potentials), and diurnal-tidal (alternating redox potential). The rate of NOSH transformation and the number of NOSHs degraded decreased in the following order: drained = tidal flooded. This indicated that sediments and associated biota exposed to tidal pulsing removed more NOSH compounds faster and to lower levels than flooded, highly reducing sediments.

INTRODUCTION

The work summarized here examined the influence of hydrology on biogeochemical processes that govern transport and transformation of selected pollutant chemicals in wetland sediment-plant systems. “Biogeochemical” refers to a group of coupled biological, e.g., microbial, plant and physicochemical, e.g., flood-drain events from tides and floods, processes that to a large extent determine the productivity, habitat quality, and regional-global significance of wetland and other ecosystems (Catallo 1999, Catallo and others 1999). Previous microcosm studies have confirmed that degradation and transformation rates and pathways of aromatic hydrocarbons (AHs) and N-, O-, and S-heterocycles (NOSHs) differed significantly in marine sediments of different particle sizes and under oxidized vs. reduced conditions (Catallo 1996b, Catallo and Gambrell 1994). In the cited work, AH and NOSH transformation was evaluated in stirred, bubbled, controlled Eh/pH reactors containing sediment slurries maintained as “oxidized, aerobic,” “moderately reducing, anoxic,” and “highly reducing, methanogenic.” AH and NOSH degradation rates generally were: oxidized ≥ moderately reducing >> methanogenic. The reactors used were first-order feedback control systems whose output domains were predefined by desired Eh ranges for each treatment. As a result, Eh vs. time outputs were sinusoidal. Fourier spectral analysis of the Eh waveforms demonstrated that periodic sampling interventions and other system perturbations were detected by the electrodes and preserved in the time series for each system. This suggested that the Eh might change rapidly and reproducibly in response to other periodic forcing, such as diurnal tidal flushing, but this had not been well documented. Further, it was clear that the terms “oxidized, aerobic” applied to wetland sediments were something of an oxymoron. Nevertheless, this treatment type was evaluated because large quantities of wetland and aquatic sediments are disposed of in upland or otherwise well-drained situations, so evaluation of these sediments was deemed important, albeit not especially compelling from the perspective of “wetlands ecology.”

Apart from the differential NOSH and AH transformation rates observed in these experiments, results strongly suggested that measured redox potential (Eh) can be a dynamic nonlinear variable in wetland sediments that can respond quickly and significantly to changing conditions, and that this information can be preserved in a time series of Eh provided adequate sampling intervals are employed. As Eh is considered a biogeochemical “master variable” in many ecosystems, the behavior of this variable should be adequately and accurately represented by the data at all scales relevant to the experimental design (Catallo 1993, Catallo and others 1999, Lindsay 1991). It was clear that the interaction of tidal water level changes with sediments should be included as a treatment in studies of biogeochemical processing of contaminants. In order to do this, effort was expended in the design, monitoring, and evaluation of hydrodynamic sediment microcosms (20 gal) and, later, mesocosms (300 to 3000 gal) containing a biocenosis comprised of sediments, plants, microbes, meiofauna, and macrofauna (Catallo 1999).

The central aims of the current study were: (1) determine the effects of static vs. dynamic hydrology on biogeochemical

[1] Associate Professor of Environmental Toxicology, Laboratory for Ecological Chemistry, School of Veterinary Medicine, Louisiana State University, Baton Rouge, LA 70803.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

behavior of salt marsh sediments, particularly with respect to major output variables, e.g., Eh dynamics, trace gas signatures, and pollutant transformation; (2) design, build, and optimize microcosms for this purpose that are reproducible, well controlled, and adequately monitored in the laboratory or greenhouse; and, (3) develop extraction/analytical approaches for NOSH target chemicals that allow for detection of small but significant concentration differences between time points and treatments. This last item was important because most common methods used for identification of chemicals in sediments are semiquantitative, i.e., provide for order-of-magnitude differentiation in analyte concentrations, and it was expected that differences between NOSH transformation profiles in different treatments of this study would be below this threshold (Catallo 1996a).

The pollutant chemicals selected for this work included representative N-, O-, and S-heterocycles (NOSHs) that occur in coal chemicals, several crude oils, and pyrogenic (combustion-generated) mixtures (fig. 1) (Turov and others 1987). These and other NOSHs are found in large quantities in pollutant mixtures near certain industrial activity, hazardous waste sites, and major harbors. They enter coastal marine and wetland ecosystems through: (a) direct dumping/spillage; (b) fluvial transport of dissolved and sediment-associated chemicals from source outfalls, deposition of combustion-generated airborne particles, semivolatiles, e.g., from the so-called *in situ* burns of spilled oils, and polar residues; and, (c) assorted natural processes, including marine oil seeps and reactions between biogenic organic matter and sub- and supercritical water in various geochemical settings. They are of continuing interest because they can accumulate to bioavailable levels in aquatic sediments, and many NOSH compounds are acutely toxic, with some, e.g., the quinolines, carbazoles, and benzoquinolines and their oxidized metabolic and environmental transformation products, being mutagens or carcinogens or both (Catallo 1996b, Catallo and others 1994, Warshawsky 1992).

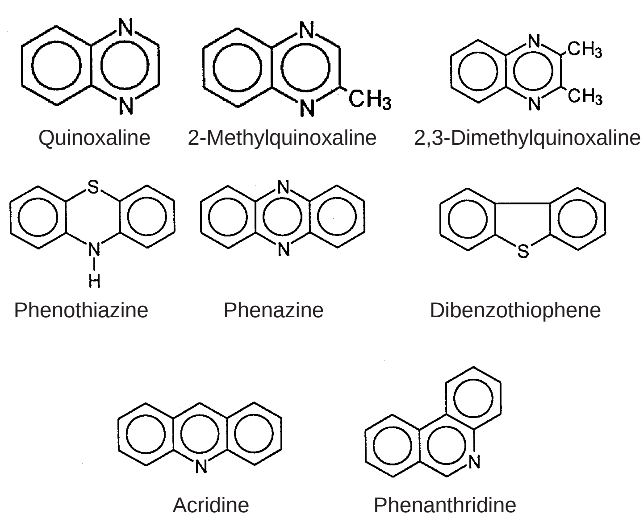


Figure 1—NOSH target analytes examined in this study.

MATERIALS AND METHODS

Hydrodynamic (Tidal) Wetland Microcosms

The microcosms for this work were comprised of 20 gal glass aquaria containing sediment columns as shown in figure 2. Sediment samples were collected from a streamside *S. alterniflora* salt marsh site in Terrebonne Parish, LA. The sediments were mixed with quartz sand and commercial seed starter (5 percent w/w) and then loaded into cylinders (26 cm i.d.) made first of wire, and ultimately of plastic mesh (0.5 cm) to a height of 38 cm. The wire mesh was discontinued after preliminary runs because it interfered with Eh measurements. The systems were covered with plexiglass containing throughputs for water lines, gas sampling, and electrode wire leads, and sealed with silicone plastic. A diurnal tide was simulated by pumping artificial seawater (15 g/L Instant Ocean) into and out of the aquarium once daily, i.e., one high and one low per 24 hour, using peristaltic pumps and controlled by battery-powered timers. The seawater reservoir was continuously aerated. At “high tide,” the sediment columns were covered with ca. 8 cm of water. At “low tide” about 5 cm of water remained in the aquarium. Two Eh electrodes (below) were positioned in the top 1 cm of sediment (surface), and two additional electrodes were placed deep within the column in the continually flooded zone near the bottom. Trace gases were analyzed by direct sampling of headspace gases under vacuum to a 10-m cell connected to a Buck Scientific FTIR spectrometer with gold optics and 4 per cm resolution. Molecular sieve 5A and Ascarite A (Aldrich) traps were used to reduce water and CO_2 spectral interference and increase sensitivity. Compounds were identified using authentic gas standards (Scott Specialty Gases) and published spectral libraries from NIST. Digital background subtractions were performed using spectra of normal laboratory air sampled immediately prior to sampling the microcosms, and handled identically; i.e., reference air samples were passed through the sampling loop of the system and through the traps. Thus, CO_2 and water vapor signals in the experimental spectra represent excesses of these materials in the microcosm headspaces vs. the laboratory air.

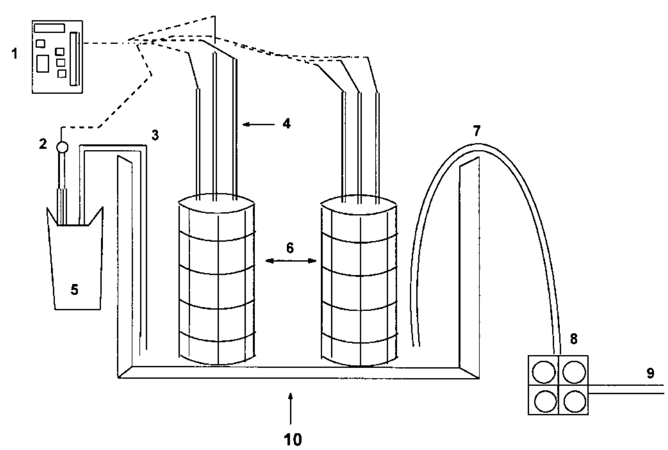
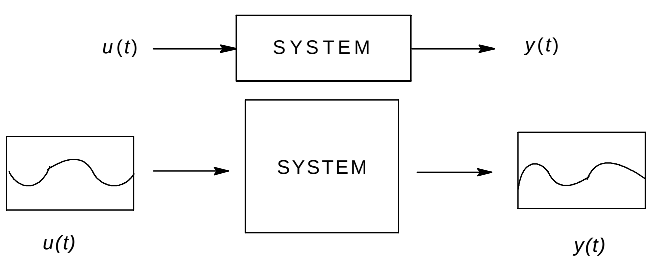


Figure 2—Schematic diagram of the tide simulation microcosm: (1) data logger, (2) calomel reference electrode, (3) salt bridge, (4) platinum electrodes, (5) concentrated KC1 solution, (6) soil column enclosures, (7) water inlet/outlet, (8) automatic peristaltic pumps, and (9) reservoir. Gas tight cover and FTIR feeds are not shown.

Redox potential (Eh) was logged using Pt wire–SCE electrode cells and a multichannel logger (Catallo 1999). The platinum electrodes were constructed by welding Cu wire to 0.5 cm-lengths of 12 gauge Pt wire. The junction and Cu lead were enclosed in glass tubes of varying lengths, and both ends were sealed using a blowtorch. Hence, a ca. 0.5-cm length of Pt was exposed to the sediments whereas junctions and leads were enclosed in sealed glass. The electrodes were polished with light emory paper, soaked in *aqua regia* and then distilled water. Each electrode was calibrated prior to use with a 1-percent quinhydrone solution. Samples were collected at rates between 1 Hz and 1 per hour, depending on the application (in general, the rates were 1 to 2 per hour).

Signal Acquisition and Analysis

A biogeochemical system can be viewed as a signal processor, with inputs, e.g., light and heat cycles, water fluctuations, inputs of organic matter and toxic chemicals, impinging on the system, influencing its behavior, e.g., aerobic vs. anaerobic microbial metabolism, and influencing the magnitude and time structure of numerous outputs (water chemistry, trace gas identities, and evolution time series). Shown schematically below is a generic system with single continuous input and output variable functions, *u(t)* and *y(t)*,



respectively. The inputs mentioned are impulses or excitation functions having importance to system processes, and all of these must be known (identified) and measured accurately with appropriate sampling frequencies. The former is important because confounding and collinear variables can invalidate normal statistical approaches and corrupt causal ascriptions (Asher 1983, Catallo and others 1999) and the latter because processes changing faster than the sampling rate applied to them can cause error (aliasing) when spectral analysis techniques are applied to the time series (Brockwell and Davis 1991, Mallat 1998). These functions, ($u(t)$), can be of many kinds, e.g., linear, spike, ramp, threshold, sinusoidal. Signals are outputs, $y(t)$ carrying information about system processes. Signals are structured in time, whereas noise is not, and increase relative to noise as the square root of the sampling repetitions; $(S/N)_n = n^{1/2}(S/N)_1$; where S is signal amplitude, N is noise amplitude, $(S/N)_1$ is signal:noise ratio after one data acquisition (scan), n is the number of sample scans, and $(S/N)_n$ is the signal:noise ratio after n scans. The states of the system can be defined by the stable and transient domains occupied by output signals *y(t).*

It frequently is assumed that biogeochemical processes change slowly, i.e., weeks to months, and therefore adequate sampling of many outputs can be performed at rates on the order of 1 per day or longer (Catallo 1999, Picek and others 2000). Unfortunately, it must first be verified that there are no cyclic outputs from the system with frequencies greater than the sampling rate, and this is rarely done. As a result, the assumption that widely spaced data points from dynamic systems can be connected by linear interpolation to accurately reconstruct system temporal behavior becomes a matter of faith rather than demonstration. Further, attempts to analyze such a time series in order to determine its component waves can be highly inaccurate and misleading. The upshot of this is that complex system variables (*u(t)*) and signals (*y(t)*) cannot be interpreted accurately if data points are gathered at rates slower than the changes in the process. For virtually all ecological system processes, classical single resolution Fourier analysis of time series suffers from major analytical drawbacks, not the least of which being its inability to accommodate nonstationarity in the data; i.e., transients and other deviations from perfect statistical regularity in the signals. Single resolution analysis of nonstationary time series leads to aliasing of power spectra (Brockwell and Davis 1991, Mallat 1998) in ways that are not readily interpreted or identified. It was clear from Eh, pH, temperature, and other time series from the microcosms that nonstationarity was a major feature of the input and output variables of concern in this work. As a result, multiresolution "wavelet" analysis was applied to the time series of signals from the microcosms.

Wavelet analysis is a recent development in applied mathematics (Mallat 1998) that has stimulated an enormous interest in many disciplines including signal processing/reconstruction, vision science, medical imaging, and physiology. The wavelet transform is a technique that allows for the acquisition of both localized time and frequency information about a signal. Its ability to dilate and shift time windows with respect to a signal allows for transient features to be accommodated, long- and short-term variations to be compensated, and periodic features to be isolated, identified, and, if needed, enhanced. In a sense, the wavelet approach has operational and conceptual affinities with the use of an oscilloscope to visualize waveforms in circuits: time sampling windows are varied by the analyst in an attempt to isolate component waveforms of complex signals as standing waves, with frequency data available as the inverse of the time window used to capture the particular wave of interest. Waveform components and their alterations by specific processors (distortion, clipping, chirping, and damping) are obtained by thoroughly permuting the available sampling windows.

The definition of the continuous time wavelet transform (CWT) entails an "analysis wavelet," $\psi(t)$, and the family of shifted and dilated versions

$$\psi_{a,b} = |a|^{-1/2}\psi\,(t-b)/a \tag{1}$$

The transform of $x(t) \in L_2(R)$ is then

$$T^w[x]\,(a,b) = |a|^{-1/2}\int x(t)\,P^*(t-b)/a\;dt \tag{2}$$

The analysis wavelet, Ψ must satisfy the admissibility condition

$$C\,p = \int \left|\Psi(w)\right|^2 /w\, dw < \infty \qquad (3)$$

This condition guarantees the existence of the inverse transform

$$x = C\psi^{-1} \int\int T^w [x]\,(a,b)\,\psi^*(t-b)/a\; da\,db; \qquad (4)$$
(intergrals evaluated ± ∞)

One way to realize the transform is using the filter bank interpretation and defining

$$h_a(t) = \psi(-t/a) \qquad (5)$$

It follows that

$$T^w [x]\,(a,b) = x(t)\, h_a(t) = \int x(t)\, h_a(b-t)dt \qquad (6)$$

So, for each scale, *a*, the CWT is comprised by the output of a filter with impulse-response $h_a(t)$. As the filters are dilated according to a scale parameter, their frequency response is modified accordingly, at the discretion of the analyst. Hence, the wavelet transform maps a function of one variable into a function of two independent variables (time, scale). There is, as a consequence, significant redundancy, and, if desired, the time/scale parameters can be rendered discrete, the signal preserved in the transform samples. A standard discretization approach employs a logarithmic step for the scale parameter and a scale-dependent discretization for the temporal parameter. Thus a discretized wavelet transform can be defined by

$$T^{w[m,n]} = T^w[x]\,(2^m, 2^m n) \qquad (7)$$

In effect, a family of wavelets of the form $\psi_{m,n} = 2^{-m/2}\psi(t/2^m - n)$ is deployed.

Even in the discretized case, the representation can be optimized by requiring the wavelet family, $\psi_{m,n}$, to be an orthonormal as a set. In this case for each value of the integer *m*, the collection of vectors $\{\psi_{m,n};\, n \in Z\}$, defines a subspace, W_m. As the dilation parameter is held constant, all wavelets in that subspace have the same scale. The projection of a function, $x(t)$, on this subspace is interpreted as the details of the function at that particular scale. Since increasing the value of the integer, *m*, introduces larger dilations in the basic wavelet, its time resolution decreases. The sum of all details for all values $m \geq m_0$ is said to give a representation of $x(t)$ up to resolution m_0. Mathematically, the representations exist in the subspace

$$V_m = \oplus_{k \geq m} W_k \qquad (8)$$

One of the bulwarks of multiresolution analysis is that a unique analysis function $\varphi(t)$ can be defined such that the collection $\{\varphi_{m,n} = 2^{-m/2}\psi(t/2^m - n)\}$ spans the subspace, V_m, and these are easy to compute recursively. If, for example, the representation of a signal up to a fine resolution level, m_0, is in the form

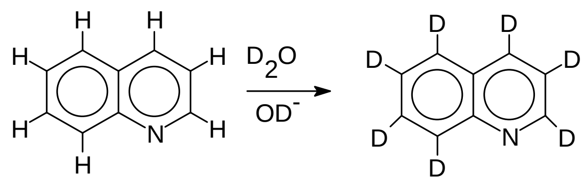
$$X^{mo} = \sum_n c^{mo}_n\, \varphi_{mo,n} \qquad (9)$$

one can define a digital filter, *H*, receiving as input the discrete signal, $\{c^{mo} = c^{mo}_n;\, n \in Z\}$, and producing as output the coefficients of the next lower resolution, c^{mo+1}. Similarly, the digital filter, *G*, can be defined as receiving the same input and producing as output the details at resolution level m_0.

The wavelet packet approach also applies to discrete time functions such as those obtained by sampling a continuous time signal. It can be considered as the discrete version of the multiresolution decomposition. Using suitable digital filter banks, a discrete time signal is decomposed into orthogonal components, which also have disjoint frequency changes. Then, by combining various orthogonal components one can develop representations of the signal at various resolution levels. For cases of nonstationary ecological signals, such as the Eh time series realized in the tidal microcosms of the current study, periodic features, transients, and long-term trends can be determined without unknown aliasing in resulting spectra. A further advantage is that the effects of low-resolution sampling can be examined, and the resulting effects of aliasing on the signal analysis evaluated. This also was applied to the Eh time series as an illustration of the perils of undersampling outputs dynamic systems and then applying spectral analytical or other analytical techniques to those data.

Synthetic Organic Chemistry

In order to quantify minor or subtle differences in transformation of target compounds between treatments, it is desirable to have stable isotope-labeled, e.g., deuterated, standards for each target compound of interest. These standards can be added to sediments before extraction and serve as internal “monitors” of extraction and analytical efficiencies for the target compounds. As these deuterated standards have properties virtually identical to the target compounds, their loss in the various steps of extraction and analysis is the same as the compounds under study. With appropriate controls and calibration, this allows for quantitative analyses (percent differences can be detected), rather than semiquantitative data (order-of-magnitude differences are detected) provided by many analytical approaches (Catallo 1996a). In addition to this advantage, isotopic dilution also allows for data streamlining: quantitation is simply a matter of integral ratios between the standard (known concentration and base peak integral) and the target analyte (concentration unknown, base peak integral known). Unfortunately, commercial availability of deuterated NOSHs is very limited. As a result, methods were developed in our laboratory to label the target compounds with deuterium either using *de novo* or postsynthetic approaches (reviewed in Junk and Catallo 1996, 1997), the latter typically using supercritical (T >374 °C/221 bar) deuterium oxide (D_2O), or “heavy water.” A basic example of this strategy is shown for the N-heterocycle, quinoline, below:



Supercritical water is extremely corrosive, and commercial stainless steel reactors leaked or exploded (destroying furnaces and part of a wall) after a few hours. As a result, a substantial amount of effort in the first year of this work was devoted to designing reactors that could withstand supercritical aqueous conditions and were large enough to permit multigram labeling of target compounds. With this accomplished, the target compounds were made in quantities sufficient for use in the transformation studies of NOSHs in the microcosms. Unfortunately, some NOSHs were unstable even in subcritical water, and novel synthetic protocols had to be devised for these compounds. Thus, quinoxaline-d6, 2-methylquinoxaline-d8, and 2,3-dimethylquinoxaline-d10 also were prepared by syntheses developed for this work (Junk and Catallo 1997, Junk and others 1997). Attempts to prepare phenazine-d8 and acridine-d9 by: (a) base-catalyzed supercritical isotope exchange at 400 ^{o}C for 6 hours following Junk and Catallo (1996); (b) acid catalyzed isotope exchange at 220 ^{o}C for 2 days; and (c) palladium-catalyzed exchange at 250 ^{o}C for 2 days, all resulted in extensive substrate decomposition. All compounds subsequently were prepared by base-catalyzed near-critical exchange at 300 ^{o}C for 4 hours. Exchange was carried out by heating 300 mg substrate, 15 mL D_2O, and 0.1 mL 40 percent NaOD (in D_2O) in a 30 mL Hastelloy C-22 autoclave. The crude products were extracted with dichloromethane (DCM), the solvent evaporated and deuterated quinoxaline, and 2,3-dimethylquinoxaline purified with a microdistillation apparatus. Phenazine and acridine were crystallized from methanol. Further purification was achieved by chromatography (200 mesh silica gel, hexane:DCM 10:1 v/v). Yields ranged from 45 to 55 percent. Dibenzothiophene-d8, phenothiazine-d8, phenanthridine-d9 were prepared using the published Supercritical Isotope Exchange technique (Catallo and Junk 1998, Junk and Catallo 1996). A 30-mL Hastelloy-C22 autoclave was charged with the 0.3 g of the respective substrates, 15 mL deuterium oxide, and 0.1-mL 40-percent deuterium deuteroxide solution. Exchange was achieved by heating to 400 ^{o}C for 6 hrs. Compounds were collected by filtration and purified by chromatography over 200-mesh silica gel using hexane as mobile phase for dibenzothiophene and phenanthridine, and DCM for phenothiazine. Yields for dibenzothiophene and phenanthridine were above 75 percent and have been reported along with a range of other AHs and NOSHs (Junk and Catallo 1996). The yield for phenothiazine was 86 percent. No attempts were made to preclude the facile back-exchange of the N-H proton of phenothiazine during work up under ambient (open-air) conditions because this analyte would be exposed to water during the sample extraction.

NOSH Transformation Studies

Three sediment columns prepared as described above were equilibrated under drained, flooded, and tidal conditions for 2 weeks. The sediments then were collected from the surface third of the enclosures from each hydrologic type. Protiated (hydrogenated) NOSH compounds in acetone were added to the sediments with mechanical mixing (2 hours) to provide uniform levels of the individual target contaminants between 200 and 400 parts per million (ppm) on a weight basis. Care was taken to maintain the electrochemical condition of the sediments (oxidized vs. reduced) during mixing by purging the system and flooding the headspace of the mixing enclosure with Ar (reduced, flooded) or air (oxidized, drained, and tidal). The sediments then were reintroduced to their respective microcosms in the appropriate column enclosure. The three hydrologic regimes were initiated and “time zero” samples were collected within 8 hours of placement using a clean glass corer. Subsequent samples were taken at intervals (1 to 4 weeks) depending on estimators of microbial activity, e.g., iodonitrotetrazolium reduction (Catallo and others 1990), and previous recovery of NOSH target analytes. After collection, the sediment-water samples (ca. 10 g) were weighed and amended with deuterated isotopic dilution standards for each NOSH target analyte at the levels near the sediment concentrations. The samples then were Soxhlet extracted with DCM for 48 hours, with the extract subsequently dried (Na_2SO_4) and concentrated under N_2. The residual sediments in the extraction thimbles were dried and weighed. The sample extracts then were subjected to GC-MS in the full-scan mode. NOSH target analyte concentrations in the extract were determined vs. the isotopic dilution standard by ratios of corresponding peak integrals. These extract concentrations were corrected for concentration factor and dry weight of sediment in the original sample.

RESULTS AND DISCUSSION

The biogeochemical behavior of the sediment columns in the microcosms was very similar to that observed in natural settings and in accord with prevailing theory (Catallo 1999, Jorgensen and Okholm-Hansen 1985, Odum 1981). With respect to the Eh, the continuously flooded and drained systems had reduced (mean = -428 mV) and oxidized (+ 73 mV) Eh values, respectively, with no evidence of daily or longer periodic variation. The routine (weekly) wetting/draining of the drained system, and water exchange in the flooded system, did not affect these trends; apparently the pre-equilibration of sediments prior to addition of the NOSH analytes afforded a degree of redox potential poising, i.e., “buffering,” that was not affected by the brief maintenance interventions. The tidal systems, however, exhibited oscillating Eh values, with significant amplitudes (40 to 180 mV) (fig. 3). Analysis of these signals using the wavelet transform confirmed the presence of strong diurnal Eh variations, with a mean value period of 23.78 ± 2.10 hours. It has been shown that these diurnal signals were: (1) reproducible in the tide microcosms; (2) found in the corresponding tidal mesocosms, which also contained the plant *S. alterniflora*; and (3) observable in tidal field sites but not impounded areas isolated from diurnal tides (Catallo 1999). Thermal and light cycles in the laboratory had no discernable effect on the Eh time signature; variation of tide stage throughout the day and at different times of the year

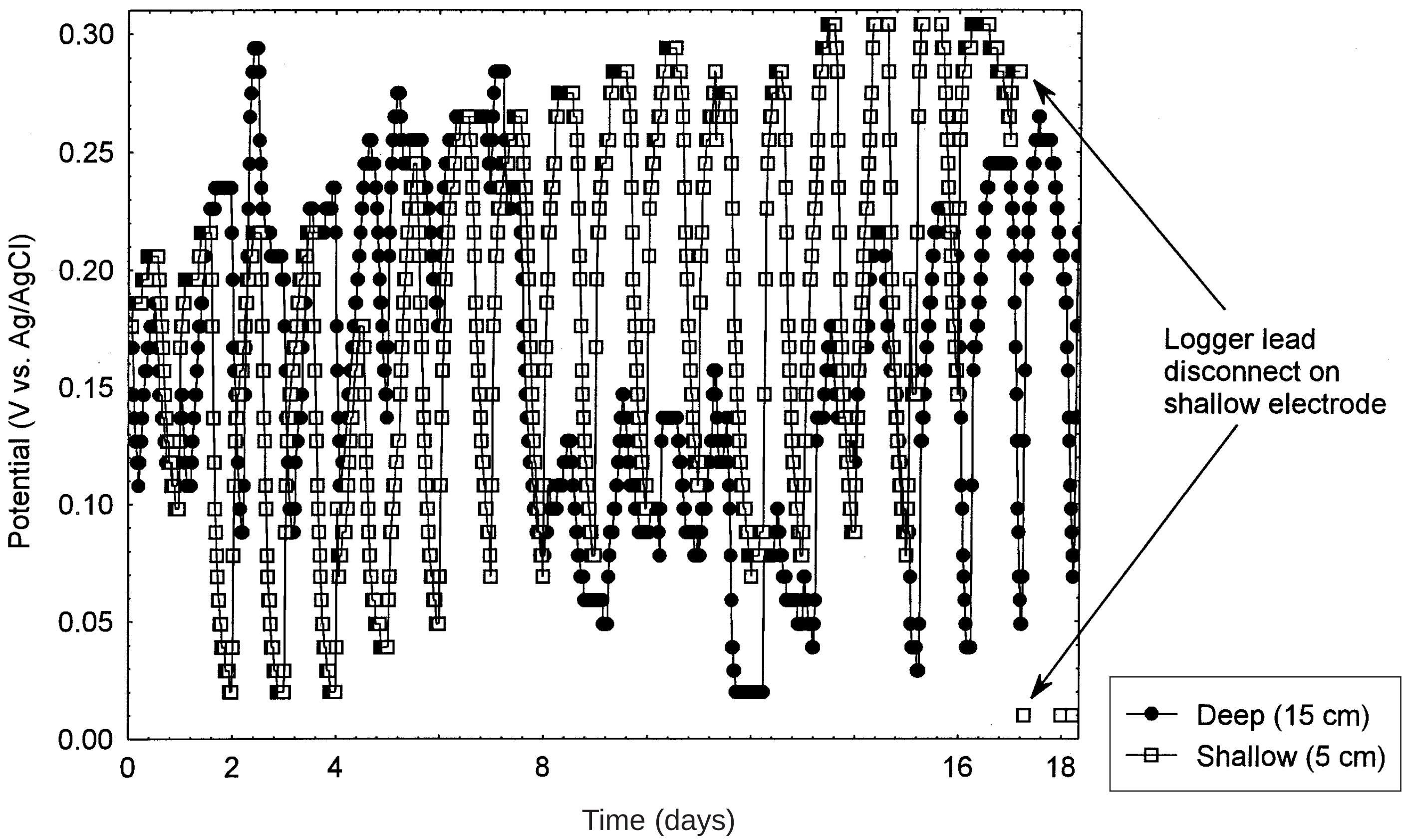


Figure 3—Time series of Eh from shallow and deep electrodes placed in tide simulation microcosms.

(the experiments were conducted in an annex not equipped with climate control) did not affect the tide-derived Eh signal period.

The Eh traces shown in figure 3 are characteristic of the tidal systems in three main respects: (1) the sinusoidal Eh oscillations continued for as long as the tides were applied, but ceased immediately when static conditions ensued; (2) the waveforms are asymmetric, exhibiting hysteresis with respect to oxidation and reduction; and (3) electrodes at the different depths were out of phase (as expected), with deeper electrodes showing smaller amplitudes (probably reflecting the effects of compaction). These features were consistent throughout the experimental runs and in subsequent work with large hydrodynamic mesocosms containing plants and invertebrates (Catallo 1999). The hysteresis of measured redox potentials (item 2, preceding) suggests that the Pt electrode-SCE cell is quasi-reversible with respect to oxidation and reduction in sediments; i.e., it is patently non-Nernstian. This almost certainly reflects compositional as well as bulk phase chemical variables *in situ*. Unpacking these factors and adequately describing the physics involved is far beyond the scope of this communication and might well be impossible given the current level of theory on heterogeneous electrochemical systems (Southampton Electrochemistry Group 1985). In spite of this, the electrodes in the tidal systems remained functional throughout the duration of the experiments and rarely had to be replaced. The electrodes in the oxidized and reduced static microcosms, however, were replaced frequently because of passivation with oxides and sulfides (respectively) and consequent loss of calibration. It would seem that dynamic Eh conditions in the tidal systems reduced the level of passivation on the working electrodes and provided for conditions most favorable for obtaining accurate potential values vs. the more static, well-poised conditions in the other treatments. “Reconditioning” of solid electrode surfaces by alternating oxidation-reduction cycles, i.e., cyclic removal of surface metal oxides and sulfides by alternating redox processes, apparently has not been reported elsewhere and represents an area of interest for further research.

Biogeochemical gas mixtures leaving the sediments also reflected the hydrological status of the treatments: the major gases observed were CO_2 from microbial respiration (drained and tidal) vs. large amounts of sulfide and methane (flooded conditions) (fig. 4). The gas profiles in each treatment were checked weekly and were consistent throughout the experiment.

Transformation of the NOSH analytes was significantly different in each of the hydrologic regimes, with the rate of transformation and the number of NOSHs degraded decreasing in the following order: drained > tidal > flooded. This can be discerned in figures 5 through 7 with respect to the number of NOSHs transformed so as to afford a recovery of under 50 ppm at 16 weeks. Except for one sample showing a spike at week 18, the drained system showed the most complete degradation of quinoxaline, 2-

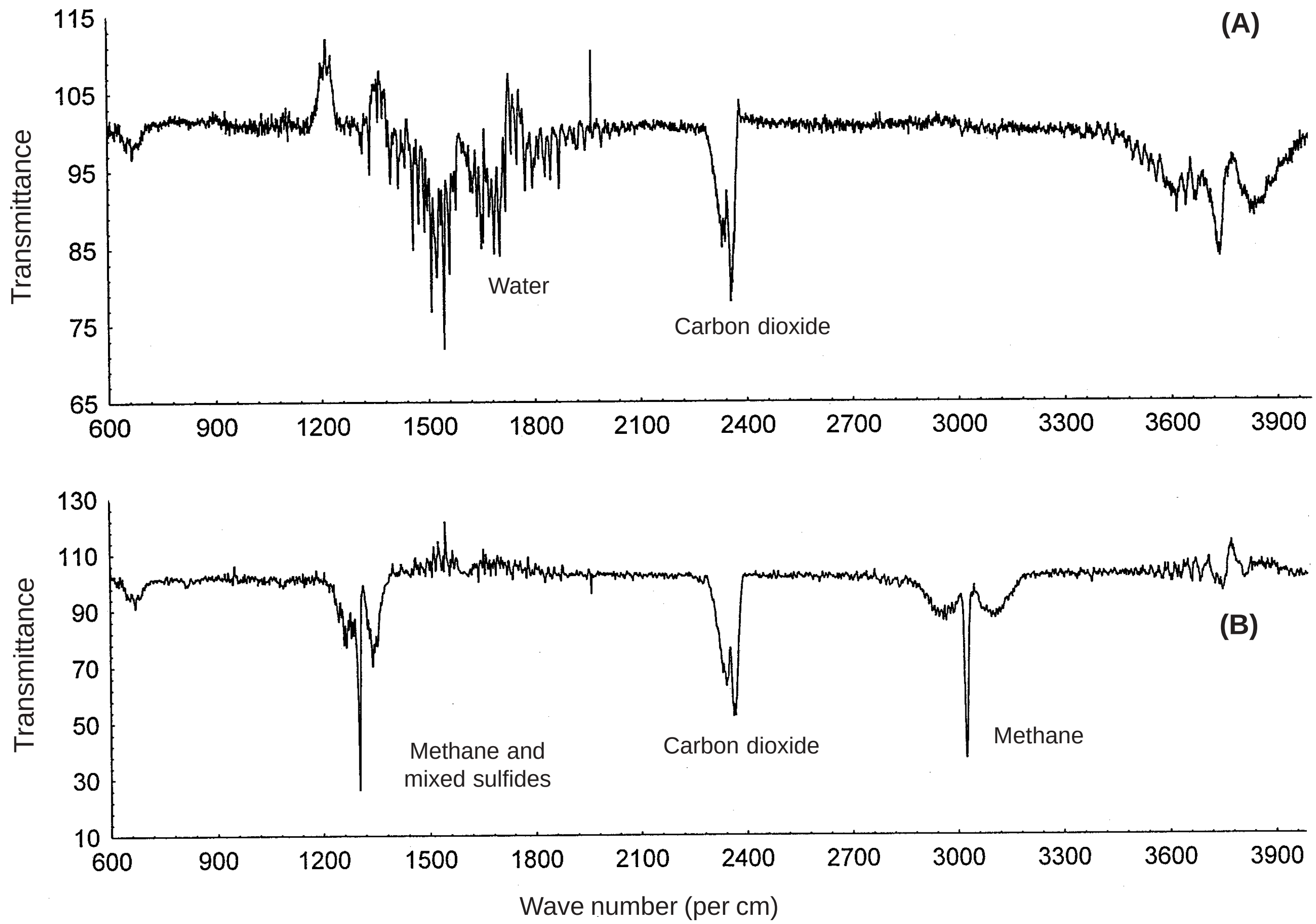


Figure 4—Headspace gases from the tidal (A) and flooded (B) microcosms by FTIR. (Gases from the drained system were similar to A.)

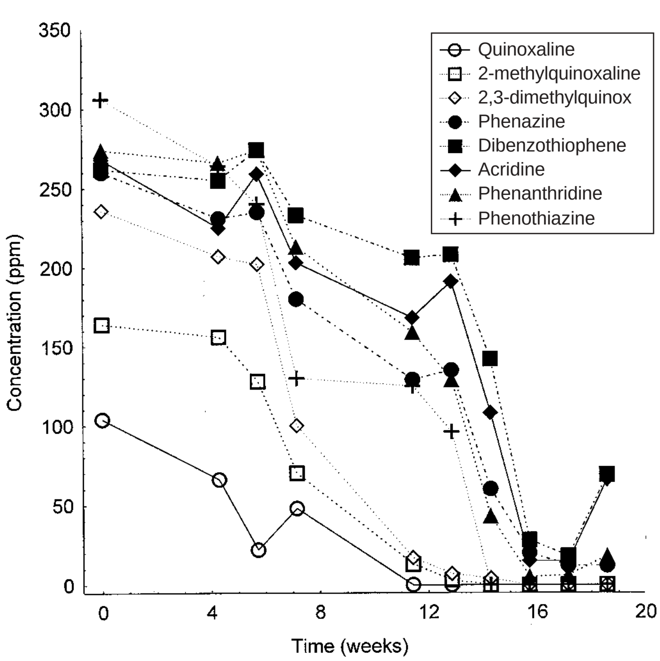


Figure 5—NOSH transformation profiles from the drained/oxidized microcosm.

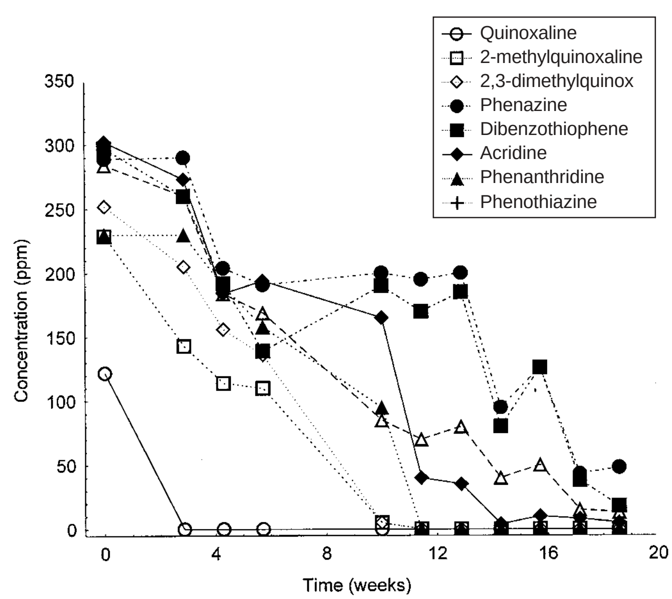


Figure 6—NOSH transformation profiles from the diurnal tidal microcosm.

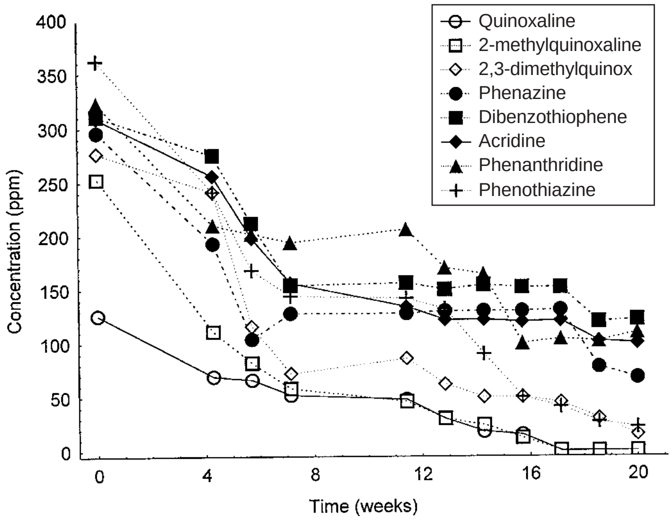


Figure 7—NOSH transformation profiles from the flooded/reduced microcosm.

methylquinoxaline, 2,3-dimenthylquinoxaline, and phenanthridine by week 16, with the recovered concentrations of the other NOSHs also diminished to or below 25 ppm, i.e., a tenfold concentration reduction. Under tidal conditions, quinoxaline, 2-methylquinoxaline, and phenothiazine were eliminated more rapidly than in any other treatment, with phenazine and phenanthridine reduced below 25 ppm by week 16. By 17.5 weeks, all NOSH analytes were transformed so that recoveries were less than 25 ppm. Under flooded conditions, only quinoxaline and dimethylquinoxaline were degraded to recovery levels below 50 ppm within 16 weeks, and these were completely eliminated by week 17. Other than these, only phenothiazine and 2,3-dimethylquinoxaline were transformed to below 50 ppm recovery by week 20. The sediment concentrations of the other NOSHs remained static, with recovery levels remaining relatively constant between 100 and 170 ppm for acridine, phenanthridine, and phenazine between weeks 7 and 20. It would seem then, that except for quinoxaline, 2-methylquinoxaline, and perhaps phenothiazine, there was a pronounced Eh effect on transformation rates for the NOSH compounds studied here.

These transformation studies, while preliminary, clearly indicated that tidal pulsing optimized the transformation of some NOSH compounds relative to flooded conditions, and that *in situ* or “passive” remediation of coal- and petro-chemical pollution in coastal wetlands should include design features that accommodate prevailing hydroperiods, including tides and seasonal events. For most settings where NOSH and AH contamination is a problem, the statically well-drained/oxidized approach is not an option unless excavation or draining/impounding of the system is envisaged. In these cases, changes in sediment physicochemical properties, e.g., acidification, resulting from such approaches are not easily reversible and would attenuate the usability of these sediments in post-treatment applications, e.g., marsh restoration. On the other hand, in many cases so called passive remediation approaches could be adopted that exploit natural hydroperiodicities and the biogeochemical processes that are coupled to them, rather than artificially manipulating them, i.e., by impounding or dredging, and further compromising the functioning of the ecosystem.

In an abstract sense, the use of hydroperiodicity for enhanced chemical remediation and ecological recovery of polluted systems seems promising in light of this and related work (Catallo 1996b). It would involve, at the least, attempts to optimize the tidal volume of the wetland without compromising its integrity and function. Obviously, this kind of undertaking in a real wetland would involve a set of engineering interventions and monitoring strategies that encompass hydrologic, sedimentologic, and biogeochemical variables in a progressive sense. The same is true of the use of marginally contaminated dredge materials and other wastes, e.g., phosphogypsum, bauxite red mud, for coastal habitat restoration projects. The author is aware of no actual cases in which the goals and approaches of chemical remediation and wetland creation/restoration have been successfully coupled. Much further ecological study is called for in microcosms and other controlled settings where variables and causal relationships can be identified and ranked with respect to holistic endpoints including, but not limited to, pollutant transformation.

ACKNOWLEDGMENTS

The author is indebted to Professors R.D. DeLaune, R.J. Gale, R.P. Gambrell, J. Aravena, T. Junk, and R. Meyhandan of LSU for discussion, critique, and substantive assistance. R.S. Murray, L.D. Civils, N. Crochet, and H.L. Dharmasena provided assistance with field work, laboratory, and greenhouse studies, logger development, and synthetic chemistry. This work was supported in part by grants from USDOI, USEPA, and USFWS.

REFERENCES

Asher, H.B. 1983. Causal modeling. SAGE Univ. Pap. 3. Newbury Park, CA: Sage Publishers. 91 p.

Brockwell, P.J.; Davis, R.J. 1991. Time series: theory and methods. New York: Springer Verlag. 577 p.

Catallo, W.J. 1993. Ecotoxicology of wetland ecosystems. Current understanding and future research needs. Annual review. Environmental Toxicology and Chemistry. 12(12): 2209–2224.

Catallo, W.J. 1996a. Bioremediation: statistical and analytical needs. In: Proffitt, C.E.; Roscigno, P.F., eds. Proceedings: Gulf of Mexico and Caribbean oil spills in coastal ecosystems: assessing effects, natural recovery, and progress in remediation research. OCS Study/MMS 95–0063. New Orleans: U.S. Department of the Interior, Mineral Management Service, Gulf of Mexico Region: 109–129.

Catallo, W.J. 1996b. Transformation of N-, O-, and S-heterocycles (NOSHs) in estuarine sediments: effects of redox potential and particle size. Chemosphere. 33/12: 2543–2563.

Catallo, W.J. 1999. Hourly and daily variation of sediment redox potential in tidal wetland sediments. Biol. Sci. Rep.: USGS/BRD/BSR–01–1999. U.S. Geological Service, Biological Resources Division.

Catallo, W.J.; Gale, R.J.; Wong, R.L.; Bender, M.E. 1990. The reducible dye 2-(p-iodophenyl)-3-(p-nitrophenyl)-5-phenyl-2H-tetrazolium chloride (INT) for use in aquatic toxicology. Notes on chemical structure, electrochemistry, and toxicity. In: Landis, W.G.; van der Schalie, W.H., eds. Aquatic toxicology and risk assessment. ASTM Spec. Tech. Publ. 1096. Philadelphia: American Society for Testing and Materials: 222–236.

Catallo, W.J.; Gambrell, R.P. 1994. Fates and effects of N-, O-, and S-heterocycles (NOSHs) from petroleum and pyrogenic sources in marine sediments. Final Grant Completion Rep. OCS Study/ MMS 674–2010. New Orleans: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico, OCS Regional Office. 72 p.

Catallo, W.J.; Gambrell, R.P.; Reddy, K.R. [and others]. 1999. Biogeochemical processes. In: Lewis, M.A.; Mayer, F.L.; Powell, R.L. [and others], eds. Ecotoxicology and risk assessment for wetlands. Pensacola, FL: SETAC Foundation for Environmental Education: 27–69.

Catallo, W.J.; Hoover, D.H.; Vargas, D. 1994. Effects of quinoline on ultrastructure in Paramecium caudatum and the protective effects of dissolved calcium. Aquatic Toxicology. 29(324): 291–303.

Catallo, W.J; Junk, T. 1998. Process for preparing deuterium tagged compounds. U.S. patent 5,830,763. Nov. 3.

Jorgenesen, B.B.; Okholm-Hansen, B. 1985. Emissions of biogenic sulfur gases from a Danish estuary. Atmospheric Environment. 19(11): 1737–1749.

Junk, T.; Catallo, W.J. 1996. Preparative supercritical deuterium exchange in arenes and heterocycles. Tetrahedron Letters. 37: 3445.

Junk, T.; Catallo, W.J. 1997. Organic reactions in supercritical polar fluids. Royal Society of Chemists (U.K.). Chemical Society Reviews. 26: 401–406.

Junk, T., Catallo, W.J.; Elguero, J. 1997. Synthesis in superheated aqueous media: preparation of fully deuterated pyrazoles and quinoxalines. Tetrahedron Letters. 38(36): 6309–6312.

Lindsay, W.L. 1991. Iron oxide solubilization by organic matter and its effect on iron availability. In: Chen, Y.; Haddar, Y., eds. Iron nutrition and interactions in plants. The Netherlands: Kluwer Academic Publishers: 29–36.

Mallat, S. 1998. A wavelet tour of signal processing. New York: Academic Press. 577 p.

Odum, H.T. 1983. Ecological and general systems. Boulder, CO: University of Colorado Press. 644 p.

Picek, T.; Simek, M.; Santruckova, H. 2000. Microbial responses to fluctuation of soil aeration status and redox conditions. Biology and Fertility of Soils. 31: 315–322.

Southampton Electrochemistry Group. 1985. Instrumental methods in electrochemistry. Chichester, West Sussex, UK: Ellis Horwood Ltd. 443 p.

Turov, Y.P.; Gerasimova, N.N.; Sagschenko, T.A.; Bieko, O.A. 1987. Group composition of the low molecular weight nitrogen bases of Samotlor crude oil. Petroleum Chemistry USSR. 27(1): 20–25.

Warshawsky, D. 1992. Environmental sources, carcinogenicity, mutagenicity, metabolism, and DNA binding of nitrogen and sulfur heterocyclic aromatics. Environ. Carcino. Ecotox. Revs. C(10): 1–71.

RESTORATION METHODS FOR DEEPWATER SWAMPS

William H. Conner, Kenneth W. McLeod, and Ellen Colodney[1]

Abstract—Planting in deepwater swamp areas is difficult and time consuming, and nursery-grown seedlings are often not suited for such conditions. Baldcypress [*Taxodium distichum* (L.) Rich.], water tupelo (*Nyssa aquatica* L.), swamp blackgum [*N. sylvatica* var. *biflora* (Walt.) Sarg.], and green ash (*Fraxinus pennsylvanica* Marsh.) have been planted at various flooded sites in South Carolina and Louisiana. One of the most effective means of planting these species in flooded situations was to heavily prune the lateral roots, grasp the seedling at the root collar, and push it into the soil. Excellent results have been obtained with baldcypress, whereas green ash was most sensitive to root pruning and water depth. Water tupelo and swamp blackgum were intermediate in response. Tree shelters are commonly used to reduce herbivory problems, and height growth inside the shelters is increased. Additional research is needed to compare operational performance of various techniques under conditions of interacting stresses such as herbivory and flooding.

INTRODUCTION

Deepwater swamps are found along rivers and streams of the Atlantic and Gulf Coastal Plains, and baldcypress [*Taxodium distichum* (L.) Rich.], pondcypress (*T. distichum* var. *nutans*), water tupelo (*Nyssa aquatica* L.), and Atlantic white-cedar (*Chamaecyparis thyoides*) are common tree species found in these areas. Both the Spanish in Florida and French in Louisiana found Indians using cypress (when used this way can be either baldcypress or pondcypress), which the Seminoles called “hatch-in-e-haw,” meaning everlasting (Neubreck 1939). Europeans quickly recognized that cypress wood was very rot resistant, strong, and easily worked, and efforts to establish a timber trade with Louisiana began around 1700 (Mancil 1980). Cypress was the staple commodity of the colonial lumber industry in Louisiana and the principal cash product for most colonists of the Lower Mississippi Valley until the 1790s, when sugar products became profitable.

The timber resource in the swamps seemed inexhaustible to early settlers with over 35.5 million m^3 (15 billion fbm) of cypress estimated in the Louisiana delta swamps alone (Kerr 1981). Harvesting in these wet swamps was seasonal in nature until the invention of the pullboat in 1889. Pullboats and the expansion of the railroad system (Sternitzke 1972), combined with a massive national campaign by cypress dealers (Burns 1980), resulted in a logging boom during the period from 1890 to 1925. Production of cypress lumber increased from 1.17 million m^3 (495 million fbm) in 1899 to over 2.36 million m^3 (1 billion fbm) in 1913 (Betts 1938, Mattoon 1915). By 1925, nearly all of the virgin timber had been cut and most of the mills closed. In 1933, only about 10 percent of the original standing stock of cypress remained (Brandt and Ewel 1989), but some cypress harvesting continued throughout the Southern United States on a smaller scale (Conner 1988).

Atlantic white-cedar logging began as early as 1700 in North and South Carolina (Frost 1987) and 1749 in New Jersey (Little 1950). Up to 50 percent of the Atlantic white-cedar area in North Carolina was cut between 1870 and 1890. As in other parts of the Southern United States, the rate at which Atlantic white-cedar swamps were logged greatly increased following the introduction of railroads, steam logging technology, portable sawmills, and dredging technology (Earley 1987, Frost 1987).

Unfortunately, the early exploitation of these swamplands occurred with little regard for sustainability. According to one logger, “We just use the old method of going in and cutting down the swamp and tearing it up and bringing the cypress out. When a man’s in here with all the heavy equipment, he might as well cut everything he can make a board foot out of; we’re not ever coming back in here again” (Van Holmes 1954). Nearly all of the virgin swamplands in the Southern United States were logged (Conner and Buford 1998).

Compared to other forest ecosystems, little silvicultural information is available for deepwater swamp forests, and management of these areas has been largely limited to clearcutting and highgrading (Johnson 1979, Williston and others 1980). Only recently have studies begun to investigate the response and recovery of these forests to harvesting practices, e.g., Aust and others 1989, 1997.

Cypress and water tupelo regenerate well in swamps where the seedbed is moist and competitors are unable to cope with flooding, but extended dry periods are necessary for the seedlings to grow tall enough to survive future flooding (Keeland and Conner 1999). Early height growth is important because seedlings can be killed in as little as 10 to 12 days of total submergence during the growing season (Demaree 1932). Although natural regeneration has also been the preferred method of regenerating Atlantic white-cedar

[1] Connor, Professor, Baruch Institute of Coastal Ecology and Forest Science, Clemson University, Box 596, Georgetown SC 29442; McLeod, Associate Professor, Savannah River Ecology Laboratory, Drawer E, Aiken, SC 29801; Colodney, Owner, Coastal Plain Conservation Nursery, 3067 Conners Drive, Edenton, NC 27932.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

(Laderman 1989), many projects now rely on artificial means to recreate Atlantic white-cedar habitat (Guidry 1999, Phillips and others 1993), and much research has been initiated to identify nursery practices to produce the best seedlings (Summerville and others 1999). Lack of seed source and herbivory problems are commonly reported as causing failure of natural regeneration projects for Atlantic white-cedar.

Coppice regeneration is also a possibility in cutover areas of cypress and water tupelo. Stumps of vigorous stock up to 60 years old can generally be counted on to send up healthy sprouts. Although many stumps sprout during the first growing season after logging, few of these sprouts survive, although results are often contradictory.

PROBLEMS LIMITING SWAMP REGENERATION

Flooding and Salinity

Human activities have inextricably altered the hydrology of almost all major alluvial floodplains in the United States within the last two centuries through the construction of dams, levees, and causeways and by channelization. Dams reduce the frequency and magnitude of downstream flooding, often extend the length of time the floodplain is inundated, and reduce the rates of erosion and sedimentation. Channelization and canal building, with associated levees or spoil banks, represent major modifications to natural hydrological patterns and often result in permanent impoundment of large areas of swamplands (Conner and Day 1989). Floodplain communities are adapted to a fairly predictable flood pulse, and alteration of the timing, duration, or magnitude of this flooding reduces diversity and productivity (Junk and others 1989). Because many swamp areas are permanently to nearly permanently flooded, natural regeneration is negligible (Conner and others 1986), and planting is difficult.

Another aspect of flooding that should be considered for vast coastal swamp areas is sea level rise and resulting increases in salinity (Conner and Brody 1989, Conner and Day 1988). While baldcypress and water tupelo can survive extended and even deep flooding (Hook 1984, Keeland and Sharitz 1995), they do not seem capable of enduring sustained flooding by water with salinity levels > 8 ppt (Conner and others 1997, McLeod and others 1996), and Atlantic white-cedar is very intolerant of salinity (Little 1950).

Hurricanes

Coastal Plain swamps have developed with windstorms as a normal, episodic part of the climatic regime (Conner and others 1989). Recent hurricanes such as Hugo (1989) in the southeast Atlantic Coastal Plain and Andrew (1992) in the northern Gulf caused extensive damage to forests in their paths. Such damage may be especially severe to the shallow-rooted hardwoods with large crowns that are common on alluvial floodplains. In the Congaree Swamp in South Carolina, 61 percent of the bottomland oaks (*Quercus* spp.) and 45 percent of the sweetgums were severely damaged by Hurricane Hugo, but only 3 percent of the baldcypress trees were affected (Sharitz and others 1993). Regeneration in hurricane-damaged areas may be limited if natural hydrological patterns have been altered.

Animals

Nutria were introduced from South America in the 1930s, and early plantings of baldcypress in Louisiana were destroyed (Blair and Langlinais 1960). The problem has not been solved (Brantley and Platt 1992, Conner 1988), and nutria have been reported to damage even mature trees (Hesse and others 1996). Beaver, deer, and feral hogs can also present a problem and can be fairly numerous in some areas. Conner and others (2000) found that clipped seedlings usually die, but baldcypress tends to resprout in many cases. Even such a small creature as the crayfish can become a problem to planted seedlings when food sources are low. Scraping of algae at the waterline can girdle a seedling and cause tip die-back (Conner 1988). Deer, field mice, and rabbits have deterimental effects on Atlantic white-cedar seedlings (Guidry 1999, Zimmerman and others 1999).

RESTORATION OF WETLAND SITES

Because of loose, unconsolidated muck commonly found in deepwater swamps, an easier method of planting seedlings was needed. The method adopted in our studies involves pruning the lateral root systems and clipping the tap root so we end up with a spear. Root systems of seedlings grown in unsaturated soils in the nursery are not appropriate to saturated soils, and large portions of this system will be lost once planted. A new root system appropriate for saturated soils will be produced. Since much of the original root system would normally be lost anyway, pruning it prior to planting generally does not cause problems.

Seedlings are bundled in plastic with damp peat moss around the tap root and either stored in coolers or transported to the field. Planting can be done along precisely laid out lines or by walking and estimating distances. By holding the seedling at the root collar, one pushes the seedling into the ground until the hand hits the soil surface. There are no tools to carry for digging holes, and one does not have to worry about filling in completely around the root. In very loose soils, the seedlings will need to be staked to keep vertical.

Root pruning does not work well with all species. While baldcypress and tupelo success rates have been high, green ash (*Fraxinus pennsylvanica* Marsh.) and swamp blackgum [*N. sylvatica* var. *biflora* (Walt.) Sarg.] success has been poor. Green ash seemed to do well in the first 1 to 2 years after planting but died in succeeding years (Conner and others 2000). One reason is that root systems never redeveloped on the seedlings when planted in wet areas.

Hand-bagged seedlings and balled and burlap seedlings have also been tried (Conner and others 1999). Balled seedlings planted on the sediment surface produced sufficient rooting down into the sediment to withstand complete drying of the surface water. However, there was no real benefit to using the hand-bagged or balled and burlapped seedlings since root-pruned baldcypress and water tupelo seedlings are less costly and easier to plant and survive just as well.

If the site is subject to drying out, a seedling with a large lateral root component is desirable. Increased root branching allows for more water and nutrient absorption, which results in more root growth and a healthier seedling. One such way to accomplish this is by way of specialized containers such as RootMaker. The pots have openings in the corners and at various levels in the sides and bottom. Roots are air-pruned and the design is such to prevent the secondary roots from becoming congested at the bottom of the pot. Seedlings are generally grown on a wire bench 20 cm above the moist soil to allow for good air circulation. Otherwise the roots of species like baldcypress and some oaks grow out of the openings, swell, and removal is awkward. The roots and secondary roots are well distributed within the pot.

Atlantic white-cedar is generally done using the same planting tools and techniques used in planting southern pines, and there is no apparent need to develop other techniques. Work by Weyerhauser has shown that rooted cuttings are more reliable and easier to grow than bare-root seedlings (Phillips and others 1998).

Tree shelters work well, especially in early years before seedlings emerge from the tube. After emerging from shelters, height growth is not as great, but the initial growth spurt keeps them above nonsheltered seedlings. Overall, baldcypress and water tupelo have grown well (Conner and others 2000). Tree shelters are generally advertised to breakdown within a few years due to ultraviolet light in the field. This has not been our experience, but the diameter growth of the seedlings has been strong enough to rip the shelters longitudinally and not girdle the trees.

CONCLUSIONS

Tree species composition in deepwater swamps remains fairly constant because so few species can tolerate extended flooding. With changes in hydrology, many of these forests are now flooded more than in the past. Regenerating swamp forests is not a simple matter of overcoming past disturbances, but is complicated by continuing disturbances and impacts, both natural and manmade. Successful regeneration is limited by flooding, and planting may be required to ensure that populations of these trees are established in areas impacted by manmade (logging) or natural (herbivory, hurricanes, salt-water intrusion) disturbances. Not all adverse site conditions can be overcome, and some solutions may not be cost effective in meeting landowner objectives. Simple root-pruning techniques allow easy planting of these areas, and survival and growth have been shown to be excellent in many areas if animal herbivory can be controlled.

ACKNOWLEDGMENTS

Many field volunteers including L. Wayne Inabinette, Michael R. Reed, and Keith T. Barnett have aided the research that has been conducted through the years on regeneration. Funding for this research has come from a variety of sources including McIntire-Stennis funds from Clemson University, the Savannah River Technology Center (Westinghouse Savannah River Company), and Financial Assistance Award Number DE–FC09–96SR18546 from the Department of Energy to the University of Georgia Research Foundation.

REFERENCES

Aust, W.M.; Mader, S.F.; Lea, R. 1989. Abiotic changes of a tupelo-cypress swamp following helicopter and rubber-tired skidder timber harvest. In: Miller, J.H., comp. Proceedings of the fifth biennial southern silvicultural conference; 1988 November 1–3; Memphis, TN. Gen. Tech. Rep. SO–74. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 545–551.

Aust, W.M.; Schoenholtz, S.H.; Zaebst, T.W.; Szabo, B.A. 1997. Recovery status of a tupelo-cypress wetland seven years after disturbance. Forest Ecology and Management. 90: 161–169.

Betts, H.S. 1938. Southern cypress. American woods. Washington, DC: U.S. Department of Agriculture, Forest Service, U.S. Government Printing Office. 6 p.

Blair, R.M.; Langlinais, M.J. 1960. Nutria and swamp rabbits damage baldcypress plantings. Journal of Forestry. 58: 388–389.

Brandt, K.; Ewel, K.C. 1989. Ecology and management of cypress swamps: a review. Gainesville, FL: Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, University of Florida.

Brantley, C.G.; Platt, S.G. 1992. Experimental evaluation of nutria herbivory on baldcypress. Proceedings of the Louisiana Academy of Science. 55: 21–25.

Burns, A.C. 1980. Frank B. Williams: cypress lumber king. Journal of Forest History. 24: 127–133.

Conner, W.H. 1988. Natural and artificial regeneration of baldcypress in the Barataria and Lake Verret Basins of Louisiana. Baton Rouge, LA: Louisiana State University. 148 p. Ph.D. dissertation.

Conner, W.H.; Brody, M. 1989. Rising water levels and the future of southeastern Louisiana swamp forests. Estuaries. 12: 318–323.

Conner, W.H.; Buford, M. 1998. Southern deepwater swamps. In: Messina, M.G.; Conner, W.H., eds. Southern forested wetlands: ecology and management. Boca Raton, FL: Lewis Publishers/CRC Press: 261–287.

Conner, W.H.; Day, J.W., Jr. 1988. Rising water levels in coastal Louisiana: implications for two forested wetland areas in Louisiana. Journal of Coastal Research. 4: 589–596.

Conner, W.H.; Day, J.W., Jr. 1989. Response of coastal wetland forests to man-made and natural changes in the environment with emphasis on hydrology. In: Hook. D.D.; Lea, R., eds. Proceedings of the symposium: forested wetlands of the Southern United States. Gen. Tech. Rep. SE–50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 34–43.

Conner, W.H.; Day, J.W., Jr.; Baumann, R.H.; Randall, J. 1989. Influence of hurricanes on coastal ecosystems along the northern Gulf of Mexico. Wetlands Ecology and Management. 1: 45–56.

Conner, W.H.; Inabinette, L.W.; Funderburke, E.L. 2000. The use of tree shelters in restoring forest species to floodplain delta: 5-year results. Ecological Engineering. 15(Suppl. 1): S47–S56.

Conner, W.H.; McLeod, K.W.; Inabinette, L.W. [and others]. 1999. Successful planting of tree seedlings in wet areas. In: Haywood, J.D., ed. Proceedings of the tenth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 201–204.

Conner, W.H.; McLeod, K.W.; McCarron, J.K. 1997. Flooding and salinity effects on growth and survival of four common forested wetland species. Wetlands Ecology and Management. 5: 99–109.

Conner, W.H.; Toliver, J.R.; Sklar, F.H. 1986. Natural regeneration of cypress in a Louisiana swamp. Forest Ecology and Management. 14: 305–317.

Demaree, D. 1932. Submerging experiments with *Taxodium*. Ecology. 13: 258–262.

Earley, L.S. 1987. Twilight for junipers. Wildlife in North Carolina. 51: 9–15.

Frost, C.C. 1987. Historical overview of Atlantic white cedar in the Carolinas. In: Laderman, A.D., ed. Atlantic white cedar wetlands. Boulder, CO: Westview Press: 257–264.

Guidry, J.I. 1999. Effects of herbivore pressure on Atlantic white-cedar rooted-cutting survival on a North Carolina coastal plain peatland. In: Shear, T.H.; Summerville, K.O., eds. Proceedings: Atlantic white-cedar: ecology and management symposium; 1997 August 6–7; Newport News, VA. Gen. Tech. Rep. SRS–27. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 18–21.

Hesse, I.D.; Conner, W.H.; Day, J.W, Jr. 1996. Herbivory impacts on the regeneration of forested wetlands. In: Flynn, K.M., ed. Proceedings of the southern forested wetlands ecology and management conference; 1996 March 25–27; Clemson, SC. Clemson, SC: Clemson University: 23–28.

Hook, D.D. 1984. Waterlogging tolerance of lowland tree species of the South. Southern Journal of Applied Forestry. 8: 136–149.

Johnson, R.L. 1979. Timber harvests from wetlands. In: Greeson, P.E.; Clark, J.R.; Clark, J.E., eds. Wetland functions and values: the state of our understanding. Minneapolis: American Water Resources Association: 598–605.

Junk, W.J.; Bailey, P.B.; Sparks, R.E. 1989. The flood-pulse concept in river floodplain systems. Canadian Special Publication of Fisheries and Aquatic Sciences. 106: 110–127.

Keeland, B.D.; Conner, W.H. 1999. Natural regeneration and growth of *Taxodium distichum* (L.) Rich. in Lake Chicot, Louisiana after 44 years of flooding. Wetlands. 19(1): 149–155.

Keeland, B.D.; Sharitz, R.R. 1995. Seasonal growth patterns of *Nyssa sylvatica* var. *biflora*, *Nyssa aquatica*, and *Taxodium distichum* as affected by hydrologic regime. Canadian Journal of Forest Research. 25: 1084–1096.

Kerr, E. 1981. The history of forestry in Louisiana. Alexandria, LA: Louisiana Forestry Association.

Laderman, A.D. 1989. The ecology of the Atlantic white cedar wetlands: a community profile. Biol. Rep. 85(7.21). Washington, DC: U.S. Fish and Wildlife Service, Office of Biological Services.

Little, S., Jr. 1950. Ecology and silviculture of whitecedar and associated hardwoods in southern New Jersey. Bull. 56. New Haven, CT: Yale University, School of Forestry.

Mancil, E. 1980. Pullboat logging. Journal of Forest History. 24: 135–141.

Mattoon, W.R. 1915. The southern cypress. Agric. Bull. 272. Washington, DC: U.S. Department of Agriculture.

McLeod, K.W.; McCarron, J.K.; Conner, W.H. 1996. Effects of inundation and salinity on photosynthesis and water relations of four southeastern Coastal Plain forest species. Wetlands Ecology and Management. 4(1): 31–42.

Neubreck, W.L. 1939. American southern cypress. Trade Promot. Ser. 194. Washington, DC: U.S. Department of Commerce.

Phillips, R.; Gardner, W.E.; Summerville, K.O. 1993. Plantability of Atlantic white-cedar rooted cuttings and bare-root seedlings. In: Brissette, J.C., ed. Proceedings of the seventh biennial southern silvicultural research conference; 1992 November 17–19; Mobile, AL. Gen. Tech. Rep. SO–93. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 97–104.

Phillips, R.W.; Hughes, J.H.; Buford, M.A. [and others]. 1998. Atlantic white cedar in North Carolina, U.S.A.: a brief history and current regeneration efforts. In: Laderman, A.D., ed. Coastally restricted forests. New York: Oxford University Press, Inc.: 156–170.

Sharitz, R.R.; Vaitkkus, M.R.; Cook, A.E. 1993. Hurricane damage to an old-growth floodplain forest in the Southeast. In: Brissette, J.C., ed. Proceedings of the seventh biennial southern silvicultural research conference; 1992 November 17–19; Mobile, AL. Gen. Tech. Rep. SO–93. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 203–210.

Sternitzke, H.S. 1972. Baldcypress: endangered or expanding species? Economic Botany. 26: 130–134.

Summerville, K.O.; Gardner, W.E.; Hinesley, L.E.; Bardon, R.E. 1999. Atlantic white-cedar plant production. In: Shear, T.H.; Summerville, K.O., eds. Proceedings: Atlantic white-cedar: ecology and management symposium; 1997 August 6–7; Newport News, VA. Gen. Tech. Rep. SRS–27. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 68–75.

Van Holmes, J. 1954. Loggers of the unknown swamp. Saturday Evening Post. 226: 32–33, 102–105.

Williston, H.L.; Shropshire, F.W.; Balmer, W.E. 1980. Cypress management: a forgotten opportunity. For. Rep. SA–FR 8. Atlanta: U.S. Department of Agriculture, Forest Service, Southeastern Area.

Zimmerman, G.; Mueller, R.; Brown, J. [and others]. 1999. The Penn Swamp experiments: an overview. In: Shear, T.H.; Summerville, K.O., eds. Proceedings: Atlantic white-cedar: ecology and management symposium; 1997 August 6–7; Newport News, VA. Gen. Tech. Rep. SRS–27. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 45–48.

WETLANDS SYSTEMS IN SOUTHERN THAILAND: THE ESSENTIAL RESOURCES FOR SUSTAINABLE REGIONAL DEVELOPMENT

Rotchanatch Darnsawasdi and Prasert Chitpong[1]

Abstract—Parts of Southern Thailand are inundated by water for months annually resulting in various wetlands including, among others, Tapi River Basin, Pak Panang River Basin, Songkhla Lake Basin, Pangnga Bay, Pattani River Basin, and Narathiwas Peat Swamp. Most wetlands perform functions such as flood retention, water filtration, bird and wildlife habitat, and tree growth. These wetlands are invaluable also for the value derived from: (a) commodity products such as timber, food, and chemicals; (b) nonconsumptive uses such as recreation and tourism; and (c) environmental attributes such as biodiversity, wildlife, and water quality. These values are derived from their hydrogeologic and biochemical functions, not dependent on the size of the wetland but on the intrinsic properties of the ecosystem, particularly their location within a watershed and positioning with respect to rivers, uplands, and seas. As many wetlands have been transformed and deteriorated, research into these wetland systems and their relationships with human activities are required. Integrated and participatory approaches to management of these wetlands are also recommended.

WETLANDS IN SOUTHERN THAILAND

Wetlands in Southern Thailand may be classified into nine categories according to their formation, location, and morphology.

Open Sea Coasts, Sandy Beaches, and Offshore Islands

These wetlands are found along the coastlines and offshore islands of the peninsula. In most cases, they occur in association with other coastal wetlands, such as mudflats and mangroves. There are many islands and small rocky outcrops off the west coast of the peninsula, most of which have never been surveyed. Some islets are known nesting sites for the Pacific Reef Egret (*Egretta sacra)* and terns such as *Stema cougalli*, *S. sumatrana, S. bergii,* and *S. anaethetus*. The Beach Thick-Knee (*Esacus magnirostris)* is also located, but is restricted to, sand beaches on offshore islands, whereas both the Malaysian Plover *Charadrius peronii* and the Little Tern *S. albifrons* breed on the sand beaches of mainland and island coasts.

Intertidal Mudflats and Mangroves

This type of wetlands is most known and is of great conservation value in Southern Thailand, having enormous importance in sustaining both inshore capture fisheries and aquaculture. The most extensive and species-rich mangrove ecosystems are found along the west coast of the peninsula, which supported 63 percent of the total mangrove area of 2871 km² at the end of 1982. There are also several important mangrove and mudflat sites on the east coast of the peninsula. Extensive areas have been converted to shrimp farms.

Intertidal mudflats are important in terms of wildlife conservation. These wetlands support a huge number of passage and wintering herons and shorebirds. The mangroves themselves still support nesting colonies of cormorants, herons, and a few Lesser Adjutants (*Leptoptilos javanicus*), together with considerable numbers of some birds of prey such as Brahminy Kites (*Haliastur indus*). Two other important species are the Brown-winged Kingfisher (Pelargopsis amauroptera) and the Mangrove Pitta (Pitta megarhyncha), both of which are restricted to the west coast.

Lower Perennial Rivers

Wetlands of this category in Southern Thailand consist of meandered rivers and riverine marshes. In most cases, these areas are deforested, except where narrow fringes of fresh and brackish water swamp woodlands remain along the riverbanks. As most riverbanks usually support a high human population density, relatively few waterfowl are found. Only certain grounds with strong conservation movement may support breeding colonies or large roosts of herons or storks, together with a few Black Kites (*Milvus migrans*). Vertical earth banks also support a few nesting Pied Kingfisher (*Ceryle rudis*), whereas mud and sand banks support passage shorebirds as well as the resident River Lapwing (*Vanellus duvaucelii*). There are a few colonies of Plain Sand martins (*Riparia paludicola*). The rare and local Jerdon’s Bushchat (*Saxicola jerdoni)* appears to be associated with stands of the tall grass (*Saccharum arundinaceum*) in riverine floodplains. The species may have decreased greatly as a result of the burning of such vegetation in order to open up seasonally inundated alluvial soils for dry season cultivation.

Large areas of seasonally inundated land also lie along many rivers in Southern Thailand, particularly along large rivers such as the Tapi River in Suratthani and the Pak Phanang River in Nakhonsrithammarat. Many such areas are utilized for the cultivation of vegetables or rice as the

[1] Darnsawasdi, Faculty of Environmental Management, and Chitpong, Faculty of Natural Resources, Prince of Songkla University, Hatyai, Songkhla, 90112.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

seasonal flooding recedes, yet they may perhaps be of considerable value to waterfowl.

Upper Perennial Rivers

Wetlands of this category are usually characterized by large streams in mountainous or hilly terrain, often in association with waterfalls and rapids. Because the upper reaches of many rivers are still largely forested, they may continue to support a considerable diversity of wildlife, including a great many nonaquatic species. Some of the important sites are already enclosed within the boundaries of national parks, forest reserves, and wildlife sanctuaries.

Several kinds of birds such as Fish-Eagle, Crested Kingfisher (*Ichthyophaga humilis*), *Sarcogyps calvus, Pavo muticus*, and *Megaceryle lugubris* are sometimes found in these wetlands. Many such sites, including many small streams and torrents, may be of critical conservation importance for various frogs and toads. There has never been any comprehensive assessment of the conservation status of Thai amphibians, however, so that many important sites remain undocumented.

Freshwater Lakes, Ponds, and Associated Marshes

These wetlands are usually related to the Lower Perennial Rivers. They comprise abandoned oxbow lakes, inundated sinkholes, and submerged flood plains. Most sites are small (less than 500 ha) and are usually surrounded by rice paddies. A great many lie within 1 to 2 km of major rivers. Like the Lower Perennial Rivers, almost all have been utilized mostly for agriculture causing habitat disturbance, which greatly limits the residence of wildlife. The vegetation is usually restricted to floating or submerged aquatic plants alternating with open water and supports such breeding waterfowl as Pheasant-tailed Jacana (*Hydrophasianus chirurgus*). The highest diversity of breeding and wintering birds is usually found at those sites, which contain extensive areas of emergent vegetation, especially Phragmites or Typha. Many such sites are of international or national importance for their wintering duck populations.

Water Storage Reservoirs

Wetlands of this category are manmade or constructed wetlands. Most larger reservoirs are constructed for hydroelectric power generation, and are situated in steep forested river valleys. They tend to support fewer native wildlife, but may lead to a new ecosystem that can support the other wildlife. Many irrigation reservoirs, on the other hand, are situated in the plains, are relatively shallow, and show considerable annual fluctuation in area. Such sites may be of value for wintering and passage wading birds. A few such sites are known to support wintering concentrations of ducks as well as such breeding species as *Porphyrio porphyrio*.

Rice Paddies

In most circumstances, rice paddies are river flood plains, which were transformed into rice fields. They constitute an important and very extensive, seasonally inundated habitat for birds. The extent to which these areas can be utilized, however, depends upon the availability of undisturbed roosting and nesting sites such as clumps of trees and permanent water bodies. Egrets and herons feed to a considerable extent in flooded paddies; cormorants utilize ditches around their margins, and the Asian Open-billed Stork (*Anastomus oscitans)* feeds both in flooded and dry paddies. Areas of hard, dry paddy stubble in the late dry season are utilized by huge numbers of nesting Oriental Pratincoles (*Glareola maldivarum)*.

Freshwater Swamp Woodlands

Detailed information on the history of such sites is usually lacking. In most cases, such swamp woodlands may be merely the degraded remnants of primary peat swamp forests. The most disturbed sites, particularly those having been subject to repeated burning, are species poor and are usually dominated by *Melaleuca leucadendron*. Some *Alstonia spathulata* may occur in the less disturbed sites, as at Thale Noi Non-Hunting Area in Peninsular Thailand. Small areas of other freshwater swamp woodland formations may occur along the banks of larger rivers and other sites, which are subject to occasional inundation. Such sites are of considerable importance for nesting and roosting colonies of larger waterbirds, such as cormorants, herons, and storks.

Peat Swamp Forests

These are wetlands dominated by a species-rich forest community growing on waterlogged peat. They are botanically very rich and may be of great conservation importance for amphibians and for some fish, such as the walking catfish *Prophagorus nieuhofi* and possibly the highly endangered Asian Bonytongue *Scleropage fonnosus*. In terms of their avifauna, they are important in supporting many arboreal members of the lowland forest community, which are scarce or absent elsewhere due to the almost complete destruction of terrestrial lowland forest. Phru Toh Daeng or Phru Sirindhom, in Narathiwat Province, is the only example of this habitat remaining in Thailand, although many other areas, now dominated by the species-poor Melaleuca woodland, may be degraded remnants of this type.

MANAGEMENT OF WETLANDS IN SOUTHERN THAILAND

Wetlands are now known for many people in Southern Thailand. So are their functions and values. Some management practices have long been imposed on these areas although not all wetlands are properly managed. These practices may be categorized roughly, according to responsible organizations, into the following five categories.

Conservation Imposed by the Government

Several wetlands, including many intertidal mangroves, forested upstream wetlands, and some primary peat forests have been designated by the government as conservation areas. Among others, national parks, nonhunting areas, and wildlife sanctuaries are most common. These practices usually target objectives such as conserving wildlife, plant species, and wetland ecology. Human activities have, to some level, been controlled by the central government through responsible departments such as the Royal Forestry Department (RFD) and the Department of Fisheries (DOF). However, ecotourism, which has been allowed and promoted in many areas, may lead to more of the general public entering these wetlands and thus posing a threat to them. One good example of wetlands with this type of management can be seen at Phru Toh Daeng in Narathiwas province.

Management Imposed by Responsible Government Agencies

Similar to the first category, this type of management, usually invented by responsible authorities in line with the government policy, aims at controlling the uses of the wetlands where such agencies have been granted special right to access and manage. Whereas the former strategy usually aims to conserve the wetlands, these agencies, Electricity Generating Authority of Thailand (EGAT) and Royal Irrigation Department (RID), for example, could seek to optimize their principal objectives while taking into account conservation practices as secondary objectives wherever possible. A constructed wetland complex at Chewlam Dam in Suratthani is a good example of wetlands with this type of management.

Management Introduced and Supported by Local People

In line with the increasing awareness of the general public, some wetlands have been conserved by various groups of local people, many of whom rely to some extent on the wetlands. Without support from the government, this type of management may lead to resource-use conflicts because some other groups sometimes want to make different uses of the same wetlands. It is, however, believed to be most cost effective and more in line with new legislation and modem resource management paradigms—the integrated and participatory approach, in particular.

Also, there are several projects funded by overseas organizations also suggesting a similar approach. Whereas the first strategy will likely remain essential, this management practice is gaining more acceptance. A good example of wetlands with this type of management is Talenoi, Phattalung. There are several other examples in Songkhla Lake Basin and some intertidal mangrove areas in Suratthani and Pattani.

Owned by the Government but Accessible Freely by the General Public

This type of management is common in most midstream wetlands such as flood plain grass swamps and some peat forests, most of which are of lower value compared to the other natural wetlands. The government usually designates these areas as public lands but has paid less attention to control of their utilization. As a result, it is considered less efficient and often leads to misuses or deterioration of these wetlands or both. These practices also result in many conflicts either among different users or between villagers and the government. Several grass wetlands in Suratthani and Nakhonsrithammarat are obvious examples of these wetlands.

Experimental Management by Researchers

Several action research projects have been conducted by many organizations including universities, international organizations, and some government agencies. Important research teams include, among others, Prince of Songkla University, Kasetsart University, Wetlands International, ASEAN, USAID-CRMP, DANCED, National Research Council of Thailand, Department of Fisheries, and Royal Department of Forestry. These research projects usually led to a set of management guidelines, some of which were adopted by responsible authorities. Some also led to actual management practices based on scientific data and reasoning. Local people also support many research outputs.

Many organizations including responsible government agencies, universities, nongovernment organizations, local administrative organizations, and local people have played different roles in wetlands management, sometimes leading to resource-use conflicts. A more integrated approach to management, which takes into account different people or organizations or both, their expectations, responsibilities, and underlying legislation have gained more acceptance and would likely result in restructuring of the region's wetlands management scheme. The Songkhla Lake management is one good example of the new paradigm.

Research into Wetlands in Southern Thailand

Numbers of research into intertidal wetlands in Southern Thailand were and have been carried out using satellite imagery, aerial photography, ground surveys, and laboratory testing to examine various aspects of major wetlands in the region. Among many, the extent of wetlands, the ecosystems, waterfowls, and other biological species have been known to some degree. Research into the only remaining primary peat forest at Phru Toh Daeng has also been carried out, and an integrated management plan for the site was developed. Several aspects of Songkhla Lake, the largest and most important lagoonal wetland complex, were and have also been studied. Many management proposals have been presented.

Apparently, past research into wetlands in this region had paid more attention to estuarine wetlands, mangroves in particular. More recent research has also extended to cover rain-forested wetlands and some peat forests. Most of the midstream wetlands, such as flooded grasslands, have been left unexplored. So have many small upstream and isolated wetlands.

Different research teams have collected various information about wetlands flora and fauna. Research interests have, however, focused upon the wetlands being habitats for wildlife, various bird species in particular. Some research into proper management has been carried out. Most research in the past aimed at each particular wetland although many are interrelated and could affect one another. A few projects applied systems approach to wetland research. Songkhla Lake management is among those who based their investigation on the wetlands system.

Just recently, Prince of Songkla University, in cooperation with Wetlands International, has set up a group of research projects aiming at wetlands inventory and management in Southern Thailand. Only a few issues have yet been handled so far.

RECOMMENDATIONS

The authors suggest that wetlands in Southern Thailand, based on recent investigation, should be categorized into three groups.

Mountainous Upstream Riverine Wetlands

These wetlands should include upper perennial rivers and hilly water storage reservoirs.

Lowland Midstream Riverine Wetlands

These wetlands should include lower perennial rivers, freshwater lakes, ponds and associated marshes, freshwater swamp woodlands, lowland water storage reservoirs, and rice paddies.

Lowland Coastal and Marine Wetlands

These wetlands should include open sea coast, sandy beaches and offshore islands, intertidal mudflats and mangroves, and peat swamp forests.

The new scheme takes into consideration the system of wetlands and thus should emphasize the interrelationships among different wetlands, which call upon more systems approach to management and further research.

Future research into wetlands should extend from lowland intertidal mudflats and mangroves and a few major upstream wetlands to peat forests, both intact and degenerated, and lowland midstream wetlands, which also support various forms of wildlife, natural conservation, and human activities. Issues for investigation should extend from biology or ecology or both to include human activities, problems and their causes, and appropriate management practices that would lead to sustainable wetlands in this region.

REFERENCES

Damsawasdi, R. 2000. Field investigation of wetlands in Southern Thailand. Hatyai, Thailand: Prince of Songkla University. 52 p.

Scott, Derek A., ed. 1989. A directory of Asian wetlands. Gland, Switzerland; Cambridge, U.K.: IUCN, The World Conservation Union: 667–736.

THE DEVELOPMENT OF A DECISION SUPPORT SYSTEM FOR PRIORITIZING FORESTED WETLAND RESTORATION AREAS IN THE LOWER YAZOO RIVER BASIN, MISSISSIPPI

Angela A. Davis, Barbara A. Kleiss, Charles G. O'Hara, and Jennifer S. Derby[1]

Abstract—The Eco-Assessor, a GIS-based decision-support system, has been developed for the lower part of the Yazoo River Basin, Mississippi, to help planners and managers determine the best locations for the restoration of wetlands based on defined ecological and geographic criteria and probability of success. To assess the functional characteristics of the potential restoration areas, the data layers are organized by hydrology, water quality, and habitat. The overall potential restorability, or the predicted physical ability of a tract of land to sustain a functional wetland, is also considered. Because an exact spatial representation of wetlands in the lower Yazoo River Basin does not exist, surrogate data layers are used to predict locations that might be restored to a functional wetland. The Eco-Assessor analyzes the following data layers by using a ranking system: geomorphology, soils, mature forest cover, farmed wetlands, flood frequency, topographic depressions, River Reach File Level 3 streams, wildlife management areas, conservation areas, primary roads, secondary roads, permanent water, and landscape factors. Various categories of each data layer are assigned a rank. A higher rank indicates that a particular geographic area has a higher probability of being restored to a functional wetland. Ranks for all the data layers are summed to result in a cumulative rank which can then be used to determine the areas that, overall, are the most likely to be successfully restored to a functional wetland. The ranking system method provides a means to analyze various restoration scenarios. A restoration scenario can be defined in a way that may focus equally on all functions, focus on one function, or focus on a particular geographic area.

INTRODUCTION

Forested wetlands, once the predominant land cover on the Mississippi River Alluvial Plain (Creaseman and others 1992), provide habitat for wildlife, water-quality benefits, flood storage, and many other ecological and environmental benefits. Ongoing efforts of many Federal, State, and local agencies and organizations to restore forested wetlands have been successful. However, the lack of quantitative methods for prioritizing the selection of wetland restoration areas has meant that a less than optimum approach has been taken in the evaluation, selection, and restoration of forested wetlands.

In the past, selecting areas for wetland restoration was conducted based largely on identifying landowners willing to sell their land. This selection method, coupled with the lack of a quantitative approach for selecting and prioritizing potential restoration sites in past efforts, caused the process of forested wetland restoration to overlook how the restoration activity occurred on the landscape. Also, forested wetland "restoration" was often undertaken with little regard as to whether wetland functions were replaced. Until recently, the evaluation of alternate forested wetland restoration scenarios was a task that was impeded by the general unavailability of input data, the cost of developing digital data, the lack of sufficient tools for developing and comparing alternate scenarios, and the difficultly of integrating results into various types of independent analysis. Recent improvements in data availability, geographic information system (GIS) applications, computer technology, and general software technology have made possible the development and use of powerful decision-support systems (DSS) that integrate data, provide flexible analysis methods, and allow the easy interchange of data between various software analysis tools.

The DSS presented in this paper is the result of an interagency agreement between the U.S. Department of the Interior, Geological Survey, Water Resources Division, and the U.S. Environmental Protection Agency. The purpose of the agreement was to develop a DSS to facilitate the rapid generation and consideration of many alternate forested wetland restoration scenarios for the study unit located in the Yazoo River backwater area.

STUDY AREA

The study area examined was the Yazoo backwater area of the lower Yazoo River Basin, which included at least a portion of the following six counties in Mississippi: Warren, Issaquena, Sharkey, Yazoo, Humphreys, and Washington. The Yazoo backwater area is in the southern portion of the Yazoo River Basin, bounded by the Mississippi River levee on the west and the valley wall on the east and south. The southern tip of the Yazoo backwater area is just north of Vicksburg, MS. The area extends north about 100 km to a latitude near Belzoni, MS (fig. 1).

[1] Davis, GIS Specialist, U.S. Department of Interior, Geological Survey, National Wetlands Research Center, 700 Cajundome Boulevard, Lafayette, LA 70506; Kleiss, Supervisory Hydrologist, and O'Hara, Hydrologist, U.S. Department of Interior, Geological Survey, 308 South Airport Road, Pearl, MS 39208; and Derby, Ecologist, U.S. Environmental Protection Agency, 61 Forsyth Street, Atlanta, GA 30303.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

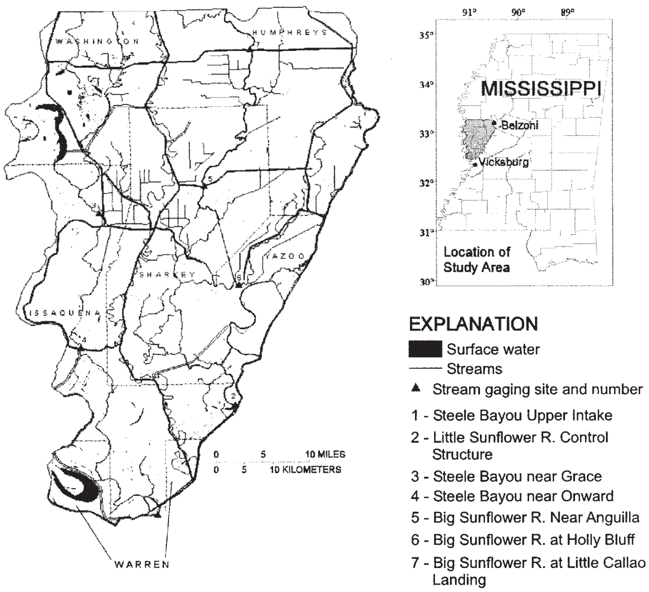


Figure 1—The boundaries of the study area located in the Lower Yazoo River Basin in Mississippi.

PROCEDURES

Ecological Rule Development

Wetland functions are often used as comparative evaluation criteria for the ecological merit of wetlands. Hydrogeomorphic Assessment (HGM) uses wetland functions to evaluate existing wetlands at a site-specific level (Brinson 1993). The Eco-Assessor DSS uses similar principles to HGM, but spatial data at a landscape level is grouped by wetland functions to evaluate whether or not a sustainable wetland could exist at a location where there currently is not one. In this study, common wetland functions have been grouped into four categories: restorability, hydrology, water quality, and habitat functions. For the purposes of the initial model generation, the hydrology, water quality, and habitat function categories were given approximately the same weight in the overall analysis, whereas restorability has a weight of approximately half that of the other three. However, the model is adaptable, using check boxes and pull-down menus, which allow the model to be run with any preferred subset of functional categories emphasized, and it also allows the input of new ranking values for any of the functions. This kind of customization can address particular resource needs or test hypotheses about the impact of ranking decisions and weights. Also in some cases, a particular function appears in more than one functional category. In these cases, the function is considered critical enough that the redundancy is justified.

Restorability—This functional category is defined as the physical ability of a parcel of land to sustain a functional wetland. The wetland restorability section of the Eco-Assessor DSS provides for the evaluation of geomorphology, soils, regeneration distance, and farmed wetlands spatial data layers. A summary of the ranks assigned to functions within this group is presented in table 1.

Geomorphology, as derived by Saucier (1994), consists of abandoned channels, backswamps, and pointbar/valley trains. Abandoned channels are considered the lowest land formations in terms of elevation, become inundated most frequently, and are given the highest rank. Backswamps are

Table 1—Summary of the ecological rules used in the Eco-Assessor decision-support system for the wetland restorability function

Wetland function	Spatial data layer	Data variables	Functional restoration ranking
Wetland restorability	Environment of deposition	Abandoned channel	5
		Backswamp	3
		Pointbar	1
	Soils	Hydric	10
		Nonhydric	1
	Regeneration distance	Within 60 m of mature forest	5
		Between 60 and 120 m	3
		Greater than 120 m	1
	NCRS farmed wetlands	Farmed wetland	5
		Other	0

NCRS = Natural Resource Conservation Service.

Table 2—Summary of the ecological rules used in the Eco-Assessor decision-support system for the wetland hydrology function

Wetland function	Spatial data layer	Data variables	Functional restoration ranking
Hydrology	Flood frequency	Within the 0.5-year flood	20
		Within the 2-year flood	10
		Beyond the 2-year flood	5
	Topographic depressions	Topographic depressions	20
		Other	1

slightly higher in elevation than abandoned channels but are still low enough to be frequently inundated. Pointbar/valley trains are slight ridges on the land's surface, are the least wet, and are given the lowest rank.

Hydric soils are defined by the U.S. Department of Agriculture, Natural Resources Conservation Service (NRCS), and modified for the Yazoo River Basin by using the U.S. Army Corps of Engineers report "Delineation of Wetlands of the Yazoo River Basin in Northwestern Mississippi," Misc. Paper EL–92–2 (Kirchner and others 1992). The presence of hydric soils is important to the sustainability of wetlands.

Restoration areas that are near existing mature forest tend to have much higher species diversity than areas that are far from an existing stand of mature forest (Allen 1990). The rapid natural regeneration of forest will occur out to a distance of 60 m from existing forest (Allen 1997), and areas within 60 m of existing forest are given the highest ranking in this model.

The criterion for NRCS "farmed wetlands" is for areas (excluding pothole, playa, and pocosin) that have a 50-percent chance of being flooded or ponded for at least 15 consecutive days during the growing season (U.S. Department of Agriculture 1996). Areas classified as NRCS "farmed wetlands" indicate places on the landscape that are inundated for a significant duration and are, therefore, very likely to maintain sustainable wetlands and are given a high rank.

Hydrology—The hydrology function is a representation of the hydrologic regime of a given cell within the landscape. Both the frequency and duration of flooding are considered. The hydrology section includes and provides for the evaluation of flood frequency and topographic sinks spatial data layers. A summary of the ranks assigned to functions within this group is presented in table 2.

Those areas indicated as flooded by a 0.5-year flood are the most frequently inundated and, therefore, most likely to sustain a wetland. The areas within the 2-year flood are not inundated as often but are still viable sites for a wetland. Those areas beyond the 2-year but within the 100-year flood are the least likely to be inundated on a regular basis and are the least likely areas for a sustainable wetland. The flood frequency data were created by compositing the nominal flood image data for the 0.5-, 2-, and 100-year nominal flood images into a composite nominal flood frequency image (fig. 2).

Topographic depressions indicate places on the landscape where water is likely to pond because they are points of low elevation surrounded by points of higher elevation. Once water enters a depression, there is no outlet through which the water is able to drain. The lack of an outlet causes the

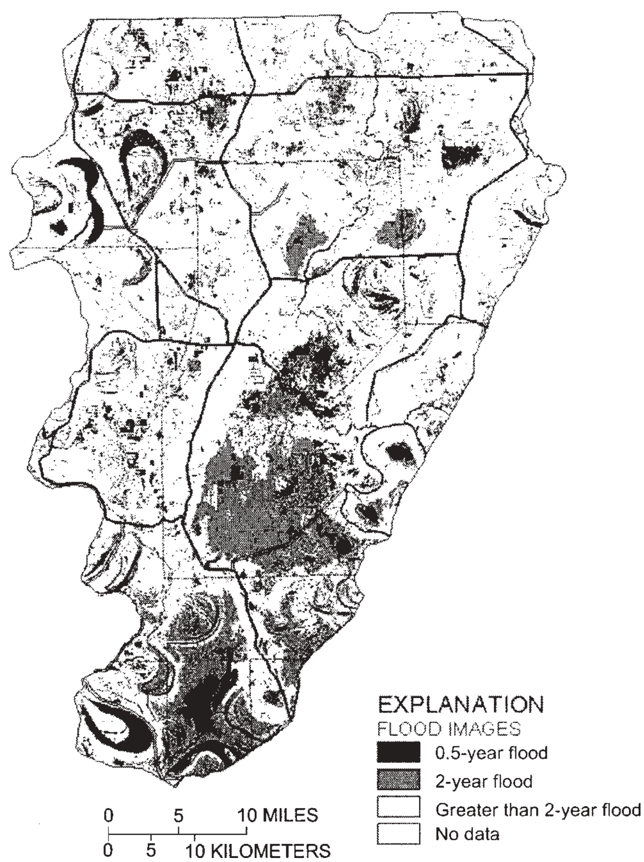


Figure 2—The spatial extent of the 0.5- and 2.0-year floods in the Lower Yazoo River Basin, as determined from satellite imagery

water to remain in the sink until evaporation or seepage or both occur, resulting in floodwater storage and possible interaction with ground-water resources.

Water quality—The water-quality function gives weight to areas on the landscape that would filter, trap, or degrade chemical components such as nitrogen and phosphorous commonly found in the water. The water-quality section includes and provides for the analysis of spatial data layers such as stream buffers, flood frequency, and topographic sinks. A summary of the ranks assigned to functions within this group is presented in table 3.

Flood frequency is a factor in both the wetland hydrology function as well as the wetland water-quality function. In the water-quality function grouping, those areas indicated as flooded by a 0.5-year flood are the most frequently inundated. Therefore, the areas within the 0.5-year flood are the most likely to sustain a wetland. The areas within the 2-year flood are not inundated as often but are still viable sites for a wetland. Those areas beyond the 2-year but within the 100-year flood are the least likely to be inundated on a regular basis and are the least likely areas for a sustainable wetland.

Stream buffers are assigned by stream level beginning with a 10-m buffer because no less than a 10-m stream buffer is a minimum necessary to filter nitrogen and phosphorous (Dillaha and others 1989, Howard and Allen 1988). Stream buffers have been shown to control the flow of nitrate, phosphorous, sediment, and sediment-borne chemicals in surface runoff and shallow ground water (Lowrance and others 1997).

Topographic depressions allow water to pond in areas with no outlet through which water can drain. If water remains trapped in a topographic depression for extended periods of time, sediments fall out and anaerobic processes begin. The amount of sediment that will be deposited in depressional areas is higher than in nondepressional areas because longer hydroperiods allow for longer settling time (Hupp and Morris 1990, Kleiss 1996). Both the trapping of the sediments and the degradation of chemicals through anaerobic processes improve overall water quality (Mitsch and Gosselink 1993).

Habitat—The habitat function gives weight to areas of the landscape in which wildlife may easily persist. The habitat section considers proximity factors, such as distance to wildlife management areas and conservation areas, distance away from primary and secondary roads, proximity to permanent water bodies, and landscape factors such as forest block size and core area. A summary of the ranks assigned to functions within this group is presented in table 4.

Proximity Functions

The public lands are divided into two categories. The first category contains the managed wildlife areas, National Wildlife Refuge And State Wildlife Management Areas. The second category contains general conservation lands, Public Land Restoration, Delta National Forest, Farmer’s Home Administration, and Wetland Reserve Program lands. Expanding existing public lands, when managed appropriately, greatly benefits wildlife by increasing the interior space available for habitat. Also, any connections that can be made between two patches of land add valuable corridors for the movement of wildlife (Allen and Kennedy

Table 3—Summary of the ecological rules used in the Eco-Assessor decision-support system for the wetland water-quality function

Wetland function	Spatial data layer	Data variables	Functional restoration ranking
Water quality	Flood frequency	Within the 0.5-year flood	15
		Within the 2-year flood	10
		Beyond the 2-year flood	5
	Stream buffers	Stream level 1: within 90 m	15
		Stream level 2: within 80 m	15
		Stream level 3: within 70 m	15
		Stream level 4: within 60 m	15
		Stream level 5: within 50 m	15
		Stream level 6: within 40 m	15
		Stream level 7: within 30 m	15
		Stream level 8: within 20 m	15
		Stream level 9: within 10 m	15
		Stream level 0: within 5 m	15
		Other	0
	Topographic depressions	Topographic depressions	15
		Other	0

Table 4—Summary of the ecological rules used in the Eco-Assessor decision-support system for the wetland habitat function

Wetland function	Spatial data layer	Data variables	Functional restoration ranking
Habitat	Wildlife management areas	Within 250 m of wildlife management areas	10
		Between 250 and 500 m	5
		Between 500 and 1000 m	1
		Beyond 1000 m	0
	Conservation areas	Within 60 m of conservation areas	5
		Between 60 and 120 m	3
		Between 120 and 500 m	1
		Beyond 500 m	0
	Primary roads	Within 50 m of primary road	0
		Between 50 and 500 m	1
		Beyond 500 m	3
	Secondary roads	Within 50 m of secondary road	1
		Between 50 and 500 m	2
		Beyond 500 m	3
	Permanent water	Within 150 m of permanent water	5
		Between 150 and 1000 m	1
		Beyond 1,000 m	0
	Forest block size	Between 1 and 10 acres	1
		Between 10 and 320 acres	5
		> 320 acres	10
	Core area ratio	Ratio of core area to total area of patch > 0.66	10
		Between 0.33 and 0.66	5
		< 0.33	1

1989). The expansion of existing wildlife management areas has an added benefit because the management of wildlife is already the top priority in this area.

Distances away from primary and secondary roads were adapted from a Louisiana Department of Natural Resources study by Kinler (1994). The study ranked human disturbances by distance and type of disturbance. For the purposes of the Eco-Assessor, primary roads are considered a constant disturbance and secondary roads are considered only a frequent disturbance.

Being near permanent water bodies is beneficial to wildlife because water is a requirement for basic living needs. In a study conducted in the same general geographic area (Wakeley and Marchi 1992), six species were chosen for a habitat evaluation of the Upper Steele Bayou area in Mississippi. The six species, which are common to bottomland hardwood forest, include the barred owl (Strix varia Barton), gray squirrel (Sciurus carolinensis Gmelin), Carolina chickadee (Parus carolinensis Audubon), pileated woodpecker (Dryocopus pileatus Linnaeus), wood duck (Aix sponsa Linnaeus), and mink (Mustela vison Schreber) (Wakely and Marchi 1992). Of these six species, the pileated woodpecker has the most quantitatively specific habitat requirements according to the U.S. Fish and Wildlife Service Habitat Suitability Index Model (HSI). Minimum distance requirements to and from permanent water bodies, as well as minimum forest block size, are given in the HSI. The HSI for the pileated woodpecker indicates that nesting habitat generally is not observed greater than 150 m from water bodies (Schroeder 1982). The habitat requirements for the pileated woodpecker are often used as a representation of the habitat requirements for other cavity nesting birds by natural resource agencies (Renken and Wiggers 1993).

Landscape Factors

The landscape can be assessed using such landscape factors as patch size, core area, and patch shape. A patch of forested land < 1 ac does not provide sufficient habitat for wildlife (Wakely and Marchi 1992); therefore, any patch < 1 ac is dissolved. The larger the patch size the more that habitat is benefited. There are two categories of wildlife species: generalists and specialists. Generalists can live in patches of many shapes and sizes because their populations are larger and they are highly mobile. It is the specialists that require the greatest conservation efforts. Specialists require large patches of forest with greater interior area and less edge (Kinler 1994). It is important to provide for the needs of the specialists by giving weight to larger patches of land.

The ratio between core area and total patch area is used to give more weight to patches of land that have a greater portion of interior area. Core area is defined by the Fragstats manual as “the area within a patch beyond some specified edge distance or buffer width” (McGarigal 1995). Any land that is in the interior of a patch more than 100 m from the edge is considered core area. For a given patch of land, the number of cells considered core area divided by the number of cells in the entire patch, results in a ratio of core area to patch shape. A long thin patch of land would receive a lower ratio, whereas a long wide patch of land would receive a higher ratio. A patch of land with a high ratio would provide wildlife habitat with fewer edge effects and more interior space. More interior space available in a given patch gives rise to the number of interior species and species diversity (Ohman and Eriksson 1998).

GIS Data Layers

The scope of this project did not include the collection of new data to develop new data layers; therefore, the data layers are the best information currently available from agencies working within the State of Mississippi. In some cases, data layers are numerically derived from existing layers. All data layers were resampled to generate a grid of 25-m cell size.

Land use image data—In 1988, the U.S. Army Corps of Engineers, Vicksburg District (USACE), collected satellite image data for the purpose of generating a land use classification data set. Land use in the study area was based on these data and was provided by the USACE in the Universal Transverse Mercator (UTM) projection, North American Datum (NAD) 27. The satellite image data were divided into the following land use categories: cotton, soybeans, corn, rice, herb1, herb2, pasture (grass), bottomland hardwood, swamp, river, lake, and pond. In areas within the satellite images where the land use was obscured by cloud cover or where the spectral response is similar to that provided by sandbars, selected pixels are classified as sandbar/clouds. For areas not classified, null data values indicate the spatial extent of the study unit.

After generating land use data, the USACE adjusted the land use. The adjusted land use data differ from the original land use classifications where it is known the land has been put into habitat or land use management programs such as wildlife management areas, wetland reserve program lands, national wildlife refuges, and conservation reserve program lands.

Flood image data—The USACE collected satellite imagery of flood scenes to accomplish several specific tasks. The primary tasks included compiling areas inundated by floods of a known stage to develop a stage-area relation and spatially characterizing areas inundated by a flood of a given frequency.

A relation between flood stage and inundated area is developed in the form of a stage-area curve by selecting images for flood scenes of various stages and determining areas inundated. The stage-area curve is useful in estimating flood extent. Dates and stages for flood scenes used by the USACE in generating a stage-area relation for the lower Yazoo River Basin are listed in table 5. By comparing those dates and times with satellite image data availability, it was possible to select images that corresponded to specific flood events for areas in the vicinity of specific gages within the study area. Image data for each gaged area for flood events of specific frequency were composited into a mosaic of images. The nominal flood image data generated from composited images provided a view of a simulated flood in which all areas are at the stage for the specific flood considered. The 2-year nominal flood image scene is shown in figure 2.

Hydric soils data—The soils data mapped and provided by the USACE show the extent of hydric, nonhydric, and riverbottom soils throughout the study area (fig. 3). The presence of hydric soils in a location provides one of the physical indicators that the location might support wetland function. Hydric soils, as defined by the NRCS were for the Yazoo River Basin in northwestern Mississippi (Kirchner and others 1992).

Geomorphology data—The base source for hydrogeo-morphology GIS data compiled at 1:250,000 scale was the report “Geomorphology and Quaternary Geologic History of the Lower Mississippi Valley” (Saucier 1994). This data layer provides an indication of the fluvial environment that gave rise to specific landforms and divides the landscape into areas that are characterized as abandoned channels, backswamps, and pointbar/valley trains.

Public lands data—The public lands data were obtained from the U.S. Department of the Interior, Fish and Wildlife Service (USFWS). The data contain six public land types, including Public Land Restoration, Delta National Forest, Farmer’s Home Administration, Wetland Reserve Program, National Wildlife Refuge, and State Wildlife Management Area lands.

Roads and transportation data—Data for primary and secondary roads and railroads were obtained from the Mississippi Automated Resources Information System (MARIS). The sources for MARIS transportation data layers are U.S. Department of the Interior, Geological Survey, digital line graph (DLG), and the Mississippi Department of Transportation. The 1:100,000-scale primary roads data include interstates, U.S. highways, and 1- and 2-digit State highways; for example, Mississippi Highway 3 and Highway 25. The secondary roads layer includes 3-digit highways; for example, Mississippi Highway 471 and the Natchez Trace Parkway.

Water bodies data—Permanent water bodies were adapted from MARIS permanent water data. This data layer was

Table 5—Stage measurements for seven stream gauging sites in the Yazoo backwater area[a]

Date of image	Steele Bayou at Grace	Steele Bayou at Onward	Steele Bayou at Upper Intake	Big Sunflower River at Little Callao Landing	Big Sunflower River at Anguilla	Big Sunflower River at Holly Bluff	Little Sunflower River at Upper Intake
03/12/73	93.3	85.5	77.2	97.1	94.1	89.5	82.2
03/31/73	98.2	92.5	89.3	102.8	99.3	96.5	95.4[b]
05/05/73	100.4	100.6	100.2	101.9	100.7	100.4	100.3
01/30/74	98.3	92.9	90.6	101.2	98.1	96.0	93.4
01/13/83	94.6	93.1[b]	91.9	100.8	98.1	95.5	93.1
02/17/84	92.6	85.8	76.1	99.4	94.3	90.5	81.4
03/05/87	91	84.9	79.5	98.7	94.7	90.0	82.4
12/02/87	86.9	73.7	66.2	87.0	83.0	79.1	70.8
03/10/89	89.7	89.7[b]	89.7	90.1	89.0	91.5	90.0
04/01/91	87.5	N/A	83.3	89.4	87.7	86.0	83.8
04/30/91	98.1	93.9	90.4	103.0	98.5	95.4	91.7
06/04/91	86.9	85.2	84.8	95.0	93.8	92.4	89.1
02/01/93	86.3	83.4	83.0	84.7	83.8	83.6	83.2

All measurements are in feet above mean sea level.
N/A = not available.
[a] Stage values are from U.S. Army Corps of Engineers published data except where indicated.
[b] Dave Johnson, U.S. Army Corps of Engineers, written communication, November 2, 1998.

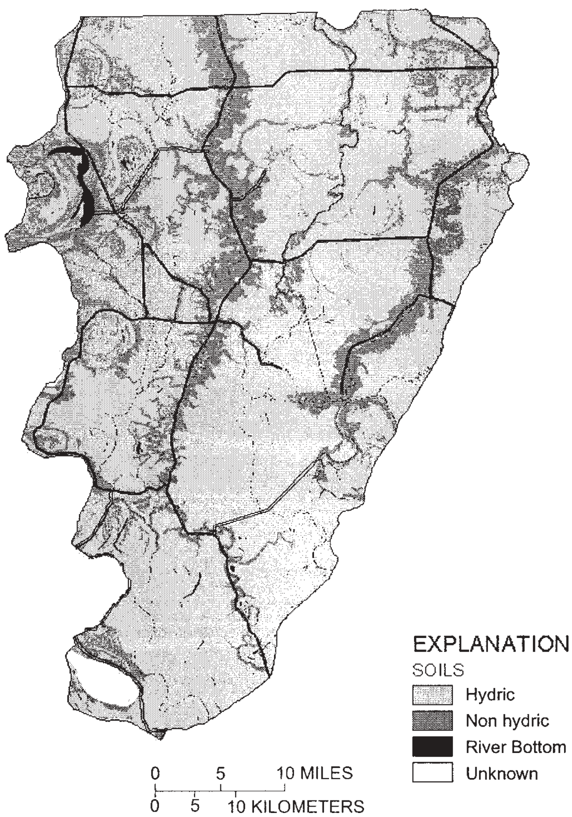


Figure 3—The spatial extent of hydric and nonhydric soil in the study area.

modified by removing areas designated as catfish ponds, for the specific needs of this effort.

Elevation contour data—Hypsographic contours were made available through MARIS as compiled from 1:24,000-scale U.S. Department of the Interior, Geological Survey, base material.

Hydrologic network data—The EPA River Reach File Level 3 (RF3) was selected to represent the hydrologic network for the area. Stream buffers were created from the RF3 streams network coverage by using stream level. Stream level ranges from 0 to 9. A level 1 stream flows to the ocean. A level 9 stream would be the size of a small creek or a ditch. A level of 0 indicates that the actual stream level is unknown in the RF3 dataset.

NRCS "farmed wetlands"—The NRCS created the "farmed wetlands" data set such that the criteria for "farmed wetlands" was for areas (not pothole, playa, or pocosin) that have a 50-percent chance of being flooded or ponded for at least 15 consecutive days during the growing season. Areas classified as NRCS "farmed wetlands," indicate places on the landscape that are inundated for a significant duration and are, therefore, very likely to maintain sustainable wetlands. The NRCS "farmed wetlands" are defined for regulatory purposes and are not indicative of farmed areas that may be considered wetlands under alternate wetland definitions.

Digital elevation data—High-resolution hypsographic data were obtained from a collaborative effort with MARIS. High-resolution, drainage-reinforced digital elevation model (DEM) data were developed by using the high-resolution hypsographic data and the ArcInfo routine TOPOGRID, which uses line information to create customized elevation grids. The hypsography data layers were combined, and a seamless elevation grid was created at a 10-m posting interval (raster cell spacing) for the entire lower Yazoo River Basin. A filled DEM layer and other hydrologic derivatives were produced to allow the analysis of hydrology within the study area. The filling of a DEM involves an automated process wherein localized depressions (which in nature fill and overflow) are digitally filled to provide a continuous hydrologic surface.

The difference between the filled and the unfilled high-resolution DEM was used to create a topographic depressions data set (fig. 4). This layer provides an analytical tool for assessing the size, distribution, and nature of areas that can be considered as topographic depressions and likely sites for water storage and functional wetlands restoration.

Topographic base data—To provide a continuous topographic base for the study area, USGS digital raster graphics (DRG) of the 1:24,000-scale quadrangle maps were collar-clipped (the white map collar or border was clipped out), edge-matched, and placed into seamless image catalogs for the study area. This topographic base layer was used to provide quality assurance for all GIS data layers used in the study.

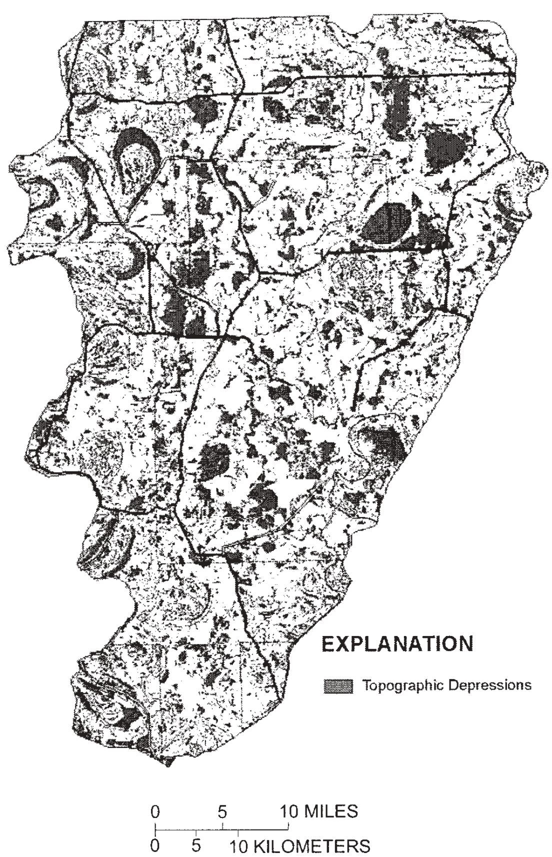


Figure 4—The spatial extent of topographic depressions in the study area.

Model Development

The Eco-Assessor program was written using Arc Macro Language programming and runs using ArcInfo in a Windows NT environment. Clickable menus are provided in order to give the user the ability to turn on and off each data layer as well as change the ranks assigned to each data layer.

RESULTS AND DISCUSSION

Generation of "Functional Restoration" Maximum

Once the Eco-Assessor has analyzed each data layer, the ranks for all data layers are summed. The summation results in a functional restoration (FR) rank for each cell of eligible land. The functional restoration rank is used as the indicator of which areas on the landscape are the most suitable for wetland restoration and would perform wetland functions well.

Functional restoration maximum is the grid generated by the Eco-Assessor that contains the total FR value for each eligible cell in the study area. The FR maximum assumes that every eligible cell within the study unit is selected for reforestation. The total FR value is the sum of the assigned rank for each data layer of a given cell. The resultant FR maximum spatial data layer has cells that have cumulative ranks that range from 15 to 140. The highest ranked areas are those that would be the most suitable for wetland restoration and would be most likely to perform wetland functions. This is depicted in figure 5.

Scenario Generation and Evaluation

Reforesting all eligible areas within the study area is an unrealistic goal; therefore, specific subsets of the study area are recommended for reforestation. These subsets of the study area or scenarios are selected by using the GIS and establishing spatial criteria. The justification for the spatial criteria may be based on a number of reasons. A scenario may be based upon targeting a particular wetland function, a certain geographic area, a certain feature on the landscape, or any other set of criteria that can be spatially determined.

The use of the FR rank becomes particularly important when considering scenarios. The FR rank allows for the comparison of scenarios on an ecological basis. The total FR rank for a given scenario provides an indicator of the ecological benefits to be gained by reforesting the area specified by the scenario. The total FR rank for a scenario is divided by the total number of acres for that scenario. This calculation results in a FR per-acre score for that scenario. The FR per-acre score can then be used to compare the ecological merits of various scenarios (fig. 6).

In the hydrology scenario, the eligible areas inundated in the USACE nominal 2-year flood scene are selected for reforestation (fig. 7).

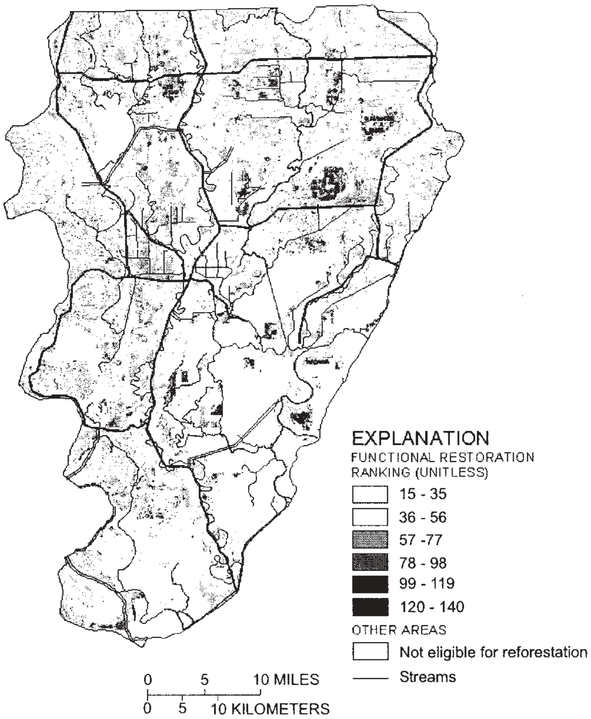


Figure 5—Functional Restoration Maximum—the data layer generated by summing the assigned ranks for all data layers for each 25-m cell in the study area. Darker areas represent areas that have a higher probability of being restored back to a functional wetland.

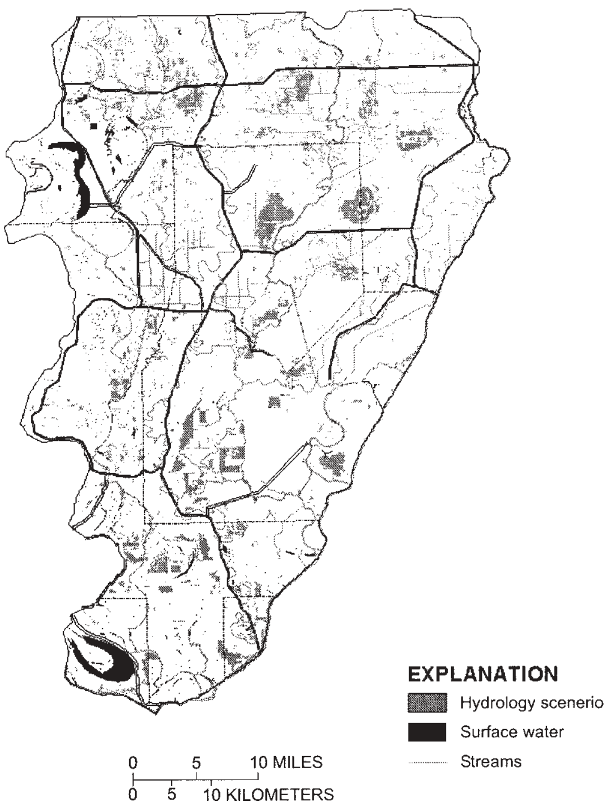


Figure 6—Various restoration scenarios can be compared by comparing their sum of functional restoration values (A) and by dividing the functional restoration value by the area to get a functional restoration value per land area estimate (B).

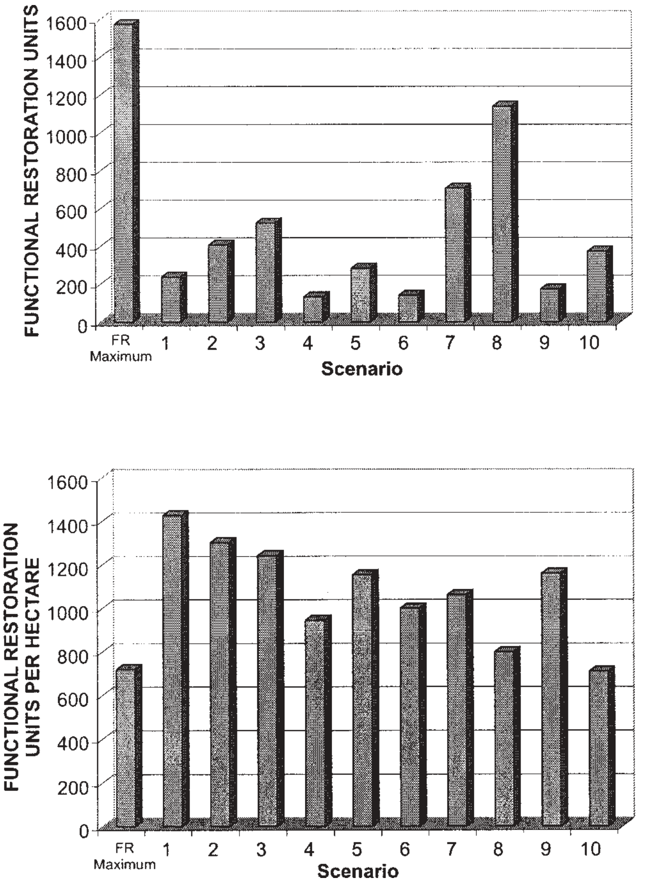


Figure 7—An example of a restoration scenario derived from the Functional Restoration Maximum layer in which all cells within the 2-year flood are selected.

SUMMARY

The Eco-Assessor DDS provides a valuable tool to prioritize the restoration of forested wetlands in the lower Yazoo River basin. The data compiled, and the tools that are included in the DSS, facilitate the rapid generation and consideration of alternate restoration scenarios. The DSS can be used to help develop reforestation plans that place wetland forest communities in locations in the landscape where each wetland has a high probability of developing into a functional wetland system. Reforestation efforts can be targeted to areas that would provide the highest ecological benefit for a given economic investment. The DSS also provides a method for systematically altering the buffer distances and ranks assigned to each data layer through the use of the interactive menus, which make up the Eco-Assessor framework. The ability to change the ecological rules and rank values allows the user to obtain the most appropriate ranking for a given wetland system.

ACKNOWLEDGMENTS

The authors would like to acknowledge the contributions of the following people to this project: Roland Viger, Physical Scientist, U.S. Department of the Interior, Geological Survey, Denver, CO; Al Rea, Hydrologist, U.S. Department of the Interior, Geological Survey, Oklahoma City, OK; Sridhar Katragadda, GIS Specialist, U.S. Department of the Interior, Geological Survey, Pearl, MS; and Leonard Shabman and Laura Zepp, Virginia Polytechnic Institute and State University, Blacksburg, VA. Reviews were provided by: Wade Bryant, Regional Biologist, U.S. Department of the Interior, Geological Survey, Norcross, GA; Craig Harvey, Geographer-GIS Specialist, Carol Moss, Editor, and Larry Slack, Supervisory Hydrologist, U.S. Department of the Interior, Geological Survey, Pearl, MS; Greg Easson, Assistant Professor of Geology and Geological Engineering and Nolan Augenbaugh, Professor of Geology and Geological Engineering, University of Mississippi, Oxford, MS; and Tammy Charron, Editor, Johnson Controls World Services, National Welands Research Center. Lance Cooper and Michael Wade provided graphics assistance. A portion of this project was used in partial fulfillment of requirements for a Master's degree at the University of Mississippi, Oxford, MS.

REFERENCES

Allen, J.A. 1990. Establishment of bottomland oak plantations on the Yazoo National Wildlife Refuge complex. Southern Journal of Applied Forestry. 14(4): 206–210.

Allen, J.A. 1997. Reforestation of bottomland hardwoods and the issue of woody species diversity. Restoration Ecology. 5(2): 125–134.

Allen, J.A.; Kennedy, H.E., Jr. 1989. Bottomland hardwood reforestation in the Lower Mississippi Valley. Slidell, LA. Stoneville, MS: U.S. Department of the Interior, U.S. Fish and Wildlife Service, National Wetlands Research Center; U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 28 p.

Brinson, M.M. 1993. A hydrogeomorphic classification for wetlands WRP–DE–4. Vicksburg, MS: U.S. Army Corp of Engineers, Waterways Experiment Station. 79 p.

Creaseman, L.; Craig, N.J.; Swan M. 1992. The forested wetlands of the Mississippi River: an ecosystem in crisis. Baton Rouge, LA: The Nature Conservancy of Louisiana: 4–5.

Dillaha, T.A.; Reneau, R.B.; Mostaghimi, S.; Lee, D. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. Transactions of the American Society of Agricultural Engineering. 32(2): 513–519.

Kinler, Q. 1994. Habitat assessment model for fresh swamp and bottomland hardwoods within the Louisiana coastal zone. Baton Rouge, LA: Louisiana Department of Natural Resources. 10 p. +3 appendices.

Howard, R.J.; Allen, J.A. 1988. Streamside habitats in southern forested wetlands: their role and implications for management. In: Hook, D.D.; Lea, R., eds. Proceedings of the symposium: the forested wetlands of the Southern United States; 1988 July 12–14; Orlando, FL. Gen. Tech. Rep. SE–50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 97–106.

Hupp, C.R.; Morris, E.E. 1990. A dendrogeomorphic approach to measurement of sedimentation on a forested wetland, Black Swamp, Arkansas. Wetlands. 10(1): 107–124.

Kirchner, W.N.; Kleiss, B.A.; Clairain E.J., Jr. [and others]. 1992. Delineation of wetlands of the Yazoo River Basin in northwestern Mississippi EL–92–2. Vicksburg, MS: U.S. Army Corps of Engineers, Waterways Experiment Station. 25 p. + 3 appendices.

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

Lowrance, R.; Altier, L.S.; Newbold, J.D. [and others]. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environmental Management. 21(5): 687–712.

McGarigal, K. 1995. Fragstats: spatial pattern analysis program for quantifying landscape structure. Gen. Tech. Rep. PNW–351. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Station. 122 p.

Mitsch, W.J.; Gosselink, J.G. 1993. Wetlands. New York: Van Nostrand Reinhold. 722 p.

Ohman, K.; Eriksson, L.O. 1998. The core area concept in forming contiguous areas for long-term forest planning. Canadian Journal of Forest Research. 28(7): 1032–1039.

Renken, R.B.; Wiggers, E.P. 1993. Habitat characteristics related to pileated woodpecker densities in Missouri. The Wilson Bulletin. 105 (1): 77–83.

Saucier, R. 1994. Geomorphology and quaternary geologic history of the Lower Mississippi Valley. Vicksburg, MS: U.S. Army Corp of Engineers, Waterways Experiment Station. 364 p.

Schroeder, R.L. 1982. Habitat suitability index models: pileated woodpecker. FWS/OBS–82/10.39. Fort Collins, CO: U.S. Department of the Interior, Fish and Wildlife Service. 15 p.

U.S. Department of Agriculture, Natural Resource Conservation Service. 1996. National food security act manual. 3rd ed. U.S. Department of Agriculture.

Wakeley, J.S.; Marchi, R.A. 1992. An evaluation of impacts to terrestrial habitats resulting from flood control projects in the Steele Bayou Basin, Mississippi. Vicksburg, MS: U.S. Army Corps of Engineers, Waterways Experiment Station. 26 p.

STATES ASSUMING RESPONSIBILITY OVER WETLANDS: STATE ASSUMPTION AS A REGULATORY OPTION FOR PROTECTION OF WETLANDS

Kristen M. Fletcher[1]

Abstract—While States have initiated their own wetland protection schemes for decades, Congress formally invited States to join the regulatory game under the Clean Water Act (CWA) in 1977. The CWA Amendments provided two ways for States to increase responsibility by assuming some administration of Federal regulatory programs: State programmatic general permits and State assumption. States are also active in conservation programs such as preserving and managing wetlands. State programmatic general permits (SPGP) allow a State to become the sole permit issuer under an existing State-permitting program for projects that have similar characteristics and will have low environmental impacts. SPGP have gained popularity whereas, in contrast, State assumption has been less popular with only two States adopting such a program. In most instances, State assumption grants more permitting authority to States but also places a heavier burden on the State with a stricter application and approval process, a greater funding obligation, and a larger regulatory responsibility.

INTRODUCTION

Last year, the Fish and Wildlife Service reported that the rate of U.S. wetland loss has slowed to a rate 60 percent below that experienced in the 1970s and 1980s. Jamie Clark, Director of the Fish and Wildlife Service, explained that the study shows that our Nation's efforts to restore and protect wetlands are making a difference. At a time when many Federal regulatory programs are criticized as too expansive, these protective efforts are increasingly occurring at the State level. Many States maintain wetland protection schemes, and their legislatures continue to consider methods of increasing control over State wetlands. Combine this factor with express invitations from the U.S. Congress and the Executive Branch, and the signs indicate that the momentum behind the State initiatives will likely continue.

While States have initiated their own wetland protection schemes for decades, Congress formally invited States to join the regulatory game under the Clean Water Act (CWA) in 1977. The CWA Amendments provided two ways for States to increase responsibility by assuming some administration of Federal regulatory programs: State programmatic general permits and State assumption. State programmatic general permits (SPGP) allow a State to become the sole permit issuer under an existing State-permitting program for projects that have similar characteristics and will have low environmental impacts. But, Federal control is maintained for permitting other projects. SPGP have gained popularity, with 27 States holding permits of this type. In contrast, State assumption has been less popular, with only two States adopting such a program. In most instances, State assumption grants more permitting authority to States: States may issue individual permits for projects that do not have to meet the general permit requirements of similar nature and low environmental impacts. But, State assumption also places a heavier burden on the State with a stricter application and approval process, a greater funding obligation, and a larger regulatory responsibility.

This paper reviews the two regulatory options available to States under the CWA, focusing on the advantages and disadvantages inherent in each. It explores the reasons why States may choose to adopt Federal regulatory responsibilities and presents a guide for States considering one or both of these regulatory options.

EVOLUTION OF WETLANDS REGULATION

In the 1700s, 221 million ac of wetlands existed in the United States. A 1995 inventory showed < 101 million ac remaining. Historically, a wetland was considered a nuisance, believed to inhibit navigation and provide habitat for little more than mosquitoes. Thus, the Federal government encouraged draining and filling of wetlands throughout the 1800s. Under the Swamp Lands Acts, the Federal government granted 15 Western States almost 65 million ac for "swamp reclamation," making drainage and filling wetlands a national policy.

In the late 1960s, the Federal government took greater notice of the benefits of wetlands. It began regulating the filling and dredging of wetlands under the authority of the Rivers and Harbors Act of 1899, which prohibited excavation from or fill to any navigable water of the United States without a recommendation by the Chief of Engineers and authorization from the Secretary of the Army. The Corps also began to consider the ecological benefits of wetlands and to make more protective decisions regarding permits.

In 1972, Congress enacted the Federal Water Pollution Control Act (Clean Water Act, CWA) to eliminate the discharge of pollutants in the waters of the United States, including wetlands, by 1985. Pursuant to this goal, the CWA prohibits all persons from discharging pollutants into waters of the United States unless they have obtained and are operating within the strictures of certain permits. If the pollutants involve dredged or fill material, the permits are issued by the Corps with the EPA maintaining a supervisory role.

[1] Director, Mississippi-Alabama Sea Grant Legal Program, 518 Law Center, University, MS 38677.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

The CWA Section 404 is the primary Federal regulatory program providing protection for the Nation's remaining wetlands. The EPA and the Corps jointly administer the program. The Corps' responsibilities include day-to-day program administration, individual permit decisions, jurisdictional determinations, development of policy and guidance, and enforcement. The EPA's responsibilities include development and interpretation of guidelines for the environmental criteria used in evaluating permit applications, identifying certain exempt activities, reviewing and commenting on individual permit applications, exercising the authority to veto Corps' permit decisions, and overseeing administrative responsibilities of the State assumption program.

The section 404 process includes a public notice and comment period. After receiving public comments, the Corps evaluates the application to decide if it contributes to conservation, economics, aesthetics, fish and wildlife values, flood protection, general public welfare, historic values, recreation, land use, water supply, water quality, and navigation. A public hearing may be held, or the Corps may grant or deny the application outright. An application for an individual permit must meet the requirements of a comprehensive inquiry through the Corps' public interest review and its analysis of compliance with the EPA guidelines. The EPA guidelines seem to require a rigorous examination of the availability of practicable alternatives, and prohibit the authorization of any project that would result in significant adverse impacts. In reality, the Corps denies < 10 percent of individual permit applications.

The CWA also authorizes general permits on a State, regional, or national basis. Rather than applying for an individual permit, a person may qualify to discharge dredged or fill material into waters of the United States under one of these permits issued for projects with small impacts. General permits decrease the administrative burden for the Federal and State regulatory agencies but have come under increasing criticism because of the potential to greatly contribute to overall loss of wetlands, little by little.

Some argue that the CWA was enacted precisely because the individual States lacked the political will to clean up their waterways and protect key resources. But, 5 years after its enactment, Congress amended the CWA authorizing substantial State regulatory participation. The amendments authorized States to take over the section 404 permitting program from the Corps and also provided for general permits to relieve pressure created by expanded Federal jurisdiction and, in part, as an acknowledgment of a practice that the Corps already was performing.

One of the reasons Congress amended the CWA was to address concerns that the section 404 program was too overwhelming for the Army Corps of Engineers to manage and that funds were insufficient. Reduction of Corps workload was a primary reason for providing delegation to the States. Even though parts of the legislative history confirm Congress' intent to delegate a greater responsibility to the States, it also indicates concern for the performance of State programs. Advocates argue that States occupy the best position to take the lead in wetlands protection because States tend to be more responsive than Federal agencies to local needs but are still removed from the influences of local politics, providing better protection than local governments.

In addition to the congressional invitation of 1977, the Executive Branch has encouraged reducing Federal involvement and replacing Federal wetlands regulators with States. Specifically, the Bush Administration preferred a minimum level of Federal involvement, citing Federal regulatory programs as burdensome. The Clinton Administration called for greater State action in its Wetlands Policy of 1993. Two of the five principles for the Federal Wetlands Policy directly related to States increasing their responsibilities: avoidance of duplication between regulatory agencies and expansion of partnerships with State and local governments.

States were given alternative opportunities to respond to pressures on wetlands. Since the passage of the CWA, States were encouraged to establish their own conservation and permitting programs and to work as partners with the Corps in order to manage wetland areas. Some State programs predated the Federal protections of wetlands, and many took action in favor of their coastal wetlands prior to extending protection to upland wetlands. Other States joined programs sponsored by the Federal government that combine Federal, State, local, and private efforts at restoring and preserving wetlands.

States may also protect against the filling of wetlands through the CWA Section 401 water-quality certification and under the consistency determinations of the Coastal Zone Management Act. Under the CWA Section 401, a State may veto or condition a Federal licensed or permitted activity that may degrade water quality or aquatic habitat, including wetlands. It requires that an applicant for a Federal permit for activities that may result in a discharge into navigable waters must receive a certification from the State to insure compliance with State water-quality requirements. Section 401 gives State water-quality control over a wide range of activities for which they otherwise might lack such authority, including wetlands preservation, protection of wildlife habitat, and protection of aesthetic and recreational values of waterways. It also allows States to limit impacts on wetlands without running its own regulatory program or operating under an SPGP or assuming section 404 authority from the Corps.

A State may also use the consistency requirement under the Coastal Zone Management Act to limit Federal permitted activities, which affect wetlands in a coastal zone. If a State has an approved coastal zone management program, then an applicant for a Federal permit whose activity may affect any land or water use or natural resource in the coastal zone, must obtain certification from the relevant State coastal resources agency that the permitted activity complies with the State program and will be conducted in a manner consistent with the program. A State may specifically designate wetlands as regulated areas if they fall within the State's coastal zone. For fill activities in these wetlands, the State may deny certification if it finds the activities inconsistent with its program. Generally, a project cannot continue without such certification. But State wetland protection schemes have powerful 404 alternatives. As regulatory tools, SPGP and State assumption must be analyzed and adapted to further wetlands protection, in addition to trying to simplify the permitting process.

STATE PROGRAMMATIC GENERAL PERMITS

The State programmatic general permit is one type of general permit issued by the Corps. Today, the Corps uses general permits to authorize 80 percent of the regulated activities. The Secretary of the Army issues general permits on a programmatic, regional, or nationwide basis. To qualify for a general permit, the project must meet the following requirements: (1) the activities authorized under the general permit must be “similar in nature;” (2) the activities may cause only minimal adverse environmental effects when performed separately; (3) the activities may have only minimal cumulative adverse effect on the environment; (4) the permit must be based on EPA guidelines; (5) the permit must be limited to a 5-year life span; (6) the Secretary can revoke or modify the permit if the authorized activities have an adverse impact on the environment, or such activities are more appropriately authorized by individual permits.

The general programmatic permit furthers the idea that State and Federal regulatory programs should complement rather than duplicate one another. Corps regulations define programmatic permits as “a type of general permit founded on an existing State, local, or other Federal agency program and designed to avoid duplication with that program.” Under the SPGP, the Corps in effect delegates to a State the primary responsibility under section 404 for permit review of the activities meeting the requirements of the SPGP. As a general permit, the SPGP must comply with the congressionally mandated requirements set forth in section 404(e).

Before authorizing an SPGP, the Corps must analyze the State program upon which it is based. Often, because of the limited scope of some State programs, an SPGP will not necessarily cover the entire State in question. Instead, the Corps refers to any general permit program based on a State program to assure the protection of wetlands as an SPGP.

The programs tend to follow two models in practice: those using fixed criteria and those using a process of consultation and review. The SPGP using fixed criteria called the “New England model,” requires the State to place potential section 404 permit applications in one of three categories: the nonreporting, screening, or individual permit categories. If the State places a permit application within the nonreporting category, the applicant does not have to inform the Corps of the activities. Instead, the applicant is only responsible for meeting State requirements: a State permit, State water-quality certification, and a consistency concurrence, if the project is in the coastal zone. An activity with higher impacts may fall in the screening category, which requires an interagency screening, or the individual permit category, which requires an applicant to go through the Corps’ individual permit application process.

The second type of SPGP uses a consultation and review structure. When a State agency receives an application for a State permit, it conducts a site visit of the area and produces a field report or evaluation. The State then produces public notices of permit applications, putting the Corps and other Federal resource agencies on notice. One of these agencies or the State agency may request that the Corps require the applicant to seek an individual permit. If not, the State agency can issue the permit. This consultation and review process is often criticized for increasing the workload and delays but potentially provides a better review of the project site and more accurate prediction of impacts.

States with SPGPs cite greater control over wetlands as a reason to take on this authority. Through an SPGP, the State can control the permitting of those actions with minimal impacts, does not have to rely on the Corps, and avoids duplication for these permit applications. Finally, the SPGP gives States an alternative avenue to control the fate of their wetlands other than by assuming the 404 permitting process.

The SPGP is a win-win for the Corps as well. By giving the State the authority to review and issue or deny these permits, the Corps can reduce duplication between State and Federal regulatory programs and reduce the Corps’ regulatory workload without compromising, at least from the Corps’ perspective, the overall effectiveness of section 404 and section 10 permit review in protecting wetlands. In addition, most SPGPs specifically exclude activities affecting sensitive areas such as endangered species habitat or historic properties and provide for kick-out provisions if necessary.

STATE ASSUMPTION

The second option for States is State assumption of the Corps’ permitting authority. Many States perceive that SPGPs provide adequate control of State wetlands without assuming the responsibility offered under State assumption whereas others view SPGPs as merely a stepping stone to assumption, which can offer a State more regulatory authority than a general permit.

Unlike SPGPs, the CWA and accompanying regulations specify the requirements for assuming section 404 authority. The EPA must approve a State’s application to assume the 404 permitting program. The statute requires the Governor of the applicant State to submit a description of the program to the EPA, along with a statement from the State attorney general that the laws of the State “provide adequate authority to carry out the described program.” A State must submit to the EPA Regional Administrator the following six items: (1) a letter from the Governor of the State requesting approval of the State program; (2) a complete program description; (3) an attorney general’s statement confirming that the laws of the State provide adequate authority to carry out the described program; (4) a memorandum of agreement with the EPA Regional Administrator; (5) a memorandum of agreement with the Secretary of the Army; and (6) copies of applicable State statutes and regulations, including those governing applicable State administrative procedures.

The attorney general’s statement must also include certification that each agency responsible for administering the State program has full authority to administer the program within its category of jurisdiction. In addition, the State as a whole must have full authority to administer a complete State program. Finally, the statement should include a legal analysis of the likelihood of a constitutional taking as a result of the successful implementation of the State's program.

In order to assume, the State will enter memorandums of agreement (MOA) with both the EPA and the Corps. The MOA with the EPA must set out State and Federal responsibilities for program administration and enforcement including provisions specifying classes and categories of permit applications for which EPA will waive Federal review authority and provisions addressing EPA and State roles and coordination with respect to compliance monitoring and enforcement activities. The MOA with the Secretary of the Army must include a description of the waters of the United States within the State over which the secretary retains jurisdiction and an identification of all general permits issued by the secretary, the terms and conditions of which the State intends to administer and enforce upon receiving approval of its program, and a plan for transferring responsibility for these general permits to the State.

The program description must include various essential elements in order to be approved. First, the description must explain the State's permitting, administrative, judicial review, and other applicable procedures. In addition, it must include a description of the funding and manpower available for program administration, a description of how the State will coordinate its enforcement strategy with the Corps and EPA for nonassumable waters or projects, a comparison of State and Federal definitions of wetlands, and the extent of the State's jurisdiction, scope of activities regulated, anticipated coordination, and the scope of permit exemptions, if any. The EPA distributes the State's program submission to the Corps, Fish and Wildlife Service, and the National Marine Fisheries Service. EPA has up to 120 days to approve or disapprove the State's program. Once the EPA approves the State application, the Corps transfers to the State those permit applications for projects in the State's jurisdiction.

The EPA retains oversight authority and receives copies of all permit applications. The State must notify the EPA of any action that it takes with respect to such applications. The EPA Administrator provides copies of the application to the Corps, the Department of Interior, and the Fish and Wildlife Service and must notify the State within 30 days if the administrator intends to comment on the State's handling of the application. The State must then await comment before it may issue the permit. If the EPA objects to the application, the State may not issue the proposed permit but may request a hearing before the EPA or alter the permit to accommodate the EPA objections. If the State does not request a hearing, the EPA transfers authority to issue the permit to the Corps. Once in the Corps' hands, jurisdiction remains there.

Finally, the statute requires that EPA review any revisions to the State wetlands program, determine whether such revisions are substantial or not substantial, and approve or disapprove the revisions. The EPA also maintains the authority to withdraw approval of the program. If the administration of the State program does not meet EPA guidelines, the EPA may take corrective action and may, within a reasonable time, withdraw approval of the program and redirect authority to the Corps.

MICHIGAN AND NEW JERSEY ASSUMPTION PROGRAMS

Two States, Michigan in 1984 and New Jersey in 1994, have assumed permitting authority with mixed results. An analysis of these two programs provides a look at permitting under State assumption.

Michigan began wetland permitting even prior to congressional authorization for State assumption of section 404 authority. In 1955, the Michigan State legislature passed the Great Lakes Submerged Lands Act, authorizing the Michigan Department of Natural Resources (MDNR) to regulate dredge, fill, and construction activities in the State's coastal zone. In 1972, the Michigan legislature acted again by passing the Inland Lakes and Streams Act authorizing the State to regulate activities occurring up to the ordinary high-water mark on Michigan's inland wetlands. Four years later, Michigan developed a MOA with the Corps for a Joint Public Notice System to expedite the issuance of wetland permits. When congressional authorization for assumption followed the next year, Michigan was well placed to assume the wetlands permitting program. The State then passed the Goemaere-Anderson Act of 1980, forming the framework for assumption and expanding the State's wetland permitting requirements to include those wetlands, which were not subject to section 404 jurisdiction.

The MOA between the EPA and Michigan named the MDNR as administrator of the Michigan wetlands program. The program's procedures are similar to those under the CWA. The Michigan DNR has 14 districts in 3 regions to handle State permitting, mitigation matters, wetland delineation, and enforcement. Upon submission of a completed application, the MDNR has 90 days to issue a determination on the proposed project; if the MDNR fails to issue a determination within that time frame, the proposed project is considered approved.

Prior to assumption, New Jersey's wetlands scheme was comprised of four acts. First, the Hackensack Meadowlands Development Commission of 1968 set up a permitting scheme for activities within district boundaries. The Wetlands Protection Act of 1970 regulated activities within the coastal and estuarine wetlands and required applicants to obtain permits from the New Jersey Department of Environmental Protection (NJDEP). In 1979, the New Jersey legislature enacted the Pinelands Protection Act to regulate wetlands and other areas within the Pinelands National Reserve. Finally, in 1987, the legislature passed the Freshwater Wetlands Protection Act developed specifically with assumption in mind. The act created the State's wetland regulatory program and allowed the State to create buffer areas adjacent to designated wetlands that are subject to active regulation.

New Jersey assumed 404 authority in the spring of 1994. The NJDEP was named as the State authority over the program. The MOA with the EPA requires the State to submit monthly and yearly reports to the EPA for review. The NJDEP is not subject to strict 90-day time constraints like the Michigan DNR. The program requires the NJDEP to act “in a timely manner”; in practice, permit processing often takes up to 4 months.

FEDERAL PERMITTING VERSUS ASSUMPTION

Improving the efficiency of permitting is a high priority for those States considering assumption. The Corps is often criticized for slow responses on permits. States can impose a strict time limit on their permitting agencies, as Michigan has done by requiring turn around by the MDNR in 90 days. States also can provide more manpower than the Corps, evidenced by Michigan’s creation of 13 field offices throughout the State, as compared with 4 Corps offices. With more field offices, decisionmakers can be more readily available to applicants and can be closer to the wetlands actually under their jurisdiction.

An element of improving efficiency of the 404 program is to reduce the ever-present regulatory duplication. Under a State-assumed program, paperwork for the applicant is reduced, and the State agency becomes directly responsible for the application. By reducing duplication, the State can also increase predictability of the application process. A State can also consolidate several different wetland statutes to reduce the burden on the regulated public and streamline the process.

Finally, advocates argue that wetlands will receive better protection under an assumed program for three reasons. First, the State is in a better position to address regional and local concerns about the conservation and use of wetlands resources. Assumption supporters claim that State agency regulators are more familiar with the treatment and use of the regulated lands. Also, because the State assumes permitting authority over a smaller square acreage of wetlands than the Corps had been responsible for, a State can potentially provide closer examination of cumulative impacts. Advocates also claim that a State regulator will be more aggressive than its Corps predecessor, and a State can avoid the inconsistency problems that often plague Corps regulators.

Once State policymakers determine that State assumption and SPGPs are a better alternative for wetlands protection in their State, they must determine which alternative to employ in their State. An analysis of the advantages and disadvantages to each alternative is essential to this determination.

Control Over Wetlands Decisions

States that have taken over some aspect of the Federal regulatory program, and those considering such action indicate that increased control over permitting decisions regarding State wetlands is a driving force in seeking more regulatory authority. But “control” over permitting decisions hinges on several factors: jurisdiction, Federal oversight, and flexibility of the program.

Jurisdiction

Jurisdiction of an SPGP varies according to the State program upon which it is based. An SPGP generally covers wetlands, waters, and activities within the Corps’ Section 404 and Section 10 jurisdiction that are regulated by a State wetlands program. Therefore, a State can design its SPGP to best match its regulatory program but can also amend this underlying program to alter its jurisdiction. SPGPs do not include those activities and areas that are considered to be of national or international concern and many exclude specific activities that have a potentially higher impact, such as new or expanded marinas, projects requiring an environmental impact statement, and wetlands fills over specific acreage. A comparison of the underlying State programs reveals the degree to which SPGPs can differ.

For instance, the Maryland SPGP covers only section 404 activities affecting < 5 ac of nontidal wetlands based upon the jurisdiction of the Maryland Nontidal Wetland Protection Act. The North Carolina SPGP applies to all section 404 and section 10 activities that receive prior approval from the State based on the State’s Coastal Area Management Act permit, a State dredge and fill permit, or a section 401 State water-quality certification when there is a discharge into U.S. waters. But the North Carolina SPGP only applies to its 20 coastal counties. Finally, the Massachusetts PGP applies to section 404 and section 10 activities that receive prior State approval under the Wetland Protection Act Final Order of Conditions, a Public Waterfront Act waterways license or permit, a section 401 State water-quality certification, or a State coastal zone consistency determination. Also, jurisdiction under the PGP is expressly limited to wetland fills less than or equal to 1 ac in size. Finally, these programs expressly exclude activities affecting navigation, national wildlife refuges, forests, parks, components of the National Wild and Scenic River System, and threatened or endangered species and their critical habitats.

The MOA between Michigan and the EPA divided jurisdiction as follows: Michigan acquired responsibility for all activities, which require dredging, placement of fill, or construction in inland waterways; the Corps (Detroit District) maintained jurisdiction over activities, which require dredging, placement of fill, or construction in the Great Lakes coastal areas, connecting waters, navigable waterways and those wetlands adjacent to navigable waterways, and under other specific circumstances. Similarly, New Jersey gained permitting authority over freshwater wetlands except that a 1,000-ft boundary from the mean high water line of navigable waterways was established as the jurisdictional boundary between “adjacent” and “nonadjacent” wetlands.

As the above examples show, the two States that assume section 404 permitting authority have jurisdiction over a greater acreage of State wetlands. This does not necessarily benefit the States, however. Jurisdiction under State assumption can be changed if a State fails to meet EPA guidelines. Thus, even though a State begins its 404 authority responsible for a large number of wetlands, the Corps can reassume permitting authority over certain areas.

Whereas SPGPs can also be subject to change, especially because they must be reapproved every 5 years, they offer more flexibility. A State can easily create a second SPGP to include wetlands in a different part of the State if it wishes authority over a greater number of acres. Even a State with an assumption program can create an SPGP to streamline routine permits with minimal impacts. New Jersey attempted to create such a general permit for the formation of cranberry bogs in wetlands. Ultimately, the general permit was rejected for lack of safeguards to pinelands in New Jersey, but the option remains even under a State-assumed permitting program.

Federal Oversight

The Corps views both the SPGP and State assumption as successful programs because they lower the workload of the Corps. Developers approve of such programs because they believe that passing authority to States may be the only way to reel in the numerous Federal regulatory arms. But, SPGPs and State assumption do not automatically shift all decisionmaking ability to the State level. Some of the control a State might gain from creating an SPGP or assuming 404 permitting authority is tempered by the remaining Federal oversight with these programs.

Under an SPGP, the Corp generally turns over to the State the primary responsibility for individual permit review of the activities included in the SPGP. This allows a State to streamline permitting applications and approval for more routine wetland disturbances but the EPA 404(b)(1) guidelines provide that an SPGP "may be revoked or modified by the Secretary of the Army if, after opportunity for public hearing, the Secretary determines that the activities authorized by such general permit have an adverse impact on the environment or such activities are more appropriately authorized by individual permits." States also object to the Corps' authority to override State decisions on a case-by-case basis. The Corps retains authority to review PGP applications individually and determine on a case-by-case basis whether or not the concerns for the environment require that the Corps override the permit and require an individual application and review. This discretionary authority is often incorporated as a condition to the programmatic permit.

Similar Federal oversight exists under State assumption. After assumption, the EPA monitors the effectiveness of the State program on individual and overall levels. The State must submit to the EPA a copy of each individual permit application and proposed general permit. The EPA may transfer permits to the Corps when deemed necessary. It also requires annual reports to evaluate the State's administration of the program. State assumption documents can also specify certain oversight authority with other Federal agencies such as the Corps or Fish and Wildlife Service. For instance, the MOA between New Jersey and the Fish and Wildlife Service specified that activities suspected of being in close proximity to sensitive areas will be cleared by the Fish and Wildlife Service.

Even with these disadvantages of Federal oversight, the State does gain more control. A State better controls the timing of permitting decisions, the execution of onsite evaluations, the drafting of permit conditions, and establishes rapport with the public, which helps to maintain public support for State wetland programs. Finally, the State may be able to benefit from the mandatory Federal oversight. When a State is faced with a tough political decision, it can remove itself from the equation and shift the decision to the Corps, leaving the Federal agency to take the resulting "heat" from an unpopular permitting decision.

Flexibility

A final element of a State's control over its regulatory decisions under an SPGP or assumed program is flexibility. Ideally, a State could experiment with parts of the program until it best fits the State's regulatory program already in existence and the State's wetland needs. Unfortunately, a number of States cite lack of flexibility as a reason not to partake in the SPGP or assumption alternatives.

Application procedures for the SPGP are less rigid than assumption, and SPGPs provide a greater potential for streamlined permit review. An SPGP can cover a limited area in the State, such as the North Carolina SPGP that covers 20 coastal counties, or can be statewide for particular activities. The SPGP process also allows delegation of a partial program to a State. The State may assume management in the form of a pilot program on a small basis before implementing the program across the State. This allows a State to institute higher levels of regulatory oversight for more fragile areas. For instance, Florida recently developed an SPGP for a single watershed.

Funding

Congress relied on a State survey when providing for State assumption in the CWA Amendments of 1977. The survey revealed that States were interested in assumption but only if adequate funding was available. Interestingly, Congress has never provided such funding and does not appear to consider it necessary. Lack of funds continues to be a top deterrent for States considering assumption of the 404 program. The State agency must incur a large share of the workload from the Corps including project review, impact assessment, program enforcement and administration, and the assumption of new responsibilities for compliance with certain Federal statutes.

With an SPGP, States do not face the heavy funding burdens associated with assumption of the section 404 program. Because the SPGP is based on a State regulatory program already in place, the State has already expended most of the necessary funding. Its earlier investment in the regulatory program will pay off when operating under an SPGP. If additional funds are necessary and a State cannot immediately fund the program, it can phase into operation under an SPGP, lessening problems with start-up funding. Or, a State can use an SPGP to strengthen an existing regulatory program, using Federal funds to complement a program limited by the shortage of State funds.

Permit Review

Both the SPGP and assumption alternatives offer a State a way to change permit reviews and procedures and offer more predictability in the permitting process. Because the Corps is criticized for delayed permit review, States often seek to simplify permitting procedures and streamline application review and negotiations.

The SPGP expedites review of permit applications. An SPGP can require accelerated action by those Federal agencies that maintain veto or supervisory power. These Federal agencies must meet State timelines that are often shorter than the Federal counterparts. For example, the North Carolina SPGP provides the Corps 45 days to distribute the SPGP application materials to Federal agencies to comment and develop a coordinated Federal agency position. The SPGP allows for extension of this peer review period, but it is discouraged. Similarly, the Maryland SPGP offers a 45-day limit to the Corps but provides that the Corps or the Maryland Department of Natural Resources may extend the 45-day limit to complete the review. Finally, in New Hampshire and Massachusetts, the Corps coordinates with the Federal resource agencies through meetings about every 3 weeks to review projects in the SPGP screening category.

Operating under an SPGP also simplifies the permitting process for applicants and avoids potential inconsistencies in Federal and State approaches. Properly implemented, a State can provide protection to wetlands while facilitating minor development proposals. This bolsters the theory that wetlands regulation is developing into a land use practice and that the historical land use actor, the State, can best define such policies.

Under State assumption, the State can design its permitting procedures as long as it continues to meet EPA requirements. States can limit delays in reviewing applications and provide an avenue for better communication between State regulators and permittees and greater predictability in the application process.

Protection

Protection of wetlands remains a thorny issue in both State and Federal regulatory programs. There is a constant struggle to determine if the ability to provide adequate protection for wetlands is at the Federal or the State level. If a State decides to participate in regulation under an assumed program or an SPGP, which alternative best protects wetlands?

SPGP proponents cite increased protection of wetlands as a benefit. Primary State control allows for increased recognition of local differences in wetland conditions and community needs. And it provides review to those smaller *de minimis* impacts that are usually never reviewed. A movement is gaining momentum to formulate an SPGP that replaces Corps nationwide permits. This may be a result of continuing criticism over the lack of examination of NWP activities. Prior to the rise of SPGPs, in order to review NWPs, States had to rely upon their authority to deny section 401 water-quality certification to the Corps for an NWP.

This approach supports delegating the NWP review to a State under an SPGP because the Corps does not exercise review over nationwide permit activities. This allows the Corps to rely on a strong State program to take up the slack and allows a State to protect against the dangers of unregulated NWP activities. Under these circumstances, an SPGP can provide a more streamlined review of minimal impact activities that is better tailored to a State's particular circumstances.

The Corps' New England Division is attempting this approach. It has revoked several nationwide permits and delegated those responsibilities to various States in its region. The New Hampshire and Massachusetts SPGPs have assumed many of these responsibilities but still ensure at least some form of Federal review for impacts over 3,000 or 5,000 ft^2, respectively. These may represent an acceptable compromise between State protection and more thorough Federal permit review.

Because the SPGP generally applies at the State government level, the effectiveness of the program is limited to that of the State agencies involved. Thus, SPGPs are criticized for restricting local and regional involvement. In addition, SPGPs may actually lower the amount of oversight of wetlands by authorizing a State agency to substitute its discretion in categorizing activities as those with minimal impacts. Even small activities that have minimal impacts cause loss of wetlands. While Corps oversight is present for certain permit applications, the goal of reduction of duplication and Corps workload may override the need for adequate supervision. In testimony last year, a Corps representative stated that "[p]rogrammatic general permits allow the State or local agency to take the lead in working with the applicant and reduce duplication among programs. If enough of these programmatic general permits are developed, the long-range benefit will be a significant workload reduction for the Corps regulatory program." The primary goal, wetlands protection, is pushed aside in favor of lessening duplication and lowering the workload for the Corps.

State assumption has the potential to improve protection for wetlands. States may offer more in-depth knowledge of local and regional wetlands values and functions than Federal agencies. In addition, a State may provide better enforcement and may even create a program more stringent than the Federal one. At a minimum, the State permits must comply with the requirements of the CWA, its regulations and the EPA 404(b)(1) guidelines, but this does not limit a State as it may adopt more stringent requirements. For example, a State may choose to require individual permits to regulate isolated wetlands less than an acre in size, often authorized under NWP 26 that would otherwise receive no review. In 1984, the Detroit District of the Corps transferred a majority of nationwide permits to the MDNR. The result is that the Detroit district no longer issues NWPs in areas that are under State jurisdiction in Michigan. There is no NWP equivalent in Michigan, a significant change as a result of the assumed program.

These benefits, however, may be outweighed by the substantial political pressure a State agency can experience when making permitting decisions previously made by the Corps. Michigan and New Jersey have both experienced some political challenges to their regulatory authority.

Michigan's program has been cited as a success as the first assumed program in the Nation. It has lowered the workloads of both the EPA and the Corps, and both agencies believe that the MDNR is doing an adequate job. The Michigan program claims greater enforcement of permitting violations than its Corps predecessor. However, wetland protection advocates support Corps regulation and cite great wetland losses due to the lack of political willpower and enforcement mechanisms at the State level. One commentator remarked that “in a large part because of its permanence, the Michigan assumption is widely regarded by conservationists to have been ultimately a disaster.”

Two examples can place this remark into context. First, the EPA overruled its own regional office objection to a permit to build a 267-acre gold course and housing development. In 1991, the Michigan DNR was set to issue a permit, but the EPA regional office objected. After meetings between the State, developer, and EPA regarding modifications, EPA Administrator William Reilly withdrew the regional office’s authority. An EPA representative explained that the action was “to send a signal to the States that, if they follow the proper procedures and are qualified and capable, EPA is not going to interfere.” It did leave tension between the MDNR, which was poised to issue a permit, and the Regional EPA.

Second, the changes in the program in recent years have caused strife within the program. Some cite Governor John Engler’s guidance as weakening the program. In 1991, Governor Engler issued Executive Order 1991–31, which abolished some State agencies and reorganized others, such as the MDNR. After litigation, the Michigan Supreme Court upheld the restructuring in House Speaker v. Governor, affirming the executive’s right to make such changes. In 1994, the principal changes set forth in the 1991 Order were codified through statute, but, in 1995, Governor Engler issued Executive Order 1995–18, further reorganizing Michigan’s environmental agencies. The National Wildlife Federation and State conservation organizations challenged the 1991 and 1994 orders for restructuring as violations of the State’s assumption requirements, suing the EPA for failing to withdraw approval of Michigan’s program due to substantial revision. The District Court for the Western District of Michigan, however, dismissed the claim, finding no subject matter jurisdiction.

The Michigan program remains in effect, but also in flux as a result of the recent restructuring challenges. Its experience shows that even at the State level, a wetlands regulatory program may be subject to political whims. Also, it highlights the hoops that a State must jump through to attain and maintain an assumption program.

The New Jersey program has received favorable reviews from the Federal agencies and many State conservationists for its protection of wetlands. The program, however, faces similar executive difficulties as the Michigan program. For instance, conservationists in New Jersey have been vigilant in opposing Governor Whitman’s attempts at creating a general permit for cranberry bog conversions. Like the struggles with reorganization of the Michigan program, New Jersey’s program is also still subject to the impulses of State political pressures and executive branches.

The experiences of these two States can assist other States in determining if State assumption can help them protect their wetlands.

LITTLE CYPRESS CREEK STUDY: A WATERSHED RESTORATION CASE STUDY

Lisa Gandy, Randy Roberson, and Tom Foti[1]

Abstract—The Little Cypress Creek watershed, which is the home of the Louisiana Purchase Historic State Park and Natural Area, is one of the only remaining examples of a headwater swamp ecosystem left in Arkansas. An increase in water elevations and a change in species composition were noticed in the park in late 1970. A study of the upper watershed of Little Cypress Creek was conducted to identify potential factors causing these changes. Water elevations, vegetative composition, and physical modifications were recorded and compared to historical information. A conceptual model of the watershed was developed summarizing current understanding and hypotheses. Although natural variability in rainfall explains some of the changes observed, roads, beaver dams, clearing, levee construction, and irrigation tail water inputs contribute to hydrologic changes in the study area. The vegetation changes observed in the study area are likely due to multiple stressors rather than any single factor.

INTRODUCTION

The Louisiana Purchase Historic State Park and Natural Area (Park) is located in eastern Arkansas at the intersection of Monroe, Lee, and Phillips Counties (fig. 1). The 37-ac Park contains a granite monument commemorating the 1803 purchase of the Louisiana Territory from France and marks the point from which all land surveys of the Louisiana Territory were referenced. The Park is not only historically significant, but it also is ecologically significant in that it preserves one of the remaining examples of a headwater swamp ecosystem left in the State.

Arkansas Department of Parks and Tourism (Parks & Tourism) and Arkansas Natural Heritage Commission (Natural Heritage) personnel noticed that water levels were increasing in the Park in the late 1970s and early 1980s. Between an 18-to 24-in. increase in surface water elevations was reported. In the early 1980s, Parks & Tourism and Natural Heritage personnel observed that a large number of overcup oaks (*Quercus lyrata* Walt.) on the slopes adjacent to the deep swamp were dying; and species such as button bush (*Cephalanthus occidentalis* L.) and smart weed (*Polygonum hydropiperoides* Michx.) were becoming more dominant.

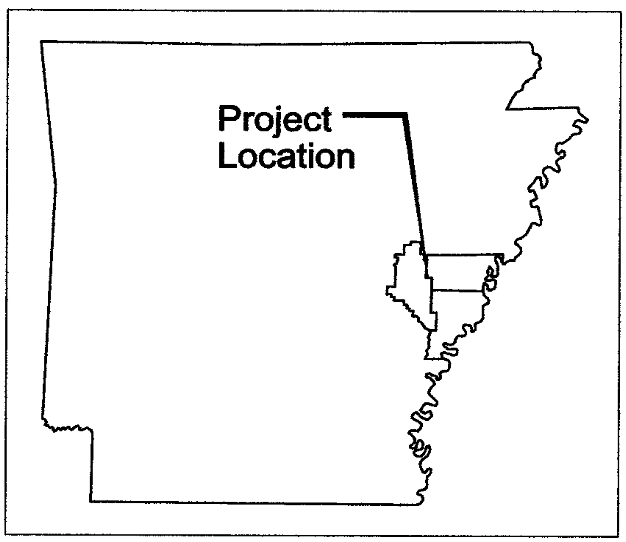


Figure 1—The Little Cypress Creek study area is located in eastern Arkansas at the intersection of Monroe, Lee, and Phillips Counties.

The ability to sustain this unique headwater swamp ecosystem and its rich heritage and values is an important goal for Parks & Tourism and Natural Heritage. Achieving this goal requires the cooperation of over 150 landowners within the Little Cypress Creek watershed. In 1998, Parks & Tourism and Natural Heritage initiated a study of the upper watershed of Little Cypress Creek as the first step in the process to achieve this goal. The objectives of this study were: (1) to identify the factors that most likely have or are contributing to changes in the upper watershed; and, (2) to build community and stakeholder interest and ownership in the restoration and long-term protection or sustainability of the swamp.

The results of phase one of the Little Cypress Creek watershed study are presented in this paper. The results of the initial historical and current data analyses and the community response to and involvement in the project are shown. A hydrologic or water-balance model and model analyses are being completed as phase 2, and the development of a conceptual plan for restoration will be completed as phase 3.

[1] Gandy, Project Scientist/Ecologist, FTN Associates, Ltd., 3 Innwood Circle, Suite 220, Little Rock, AR; Roberson, Natural Resources Planner, AR Department of Parks and Tourism, One Capitol Mall, Little Rock, AR; Foti, Director of Research, AR Natural Heritage Commission, 323 Center Street, Little Rock, AR.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

PROJECT APPROACH AND METHODS

Community Involvement

The ultimate goal of this project is to develop a community-based restoration and protection plan that will be implemented by landowners and other stakeholders within the watershed. Community interest and involvement in the study is a critical component in the overall success of this project. Figure 2 shows the approach used to initiate and foster community involvement in the project. The approach consists of three interrelated and iterative steps: (1) community and stakeholder outreach; (2) education and community participation; and (3) transfer of project ownership to stakeholders. Steps one and two were initiated in this project.

Watershed Data and Analyses

The upper watershed of Little Cypress Creek was identified as the limits of the study (fig. 3) to make best use of project funding. As additional funding potentially becomes available, data collection efforts will be expanded to the lower parts of the watershed.

Available data and information describing historical and current conditions in the study area were obtained from a number of sources and through field surveys. These data and information were compared to current data to identify potential changes or factors within the study area that might have impacted water levels and plant communities.

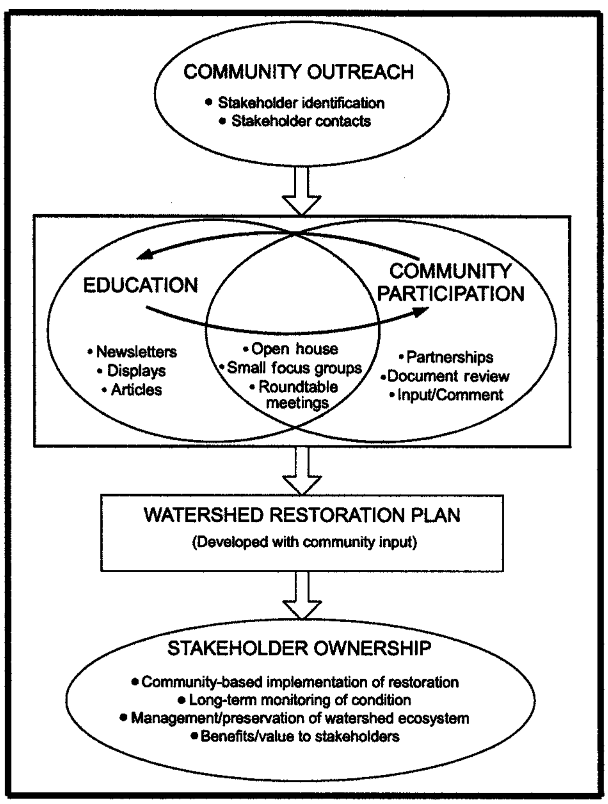


Figure 2—A three-stepped approach is being used to initiate and foster community involvement in the project.

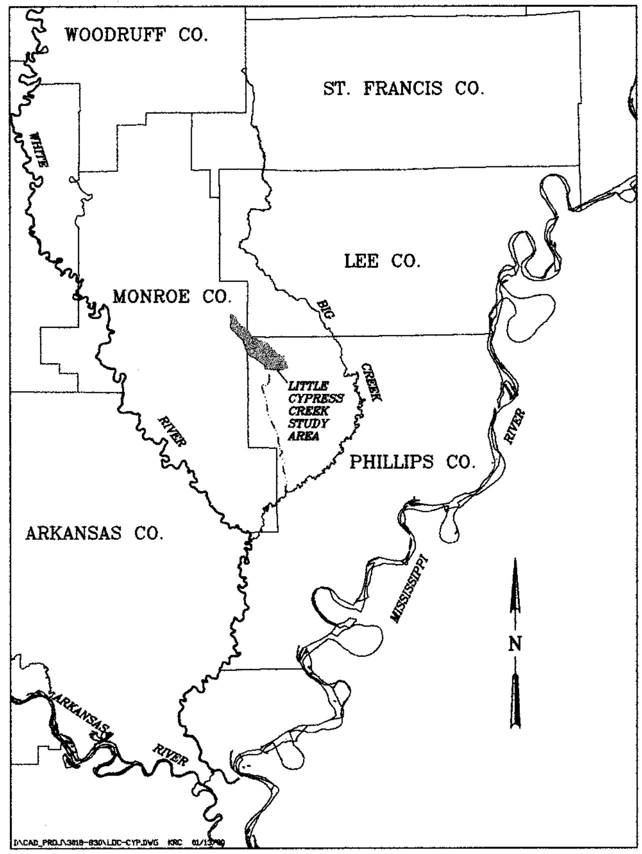


Figure 3—The Little Cypress Creek study area is located within the drainage basin of Big Creek within the White River watershed. The upper drainage area of Little Cypress Creek was the limits of phase 1 study area.

Drainage basin delineation—A preliminary delineation of the study area basin was made based on 1971 USGS topographic quadrangle maps (Monroe, Pine City, and Marvell, ARK 7.5 minute series maps). Field surveys were completed in April 1999 to confirm the preliminary drainage area delineation. Due to the flatness of the area, the drainage basin delineation will be refined as additional information on drainage patterns in the study area are collected.

Aerial photographs—Historical and current aerial photographs were used to identify changes in land use, drainage patterns, hydrologic modifications, and vegetation within and adjacent to the study area. Several sources for historical aerial photographs were investigated including the U.S. Department of Agriculture (USDA) Farm Service Agency (FSA) for Monroe and Phillips Counties, Arkansas State Highway and Transportation Department, and the USDA FSA Aerial Photography Field Office. The oldest aerial photographs of the site were found at the USDA FSA county offices. Black and white aerial photographs were obtained for Phillips County for the period covering 1969 through 1972. Black and white photos dated 1982 were obtained for Monroe County. No aerial photos were found for Monroe County before 1982. No aerial photographs were needed for Lee County because it was covered in the other photographs.

The USDA FSA Aerial Photograph Field Office obtained a set of color infrared photographs of the study area dated 1992. The watershed was flown in May 1999 to obtain current aerial photographs of the study area. Color infrared aerial photographs were produced.

Hydrometeorological data—Historical and recent precipitation and temperature data were available from the National Weather Service monitoring stations at Brinkley, Clarendon, and Marianna, AR. Monthly rainfall and temperature data for these stations were downloaded from CLIMATEDATAÔ (Hydrosphere 1998) for the period of record covering January 1948 through December 1998. Data for January 1999 through December 1999 were obtained from the climatological reports issued by the National Climatic Data Center. Daily rainfall and temperature data for the study period, i.e., February 1999 through August 1999, were obtained from the local observers. Based on the relative location to the study area, data from Clarendon and Marianna were averaged to develop mean annual and monthly precipitation totals for the study area. Data from Brinkley were only used to fill in missing data from the other two stations.

Runoff data—There are relatively few gauging stations within the immediate vicinity of the study area. The closest station is a discontinued station on Big Creek at Poplar Grove (USGS station 07077950) located southeast of the Little Cypress Creek study area. Data were available from this station for the period from 1970 through 1993. These data were supplemented with data from the L'Anguille River at Palestine (USGS station 07047950) to develop a historical record. This station was the closest station in the immediate vicinity that had similar land use. A comparison of annual runoff values from the L'Anguille River station with the Big Creek at Poplar Grove station for the overlapping period justified the use of the L'Anguille River data. The flow data were expressed as runoff in inches per year. The annual totals for the historical period and the mean monthly totals were obtained.

Government Land Office record review—Government Land Office survey records, housed at the Arkansas Geologic Commission office, document early land surveys throughout Arkansas. Records dating back to the original establishment of the Baseline and the 5th Principle Meridian, i.e., the location of the granite monument were reviewed. Information on the historical composition of vegetation in the watershed was obtained. Other vegetation accounts specifically relating to the watershed are practically nonexistent.

Arkansas Department of Heritage—Historical records maintained at the Arkansas Department of Heritage's Arkansas History Commission and Natural Heritage Commission offices also were reviewed to obtain available information on Little Cypress Creek upper watershed and the Louisiana Purchase Historic State Park. The available records spanned a time frame over the past 100 years.

Arkansas Department of Parks and Tourism—A review of files and records at Parks & Tourism's Planning and Development office was completed to obtain additional information on the historical condition or on activities and changes that have been documented for the Park. Information in these files covered a 30-year time frame from mid-1960 to mid-1990.

Water surface elevation measurements—No historical water-surface elevations were available or are known from the study area. To document current conditions within the study area, water-surface elevation data were collected at five locations within the study area (fig. 4). Three types of recording devices were used to measure water-surface elevations within the watershed: continuous stage recorders, staff gauges, and crest gauges.

Continuous stage recorders—Stevens™ stage recorders were placed at three locations within the Little Cypress Creek study area (fig. 4) to record water level data on a continuous basis during the study. One recorder (recorder 7) was placed adjacent to the granite monument to record water levels within the Park. A second recorder was placed on the upstream side of the road at the Cotton Trailer Road (recorder 3). This road is located in the approximate middle of the study area. The third recorder (recorder 9) was placed in the lower portion of the study area on the downstream side of Rogers Road. The monument and Cotton Trailer Road recorders were installed in the watershed in February 1999. The third recorder at Rogers Road was installed in June 1999 after a review of water-surface elevation data indicated that a recorder was needed in the lower part of the study area.

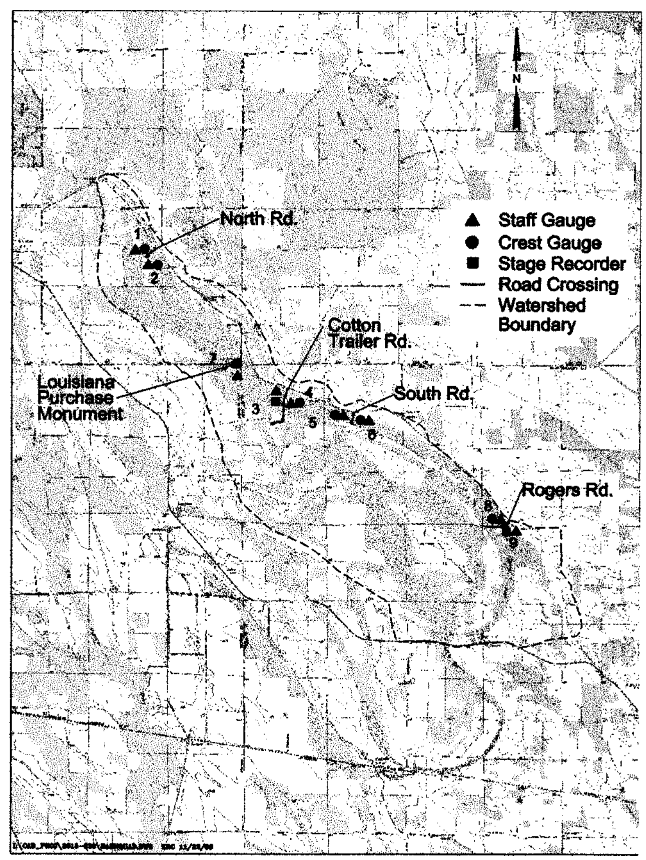


Figure 4—Locations of road crossings and water-surface elevation recorders in the study area.

The stage recorders were set to record changes in water-surface elevations over a 16-day period. Charts were replaced every 2 weeks from February 4, 1999, through August 9, 1999. Electronic data loggers were initially placed on the recorders at the monument and at the Cotton Trailer Road. Continuous water-elevation data were captured with the data loggers for a period of approximately 1 to 2 months, and the data were cross-referenced to the chart data to verify the operation of the recorders.

Staff gauges—A total of nine staff gauges were installed at five locations throughout the study area (fig. 4) to obtain instantaneous water-surface elevations. Only one staff gauge (staff gauge 7) was placed in the Park near the granite monument. Staff gauges were placed on the upstream and downstream sides of the North Road (staff gauges 1 and 2, respectively), the Cotton Trailer Road (staff gauges 3 and 4, respectively), the South Road (staff gauges 5 and 6, respectively), and Rogers Road (gauges 8 and 9, respectively). Staff gauges were surveyed at the beginning and end of the study. Staff-gauge readings were taken on a biweekly basis from February 4, 1999, to August 9, 1999.

Crest gauges—A total of six crest gauges were installed at three locations throughout the study area (fig. 4) to obtain the maximum surface-water elevation at designated locations within a 2-week period. Readings were taken on a biweekly basis from February 4, 1999, through August 9, 1999. Crest gauges were surveyed at the beginning and end of the study period.

Spot elevations—No historical survey data documenting land-surface elevations within the headwater swamp or elevation of the granite monument were found. The elevations of the recorders and gauges, top of the granite monument in the Park, and the ordinary high-water mark on the granite marker were surveyed. Road profiles were shot for the North Road, Cotton Trailer Road, South Road, and Rogers Road to determine the elevation of the lowest point along each road. Where culverts were present under the road, the elevation was obtained for the invert of each culvert.

Surface water withdrawals and irrigation—The Arkansas Soil and Water Conservation Commission was contacted to identify whether any registered riparian or nonriparian water users were located within the study area. Registered users are persons that withdraw greater than or equal to 1 ac ft of water per year from a stream or other waterbody within the State. The USDA Natural Resources Conservation Service County offices in Monroe and Phillips Counties were contacted to identify any water withdrawals within the study area.

Groundwater withdrawals—The U.S. Geologic Survey (USGS) was contacted to obtain a list of irrigation or other wells within the watershed. Although there are several irrigation wells in the study area (USGS 1968, 1984), no information on the amount of groundwater withdrawn from the study area was available from the USGS.

Vegetation surveys—Vegetation surveys were completed in the Little Cypress Creek study area on March 30, April 13, and May 13, 1999, to make observations of existing vegetation. These surveys also were completed to identify potential reference areas within the watershed for restoration. Dominant species, community types, and condition of vegetation were observed and recorded.

RESULTS

Aerial Photograph Review

Land use changes—Approximately 65 percent of the approximately 7,680-ac study area, i.e., approximately 12 mi^2, is currently in agricultural crop production with the remaining 45 percent in forests. Historically, hardwood forests were extensive within the watershed. The majority of land clearing in the study area occurred prior to 1969. Land clearing has been minimal in the Little Cypress Creek study area since that time. A total of 440 ac were cleared in the lower half of the study area, i.e., below the Park, between 1969 and 1982. No additional land clearing of significance was identified between 1982 and 1999. Overall, < 10 percent of the land within the study area had been converted from forest to agricultural land use between 1969 and 1999.

Drainage—The comparison of historical and current aerial photographs did not reveal any obvious changes in drainage patterns within the study area from 1969 to the present. However, discussions with landowners indicated that large changes in drainage patterns have occurred within the watershed, including an increase in pumped drainage and rerouting of flows within the watershed. Detailed mapping of drainage patterns was not conducted as part of this phase of the project but general patterns were recorded. Fields on the far-west side of the study area generally pump water into ditches that ultimately drain to the lower part of the study area. Most of the fields close to or adjacent to Little Cypress Creek either pump or discharge directly into the creek. Several ditches along the west and east sides of the study area carry water directly into Little Cypress Creek.

Levees—Levees have been constructed throughout the watershed to eliminate inundation of adjacent farmland or to hold water during the winter for hunting leases. Landowners in the study area indicate that since the early 1950s, a significant number of acres that historically provided storage of water within the system have been cleared and leveed. Approximately 320 ac were cleared and leveed between 1979 and 1982. Without additional historical and current information, the reduction in storage capacity within the study area could not be adequately quantified, but it is estimated that approximately one-third of the historical storage capacity of the study area has been lost.

Irrigation—Based on conversations with landowners, there has been an increase in the acres of irrigated lands within the study area since the early 1980s due to the conversion from soybeans to rice production. Within the study area, groundwater is continuously pumped onto the fields to eliminate stagnation of water and increased incidence of pests. Consequently, irrigation tail water is continuously discharged into Little Cypress Creek from April through September. Rapid discharges of large volumes of water into Little Cypress Creek also occur at the end of the rice-growing season in preparation for harvest. The volume of water entering the system from irrigated fields was not quantified as part of this phase of the study.

Roads—The historical aerial photographs showed that a number of road crossings existed within the study area prior to 1969. Figure 4 shows the locations of three major crossings south of the Park, i.e., Cotton Trailer Road, South Road, and Rogers Road, and one north of the Park, i.e., North Road, that were present in the study area in 1969 and that currently exist. No new road crossings were constructed in the watershed since 1969. The roadbed of Cotton Trailer Road was elevated some time in the 1990s to its present height.

Hydrometeorological Data

Precipitation—Annual precipitation totals in the study area range from 35.3 to 71.5 in. over the approximately 42-year period of record (table 1), and the long-term mean annual precipitation total is 51.1 in. A comparison of rainfall data for the study period, i.e., 1999, to historical data indicates that total precipitation for 1999 was approximately 4 in. less than the long-term average. The highest total annual rainfall of 71.5 in. was observed in 1979 and was approximately 20 in. greater than the mean annual total rainfall. Observations of an approximate 18-to 24-in. increase in water elevations in the Park made by Parks & Tourism and Natural Heritage personnel in the late 1970s are consistent with this data.

Average monthly precipitation totals for the study area (fig. 5) for the period of record range from approximately 3.0 to 5.3 in. The greatest amounts of precipitation typically fall during the months of November through May. The driest months are June through October. During the study period, conditions were much drier than the average during the fall of 1998. With the exception of February 1999, conditions during the study period were wetter in the winter and spring than long-term monthly mean values.

Runoff—The delineation for the upper Little Cypress Creek drainage area was based on historical topographic maps

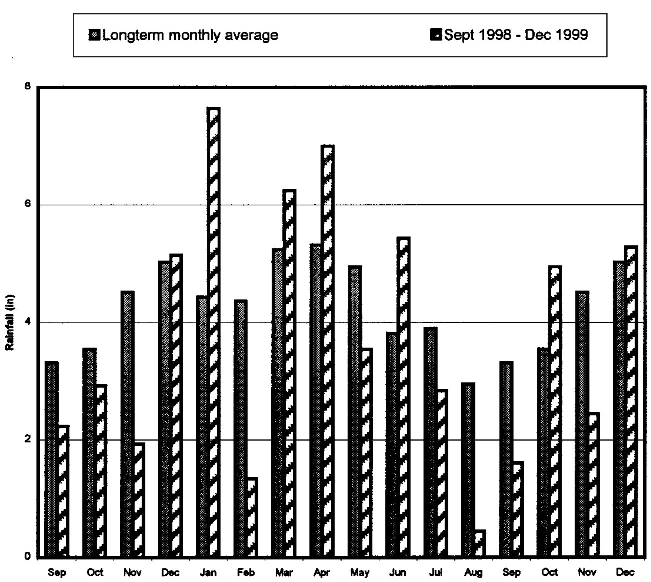


Figure 5—Comparison of monthly rainfall totals during the study period with long-term monthly average rainfall.

and a ground reconnaissance. Due to the flatness of the area and the many anthropogenic modifications for drainage, the basin boundaries are difficult to define and probably have changed over the years. Runoff averaged 18.3 in. per year or approximately 36 percent of precipitation. On a monthly basis, runoff varied from 0.46 in. or 12 percent of precipitation in July to 2.54 in. or 58 percent of precipitation in February.

Groundwater

Although irrigation wells exist within the study area (USGS 1968, 1984), there is no published groundwater level data to characterize groundwater levels within the Little Cypress Creek watershed or to confirm whether groundwater discharges to the Little Cypress Creek wetland ecosystem. Papers describing the geomorphology of the Little Cypress Creek drainage area and the HGM classification of wetlands in Arkansas (Klimas 1999, Saucier 1996) indicate that there is the potential for groundwater discharge into Little Cypress Creek. Based on groundwater data (USGS 1998) collected from a well located less than one-fourth of a mile south of the study area (local well 1S01E20DDB1), the water table may be as close as 10 to 20 ft below ground surface at ground surface elevations of 185 ft.

Hydrographs from wells located throughout the region show a general decrease in groundwater levels, since the mid-1950s (USGS 1998). Landowners with irrigation wells within the study area indicated that groundwater within the study area has not exhibited this trend. These observations are consistent with groundwater elevation data collected from a local well several miles south of the study area in Phillips County (local well 02S01E28CCB1). Data collected since 1955 from this well show that groundwater levels have varied over the 40-year period of record but have not been decreasing.

Water-Surface Elevations

Time series plots of water-surface elevations at the monument and at Cotton Trailer Road indicate that water-surface elevations within the wetland ecosystem follow seasonal rainfall patterns, increasing during the winter and early spring, then dropping during late summer and early fall. Early in the season, water levels in the swamp also rise in response to individual storm events (fig. 6). The corresponding rise in surface-water elevations with large storm events occurs mostly before leaf out and disappears during the latter part of the growing season when evapotranspiration is at its highest. The measured increase in water-surface elevations is greater than total precipitation inputs into the basin, indicating that runoff is an important hydrologic input into the system.

During the study period, maximum water-surface elevations in the Park were approximately 10 in. lower than the high-water mark on the granite marker. The high water levels observed in the late 1970s and 1990s were not observed during this study. Staff gage readings taken on the upstream and downstream sides of four major roads show that surface-water elevations on the upstream sides of the roads typically were higher than the downstream sides.

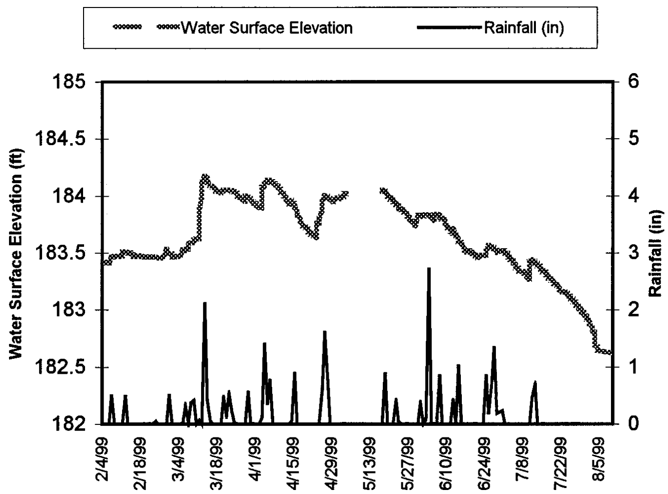


Figure 6—Continuous stage recorded within the Park (recorder 7), and daily rainfall from January 1999 through August 1999.

Surface-water profiles of the study area for the months of March, June, and July 1999 provide a graphical depiction of surface-water elevations relative to location in the watershed and the roads (figs. 7, 8, 9). Water-surface elevations drop over the season and typically are higher on the upstream side than the downstream side of the road. The profile from July shows an increase in surface-water elevations between the South Road and Rogers Road during a period when the water levels are dropping in other parts of the ecosystem. Landowners indicate that at times, water flows backwards, i.e., upstream, over the South Road.

Vegetation Assessment

The original survey notes of the Louisiana Purchase Monument area provide strong evidence that the basic vegetation and landscape features immediately surrounding the granite monument probably have changed little since 1815. The survey notes do not adequately distinguish between certain critical species. However, the records clearly denote that areas presently occupied by tupelo swamp were occupied by swamp in 1815. From the historical accounts, it appears that the area surrounding the granite monument ranged from being flooded to dry during the summer months. No historical information was found describing the vegetation in other areas of the Park or within the study area.

The upper watershed of Little Cypress Creek contains a diversity of upland and wetland plant communities. The central core of the wetland ecosystem supports the deepest water levels, i.e., 2 to 5 ft deep, and is dominated by water tupelo (*Nyssa aquatica* L.), although, in some cases, bald cypress (*Taxodium distichum* L.) also occurs. Historically, bald cypress was codominant in the deep swamps. With the exception of a few depressions within the deep swamp, timber harvest activities have removed most of the bald cypress from this community.

The shallowly flooded areas on the west side of the Park were dominated by overcup oak, red maple [*Acer rubrum* L. var. *rubrum* (BB) and *A. rubrum* L. var. *drummonii* (H. & A.) Sarg.], sweet gum (*Liquidambar styraciflua* L.), and button bush. Several dead snags of overcup oak are present within this community, and those that are alive exhibit morphological features indicative of stress. Tree branches are short, thickened, and knarled, and many exhibit epicormic branching. Almost all of the dead trees appear to be overcup oak and approximately the same age. The 1999 field surveys also identified that many overcup oaks near the entrance of the Park appear to be infected with a canker rot fungus. A large percentage of the oaks in this part of the Park and along the road leading into the Park are heavily infested with insect galls.

Beaver Impacts

Beaver dams exist throughout the watershed. Large beaver dams were recorded at the North Road, at the Cotton Trailer Road, in the vicinity of the South Road, and at the outlet end of the study area where Little Cypress Creek crosses Arkansas State Highway 49. Other beaver dams likely exist within the study area. Most of the beaver activity appears to take place in the more shallowly flooded areas and where there has been disturbance. The beaver-impacted area upstream of the North Road appears to have been in existence for approximately 30 years based on the size and appearance of the black willow (*Salix nigra* Marsh.) trees. Some of these areas also appear to have been cut over repeatedly by beaver. Beaver activity has resulted in a degradation of bottomland hardwood forests and loss of overstory and resulted in increased scrub/shrub communities dominated by black willow and buttonbush.

Timber Harvest

Historical and recent timber-harvesting activities have occurred throughout the study area. In drier portions of the swamp such as the area adjacent to the Park on the east, there is ample evidence of former timber-harvest activities in the form of stumps and downed logs. An open area within the swamp just upstream of the South Road has been impacted by timber harvest of bald cypress. No pole-sized timber or seedlings have become reestablished in this area.

DISCUSSION

Over the past decade there has been a shift away from the management and protection of natural resources on a site-by-site basis to the management and protection of natural resources on an ecosystem basis, i.e., watershed, landscape, or regional scale (Sparks 1995). The assessment, management, and protection of resources is being conducted on spatial scales ranging from millions of acres, such as in the Chesapeake Bay and the Florida Everglades, to hundreds of acres in individual drainage basins and watersheds. Current programs and regulatory mandates, e.g., unified watershed assessments, total maximum daily load studies, etc., are evidence that policy and decisionmakers, natural resource managers, scientists, regulators, and the public recognize that the sustainability of natural resources is dependent on the management of these resources at larger scales.

The attempt to manage resources and water within the Little Cypress Creek study area on an individual-by-individual farm basis has failed. As landowners have cleared, drained,

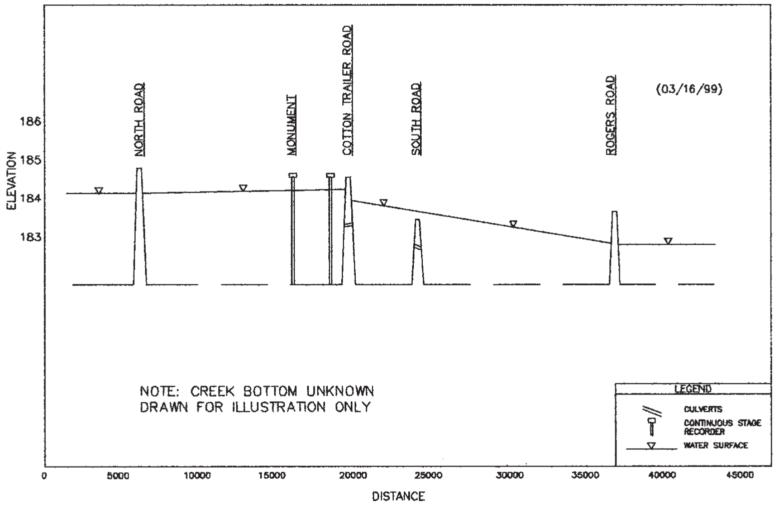


Figure 7—Water-surface profile in the Little Cypress Creek upper watershed study area on March 16, 1999.

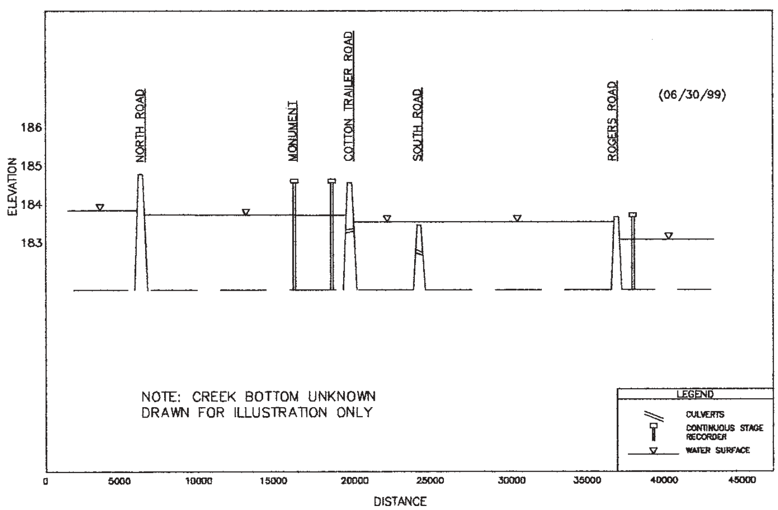


Figure 8—Water-surface profile in the Little Cypress Creek upper watershed study area on June 30, 1999.

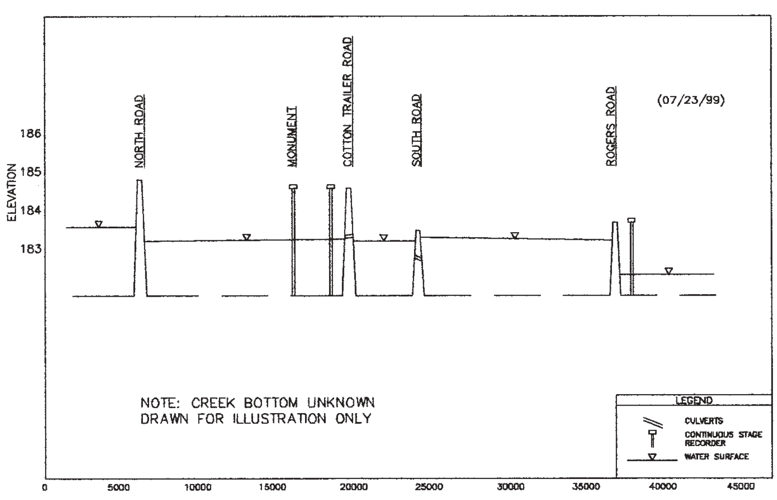


Figure 9—Water-surface profile in the Little Cypress Creek upper watershed study area on July 23, 1999.

or held water on one property, others have been flooded and lost timber on other parcels. Although small relative to other large-scale restoration and management efforts in the United States., a comprehensive watershed approach is needed to manage, restore, and sustain the values and condition of the Little Cypress Creek headwater swamp ecosystem.

Because little historical data is available to compare to current data and conditions, it is difficult to clearly identify and quantify the changes within the headwater swamp ecosystem. In the absence of historical data, the data collected in the phase 1 study were summarized in the context of a conceptual model (fig. 10). This conceptual model focuses on the changes in hydrology within the upper watershed, which can then be linked or correlated to other changes within the watershed, e.g., changes in plant community composition, changes in wildlife habitat and usage, etc. This conceptual model summarizes the hypotheses and assumptions that can be tested or evaluated and serves as a framework for identifying data gaps and future data needs or studies. The conceptual model will be modified and expanded over time as new information and data are gathered, the natural variability within the system is better quantified, and as changes or perturbations continue to occur within the system.

Changes in Storage

Alterations in the Little Cypress Creek upper watershed have had an influence on the historical hydrologic regime of the headwater swamp ecosystem. These changes can be expressed as a change in the water balance within the system. Although alterations within the watershed can either increase or decrease water stored within the swamp ecosystem, the direction of change in the Little Cypress Creek study area has been weighted more toward an increase in the amount of water stored within the Little Cypress Creek headwater swamp ecosystem (fig. 10).

Like most watersheds in the Mississippi Alluvial Plain of the Lower Mississippi Valley, bottomland and upland forests of the Little Cypress Creek watershed have been cleared for agricultural production. Approximately 65 percent of the Little Cypress Creek study area has been cleared of its original forest cover and is in agricultural production. Clearing within the Little Cypress Creek study area likely has changed the amount and timing of runoff into the Little Cypress Creek headwater swamp. Continuous water-surface elevation data collected at the stage recorders indicate that changes in stage are greater than the total rainfall of any single storm event. The data suggest surface runoff during a storm event is an important input into the system.

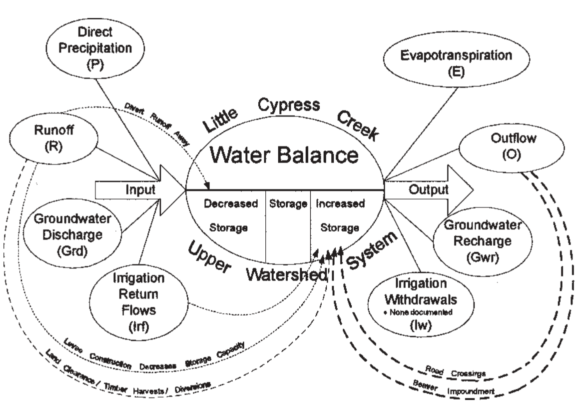


Figure 10—Conceptual model of potential factors affecting water storage within the Little Cypress Creek upper watershed study area.

Crop changes from soybeans to rice also have changed the hydrology of the Little Cypress Creek ecosystem. The timing of hydrological inputs has been altered, and inputs of water into the system are currently being extended into late summer and fall when natural precipitation and runoff inputs typically begin to decrease. Groundwater has now become a surface-water input into the system as it is pumped onto fields and discharged into the creek on a continuous basis from April through September.

Levee construction within the study area has largely been successful in keeping water off many parcels that historically were wet and supported bottomland hardwood forests. Levee construction, however, has effectively reduced the historical storage capacity, constricted drainage, and reduced total storage area within the watershed. Belt (1977) and Myers and White (1993) have documented that the constriction of the floodway has increased flood stage on the Mississippi River. Although the hydraulics of the Little Cypress Creek system are very different from the Mississippi River, the loss of storage capacity and constriction of drainage by levee construction can have the same effect. Without a concomitant reduction in hydrologic inputs, the amount of water stored within the remaining headwater swamp ecosystem and water levels will increase on a per-acre basis.

Road construction and the introduction of beaver have also contributed to the alteration of the natural hydrology of the system. The roads and the beaver dam at the North Road have compartmentalized the ecosystem and impeded natural flows within and discharge out of the system. In this low gradient, flat ecosystem where there is < 1 ft difference in elevation from the top of the watershed to the outlet end, water is easily impounded on the upstream sides of the roads increasing both the aerial extent of inundation and water depths within each compartment. The total acres inundated and depth of water within each compartment will depend on the amount, timing, and duration of the various hydrologic inputs into each of these compartments.

Above normal rainfall totals and increased runoff that occurred in the late 1970s and in the early 1990s are consistent with increased storage and increased surface-water elevations observed in the Park since the late 1970s. Based on these data alone, it would suggest that some of the increases in surface-water levels observed in the Park in the late 1970s through the early 1990s were due to natural variability in rainfall. Although no data are available to document whether water storage has increased in the Little Cypress Creek headwater swamp, it is indicated by this study that the cumulative effects of increased runoff, loss of storage capacity, constriction and impediments in the drainageway, and new hydrologic inputs should increase the amount of storage within the remaining headwater swamp ecosystem.

Vegetation Changes

Beaver impacts have become widespread throughout Arkansas since the reintroduction of this species in the State in the 1950s (Selander 1979). Beaver activities within the study area have led to a marked increase in sedimentation, loss of forest cover, and changes in vegetative composition. Some of these areas appear to have been affected by beaver for at least 30 years.

Changes in the duration, depth, and timing of flooding or inundation of bottomland hardwood systems are documented to have an effect on the survival of many bottomland hardwood trees (Theriot 1993, Whitlow and Harris 1979). Although it is reported that major changes in water level may cause mortality in water tupelo (Johnson and Beaufait 1965), no evidence of dead or stressed tupelo was observed within the Park or in the rest of the study area.

There is evidence of mortality in overcup oak near the entrance of the Louisiana Purchase Historic State Park. The oak mortality at the entrance to the Park appears, however, to represent an episodic event rather than a continuing occurrence at the site. Almost all of the dead trees appear to be overcup oak, and most appear to be of approximately the same age. In addition to mortality, many of the overcup oak trees that remain in this part of the Park are infected with what appears to be a canker rot fungus. Canker rot fungus has been reported to occur in bottomland hardwood forests in the South and to cause mortality in many wetland species (McCracken 1978), but overcup oak is not reported to be susceptible to canker rot (Morris 1965, Solomon 1990).

The reported flood tolerance of overcup oak varies from being very tolerant of deep flooding, i.e., flooding for more than 1 year (Theriot 1993, Whitlow and Harris 1979) to moderately tolerant; i.e., able to tolerate saturated or flooded soils for several months during the growing season but susceptible to high mortality if flooding persists or reoccurs several consecutive years (Hook 1984, McKnight and others 1981). A few reports indicate that high mortality in overcup oak is likely if flooding persists or reoccurs for a period greater than 3 years (Teskey and Hinckley 1978).

Historical precipitation data, however, show an approximate 10-year period of below average rainfall conditions from 1962 to 1972. The study suggests that dry conditions in the ecosystem from 1962 to 1972 could be as important a factor in stressing overcup oak as increased storage of water in the late 1970s. Periodic occurrences of oak decline have been reported in many areas of the southeast (Wargo and others 1983). The causes of oak decline are very complex and often are due to the interaction of multiple stressors (Leininger 1998, McCracken and others 1991, Wargo and others 1983). A combination of drought, flooding, changes in hydroperiods, and pest infestation could be important factors in the decline of overcup oak-dominated forest in the Park.

Community Involvement

Initial public participation in and response to this study has been positive. Many landowners cooperated with the study, and a partnership with the local high school was formed. Over 40 individual stakeholders have provided support to the project in the form of interest in the project, information, permission to enter property, permission to install scientific equipment on private property, and collection of data.

Involved landowners and stakeholders would like to see the Little Cypress Creek watershed restored to: (1) reduce flooding on their properties; (2) increase ability to farm marginal property; (3) restore lost bottomland hardwoods to increase income from hunting leases; and, (4) restore bottomland hardwood values for personal recreational opportunities. With a core group of landowners showing interest in the management and restoration of the Little Cypress Creek ecosystem, the initial steps in gaining community involvement in the project were considered successful. Like other community-based restoration programs that have been developed throughout the country to sustain natural resources (EPA 1997, French 1999), it is anticipated that it will take years to build a community-based alliance focused on the restoration of the Little Cypress Creek watershed.

The information collected in this first phase of the Little Cypress Creek watershed study has only begun to scratch the surface of the information needed to assess condition, determine restoration alternatives, and develop management plans for the watershed. Nevertheless, this study of the Little Cypress Creek upper watershed has lead to a number of important findings relative to the project goals and objectives; and community awareness of the issue has increased. A more organized or formal vehicle for stakeholder involvement has been established as a result of the study along with identification of those areas that are critical for continued investigation.

ACKNOWLEDGMENTS

The Little Cypress Creek Upper Watershed Hydrologic and Environmental Study was funded through the U.S. Environmental Protection Agency Region 6, State Wetlands Grants program. The Arkansas Department of Parks and Tourism and the Arkansas Natural Heritage Commission provided matching funds for the project. The authors wish to acknowledge the following individuals who contributed greatly to the project: Dr. Dennis E. Ford, Shannon P. Holbrook, Dr. Gary E. Tucker, and Gerry T. Conley of FTN Associates, Ltd.; Brinkley High School East lab students; and the Little Cypress Creek landowners.

REFERENCES

Belt, C.B. 1977. The 1973 flood and man's constriction of the Mississippi River. Science. 189: 681–684.

French, L.J. 1999. A case study in forming a grassroots watershed organization. DE: Nanticoke Watershed Alliance. 3 p.

Hook, D.D. 1984. Waterlogging tolerance of lowland tree species in the South. Southern Journal of Applied Forestry. 3: 136–149.

Hydrosphere [Corporation]. 1998. CLIMATEDATA™ CD-ROM, NCDC summary of the Day-West2. Boulder, CO: Hydrosphere Inc. Vol. 9.3.

Johnson, R.L.; Beaufait, W.R. 1965. Water tupelo (*Nyssa aquatica* L.). In: Fowells, H.A. comp. Silvics of forest trees of the United States. Agric. Handb. 271. Washington, DC: U.S. Department of Agriculture: 284–286.

Klimas, C. 1998. Preliminary classification of wetlands in Arkansas based on hydrogeomorphic criteria. Little Rock, AR: Arkansas Multiagency Wetland Planning Team.

Leininger, T.D. 1998. Effects of temperature and drought stress on physiological processes associated with oak decline. In: Mickler, R.; Fox, T., eds. The productivity and sustainability of southern forest ecosystems in a changing environment. New York: Springer Verlag, Inc.: 647–661.

McCracken, F.I. 1978. Canker rots in southern hardwoods. Forest Insect and Disease Leafl. 33. Stoneville, MS: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 4 p. In cooperation with: Mississippi Agricultural and Forestry Experiment Station; Southern Hardwood Forest Research Group. [Internet version].

McCracken, F.I.; Ammon, V.; Solomon, J.D.; Nebeker, T.E. 1991. Oak decline in the Lower Mississippi River Valley. In: Coleman, S.S., Neary, D.G., comps. Proceedings of the sixth biennial southern silvicultural research conference; 1990 October 30–November 1; Memphis, TN. Gen. Tech. Rep. 70. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 299–306. Vol. 1.

McKnight, J.S.; Hook, D.D.; Langdon, O.G.; Johnson R.L. 1981. Flood tolerance and related characteristics of trees of bottomland forests of the Southern United States. In: Clark, J.R.; Benforado, J., eds. Wetlands of bottomland hardwood forests. New York: Elsevier Press: 26–69.

Morris, R.C. 1965. Overcup oak (*Quercus lyrata* Walt.). In: Fowells, H.A., comp. Silvics of forest trees of the United States. Agric. Handb. 271. Washington, DC: U.S. Department of Agriculture: 600–602.

Myers, M.A.; White, G.E. 1993. The challenge of the Mississippi flood. Environment. 10(12): 6–35.

Saucier, R.T. 1994. Geomorphology and quarternary geologic history of the Lower Mississippi Valley. Vicksburg, MS: U.S. Army Engineers Waterways Experiment Station. 364 p. Vol. 1.

Selander, J. 1979. A guide to Arkansas mammals. Conway, AR: River Road Press. 313 p.

Solomon, J.D. 1990. Overcup oak (*Quercus lyrata* Walt.). In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service. 877 p. Vol. 2.

Sparks, R.E. 1995. Need for ecosystem management of large rivers and their floodplains. BioScience. 45(3): 168–182.

Teskey, R.O.; Hinckley, T.M. 1978. Impact of water level changes on woody riparian and wetland communities. southern forest region. FWS/OBS–78/87. Columbia, MO: U.S. Department of the Interior. U.S. Fish and Wildlife Service, Office of Biological Services. 46 p. Vol.2.

Theriot, R.F. 1993. Flood tolerance of plant species in bottomland forests of the Southeastern United States. Wetlands Res. Program Tech. Rep. WRP–DE–6. Vicksburg, MS: U.S. Army Engineers Waterways Experiment Station, Environmental Laboratory. 50 p.

U.S. Environmental Protection Agency. 1997. Top 10 watershed lessons learned. EPA/840–F–97–001. Washington, DC: U.S. Environmental Protection Agency, Office of Water, Office of Wetlands, Oceans, and Watersheds. 59 p.

U.S. Geological Survey. 1968. Quaternary aquifers in the Mississippi Embayment. Geol. Sur. Prof. Pap. 448–E. Washington, DC: U.S. Government Printing Office. 15 p.

U.S. Geological Survey. 1984. Maps showing altitude of the potentiometric surface and changes in water levels of the alluvial aquifer in eastern Arkansas, spring 1983. Water Res. Invest. Rep. 84–4264. Little Rock, AR: U.S. Department of the Interior, U.S. Geological Survey. 1 p. In cooperation with: Arkansas Geological Commission and the Arkansas Soil and Water Conservation Commission.

U.S. Geological Survey. 1998. Status of water levels and selected water quality conditions in the Mississippi River Valley alluvial aquifer in eastern Arkansas, 1994–96. Water–Res. Invest. Rep. 98–4131. Little Rock, AR: U.S. Department of the Interior, U.S. Geologic Survey. 72 p. In cooperation with: Arkansas Soil and Water Conservation Commission and the Arkansas Geological Commission.

Wargo, P.M.; Houston, D.R.; LaMadeleine, L.A. 1983. Oak decline. Forest Insect and Disease Leafl. 65. Hamden, CT: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station; Northeastern Area State and Private Forestry. 6 p. [Internet version].

Whitlow, T.H.; Harris, R.W. 1979. Flood tolerance in plants: a state-of-the-art review. Tech. Rep. E–79–2. Vicksburg, MS: U.S. Army Engineers Waterways Experiment Station, Environmental Laboratory. 161 p.

BOTTOMLAND HARDWOOD AFFORESTATION: STATE OF THE ART

Emile S. Gardiner, D. Ramsey Russell, Mike Oliver, and Lamar C. Dorris, Jr.[1]

Abstract—Over the past decade, land managers have implemented large-scale afforestation operations across the Southern United States to rehabilitate agricultural land historically converted from bottomland hardwood forest cover types. These afforestation efforts were initially concentrated on public land managed by State or Federal Government agencies, but have later shifted towards private holdings that qualified for governmental assistance or cost-share programs. Traditional silvicultural practices dominate bottomland hardwood afforestation schemes in the South, with 1-0 bare-root oak (*Quercus* spp.) seedlings comprising the balance of planting stock mixtures. However, traditional methods do not always yield successful afforestation, especially when applied on an operational scale across a landscape of heterogeneous site types and ownership objectives. This manuscript summarizes bottomland hardwood afforestation techniques and compares the knowledge base with current practices. Additionally, this manuscript reviews new silvicultural systems to enhance establishment success on adverse sites, to enhance ecological benefits of afforestation, and to address multiple objectives of landowners. We identify four vital components of afforestation that are generally lacking in most regeneration activities in the Lower Mississippi River Alluvial Valley.

INTRODUCTION

Afforestation activities in the Lower Mississippi River Alluvial Valley (LMRAV) are currently peaking after the past four decades of extensive deforestation (Allen 1997, King and Keeland 1999, Stanturf and others 1998). Different forces drive the afforestation efforts of public resource agencies, conservation organizations, private corporations, and private landowners. Their interests include conversion of economically marginal agricultural land to forest cover for ecosystem rehabilitation, soil conservation, aesthetics, and recreation. They may want to establish intensively managed fiber farms and carbon sequestration banks, and to mitigate forested wetlands destroyed elsewhere in development projects.

Since 1992 on more than 250,000 ac in Louisiana, Mississippi, and Arkansas, governmental incentive programs have defrayed plantation establishment costs and/or purchased long-term conservation easements from private landowners (King and Keeland 1999, Stanturf and others 1998). Since program dollars largely support these afforestation activities, associated enrollment deadlines often create a sense of urgency about plantation establishment. The urgency with which managers have implemented recent afforestation activities leads some to question the validity of afforestation decisions and practices at the stand, forest, and landscape levels (Allen 1997; Stanturf and others, in press; Twedt and Portwood 1997; Wilson and others, in press).

A sustained, formal research initiative on bottomland hardwood plantation establishment in the LMRAV began as early as the 1960s (Kennedy 1993). This manuscript summarizes current afforestation techniques and compares the knowledge base with current practices to emphasize aspects of afforestation where existing knowledge is not incorporated into operational practice. This manuscript also reviews new silvicultural systems to enhance success on adverse sites and enhance ecological benefits, while addressing multiple objectives of landowners. In addition to available literature, a large portion of this manuscript is based on the authors' combined observations and experiences in the LMRAV.

REVIEW OF CURRENT AFFORESTATION TECHNIQUES

Management Objectives

Defining clear management objectives in terms of what outputs are to be achieved is prerequisite to directing successful establishment and future management of afforestation sites. Landowner objectives will ultimately influence or govern all system decisions involving stand establishment (species suitability, site preparation requirements, planting density, postplanting operations), intermediate stand management (precommercial thinning, timber stand improvement, stand health and sanitation practices, improvement cutting and thinning, control of fire, disease, insects, or other damaging agents), and regeneration harvesting (including regeneration methods) in order to regulate forest structure for the desired outputs (Daniel and others 1979). Management objectives can be simple or complex encompassing multiple aspects of watershed or wildlife management, wood, fiber or forage production, and aesthetics (Daniel and others 1979, Smith 1986). Rehabilitating functions and health of the bottomland hardwood forest ecosystem always underlies other management objectives.

[1] Gardiner, Research Forester, Center for Bottomland Hardwoods Research, Southern Research Station, USDA Forest Service, Stoneville, MS 38776; Russell, Administrative Forester, North Mississippi Refuges Complex, US Fish and Wildlife Service, Grenada, MS 38901; Oliver, Project Biologist, Ducks Unlimited Inc., Vicksburg, MS 39180; Dorris, Administrative Forester, Yazoo National Wildlife Refuge Complex, US Fish and Wildlife Service, Hollandale, MS 38748.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

Government monetary incentives for landowners to establish forest cover drive objectives on the vast majority of afforestation sites in the LMRAV (Haines 1995, Kennedy 1990, Stanturf and others 1998). Although a multitude of cost-share programs are available to private landowners, most incentives target public environmental goals such as conservation of highly erodible or otherwise delicate soil (Haines 1995, Kennedy 1990). Even when program objectives allow for "creation of wildlife habitat," "promotion of biodiversity," or "production of sustainable timber harvest," these objectives often offer insufficient focus to ensure optimal management of future stands for specific outputs (Wilson and others, in press). The landowner should clearly define singular or multiple objectives; for example, a management objective for carbon sequestration would describe the desired accumulated carbon or biomass output in tons per acre per year over the specified rotation. Without explicit objectives, the forester cannot properly determine planting density, species assignments, site preparation requirements, or postplanting cultural practices; nor can he or she evaluate success of the afforestation effort. The failure to specify management objectives has led to wholesale establishment of stands without clear description of future management pathways. Our observation suggests that some landowners may be making uninformed decisions. The current approach does not address sustainability issues and could prove costly if the future forest structure is not compatible with desired outputs.

Species–Site Selections

Alluvial floodplain forests exhibit high species richness and spatial diversity of vegetational communities (Kellison and others 1998, Meadows and Nowacki 1996). Bottomland hardwood forests are comprised of more than 70 endemic tree species along with numerous vines, shrubs, and herbaceous species (Carter 1978, Putnam and others 1960, Tanner 1986). A wide array of edaphic and hydrologic conditions sculpted by the erosional and depositional processes of rivers provide the foundation for vegetational diversity in alluvial floodplains. Site types range from permanently inundated sloughs with very poorly drained, heavy clay soils to rarely inundated ridges of well-drained, sandy loams (Stanturf and Schoenholtz 1998). Since the early 1900s, studies have associated tree species with various site types (Meadows and Nowacki 1996, Putnam and others 1960, Tanner 1986). It follows that the suitability of the species assigned to a given site will largely determine initial and long-term afforestation success, the trajectory of stand development, site productivity, future management opportunities, and costs. Some of these relationships appear in reports by Baker (1977), Dicke and Toliver (1987), Krinard and Johnson (1985), Stine and others (1995), and Williams and others (1993).

Afforestation information is available to assist the forester with species-site prescriptions. Useful sources include "A Practical Field Method of Site Evaluation for Commercially Important Southern Hardwoods" by Baker and Broadfoot (1979), "Hardwood Suitability for and Properties of Important Midsouth Soils" by Broadfoot (1976), and county soil series manuals published by the Natural Resource Conservation Service. In addition, basic information characterizing physical and chemical soil properties can identify soil texture and drainage classes, plow pan development, nutrient deficiencies or factors such as pH that regulate nutrient uptake. Surveys of adjacent forested stands to determine local abundance of desirable species can inform decisions on species assignments (Groninger and others 1999). In practice, though, market availability of planting stock is probably the most prevalent factor-driving species assignments on afforestation sites. Fewer than 25 of the 70-plus native bottomland hardwood species are available through established commercial nurseries on a yearly basis [(Personal communication. 2000. Sam Campbell, Nursery Manager, Molpus Timberlands Mgmt. L.L.C., 29650 Comstock, Elberta, AL 36530); (Personal communication. 2000. David McCain, Nursery Manager, Delta View Nursery, Route 1 Box 28, Old Highway 61 South, Leland, MS 38756); (Personal communication. 2000. Randy Rentz, Nursery Manager, Columbia Nursery, P.O. Box 647, Columbia, LA 71418); (Personal communication. 2000. Gary Schaefer, Nursery Manager, Winona Nursery, Route 3, Box 83, Winona, MS 38967)]. However, some nursery managers will raise custom seedlings of other species if contracted.

Site Preparation

Site preparation can be vital to afforestation of former agricultural land. Treatments can condition the seed or seedling bed; decrease competing or undesirable vegetation, such as exotic pests; reduce herbivore habitat; improve nutrient availability; and improve access on the site for the planting operation (Baker and Blackmon 1978; Kennedy 1981a, 1993). Benefits are typically realized though increased survival and improved early growth of hardwood planting stock (Baker and Blackmon 1978, Ezell and Catchot 1998, Russell and others 1998). The wide array of conditions on former agricultural fields precludes wholesale prescription of site preparation practices. Rather, the landowner's objectives and the condition of the field determine the appropriate level of site preparation for a given tract. For fields immediately out of crop production, site preparation is generally not necessary unless objectives and site conditions make it desirable to break up a hard pan or compacted soil, broadcast a pre-emergent herbicide application for weed control, or incorporate fertilizer into the planting site. Fertilization, for example, can consistently boost growth of hardwood reproduction on former agricultural sites, because long-term agricultural production significantly depletes soil organic matter and associated nutrients (Francis 1985, Houston and Buckner 1989). Such practices are common if fiber production, timber production, or biomass production are identified as primary management objectives (Joslin and Schoenholtz 1998, Kennedy 1981a, Thornton and others 1998, Yeiser 1999), but also have merit where other objectives target early stand growth and development.

Depending on the length of the uncultivated period and the rate of succession, fields removed from cultivation for more than a year prior to planting will present a range of herbaceous and woody vine, shrub, or tree competition. It may be desirable to control advance vegetation prior to planting, and site preparation practices for such fields can be accomplished with mechanical or chemical methods. In the LMRAV, multiple-pass disking has been effective to bust dense sod, improve soil aeration, and promote water infiltration (Baker and Blackman 1978, Kennedy 1990, McKnight 1970). Following years of cultivation, subsoil or deep plowing to 16 to 20 in. is effective in breaking plow pans that may develop. This practice, which is generally necessary for establishment of fast-growing species, such as eastern cottonwood (*Populus deltoids* Bartr. Ex Marsh) (McKnight 1970), improves aeration and allows the regeneration to exploit a greater soil volume.

Chemical site preparation and new dormant season weed control applications show promise for relatively inexpensive early control of herbaceous weeds for improved survival and growth in hardwood plantations (Ezell 1995, 1999; Ezell and Catchot 1998; Ezell and others 1999). Chemical methods of site preparation offer the forester an ability to apply weed control during periods when site conditions prevent use of mechanical practices. Prior to herbicide application, mowing or burning the field and allowing for a uniform regrowth of vegetation can improve efficacy (Miller 1993).

In the LMRAV, mowing is a common site preparation technique on afforestation projects sponsored by governmental cost-share programs. This practice improves planter access on afforestation sites that have not received cultivation for several years, but mowing probably does little to reduce weed competition or herbivory (Houston and Buckner 1989). In fact, empirical studies rarely demonstrate improved survival or growth of hardwood regeneration following mowing for site preparation and/or subsequent weed control (Houston and Buckner 1989, Kennedy 1981b, Schweitzer and others 1999). Kennedy (1981b) reasoned that mowing is not effective for improving growth or survival because it does not reduce competition for soil water or nutrients. Prescribed burning, a more economical practice than mowing, can also be used to improve planter access on afforestation sites. However, the use of prescribed fire requires training, and liability related to smoke management may limit the use of burning in some regions.

In the LMRAV, it is common to omit site preparation on some sites to accommodate use of heavy equipment in machine planting. To reduce site damage on saturated soil, competing vegetation and the increased risk of herbivory are accepted in a tradeoff with improved trafficability and planting machine function.

Planting Stock Types, Size, and Procurement

Planting stocks used for afforestation in the LMRAV include seed, bare-root seedlings, containerized seedlings, and cuttings. Hard mast species, which can be successfully established with seed on an operational scale include several of the red oak species (*Quercus nigra* Linnaeus, *Q. phellos* Linnaeus, *Q. shumardii* Buckley, *Q. pagoda* Rafinesque, *Q. nuttallii* Palmer); white oak species (*Q. lyrata* Walter, *Q. michauxii* Nuttall); common persimmon (*Diospyros virginiana* Linnaeus); and sweet pecan [*Carya illinoensis* (Wang.) K. Koch] (Johnson and Krinard 1985, Stanturf and others 1998). Bottomland species that can be established vegetatively with cuttings include eastern cottonwood, American sycamore (*Platanus occidentalis* Linnaeus), green ash (*Fraxinus pennsylvanica* Marshall), and black willow (*Salix nigra* Marshall) (Kennedy 1977, McKnight 1970, Stanturf and others 1998). The species listed above and others not listed can be established as bare-root or containerized seedlings. Choice of stock type should be determined by management objectives, site preparation practices, species decisions, planting window, and market availability. King and Keeland (1999) estimated that over 67 percent of the public land and cost-share plantings in the LMRAV have been established with 1-0, bare-root seedlings.

Size and quality of bare-root planting stock can be of major importance in determining establishment success and early growth of tree seedlings (Land 1983, Thompson and Schultz 1995). Because of differing growth rates, growth habits, i.e. indeterminate, semi-determinate, or determinate, and biomass accumulation patterns (Dickson 1994; Hodges and Gardiner 1993; Long and Jones 1993, 1996), bottomland hardwoods exhibit a wide range of interspecific seedling morphologies. Early researchers working on bottomland hardwood regeneration identified desirable seedlings as having a shoot length of 30 to 36 in. and a root-collar diameter of 1/4 to 3/8 in. or larger (Kennedy 1981a, McKnight and Johnson 1980). However, definitive guidelines defining optimal seedling dimensions for bottomland hardwood species, particularly concerning factors that reduce survival and growth such as competing vegetation and flooding, have not been developed or published.

Research on quality seedling production in nursery beds has revealed that certain practices can improve outplanting performance, especially on harsh sites. In his review of existing literature, South (1998) concluded that proper top pruning of hardwood seedlings can significantly boost outplanting survival (range 3 to 42 percent). Top pruning may benefit the seedling by improving its root-weight ratio, while it also helps the planter because top-pruned seedlings are easier to handle. In addition to potential gains in survival, stimulated height growth of seedlings from moderate top pruning quickly compensates for the lost height from pruning (Adams 1985, Meadows and Toliver 1987). Moderate root pruning can also facilitate planting without significantly reducing survival or growth (Toliver and others 1980). Yet, root pruning should be approached cautiously because excessive pruning will negatively alter root-weight ratio and reduce carbohydrate reserves needed by the seedling to survive lifting and transplanting. Kennedy (1993) suggested that root systems of oak seedlings should be pruned to no shorter than 8 in.

In addition to seedling size and handling practices, morphological traits including the number of first-order lateral roots can have a profound effect on early survival and growth of hardwood seedlings (Thompson and Schultz 1995). Research conducted by Kormanic and colleagues provides clear evidence linking seedling out-planting performance and the inherited expression of first-order lateral root proliferation (Kormanik 1986, 1989; Kormanik and Ruehle 1987; Kormanik and others 1998; Thompson and Schultz 1995). Based on their observations, Clark and others (2000), Kormanik and Ruehle (1987), and Johnson (1984) suggested that fewer than 40 percent of oak seedlings lifted from nursery beds are suitable planting stock based on lateral root development. Although first-order lateral roots are strongly controlled by genetics, research has demonstrated that their development can be increased by growing seedlings at relatively low nursery bed densities (Dey and Buchanan 1995). Most planting operations in the LMRAV do not consider seedling morphology. Operational programs generally target a shoot length of 18 to 24 in. and a root-collar diameter of 3/8 in. as the minimal seedling size. Clearly, managers need empirical data defining optimal seedling dimensions and morphological traits to support efficient planting of a diverse array of bottomland hardwood species.

Few studies have examined the transfer of seed within the southern hardwood region, but available evidence has revealed provenance and family-within-provenance differences for survival and growth of common species, including cherrybark oak (*Q. falcata* var. *pagodifolia* Ell.), American sycamore, and eastern cottonwood (Greene and others 1991, Jokela and Mohn 1976, Land 1983). These studies suggest that survival and growth can be increased through provenance selections, but they also illustrate the hazards of indiscriminate seed transfer. For example, Dicke and Toliver (1987) observed a 30-percent range in survival within cherrybark oak families at age 5. Transfer of seed to different regions may be a concern, as well as establishing upland ecotypes on bottomland soils. For example, Keeley (1979) demonstrated that blackgum (*Nyssa sylvatica* Marsh.) ecotypes selected along a flooding gradient exhibited differing physiology, biomass accumulation patterns, and survival rates. On the other hand, short-term data presented by Yuceer and others (1998) revealed no distinct differences in survival or growth of upland versus bottomland sources of cherrybark oak. In practice, few foresters in the LMRAV specify seed source constraints in purchasing agreements. This lack of quality control or use of certified seed in afforestation projects could potentially reduce establishment success, productivity, and forest health. Ideally, foresters should avoid transfer of seed collected from other regions and site types until adequate protocols for seed transfer are established. Morgenstern (1996) provides conceptual details for establishing seed transfer protocols for forest tree species. Interestingly, most other developed countries and large companies have in place transfer protocols and seed-certification programs for the forests they manage.

Seed and Seedling Storage

Because seedlings are the predominant stock type currently used in afforestation projects of the LMRAV, the remainder of this manuscript will concentrate on practices and techniques appropriate for them. Bonner and Vozzo (1987), Bonner and others (1994), and Schopmeyer (1974) have thoroughly discussed suitable techniques for collecting and storing seed of bottomland hardwood species. Allen and Kennedy (1989) and Stanturf and others (1998) describe direct seeding techniques.

Bare-root seedlings should be lifted when dormant and directly transferred to storage under refrigeration at 34 to 38 ^{o}F. To maintain seedling viability in cold storage, seedling bags should receive ample ventilation and moisture-content monitoring. Mobile cold-storage facilities are readily available for lease and most large-scale contractors maintain on-site cold storage facilities during active planting. Such practices enable operators to maintain seedling dormancy and viability until time of planting.

Planting Seedlings

To establish reproduction on afforestation sites, contractors operating in the LMRAV utilize crews of hand and machine planters. Few studies have directly compared establishment success rates between hand planting and machine planting (Russell and others 1998). However, the authors have observed that either method can be sufficiently effective if experienced, conscientious personnel oversee the planting job. We discuss techniques, advantages, and disadvantages of each method below.

Hand planting—Hand-planting techniques originally employed to establish large-scale hardwood plantations were generally borrowed from conifer plantations. These practices were generally not applicable to hardwood plantation establishment in the LMRAV because of the relatively large root system characteristic of most hardwood seedlings and the often saturated, heavy clay alluvial soils. Hardwood seedlings should be planted such that the apparent root collar is at least 1 to 2 in. below the soil surface. This practice helps ensure that all lateral roots are sufficiently covered, and it can improve the sprouting potential of seedlings, primarily oaks, that herbivores have clipped. Hardwood seedlings typically have the taproot pruned to about 8 in. and the laterals to 4 to 6 in. A planting tool is needed with a blade at least 10- to 12-in. long by 6- to 8-in. wide. The type of dibble or planting shovel varies among contractors, and often the same tool will not work well on all sites due to soil and/or moisture conditions. Because of the time and care required to plant large seedlings properly in saturated soil, some contractors pay their planters by the hour rather than by the number of seedlings planted. A hand-planting crew of 20 people can usually plant over 130 ac per day (about 2,000 seedlings per planter at 300 seedlings per acre). This rate is quicker than a machine-planting crew with one tractor. However, because hand planting is labor intensive and requires more administrative supervision and logistical planning to keep planting crews active, it can be more expensive than machine planting.

Machine planting—Machine planters for hardwoods are largely similar to conifer planters, with modified packing wheels and coulters to allow for planting of larger seedlings. In addition, most operators modify stock planters to accommodate their specific planting needs. Planting machines are normally pulled by four-wheel drive, rubber-tired tractors with a minimum rating of 175 horsepower. If soil conditions are favorable, machine planting can be more consistent for large seedlings with well-developed root systems, and machine planting is generally not as expensive as hand planting based on cost per seedling. A single-machine planter crew can plant about 15 to 20 ac per day if soil conditions are ideal. However, water saturated, heavy clay soil typical of some alluvial floodplain sites can hamper progress of machine planting, and the heavy equipment required for machine planting can damage afforestation sites by creating ruts. Furthermore, if soil conditions are not ideal, the slit created by the planting machine is often never closed near the lower reaches of the foot or coulter blade and may serve as a site of cracking under dry conditions in the smectitic soils of the LMRAV. Machine planting also increases the minimal distance between rows, and may damage growing stock if seedlings are being planted supplemental to partial failures or volunteer regeneration.

Planting Job Inspections

Ongoing inspection is necessary to ensure proper seedling handling, planting, and spacing. Viability problems from improper handling or storage are near impossible to detect after planting. Inspections also allow for real-time correction of planting and spacing mistakes. Walk-through inspections enable the forester to verify seedling condition and appropriate root pruning. Establishing fixed-radius plots behind the planting crew, the forester can monitor planting density, seedling size, planting depth, and general quality. Some choose to routinely sample one 0.02-ac plot for every 10 ac planted. However, seedling spacing should be considered when determining the size of fixed-radius plots, while sampling intensity will depend on the project area, site heterogeneity, and the consistency of the planting crew.

Postplanting Cultural and Protection Practices

Postplanting cultural and protection practices improve seedling survival, early growth, and plantation integrity. Postplanting cultural practices primarily target competition control as a means of boosting survival and improving seedling growth, but irrigation and fertilization practices may increase as future demands for hardwood fiber increase (Francis 1985; Houston and Buckner 1989; Kennedy 1981a, 1981b, 1993; Schweitzer and others 1999; Yeiser 1999). In spite of the demonstrated biological benefits, cost-benefit analyses of postplanting operations have not been conducted to project their financial benefits. However, the additional costs of the practices may be justified if they prevent plantation failure during drought or herbivore damage, or if they significantly decrease rotation length as in the case of disking operations in short-rotation woody crops. In practice, few afforestation foresters prescribe postplanting cultural treatments unless fiber or timber production is a primary management objective.

Competition control in hardwood plantations can be accomplished with mechanical or chemical methods, or with using mulch material. Mechanical methods of competition control primarily include mowing and disking. Because mowing does not reduce belowground competition for soil water and nutrients, hardwood reproduction generally does not respond (Houston and Buckner 1989, Kennedy 1981b, Schweitzer and others 1999). Mowing may only be practical where the forester wishes to slow down development of invasive woody species.

Although both share similar costs, disking is generally more effective than mowing for controlling competing vegetation. Several bottomland species including sweet pecan, Nuttall oak, green ash, American sycamore, eastern cottonwood, and sweetgum (*Liquidambar styraciflua* L.) respond favorably to disking (Houston and Buckner 1989, Kennedy 1981b, Schweitzer and Stanturf 1999). In addition to increasing aeration and moisture infiltration into soil, disking improves nutrient status and subsequently growth of hardwood reproduction (Kennedy 1981b). Gains in survival and growth derived from disking often come from early stand development, e.g., quicker advancement to canopy closure and self-pruning. However, excessive disking or disking too deep can prune roots excessively and reduce tree growth (Schweitzer and Stanturf 1999).

Recent research on plantation establishment has identified several herbicide tank mixes suitable for use with bottomland hardwood species (Ezell 1999, Ezell and Catchot 1998, Ezell and others 1999, Russell and others 1998). Vegetation control with herbicides can effectively increase growth of bottomland hardwood seedlings (Miller 1993, Russell and others 1998) and may provide the most cost-effective control of competing vegetation in relatively large, hardwood plantations. However, most tank mixes are best suited for controlling grass and some broadleaf herbaceous species, and chemical technology is not available for woody vines, shrubs, or trees in established plantations. Chemical control of undesirable woody species can only be attained with directed applications of suitable herbicides with appropriate measures taken to minimize spray drift and contact with crop species (Leininger and McCasland 1998, Miller 1993). Sites occupied by resilient vine species, such as ladies'-eardrops (*Brunnichia cirrhosa* Banks), trumpet creeper [*Campsis radicans* (L.) Seemann], and peppervine [*Ampelopsis arborea* (L.) Koehne], may require 2 or more years of treatment before afforestation.

Mulching is generally more expensive and more cumbersome than other methods of vegetation control, but it can provide long-term efficacy resulting in dramatic gains in survival and growth during the initial stages of stand development (Adams 1997, Windell and Haywood 1996). Limited research has demonstrated promising gains in early growth for mulched common persimmon, green ash, Nuttall oak, cherrybark oak, and water oak (Adams 1997, Schweitzer and others 1999). Various organic and synthetic mulch materials are commercially available, but a manager should consider ease of application, durability of the material, maintenance requirements, effectiveness, and cost (Haywood 1999, Windell and Haywood 1996). Mulch use may increase on wetland sites not amenable to mechanical or chemical control.

In addition to improving early survival and growth of seedlings, control of herbaceous vegetation can reduce herbivory by modifying herbivore use of old field habitats (Paul B. Hamel. 1995. Files/Sharkey/mammals. On file with: U.S. Department of Agriculture, Forest Service, Southern Research Station, Southern Hardwoods Laboratory, P.O. Box 227, Stoneville, MS 38776). White-tailed deer (*Odocoileus virginianus* Zimmerman), rodents (including *Sigmodon hispidus* Say and Ord), rabbits (*Sylvilagus* spp.), beaver (*Castor canadensis* Kuhl), and nutria (*Myocastor coypus* Molina) can be primary damaging agents in bottomland hardwood plantations (Burkett and Williams 1998, Conner and Toliver 1990, Conner and others 1999, King and Keeland 1999, McKnight 1970). Animal damage can range from mild, with little effect on planted seedlings, to severe, in which high densities of herbivores decimate young tree plantations (Conner and Toliver 1990). Aside from modification of habitat, which is effective on rodents, seedling protection or herbivore eradication practices may discourage herbivory.

Shelters can increase seedling survival where herbivory limits establishment (Conner and others 1999, Graveline and others 1998, Strange and Shea 1998). Several different styles of seedling shelters are available commercially, and selection of style and size will depend on the size of seedlings, expected herbivory type, costs, and assembly and installation requirements (Windell 1991). Some tree shelters also provide a favorable microclimate for improved early tree growth (Schweitzer and others 1999, Tuley 1985). Shelters can facilitate growth by moderating the light environment, reducing seedling transpiration rates, increasing temperature, and increasing carbon dioxide (Tuley 1985, Windell 1991). However, early gains in height growth are often due to temporary shifts in biomass accumulation and are not always maintained after seedlings grow above the shelters (Clatterbuck 1999, Mullins and others 1998). Besides their high costs, shelters are easily knocked down or swept away by floodwaters. These drawbacks limit the use of shelters to sites of severe herbivory. Perpetual eradication practices may most effectively curtail severe herbivory by beaver and nutria.

Other protection in established plantations involves control of insect or disease pests, fire prevention and suppression, and floodwater management. Insects and diseases can reduce plantation health and can render planted stock vulnerable to other stress. For example, young plantations of eastern cottonwood cultured for rapid biomass production may require control of several pests including the cottonwood leaf beetle (*Chrysomela scripta* Fabricius) and the cottonwood borer [*Plectrodera scalator* (Fabricius)] (Solomon 1985). Preventative practices such as selection of resistant seed sources or clones may reduce damage by insects and disease (Cooper and others 1977, Kellison 1994, Nebeker and others 1985); direct cultural, chemical, or microbial techniques may also eradicate pests (Solomon 1985, Solomon and others 1997). In several useful handbooks, Solomon and his colleagues describe major insect pests and diseases of common bottomland tree species including cottonwood, green ash, sycamore, and the oak species (Leininger and others 1999; Morris and others 1975; Solomon 1995; Solomon and others 1993, 1997).

Wildfire can destroy young hardwood plantations and reduce stem quality on stump spouts. As a precautionary measure against wildfire, Kennedy (1993) suggested maintenance of fire lanes around all plantations. If fire sweeps though a hardwood plantation, a site inventory must determine the extent of damage and the necessary management.

Although most bottomland hardwood species exhibit some level of tolerance to anaerobic soil, long-term inundation during the growing season can harm all but the most flood-tolerant species (Baker 1977, Hook 1984). Monitoring and control of floodwater depth and duration are necessary if survival of young hardwood seedlings is at stake. Where flooding is desirable for waterfowl habitat, floodwater removal before the active growing season will usually reduce stress on seedlings. Additionally, by increasing soil moisture availability during the potentially dry summer months, well-managed impoundments may improve seedling survival or growth (Broadfoot 1967).

Postplanting Survival and Growth Monitoring

Comparing seedling survival and growth to the *A Priori* definition of success can determine success of the planting effort. The landowner's management objectives, the type of plantation, e.g., pure versus mixed species, availability of preexisting data, and the costs of acquiring new data will help determine sampling interval, timing, and measurement intervals (Curtis 1983). However, prior to postplanting assessments, baseline information on plantation establishment will be vitally important to the afforestation forester. Information such as seed source, seedling size and condition, seedling lifting, shipment and storage history, soil and atmospheric conditions during planting, planting methods, planting contractor, site preparation activities, and planting date can identify the source of problems or successes. Postplanting assessment and monitoring techniques vary widely among landowners and public agencies, but they may often include sample transects, permanent sample plots, photodocumentation, and periodic aerial photography.

PLANTATION DESIGN

Hardwood plantations on former agricultural fields in the LMRAV range from single-species to mixed-species plantings. The afforestation forester should select a particular plantation type based on the desired outputs defined by management objectives (Daniel and others 1979). Single-species plantations, or monocultures, are often the most efficient plantation type for optimizing a single output, e.g., fiber production or soil amelioration. Single-species stands allow efficient cultural practices, more predictable stand-development patterns, and more predictable yields (Smith 1986). In the LMRAV, the native soft broadleaf species that exhibit indeterminate growth patterns are well suited for single-species stands. Perhaps eastern cottonwood plantations, cultivated for high-quality printing fiber, are the most extensive single-species plantations in the LMRAV (Krinard and Johnson 1980). In recent years, scientists in other regions have demonstrated the value of fast-growing, single-species plantations as catalysts for rehabilitating degraded forest ecosystems (Parrotta and others 1997). In this role, rapidly grown above- and belowground biomass stabilize soil, increase soil organic matter, nutrient or water-holding capacity, develop an understory microclimate that promotes establishment of native species, and develop habitat for native fauna (Fisher 1995; Lugo 1997; Mapa 1995; Parrotta 1992, 1999). Single-species plantations often do not produce high-quality sawtimber because most valuable species such as the oaks generally develop their highest vigor and quality in stands providing interspecific competition (Lockhart and Hodges 1998). Some managers may assume that single-species stands provide poor wildlife habitat, but homogeneous stands of eastern cottonwood, black willow, sandbar willow (*S. exigua* Nutt.), and baldcypress [*Taxodium distichum* (L.) Rich] occurring naturally along the Mississippi River contribute to landscape diversity and provide critical habitat for various wildlife species.

Mixed-species plantations can include various arrangements of multiple species in true mixtures or intercropping mixtures (Goelz 1995a). Potential benefits of mixed-species stands versus single-species stands can increase pest resistance, productivity in a vertically stratified stand, product diversity, crop tree quality, and canopy species diversity (Goelz 1995b, Smith 1986). True mixtures generally consist of randomly or systematically assigned species combinations established at the same time. Some mixed plantations are established with species of similar growth rates and developmental patterns (Goelz 1995a), but most successful mixtures require species that will stratify within the forest canopy (Smith 1986). Stressing these points for bottomland hardwood plantations, Lockhart and Hodges (1998) cited work on mixed-species stand development by Clatterbuck and Hodges (1988) and Clatterbuck and others (1987). Lockhart and others (1999) also indicated that stand development processes in well-designed species mixtures will be similar to developmental tracts observed in natural patterns. Most current afforestation practices under governmental cost-share programs attempt to establish true species mixtures to provide stand-level species diversity. Unfortunately, many plantations are established without consideration for the developmental trajectories and competitive interactions of individual species comprising the mixed plantation (Lockhart and Hodges 1998).

Establishing species that exhibit very different growth rates can create intercropping mixtures. Such mixtures may provide different products such as a commercial timber species intercropped with a nitrogen-fixing species (Goelz 1995a). In the LMRAV, scientists and land managers have developed an intercropping scheme using the early-successional eastern cottonwood as a nurse species for the slower growing, disturbance-dependent Nuttall oak (Schweitzer and others 1997, Twedt and Portwood 1997). Its very fast early growth, sparse crown architecture, and its suitability to intensive culture make eastern cottonwood a viable candidate as a nurse species. Potential benefits of the eastern cottonwood-Nuttall oak intercropping could include rapid rehabilitation of soil quality, rapid development of vertical structure for faunal habitat, early financial return on the rehabilitation investment, and development of an understory favorable for oak seedlings and other native woody species. Intercropping systems show potential for providing multiple ecological and landowner benefits in the LMRAV. Future research scheduled by the lead author and cooperators will examine development of other intercropping systems to extend application on a variety of bottomland site types, e.g., use of black willow as a nurse for other species on hydric sites.

MANAGEMENT CONSIDERATIONS

The LMRAV is currently experiencing extensive afforestation of former agricultural fields on sites that historically supported bottomland hardwood forests. Projections indicate that the current pace may be maintained through the next decade, resulting in hundreds of thousands of acres in bottomland hardwood plantations. In summarizing our review of literature, techniques, and practices, it became apparent that several fundamental components of afforestation were generally lacking in most regeneration practices currently performed in the LMRAV. Developing some of these missing components will require additional research, but others will require only an extension of current knowledge or application of conservation principles. Four fundamentally vital components should be more deeply incorporated into 21st century, state-of-the-art afforestation activities in the LMRAV:

(1) definition of specific landowner management objectives,
(2) establishment of stock size and quality guidelines,
(3) development of protocols for transfer of genetic material, and
(4) application of silvicultural and ecological principles in plantation establishment.

Incorporating these basic components will enable landowners, natural resource managers, and the general public a method of evaluating success of these afforestation activities and should improve afforestation efficiency, ecosystem health, and resource sustainability in the LMRAV.

LITERATURE CITED

Adams, J.C. 1985. Severe top pruning improves water oak seedling growth. In: Shoulders, Eugene, ed. Proceedings of the third biennial southern silvicultural research conference; 1984 November 7–8; Atlanta. Gen. Tech. Rep. SO–54. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 1–3.

Adams, J.C. 1997. Mulching improves early growth of four oak species in plantation establishment. Southern Journal of Applied Forestry. 21: 44–46.

Allen, J.A. 1997. Reforestation of bottomland hardwoods and the issue of woody species diversity. Restoration Ecology. 5: 125–134.

Allen, J.A.; Kennedy, H.E., Jr. 1989. Bottomland hardwood reforestation in the Lower Mississippi Valley. Slidell, LA: U.S. Department of the Interior, Fish and Wildlife Service, National Wetlands Research Center; Stoneville, MS: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 28 p.

Baker, J.B. 1977. Tolerance of planted hardwoods to spring flooding. Southern Journal of Applied Forestry. 1: 23–25.

Baker, J.B.; Blackmon, B.G. 1978. Summer fallowing - a simple technique for improving old-field sites for cottonwood. Res. Pap. SO–142. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 5 p.

Baker, J.B.; Broadfoot, W.M. 1979. A practical field method of site evaluation for commercially important southern hardwoods. Gen. Tech. Rep. SO–26. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 51 p.

Bonner, F.T.; Vozzo, J.A. 1987. Seed biology and technology of *Quercus*. Gen. Tech. Rep. SO–66. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 21 p.

Bonner, F.T.; Vozzo, J.A.; Elam, W.W.; Land, S.B., Jr. 1994. Tree seed technology training course. Gen. Tech. Rep. SO–107. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 81 p.

Broadfoot, W.M. 1967. Shallow-water impoundments increases soil moisture and growth of hardwoods. Soil Science Society of America Proceedings. 31: 562–564.

Broadfoot, W.M. 1976. Hardwood suitability for and properties of important midsouth soils. Res. Pap. SO–127. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 84 p.

Burkett, V.; Williams, H. 1998. Effects of flooding regime, mycorrhizal inoculation and seedling treatment type on first-year survival of Nuttall oak (*Quercus nuttallii* Palmer). In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 289–294.

Carter, J.R., Jr. 1978. A floristic study of the Delta National Forest and adjacent areas. Mississippi State, MS: Mississippi State University. 79 p. M.S. thesis.

Clark, S.L.; Schlarbaum, S.E.; Kormanik, P.P. 2000. Visual grading and quality of 1-0 northern red oak seedlings. Southern Journal of Applied Forestry. 24: 93–97.

Clatterbuck, W.K. 1999. Effects of tree shelters on growth of hardwood seedlings after seven growing seasons. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 43–46.

Clatterbuck, W.K.; Hodges, J.D. 1988. Development of cherrybark oak and sweet gum in mixed, even-aged bottomland stands in central Mississippi, U.S.A. Canadian Journal of Forest Research. 18: 12–18.

Clatterbuck, W.K.; Oliver, C.D.; Burkhardt, E.C. 1987. The silvicultural potential of mixed stands of cherrybark oak and American sycamore: spacing is the key. Southern Journal of Applied Forestry. 11: 158–161.

Conner, W.H.; McLeod, K.W.; Inabinette, L.W. [and others]. 1999. Successful planting of tree seedlings in wet areas. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 201–204.

Conner, W.H.; Toliver, J.R. 1990. Observations on the regeneration of baldcypress (*Taxodium distichum* [L.] Rich.) in Louisiana swamps. Southern Journal of Applied Forestry. 14: 115–118.

Cooper, D.T.; Filer, T.H., Jr.; Wells, O.O. 1977. Geographic variation in disease susceptibility of American sycamore. Southern Journal of Applied Forestry. 1: 21–24.

Curtis, R.O. 1983. Procedures for establishment and maintaining permanent plots for silvicultural and yield research. Gen. Tech. Rep. PNW–155. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 56 p.

Daniel, T.W.; Helms, J.A.; Baker, F.S. 1979. Principles of silviculture. 2d ed. St. Louis: McGraw-Hill, Inc. 500 p.

Dey, D.; Buchanan, M. 1995. Red oak (*Quercus rubra* L.) acorn collection, nursery culture and direct seeding: a literature review. For. Res. Inf. Pap. 122. Sault Ste. Marie, Ontario: Ontario Ministry of Natural Resources, Ontario Forest Research Institute. 46 p.

Dicke, S.G.; Toliver, J.R. 1987. Response of cherrybark oak families to different soil-site conditions. In: Phillips, Douglas R., comp. Proceedings of the fourth biennial southern silvicultural research conference; 1986 November 4–6; Atlanta. Gen. Tech. Rep. SE–42. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 260–263.

Dickson, R.E. 1994. Height growth and episodic flushing in northern red oak. In: Isebrands, J.G.; Dickson, R.E., comps. Biology and silviculture of northern red oak in the north central region: a synopsis. Gen. Tech. Rep. NC–173. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station: 1–9.

Ezell, A.W. 1995. Importance of early season competition control in establishing eastern cottonwood (*Populus deltoides*) plantations. In: Edwards, M. Boyd, comp. Proceedings of the eighth biennial southern silvicultural research conference; 1994 November 1–3; Auburn, AL. Gen. Tech. Rep. SRS–1. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 94–97.

Ezell, A.W. 1999. Crop tolerance of Nuttall, water, and willow oaks to preemergent applications of DPX–6447: second year results. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 142–144.

Ezell, A.W.; Catchot, A.L., Jr. 1998. Competition control for hardwood plantation establishment. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 42–43.

Ezell, A.W.; Portwood, J.; Quicke, H. 1999. Pre- and postemergent applications of imazaquin for herbaceous weed control in eastern cottonwood plantations: second year results. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 73–75.

Fisher, R.F. 1995. Amelioration of degraded rain forest soils by plantations of native trees. Soil Science Society of America Journal. 59: 544–549.

Francis, J.K. 1985. Bottomland hardwood fertilization – the Stoneville experience. In: Shoulders, Eugene, ed. Proceedings of the third biennial southern silvicultural research conference; 1984 November 7–8; Atlanta. Gen. Tech. Rep. SO–54. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 346–350.

Goelz, J.C.G. 1995a. Establishment of mixed-species plantations of bottomland hardwoods. In: Landin, M.C., ed. National interagency workshop on wetlands: technology advances for wetlands science; 1995 April 3–7; New Orleans. Vicksburg, MS: U.S. Army Corps of Engineers, Waterways Experiment Station: 178–180.

Goelz, J.C.G. 1995b. Experimental designs for mixed-species plantations. In: Edwards, M. Boyd, comp. Proceedings of the eighth biennial southern silvicultural research conference; 1994 November 1–3; Auburn, AL. Gen. Tech. Rep. SRS–1. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 559–563.

Graveline, B.D.; Wells, G.R.; Schlarbaum, S.; Fribourg, H.A. 1998. Growth and protection of selected northern red oak seedlings planted on old field sites. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 257–262.

Greene, T.A.; Lowe, W.J.; Stine, M. 1991. Volume production of six cherrybark oak provenances in the western gulf region. In: Coleman, Sandra S.; Neary, Daniel G., comps., eds. Proceedings of the sixth biennial southern silvicultural research conference; 1990 October 30–November 1; Memphis, TN. Gen. Tech. Rep. SE–70. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 736–743.

Groninger, J.W.; Aust, W.M.; Miwa, M.; Stanturf, J.A. 1999. Tree species-soil relationships on marginal soybean lands in the Mississippi Delta. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 205–209.

Haines, T. 1995. Federal and State forestry cost-share assistance programs: structure, accomplishments, and future outlook. Res. Pap. SO–295. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 18 p.

Haywood, J.D. 1999. Durability of selected mulches, their ability to control weeds, and influence growth of loblolly pine seedlings. New Forests. 18: 263–276.

Hodges, J.D.; Gardiner, E.S. 1993. Ecology and physiology of oak regeneration. In: Loftis, David L.; McGee, Charles E., eds. Oak regeneration: serious problems, practical recommendations: Symposium proceedings; 1992 September 8–10; Knoxville, TN. Gen. Tech. Rep. SE–84. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 54–65.

Hook, D.D. 1984. Waterlogging tolerance of lowland tree species of the South. Southern Journal of Applied Forestry. 8: 136–149.

Houston, A.E.; Buckner, E.R. 1989. Cultural treatments for the establishment of bottomland hardwoods in west Tennessee. In: Miller, James H., comp. Proceedings of the fifth biennial southern silvicultural research conference; 1988 November 1–3; Gen. Tech. Rep. SO–74. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 285–288.

Johnson, P.S. 1984. Responses of planted northern red oak to three overstory treatments. Canadian Journal of Forest Research. 14: 536–542.

Johnson, R.L.; Krinard, R.M. 1985. Oak seeding on an adverse field site. Res. Note SO–319. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 4 p.

Jokela, J.J.; Mohn, C.A. 1976. Geographic variation in eastern cottonwood. In: Thielges, B.A.; Land, S. B., eds. Proceedings of the symposium on eastern cottonwood and related species; 1976 September 28–October 2; St. Paul, MN. Sci. J. Ser. Pap. 1638. St. Paul, MN: University of Minnesota Agricultural Experimental Station: 109–125.

Joslin, J.D.; Schoenholtz, S.H. 1998. Measuring the environmental effects of converting cropland to short-rotation woody crops: a research approach. Biomass and Bioenergy. 15: 301–311.

Keeley, J.E. 1979. Population differentiation along a flood frequency gradient: physiological adaptations to flooding in *Nyssa sylvatica*. Ecological Monographs. 49: 89–108.

Kellison, R.C. 1994. Genetic improvement of southern hardwoods - 1994 update. In: Landis, T.D.; Dumroese, R.K., tech. coords. National proceedings, forest and conservation nursery associations. Gen. Tech. Rep. RM–257. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 20–25.

Kellison, R.C.; Young, M.J.; Braham, R.R.; Jones, E.J. 1998. Major alluvial floodplains. In: Messina, M.G.; Conner, W.H., eds. Southern forested wetlands - ecology and management. Boca Raton, FL: CRC Press LLC: 291–341.

Kennedy, H.E., Jr. 1977. Planting depth and source affect survival of planted green ash cuttings. Res. Note SO–224. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 3 p.

Kennedy, H.E., Jr. 1981a. Bottomland hardwoods research on site preparation, plantation establishment, and cultural treatments, at the Southern Hardwoods Laboratory. In: Barnett, James P., ed. Proceedings of the first biennial southern silvicultural research conference; 1980 November 6–7; Atlanta. Gen. Tech. Rep. SO–34. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 75–78.

Kennedy, H.E., Jr. 1981b. Foliar nutrient concentrations and hardwood growth influenced by cultural treatments. Plant and Soil. 63: 307–316.

Kennedy, H.E., Jr. 1990. Hardwood reforestation in the South: landowners can benefit from Conservation Reserve Program incentives. Res. Note SO–364. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 6 p.

Kennedy, H.E., Jr. 1993. Artificial regeneration of bottomland hardwoods. In: Loftis, David L.; McGee, Charles E., eds. Oak regeneration: serious problems, practical recommendations. Symposium proceedings; 1992 September 8–10; Knoxville, TN. Gen. Tech. Rep. SE–84. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 241–249.

King, S.L.; Keeland, B.D. 1999. Evaluation of reforestation in the Lower Mississippi River Alluvial Valley. Restoration Ecology. 7: 348–359.

Kormanik, P.P. 1986. Lateral root morphology as an expression of sweetgum seedling quality. Forest Science. 32: 595–604.

Kormanik, P.P. 1989. Frequency distribution of first-order lateral roots in forest tree seedlings: silvicultural implications. In: Miller, James H., comp. Proceedings of the fifth biennial southern silvicultural research conference; 1988 November 1–3; Gen. Tech. Rep. SO–74. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 101–105.

Kormanik, P.P.; Ruehle, J.L. 1987. Lateral root development may define nursery seedling quality. In: Phillips, Douglas R., comp. Proceedings of the fourth biennial southern silvicultural research conference; 1986 November 4–6; Atlanta. Gen. Tech. Rep. SE–42. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 225–229.

Kormanik, P.P.; Sung, S.S.; Kass, D.J.; Schlarbaum, S. 1998. Effect of seedling size and first-order-lateral roots on early development of northern red oak on mesic sites. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 247–252.

Krinard, R.M.; Johnson, R.L. 1980. Fifteen years of cottonwood plantation growth and yield. Southern Journal of Applied Forestry. 4: 180–185.

Krinard, R.M.; Johnson, R.L. 1985. Eighteen-year development of sweetgum (*Liquidambar styraciflua* L.) plantings on two sites. Tree Planters' Notes. 36: 6–8.

Land, S.B., Jr. 1983. Performance and G-E interactions of sycamore established from cuttings and seedlings. In: Jones, Earle P., Jr., ed. Proceedings of the second biennial southern silvicultural research conference; 1982 November 4–5; Atlanta. Gen. Tech. Rep. SE–24. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 431–440.

Leininger, T.D.; McCasland, C.S. 1998. Does paraquat cause stem swellings in first-year cottonwood saplings? In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 44–48.

Leininger, T.D.; Solomon, J.D.; Wilson, A.D.; Schiff, N.M. 1999. A guide to major insects, diseases, air pollution injury, and chemical injury of sycamore. Gen. Tech. Rep. SRS–28. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 44 p.

Lockhart, B.R.; Ezell, A.W.; Hodges, J.D.; Clatterbuck, W.K. 1999. Development of mixed cherrybark oak-sweetgum plantations planted at different spacings in east-central Mississippi after 17 years. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 103–106.

Lockhart, B.R.; Hodges, J.D. 1998. Bottomland red oak stand development with implications for management and future research. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 548–554.

Long, T.J.; Jones, R.H. 1993. Biomass partitioning in seedlings of ten *Quercus* species native to the Southeastern United States. In: Brissette, John C., ed. Proceedings of the seventh biennial southern silvicultural research conference; 1992 November 17–19; Mobile, AL. Gen. Tech. Rep. SO–93. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 257–261.

Long, T.J.; Jones, R.H. 1996. Seedling growth strategies and seed size effects in fourteen oak species native to different soil moisture habitats. Trees. 11: 1–8.

Lugo, A.E. 1997. The apparent paradox of reestablishing species richness on degraded lands with tree monocultures. Forest Ecology and Management. 99: 9–19.

Mapa, R.B. 1995. Effect of reforestation using *Tectona grandis* on infiltration and soil water retention. Forest Ecology and Management. 77: 119–125.

McKnight, J.S. 1970. Planting cottonwood cuttings for timber production in the South. Res. Pap. SO–60. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 17 p.

McKnight, J.S.; Johnson, R.L. 1980. Hardwood management in southern bottomlands. Forest Farmer. 39: 30–39.

Meadows, J.S.; Nowacki, G.J. 1996. An old-growth definition for eastern riverfront forests. Gen. Tech. Rep. SRS–4. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 7 p.

Meadows, J.S.; Toliver, J.R. 1987. Three-year response of pecan to six methods of seedling establishment. Southern Journal of Applied Forestry. 11: 56–59.

Miller, J.H. 1993. Oak plantation establishment using mechanical burning, and herbicide treatments. In: Loftis, David L.; McGee, Charles. E., eds. Oak regeneration: serious problems, practical recommendations: Symposium proceedings; 1992 September 8–10; Knoxville, TN. Gen. Tech. Rep. SE–84. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 264–289.

Morgenstern, E.K. 1996. Geographic variation in forest trees: genetic basis and application of knowledge in silviculture. Vancouver, BC: UBC Press, University of British Columbia. 209 p.

Morris, R.C.; Filer, T.H.; Solomon, J.D. [and others]. 1975. Insects and diseases of cottonwood. Gen. Tech. Rep. SO–8. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Research Station. 37 p.

Mullins, J.A.; Buckner, E.R.; Hopper, G.; Percell, G.G. 1998. Early growth of sheltered and unsheltered cherrybark oak established by planting 1-0 bareroot and 1-0 containerized seedlings, and by direct-seeding. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 234–237.

Nebeker, T.E.; Schmitt, J.J.; Solomon, J.D.; Honea, C.R. 1985. Clonal resistance to and incidence of the poplar borer in southern cottonwood plantations. In: Shoulders, Eugene, ed. Proceedings of the third biennial southern silvicultural research conference; 1984 November 7–8; Atlanta. Gen. Tech. Rep. SO–54. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 247–251.

Parrotta, J.A. 1992. The role of plantation forests in rehabilitating degraded tropical ecosystems. Agriculture, Ecosystems and Environment. 41: 115–133.

Parrotta, J.A. 1999. Productivity, nutrient cycling, and succession in single- and mixed-species plantations of *Casuarina equisetifolia*, *Eucalyptus robusta*, and *Leucaena leucocephala* in Puerto Rico. Forest Ecology and Management. 124: 45–77.

Parrotta, J.A.; Turnbull, J.W.; Jones, N. 1997. Catalyzing native forest regeneration on degraded tropical lands. Forest Ecology and Management. 99: 1–7.

Putnam, J.A.; Furnival, G.M.; McKnight, J.S. 1960. Management and inventory of southern hardwoods. Agric. Handb. 181. Washington, DC: U.S. Department of Agriculture, Forest Service. 102 p.

Russell, D.R., Jr.; Hodges, J.D.; Ezell, A.W. 1998. An evaluation of hardwood reforestation methods on previously farmed lands in central Alabama. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 272–276.

Schopmeyer, C.S. 1974. Seeds of woody plants in the United States. Agric. Handb. 450. Washington, DC. U.S. Department of Agriculture, Forest Service. 883 p.

Schweitzer, C.J.; Gardiner, E.S.; Stanturf, J.A.; Ezell, A.W. 1999. Methods to improve establishment and growth of bottomland hardwood artificial regeneration. In: Stringer, Jeffrey W.; Loftis, David L., eds. Proceedings, 12th central hardwood forest conference; 1999 February 28, March 1–2; Lexington, KY. Gen. Tech. Rep. SRS–24. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 209–214.

Schweitzer, C.J.; Stanturf, J.A. 1999. A comparison of large-scale reforestation techniques commonly used on abandoned fields in the Lower Mississippi Alluvial Valley. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 136–141.

Schweitzer, C.J.; Stanturf, J.A.; Shepard, J.P. [and others]. 1997. Large-scale comparison of reforestation techniques commonly used in the Lower Mississippi Alluvial Valley: first year results. In: Pallardy, S.G; Cecich, R.A.; Garrett, H.G.; Johnson, P.S., eds. Proceedings, 11th central hardwood forest conference; 1997 March 23–26; Columbia, MO. Gen. Tech. Rep. NC–188. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station: 313–320.

Solomon, J.D. 1985. Impact of insects on growth and development of young cottonwood plantations. Res. Pap. SO–217. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 6 p.

Solomon, J.D. 1995. Guide to insect borers of North American broadleaf trees and shrubs. Agric. Handb. 706. Washington, DC: U.S. Department of Agriculture, Forest Service. 735 p.

Solomon, J.D.; Leininger, T.D.; Wilson, A.D. [and others]. 1993. Ash pests - a guide to major insects, diseases, air pollution injury, and chemical injury. Gen. Tech. Rep. SO–96. New Orleans: U.S. Department of Agriculture, Forest, Service, Southern Forest Experiment Station. 45 p.

Solomon, J.D.; McCracken, F.I.; Anderson, R.L. [and others]. 1997. Oak pests - a guide to major insects, diseases, air pollution and chemical injury. Prot. Rep. R8–PR7. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region. 69 p.

South, D.B. 1998. Effects of top-pruning on survival of southern pines and hardwoods. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 3–8.

Smith, D.M. 1986. The practice of silviculture. 8th ed. New York: John Wiley. 527 p.

Stanturf, J.A.; Schoenholtz, S.H. 1998. Soils and landforms. In: Messina, M.G.; Conner, W.H., eds. Southern forested wetlands - ecology and management. Boca Raton, FL: CRC Press LLC: 123–147.

Stanturf, J.A.; Schoenholtz, S.H.; Schweitzer, C.J.; Shepard, J.P. 2001. Achieving restoration success: myths in bottomland hardwood forests. Restoration Ecology. 9(2): 189–200.

Stanturf, J.A.; Schweitzer, C.J.; Gardiner, E.S. 1998. Afforestation of marginal agricultural land in the Lower Mississippi River Alluvial Valley, U.S.A. Silva Fennica. 32: 281–297.

Stine, M.; Chambers, J.L.; Wilson, M.; Ribbeck, K. 1995. Twenty-year survival and growth of six bottomland hardwood species. In: Edwards, M. Boyd, comp. Proceedings of the eighth biennial southern silvicultural research conference; 1994 November 1–3; Auburn, AL. Gen. Tech. Rep. SRS–1. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 500–502.

Strange, E.E.; Shea, K.L. 1998. Effects of deer browsing, fabric mats, and tree shelters on *Quercus rubra* seedlings. Restoration Ecology. 6: 29–34.

Tanner, J.T. 1986. Distribution of tree species in Louisiana bottomland forests. Castanea. 51: 168–174.

Thompson, J.R.; Schultz, R.C. 1995. Root system morphology of *Quercus rubra* L. planting stock and 3-year field performance in Iowa. New Forests. 9: 225–236.

Thornton, F.C.; Joslin, J.D.; Bock, B.R. [and others]. 1998. Environmental effects of growing woody crops on agricultural land: first year effects on erosion, and water quality. Biomass and Bioenergy. 15: 57–69.

Toliver, J.R.; Sparks, R.C.; Hansbrough, T. 1980. Effects of top and lateral root pruning on survival and early growth - three bottomland hardwood tree species. Tree Planters' Notes. 31: 13–15.

Tulley, G. 1985. The growth of young oak trees in shelters. Forestry. 58: 181–195.

Twedt, D.J.; Portwood, J. 1997. Bottomland hardwood reforestation for neotropical migratory birds: are we missing the forest for the trees? Wildlife Society Bulletin. 25: 647–652.

Williams, H.M.; Kleiss, B.A.; Humphrey, M.N.; Klimas, C.V. 1993. First-year field performance of oak species with varying flood tolerance planted on hydric and non-hydric soils. In: Brissette, John C., ed. Proceedings of the seventh biennial southern silvicultural research conference; 1992 November 17–19; Mobile, AL. Gen. Tech. Rep. SO–93. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 409–414.

Wilson, R.R.; Oliver, J.M.; Twedt, D.J.; Uihlein, W.B., III. [In press]. Bottomland hardwood restoration in the Mississippi Alluvial Valley: looking past the trees to see the forest. In: Fredrickson, L., ed. Ecology and management of bottomland hardwood systems symposium; 1999 March 11–13; Memphis, TN. Columbia, MO: University of Missouri.

Windell, K. 1991. Tree shelters for seedling protection. Tech. Rep. 9124–2834– MTDC. Missoula, MT: U.S. Department of Agriculture, Forest Service, Technology and Development Program. 142 p.

Windell, K.; Haywood, J.D. 1996. Mulch mat materials for improved tree establishment. Tech. Rep. 9624–2811–MTDC. Missoula, MT: U.S. Department of Agriculture, Forest Service, Technology and Development Center. 124 p.

Yeiser, J.L. 1999. Irrigated and unirrigated eastern cottonwood and water oak in a short rotation fiber system on a former agricultural site. In: Haywood, James D., ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 76–79.

Yuceer, M.C.; Hodges, J.D.; Land, S.B., Jr.; Friend, A.L. 1998. Upland vs. bottomland seed sources of cherrybark oak. In: Waldrop, Thomas A., ed. Proceedings of the ninth biennial southern silvicultural research conference; 1997 February 25–27; Clemson, SC. Gen. Tech. Rep. SRS–20. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 215–221.

WETLAND EDUCATION THROUGH COOPERATIVE PROGRAMS BETWEEN COASTAL CAROLINA UNIVERSITY AND HORRY COUNTY PUBLIC SCHOOLS

Sharon L. Gilman[1]

Horry County, in the Coastal Plain of South Carolina, is approximately 50 percent wetlands. The Waccamaw Region (Horry, Georgetown, and Williamsburg Counties) has experienced a 58-percent population increase during the period from 1960 to 1990. Population growth trends suggest that from 1990 to 2020, the total daily population will increase by 125 percent, representing an additional 510,000 people (Horry County Comprehensive Plan, 1998). The impact of this on the remaining wetlands will be substantial. In addition to this problem, S.A.T. scores of South Carolina students, whereas certainly not the only measure of academic achievement, do consistently rank below national averages (South Carolina Dept. of Educ., 2000). For these reasons, it is necessary to create new enthusiasm for science education in general and for the value of wetlands in particular.

In an effort to address this issue, Coastal Carolina University (CCU) has initiated three programs. In 1998, the School of Science received an award for Integration of Research and Education from the National Science Foundation (NSF). This has provided funding for two of these programs. The first is a cooperative effort with the Playcard Environmental Education Center (PEEC), a local nonprofit organization run mostly by Horry County School District and located along a blackwater swamp. The grant has provided funding for me to work with science education students to assist with educational programs and conduct wetland research. It also funds minigrants for local high school teachers to bring their classes to PEEC for research.

The second NSF-funded program is The River Project, operated by CCU's Environmental Quality Lab in the Center for Marine and Wetland Studies. All of our local high schools are within a 5-minute drive of a river, and, in this program, they monitor water quality with the assistance of CCU faculty and students. They report their data on a university Web site and discuss their findings at an annual student congress. Funding from Wal-Mart's Clean Air/Clean Water Program also supports this program. The Web address is: http://www.coastal.edu/nsf-aire/river.htm.

Finally, with assistance from the NSF-sponsored State Systemic Initiative, CCU offered a graduate course last summer entitled "Wetland Ecology for Teachers". This was an activity oriented course designed to give teachers the knowledge to incorporate wetland education into their classes in a way that meets the new South Carolina State Science Education Standards. We also took field trips to local wetlands, including PEEC, in order to show teachers how to take advantage of these environments to make science more alive and interesting for their students. Mostly middle and high school teachers attended the course. I am currently researching the extent to which the teachers were able to incorporate wetlands into their curriculum this year. The Web address for this course is: http://www.coastal.edu/science/biology/sgilman/bio778.htm.

ACKNOWLEGMENTS

Thanks to the authors of the NSF-AIRE Grant, Drs. John Idoux and Val Dunham, the Directors of CCU's Environmental Quality Lab, Drs. Susan Libes and Kim Weaver, Student Coordinator of The River Project, Mr. Matthew Cline, and the Director of the Waccamaw Math and Science Hub, Ms. Pam Barnhill.

[1] Instructor of Biology, Department of Biology, Coastal Carolina University, Conway, SC 29528.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

WETLANDS AND AGRICULTURE: ARE WE HEADING FOR CONFRONTATION OR CONSERVATION?

Brij Gopal[1]

Abstract—Wetlands and agriculture are closely linked. Historically, agriculture had its beginning in riparian wetland habitats and expanded into other wetlands. Later, large areas of riverine, palustrine, and coastal wetlands were converted into paddy fields or drained for agriculture. Agriculture has grown most at the expense of natural wetlands. Today, the intensive agriculture depends upon heavy inputs of water (irrigation) and agrochemicals. Thus, agriculture now threatens the remaining wetlands through alteration of hydrological regimes, siltation, and pollutants. Conservation of wetlands requires an integrated, balanced, and coordinated approach to the management of water resources whereby the impacts of agriculture on wetlands are minimized without compromising agricultural production.

INTRODUCTION

Wetlands and agriculture are closely linked together, dating back to the prehistoric period. Both wetlands and agriculture have greatly influenced humankind. Available evidence suggests that human settlements started in and around the wetlands (Williams 1990). Long before humans learned to grow food, they depended, at least partly, on wetlands for their sustenance. Agriculture had its beginning in the wetlands and grew at the expense of wetlands (and forests). Today, about 12 percent of the earth's land area is under agriculture. Wetlands occupy < 6 percent of the land area. Of this, rice culture accounts for about 15 percent and provides staple food for about 40 percent of the world's human population (Hook 1993).

The concern for wetlands is rather recent—only about three decades old. Until recently, wetlands were treated with contempt as wastelands, worthy of drainage and reclamation for agriculture and other land uses. Now that many functions and values of wetlands have been recognized (Mitsch and Gosselink 1993), efforts are made to conserve and restore them. Wetlands, which have been the victim of agriculture, are also projected today as protectors of water resources against the impacts of agriculture. In this paper, I try to trace the linkages between wetlands and agriculture through the millennia and show how agriculture continues to impact upon the wetlands. I conclude with a call for an appropriate policy to strike a balance between agriculture and wetland conservation.

AGRICULTURE IN WETLANDS

Agriculture is deeply rooted in wetlands. Since prehistoric times, agriculture has been practiced in and sustained by wetlands. Agriculture is known to have started in the Middle East where seasonally flooded wetlands—the riverine floodplains—provided the ideal environment. The crops required no subsidy of energy and nutrients. Seeds were sown after the floods receded exposing the wet soils enriched with fresh sediments, and the decaying terrestrial vegetation provided the nutrients. Such low-subsidy agriculture in floodplains continues today in many parts of the world. In India, the local communities raise a variety of crops, such as cucurbits, chillies, tomatoes, etc., not only in the riparian fringes, but even on the riverbed. The seasonal wetlands throughout the arid and semi-arid regions of the world are widely used for raising a variety of crops during the low water period; for example, dambos are valuable habitats for agriculture and grazing in Zimbabwe (Scoones and Cousins 1994). Such agriculture relies heavily on the timing, frequency, and intensity of flood events.

As the human societies depended increasingly more on agriculture and developed settlements around wetlands, agriculture was diversified greatly to exploit many kinds of wetlands. Humans also recognized the food value of many of the wetland plants, which were cultivated in the wetlands by excluding other competitors. Rice (including deep-water rice) and taro are well-known examples. *Trapa bispinosa, Nelumbo nucifera, Euryale ferox*, several sedges (*Cyperus esculentus*) and grasses (*Panicum, Echinochloa*, etc.) are cultivated extensively in seasonal wetlands of South and Southeast Asia for food.

At the same time, in several parts of the world, agriculture expanded further into the waterbodies. Deep-water rice, for example, is grown in India, Bangladesh, Myanmar, Thailand, Vietnam, Mali, and Niger in waters where depths may exceed 6 m (Vergara 1992). In shallow marshes and littorals of large lakes, rapid vegetative growth of certain macrophytes, such as *Vossia, Cyperus papyrus, Salvinia* spp., *Eichhornia crassipes*, *Echinochloa, Paspalum*, etc., produces huge biomass, which decays very slowly resulting in the accumulation of thick layers of detritus mixed with mineral matter. This decaying mass of organic matter becomes afloat with the rise in water level following the rains. These floating islands also become colonized with a variety of terrestrial and wetland plants and often grow in size with further accumulation of organic and mineral matter. Floating islands occur widely throughout the tropics and are known

[1] Associate Professor, School of Environmental Sciences, Jawaharlal Nehru University, New Delhi 110067, India.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

variously as camalotes in Brazil, embalsados in Argentina, Batumes in Paraguay, sudds in Africa, and phumdis in eastern India (John 1986, Por 1995, Tombi Singh and Singh 1995). They are known also from outside the tropics; e.g., in the Mississippi River Delta in Louisiana (Sasser and others 1995) and in Lake Posta Fibreno in Italy (V. Chet, personal communication). Probably taking cue from these natural floating islands, and apparently as a response to the uncertainty of availability of water or to the threats from flood, and recognizing the freedom of moving them around at their will, humans started cultivating upon these floating islands. The chinampas (floating gardens) in Mexico are a well-known example (Coe 1964, West and Armillas 1950). In China, rice cultivation on artificial floating islands in lakes and rivers is known to have been practiced for more than 1,600 years (You Xiuling 1991). Some of the floating islands on the Yangtze River are reported to have been so large that several families lived on them together with their pets. The practice of creating floating islands from lake mud, decaying organic matter, and macrophytes and cultivating vegetables on them continues even today in several parts of South and Southeast Asia. Elaborate descriptions of these floating islands are available for Malaysia (Department of Agriculture 1939), New Guinea (Serpenti 1965), Myanmar (Annandale 1918), and Kashmir in India (Kaul and Zutshi 1966, Moorcroft and Trebeck 1841, Sahni 1927, Sturtevant 1970).

CONVERSION OF WETLANDS FOR AGRICULTURE

The most common form of agriculture in wetlands is, however, paddy cultivation. Evidence of rice culture dates back to the earliest age of humans (Hook 1993). Domestication of rice started in shallow swamps (Chang 1976, Harlan 1977), and probably independently in China, Thailand, and India (Gorman 1977, Ho 1977, Vishnu-Mittre 1977). With the growing demand for food, seasonal marshes throughout South and Southeast Asia and in China were modified into paddy fields as man-managed wetlands. It was only after the European colonization in the 18th and 19th centuries that the coastal swamps and mangroves were rapidly converted to paddy fields (Richards 1990). The conversion of extensive mangroves (Sunderbans) in the Ganga-Brahmaputra Delta started at the end of the 18th century by then the East India Company, which required the private landowners to clear and reclaim Sunderban forests and swamps for rice cultivation. By the 1870s, 2790 km^2 of mangroves had been converted to rice fields. The pace of conversion got further accelerated, and though enormous deposits of sediments transported by the two rivers had resulted in the expansion of Sunderbans, 2,750 km^2 of Sunderbans were reclaimed between 1880 and 1940 and another 5,230 km^2 in the next 40 years (Richards 1990). Similarly, the conversion of mangrove swamps in the Irrawady River Delta started after the British began colonizing Burma in 1852. During the period from 1880 to 1920, the area under rice grew from 3,979 km^2 to 12,059 km^2, and wetlands declined from 9,059 km^2 to 4,698 km^2. The story of conversion of wetlands including mangroves in the Chao Phraya Delta under the British colonial rule is also similar, though it required a great deal of water management and flow regulation. The independence of nations in the region in the middle of the 20th century did not help check the loss of wetlands, particularly the mangroves. The rapid growth of human population and the need for self-sufficiency in agricultural production led to expansion of cultivated land area and intensification of agriculture. This resulted in further decline of mangroves in the deltas of all major rivers in the Indian subcontinent. It is estimated that recent losses of mangroves have been 6 percent in Indonesia, 8 percent in Malaysia, 20 percent in Thailand, and 50 percent in the Philippines (Gosselink and Maltby 1990). Mangroves in the Mekong River Delta in Vietnam suffered extensive destruction (estimated 24 000 ha) by defoliants during the war, and, later, they have been rapidly converted to paddy fields and aquaculture (Khoa and Roth-Nelson 1994).

Other inland marshes and swamps have also been extensively converted to paddy fields and fishponds. In India, the fisheries department promoted conversion of marshes into fishponds throughout the Gangetic Plains (Jhingran 1992). A recent example is that of Lake Kolleru—a shallow lake covering 900 km^2 between the floodplains of River Krishna and Godavari. More than 90 percent of the lake has been converted into large fishponds—an activity promoted by the government to increase fish production (Gopal 1991).

Further losses of wetlands have occurred throughout the tropics for agriculture in the floodplains. The rivers have been regulated extensively by levees and dykes to isolate the floodplains. There are no published estimates of the loss of floodplain wetlands. However, it must be pointed out that the growing needs of water for irrigated agriculture have also resulted in an equally large area of man-made wetlands in the form of tens of thousands of irrigation tanks and reservoirs (Gopal and Krishnamurthy 1993). Vast areas of wetlands have also developed by waterlogging due to seepage (or rise in water table) from the irrigation network.

In the temperate regions, however, the wetlands have been extensively drained and converted into terrestrial systems. It is estimated that over 1.6 million km^2 of wetlands had been drained until 1985 (L'vovich and White 1990) of which three-fourths were drained in the temperate regions. Williams (1990) and Gosselink and Maltby (1990) have discussed in detail the wetland drainage for agriculture with detailed examples from U.S., Europe, and Australia. The extent and history of wetland reclamation in Holland is indeed impressive. People used coastal salt marshes as early as 500 B.C. Reclamation with polders proceeded with increasing pace until around the 14th century, and after 1500, agriculture fuelled the reclamation at a much faster rate. The limited land space motivated a larger plan in 1918 to reclaim Zuider Zee to add 2,050 km^2 of agricultural land. The drainage was accompanied by extensive peat mining and elimination of peat bogs. In the Netherlands alone, 20,000 km^2 (two-thirds of the present-day Netherlands) of wetlands have been reclaimed from the sea, freshwater lakes, low-lying riverine silts, and peat areas. Similar drainage of peatlands occurred in other parts of Europe much earlier.

Drainage has been far more extensive in the United States, though it started only in the mid-19th century. As early as 1921, the U.S. Department of Agriculture had identified about 37 million ha of wetlands that were "in need of drainage" in the eastern parts of the country. The wetlands in the presettlement United States are estimated to cover

74.87 million ha (OTA 1984) of which more than 50 percent had been lost until 1975. More than 80 percent of the total wetland loss has occurred due to conversion to agriculture. Extensive drainage has occurred in bottomland forests and wet prairie regions where a 13.7-percent wetland loss occurred between the 1950s and 1970s (Gosselink and Maltby 1990, Williams 1990). The riparian and coastal wetlands and peatlands have also been lost similarly though to a much lesser extent. Wetland loss to agriculture by conversion continues even today. Bernert and others (1999) report that in the Willamette Valley (Oregon) about 3800 ha (2.1 percent of the wetlands in the 1980s) of wetlands had been lost to uplands during 1980 to 1994, of which a 70-percent loss was associated with agriculture and only 6 percent was lost to urbanization.

IMPACTS OF WETLAND CONVERSION

Conversion of wetlands to agriculture means more than just a loss. The resulting land use changes as well as the agricultural practices in and around the remaining wetlands, which have many impacts upon the wetlands. Several recent studies have shown that the conversion of wetlands to agriculture or other land uses (including forestry) impacts upon the biodiversity though various taxonomic groups respond differently and at different spatial scales. Mensing and others (1998) have observed that in the riparian wetlands of the Northern temperate United States, shrub carr vegetation, amphibians, and birds are influenced by land use at relatively smaller scales (500 and 1000 m), whereas fish respond to land use at landscape level (2500 m or more). Diversity and richness of shrub carr vegetation, birds, and fish generally decrease with increasing cultivation in the landscape. A decrease in the proportion of open water to rangeland results in an increase of amphibian abundance but a decline of fish abundance. They also reported that wet meadow vegetation, aquatic macro-invertebrates, amphibians, and fish respond to local disturbances. Galatowitsch and others (1999) examined changes in floristic composition corresponding to land use differences at site to landscape levels in wet meadows associated with prairie glacial marshes in Minnesota. They observed that under the impact of agriculture, together with urbanization, the vegetation composition shifted from native graminoid and herbaceous perennial abundance to annuals or introduced perennials. Bethke and Nudds (1995), who analyzed data on duck abundance in Canadian prairie-parklands or the period 1975–89, observed that the recent decline in the number of breeding ducks, particularly in the West, have been partly due to loss of habitat to agriculture, in addition to loss to climatic change (drought). Agriculture can impact on aquatic invertebrates also in temporary wetlands. Based on a study of the resting eggs, shells, and cases remaining after wetlands dried in the prairie pothole region, Euliss and Mushet (1999) observed that the aquatic invertebrates were negatively impacted by intensive agriculture. There were more taxa and greater numbers of cladoceran resting eggs (ephippia), planorbid, and physid snail shells, and ostracod shells in wetlands within grasslands than in croplands.

The changes in agricultural wetlands, such as paddy fields, depend upon the agricultural practices involving removal of plants other than the crop, water management, and the use of agrochemicals. In less intensive paddy cultivation, the paddy fields support large biodiversity and their productivity, taking into account that the production of all consumable biota is substantially high (Heckman 1979). Under intensive cultivation, the biodiversity is greatly reduced. Aquaculture is another form of agricultural activity that has adversely impacted the coastal wetlands in many countries through loss of biodiversity and pollution.

The agricultural impacts on wetlands are far more complex. The requirements of water for irrigation directly impinge on the wetlands as water flows are regulated and diverted. Shallow and smaller wetlands in drier climates are more severely affected as their water is used for irrigation. In Greece, irrigation is reported to be the most important activity negatively influencing all functions and values of Ramsar wetlands (Gerakis and Kalburtji 1998). Diversion of water and reduced freshwater flows to estuarine areas have affected the mangroves worldwide by way of changes in species composition due to increased salinity. The fertilizers and pesticides applied in the field find their way through surface runoff and subsurface flow into adjacent wetlands. Furthermore, the agricultural activities enhance erosion resulting in increased input of sediments into the wetlands. Thus, changes in the hydrological regimes and an increase in the sediment and nutrient loading impact upon the biota and ecosystem processes in wetlands.

Zalidis and others (1997) identified the four most frequent factors that caused change in the ecological character of Greek wetlands. Of these, agricultural and municipal pollution, causing changes in water quality, accounted for 54 percent, construction of irrigation schemes and diversion of water courses, for 12 percent, and the expansion of agriculture and settlement for 32 percent of the damage to wetlands. Change in water regime affected 50 percent of the springs and 40 percent of the rivers; loss of wetland area affected 60 percent of the marshes and 52 percent of the estuaries, whereas all deltas and 75 percent of rivers had their water quality impacted. Wetlands in the Evros River Delta are also affected by pollutants transported by the river from agricultural area in the catchment lying in Bulgaria, Turkey, and Greece (Angelidis and Athanasiadis 1995).

In Southern Ontario (Canada), cumulative effects of agricultural land drainage (runoff, subsurface flow, and nutrient loss) have been estimated to account for the loss of 47 percent of wetlands from 1800 to 1990 (Spaling 1995). In northern prairie wetlands in North Dakota, Freeland and others (1999) observed higher sedimentation and fertilization rates in wetlands next to cultivated fields as indicated by higher phosphorus, organic matter, and nitrate-N concentrations in subsoils (15–60 cm) of wetlands surrounded by cultivated land than in those surrounded by grasslands.

In a study of Dives marshes (France), Granval and others (1993), however, observed that agricultural practices such as fertilizer use and grazing are beneficial to earthworms, and, therefore, to their various predators such as snipes (*Gallinago gallinago*). The biomass of earthworms in grazed meadows was more than 10 times higher than in the adjacent reedbed (*Arundo phragmites*).

Data on agricultural impacts on wetlands in the tropics are rare. Kassenga (1997) identified irrational use for agriculture and pollution to be responsible for the degradation and loss of wetlands in the basin of Lake Victoria (Tanzania). He also reported subsidence of wetlands due to excessive extraction of water. In India, practically all wetlands are affected by agriculture though there are no specific data. Pesticides in the fields surrounding the Keoladeo National Park (Bharatpur) are considered to be a major threat because the park depends entirely on water from outside (Vijayan 1995). In Kerala (South India), intensive use of fertilizers for rice cultivation has caused widespread eutrophication of backwaters.

WETLANDS AS REGULATORS OF NUTRIENT FLUX

Natural wetlands, which lie at the interface between agricultural uplands and the deep open waters, act as recipients of sediments and agrochemicals and are known to regulate their flux to the lakes and rivers. As pointed out above, the agrochemicals entering the wetlands are recognized to be among the factors causing degradation of wetlands. Yet, these very wetlands are valued for the same functions of intercepting nutrients and other substances, and, hence, for protecting the downstream waters (Lowrance and others 1984, Schlosser and Karr 1981, Weller and others 1994, Whigham and others 1988). Hillbricht-Ilkowska and Kostrzewska-Szlakowska (1993) have shown that the lake littorals are more efficient in removing nutrients (particularly nitrogen) than the riparian zones. However, there is growing realization that the natural wetlands will be quite limited in absorbing agricultural wastewater (Peterson 1998). Natural wetlands have in most places either been lost or are highly degraded. Furthermore, the natural wetlands in the tropics, e.g., in the Indian subcontinent, also appear to receive wastewaters and runoff from nonpoint sources (including agriculture) far beyond their buffering capacity. In this context, the constructed wetlands technology is being extended to intercept agricultural runoff and remove pollutants from it (Hammer 1992, Kern and Idler 1999, Peterson 1998, Rodgers and Dunn 1992). Thus, constructed wetlands may replace the natural wetlands and become partners with agriculture in integrated water and nutrient management.

CONCLUSION

The continuing loss of natural wetlands to agriculture and the increasing emphasis on intensive agriculture, which causes wetland degradation, raise doubts over the future of wetlands, including agricultural wetlands. Can the man-made wetlands replace the functions and values of natural wetlands? Can wetlands be really conserved without checking agricultural impacts? The sustainability of wetlands and agriculture is interlinked with that of the water resources. The sustainability of agro-ecosystems also depends upon wetlands because the latter provide, besides irrigation water, also crop pollinators, some frost protection, and predators of crop pests (Gerakis and Kalburtji 1998). The future scenario appears to be worse. Discussing the loss of wetlands to drainage in Eastern Europe, Hartig and others (1997) have pointed out that following climate change, higher temperatures and greater evapotranspiration may alter the hydrologic regime such that freshwater wetlands are further encroached upon by agricultural land use. Wetland conservation seemingly faces confrontation with agriculture.

The problem lies in the national policies and their implementation. In recent years, several countries have adopted national wetland policies, but, only rarely, agriculture is recognized as a major sector competing with wetlands for water or agriculture leading to degradation of wetlands. There has been no effort to resolve the conflict with the agricultural policy. Within the United States, Nelson (1986a, 1990) had discussed various aspects of the agricultural policy in some detail. He points out the policy crisis results from contradictory incentives of central government to both drain and preserve wetlands. A similar situation exists also in the United Kingdom (Nelson 1986b).

In Greece, Zalidis and others (1997) pointed out the inefficacy of the agricultural policy, which could not protect small wetlands or deal with the loss of wetlands from intensified use of existing farmland. However, Pyrovetsi and Daoutopoulos (1998) observed that the wetland farmers had a more negative attitude toward the wetland resources and were more ignorant of conservation issues or the impact of their practices on the environment than plain farmers.

Most of the developing countries, with great population pressures, are compelled to focus on increasing agricultural production despite declining returns from the degraded land resources. In the process, there are ever-increasing demands on water resources, which have depleted and degraded rapidly. Both agriculture and wetlands are threatened. Therefore, the need for an integrated, balanced, and coordinated approach to water resource management is greater than ever before. We need an integrated policy, which takes care of both natural wetlands and agriculture so as to minimize the impacts of agriculture on wetlands without compromising agricultural production.

ACKNOWLEDGMENTS

I am grateful to Dr. Marjorie Holland and Dr. Charles M. Cooper for their invitation to the conference, and to the Croft Institute for International Studies at the University of Mississippi and the Society of Wetland Scientists for providing necessary financial support for my participation. I thank Professor D.P. Zutshi and Mr. V. Chet for valuable information on the cultivation of floating islands.

REFERENCES

Angelidis, M.O.; Athanasiadis, A.I. 1995. Pollution mechanisms. In: A Ramsar wetland—delta of the River Evros, Greece. Aquatic Sciences. 57(2): 161–171.

Annandale, N. 1918. Aquatic molluscs of the Inle Lake and connected water. Records of the Indian Museum. 14: 103–182.

Bernert, Joseph A.; Eilers, Joseph M.; Eilers, Benn J. [and others]. 1999. Recent wetlands trends (1981/82–1994) in the Willamette Valley, Oregon, U.S.A. Wetlands. 19(3): 545–559.

Bethke, R.W.; Nudds, T.D. 1995. Effects of climate change and land use on duck abundance in Canadian Prairie–Parklands. Ecological Applications. 5(3): 588–600.

Chang, T.T. 1976. Rice: *Oryza sativa* and *Oryza glaberrima* (Graminae: Oryzae). In: Simmonds, N.W., ed. Evolution of crop plants. London: Longman: 98–104.

Coe, Michael D. 1964. The chinampas of Mexico. Scientific American. 211 (1): 90–98.

Department of Agriculture (Economics Branch). 1939. Padi planting methods in Malaya. Malayan Agricultural Journal. 27(2): 40–59.

Euliss, Ned H., Jr.; Mushet, David M. 1999. Influence of agriculture on aquatic invertebrate communities of temporary wetlands in the prairie pothole region of North Dakota, U.S.A. Wetlands. 19(3): 578–583.

Freeland, John A.; Richardson, Jim L.; Foss, Lynn A. 1999. Soil indicators of agricultural impacts on northern prairie wetlands: Cottonwood Lake research area, North Dakota, U.S.A. Wetlands. 19(1): 56–64.

Galatowitsch, S.M.; Whited, D.C.; Lehtinen, R. [and others]. The vegetation of wet meadows in relation to their land use. Environmental Monitoring & Assessment. 60(2): 121–144.

Gerakis, Argyrios; Kalburtji, Kiriaki. 1998. Agricultural activities affecting the functions and values of Ramsar wetland sites of Greece. Agriculture Ecosystems & Environment. 70(2–3): 119–128.

Gopal, B. 1991. Wetland (mis)management by keeping people out: two examples from India. Landscape and Urban Planning. 20: 53–59.

Gopal, B.; Krishnamurthy, K. 1993. Wetlands of South Asia. In: Whigham, D.F.; Dykyjova, D.; Hejny, S., eds. Wetlands of the world. I. Inventory, ecology and management. Handbook of vegetation science. Dordrecht, Kluwer: 15/2: 345–414.

Gorman, C. 1977. A priori models and Thai prehistory: a reconsideration of the beginnings of agriculture in Southeastern Asia. In: Reed, C.A., ed. Origins of agriculture. The Hague: Mouton: 321–355.

Gosselink, J.G.; Maltby, E. 1990. Wetland losses and gains. In: Williams, M., ed. Wetlands: a threatened landscape. Oxford, U.K.: Blackwells: 296–322.

Granval, P.; Aliaga, R.; Soto, P. 1993. The impact of agricultural management on earthworms (*Lumbricidae*), common snipe (*Gallinago gallinago*), and the environmental value of grasslands in the Dives marshes (Calvados); Gibier Faune Sauvage. 10(March): 59–73. In French.

Hammer, D.A. 1992. Designing constructed wetland systems to treat agricultural nonpoint source pollution. Ecological Engineering. 1: 49–82.

Harlan, J.R. 1977. The origins of cereal agriculture in the Old World. In: Reed, C.A., ed. Origins of agriculture. The Hague: Mouton: 357–383.

Hartig, Ellen K.; Grozev, Ognyan; Rosenzweig, Cynthia. 1997. Climate change, agriculture, and wetlands in Eastern Europe: vulnerability, adaptation, and policy. Climatic Change. 36(1–2): 107–121.

Heckman, C.W. 1979. Rice field ecology in northeastern Thailand. The Hague: W. Junk, bv. 228 p.

Hillbricht-Ilkowska, A.; Kostrzewska-Szlakowska, I. 1993. Transport of dissolved forms of nitrogen derived from arable land along the river-lake system in a post-glacial landscape. In: Gopal, B.; Hillbricht-Ilkowska, A.; Wetzel, R.G., eds. Wetlands and ecotones: studies on land-water interactions. New Delhi: National Institute of Ecology and International Scientific Publications: 61–74.

Ho, P.T. 1977. The indigenous origins of Chinese agriculture. In: Reed, C.A., ed. Origins of agriculture. The Hague: Mouton: 413–484.

Hook, Donald. 1993. Wetlands: history, current status, and future. Environmental Toxicology & Chemistry. 12(12): 2157–2166.

Jhingran, V.G. 1992. Fish and fisheries of India. 3rd ed. New Delhi: Hindustan Publishing Corp.

John, D.M. 1986. The inland waters of tropical West Africa. Archiv fuer Hydrobiologie, Ergebnisse der Limnologie. 23: 1–244.

Kassenga G.R. 1997. A descriptive assessment of the wetlands of the Lake Victoria Basin in Tanzania. Resources Conservation & Recycling. 20(2): 127–141.

Kaul, V.; Zutshi, D.P. 1966. Some ecological consideration of floating islands in Srinagar lakes. Proceedings of the National Academy of Sciences of India; 36B: 273–281.

Kern, J.; Idler, C. 1999. Treatment of domestic and agricultural wastewater by reed bed systems. Ecological Engineering. 12: 13–25.

Khoa, L.V.; Roth-Nelson, W. 1994. Sustainable wetland use for agriculture in the Mekong River Delta of Vietnam. In: Mitsch, W.J., ed. Global wetlands: Old World and new. Amsterdam: Elsevier: 737–748.

L'Vovich, M.I.; White, G.F. 1990. Water. In: Burner, B.L., III; Clark, W.C.; Kates, R.W. [and others], eds. The earth as transformed by human action. New York: Cambridge University Press.

Lowrance, R.; Todd, R.; Fail, J. [and others]. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience. 34: 374–377.

Mensing, D.M.; Galatowitsch, S.M.; Tester, J.R. 1998. Anthropogenic effects on the biodiversity of riparian wetlands of a northern temperate landscape. Journal of Environmental Management. 53(4): 349–377.

Mitsch, W.J.; Gosselink, J.G. 1993. Wetlands. 2nd ed. New York: Van Nostrand-Reinhold. 722 p.

Moorcroft, William; Trebeck, G. 1841. Travels in the Himalayan provinces of Hindustan and the Panjab; in Ladakh and Kashmir. London: John Murray.

Nelson, R.W. 1986a. Wetland management strategies balancing agriculture with conservation: a national survey of expert opinion. Environmental Professional. 8(1): 54–69.

Nelson, R.W. 1986b. Wetlands policy crisis U.S.A and U.K. Agriculture Ecosystems & Environment. 18(2): 95–122.

Nelson, R.W. 1990. Can experts agree on policy directions toward wetlands and agriculture. Environmental Professional. 12(2): 131–143.

OTA (Office of Technology Assessment). 1984. Wetlands: their use and regulation. Washington, DC: U.S. Government Printing Office.

Peterson, H.G. 1998. Use of constructed wetlands to process agricultural wastewater. Canadian Journal of Plant Science. 78(2): 199–210.

Por, F.D. 1995. The pantanal of Mato Grosso (Brazil): world's largest wetlands. Monographiae Biologicae 73. Dordrecht: Kluwer. 122 p.

Pyrovetsi, M.; Daoutopoulos, G. 1999. Farmers' needs for nature conservation education in Greece. Journal of Environmental Management. 56(2): 147–157.

Richards, J.F. 1990. Agricultural impacts in tropical wetlands: rice paddies for mangroves in South and Southeast Asia. In: Williams, M., ed. Wetlands: a threatened landscape. Oxford, U.K.: Blackwells: 217–233.

Rodgers, J.H., Jr.; Dunn, A. 1992. Developing design guidelines for constructed wetlands to remove pesticides from agricultural runoff. Ecological Engineering. 1: 83–95.

Sahni, B. 1927. A note on the floating islands and vegetation of Khajiar, near Chamba in the N.W. Himalayas. Journal of the Indian Botanical Society. 6: 1–7.

Sasser, C.E.; Gosselink, J.G.; Swanson, E.M.; Evers, D.E. 1995. Hydrologic, vegetation and substrate characteristics of floating marshes in sediment rich wetlands of the Mississippi River Delta Plain, Louisiana, U.S.A. Wetlands Ecology and Management. 3: 171–187.

Schlosser, I.J.; Karr, J.R. 1981. Riparian vegetation and channel morphology impact on spatial patterns of water quality in agricultural watersheds. Environmental Management. 5: 233–243.

Scoones, I.; Cousins, B. 1994. Struggle for control over wetland resources in Zimbabwe. Society & Natural Resources. 7(6): 579–594.

Serpenti, L.M. 1965. Cultivators in the swamps: social structure and horticulture in a New Guinea society (Frederik-Hendrik Island West New Guinea). Assen, Netherlands: Van Gorcum-Prakke.

Spaling, H. 1995. Analyzing cumulative environmental effects of agricultural land drainage in Southern Ontario, Canada. Agriculture Ecosystems & Environment. 53(3): 279–292.

Sturtevant, W.C. 1970. Agriculture on artificial islands in Burma and elsewhere. Eighth International Congress of Anthropological and Ethnological Sciences; Tokyo and Kyoto; 1968: 11–13. Vol. 3.

Tombi Singh, H.; Singh, R.K.S. 1995. Loktak Lake, Manipur. WWF-India, New Delhi. 69 p.

Vergara, B.S. 1992. Tropical wet rice systems. In: Pearson, C.J., ed. Field crop ecosystem. Amsterdam: Elsevier: 167–182.

Vijayan, L. 1995. Keoladeo National Park. WWF-India, New Delhi. 77 p.

Vishnu-Mittre. 1977. Changing economy in ancient India. In: Reed, C.A., ed. Origins of agriculture. The Hague: Mouton: 570–588.

Weller, D.E.; Correll, D.L.; Jordan, T.E. 1994. Denitrification in riparian forests receiving agricultural discharges. In: Mitsch, W.J., ed. Global wetlands: Old World and new. Amsterdam: Elsevier: 117–131.

West, Robert C.; Armillas, P. 1950. Las chinampas de México: Poesia y realidad de los "jardines flotantes." Cuadernos Americanos. Año 9, No. 2: 165–182. Vol. 50.

Whigham, D.F.; Chitterling, C.; Palmer, B. 1988. Impacts of freshwater wetlands on water quality: a landscape perspective. Environmental Management. 12: 663–671.

Williams, Michael, ed. 1990. Wetlands: a threatened landscape. Oxford, U.K.: Blackwells. 419 p.

You Xiuling. 1991. Rice planting on the water surface in ancient China. Chinese Journal of Rice Science. 5.2: 79–82. In Chinese.

Zalidis, George C.; Mantzavelas, Antonis L.; Gourvelou, Eleni. 1997. Environmental impacts on Greek wetlands. Wetlands. 17(3): 339–345.

THREE MISSISSIPPI ECOTYPES OF WETLAND PLANTS

Janet M. Grabowski[1]

Abstract—In 1996, the USDA-Natural Resources Conservation Service (NRCS) Jamie L. Whitten Plant Materials Center (PMC) released three locally collected, source-identified wetland plants. Indian Bayou source powdery thalia (*Thalia dealbata* Fraser ex Roscoe) and Leflore source creeping burhead [*Echinodorus cordifolius* (L.) Griseb.] were collected in the Mississippi Delta and Leaf River source woolgrass [*Scirpus cyperinus* (L.) Kunth] from southeastern Mississippi. Both vegetative propagules and seedlings can be planted. PMC seed-germination studies have shown that Indian Bayou and Leaf River germinate readily after exposure to cold, moist conditions (stratification). Indian Bayou germination and seedling growth was best in a moist, not saturated, growing medium, whereas best germination of Leaf River was in a saturated medium, and seedling growth was better in a moist medium. Germination of Leflore was very poor for all seed treatments in the test, even though seedlings have established in PMC plant-production ponds.

INTRODUCTION

Current interest in wetland restoration, enhancement, and mitigation has led to an increased need for sources of plants that are well adapted to local environmental and soil/water conditions. Most plant materials available commercially are from widely varying geographic regions. Plant materials originating from areas with vastly different physical and environmental characteristics are often not well suited to conditions on the planned planting site, and use of these plants can lead to less than satisfactory performance. McNaughton (1966) found significant differences in growth habit, environmental tolerances, and dormancy patterns of several ecotypes of three *Typha* L. species collected from diverse sites in the United States. Growth patterns of each ecotype were matched for optimum survival and productivity to the environmental conditions common to their original source location. Although the ideal recommendation would be to use planting materials collected from local ecotypes, this is often impractical because availability may be lacking and costs prohibitive (U.S. Army Engineer Waterways Experiment Station 1978). Therefore, an alternate method of obtaining adapted plant materials is required.

Awareness of genetically controlled ecotypic differences between plant populations has led to development of a new type of plant release within the USDA-Natural Resources Conservation Service (NRCS) Plant Materials Program called a source-identified release. A source-identified release was collected from a natural plant population and has not undergone any testing or selection prior to its release.

Staff at the Jamie L. Whitten Plant Materials Center (PMC), Coffeeville, MS, collected plants of several emergent wetland species within the State and released three of these for use in various types of wetland plantings. Although these ecotypes are best suited to areas within Mississippi, ecological differences within the State and among adjoining States are much smaller than in some other parts of the country. This fact along with the fairly broad natural geographic range of these species suggests that they may also be planted successfully in other areas in the Southeastern United States with comparable climatic, hydrologic, topographic, and soil features.

Plant material of these releases was provided to a limited number of local plant producers for increase and sale to the public. At present, vegetative propagules (clonal material) are being marketed; however, production of seedlings may be more desirable because genetic recombination could lead to greater diversity in the planting population and increased ability to withstand environmental stresses. This paper presents descriptive information on each of these releases as well as results of a study conducted to determine seed propagation methods for these plants.

PLANT COLLECTIONS

Indian Bayou

Thalia dealbata is a member of the Maranthaceae or arrowroot family. It is a rare inhabitant of shallow wetlands in the Mississippi Delta and other Southeastern States. The Indian Bayou source of powdery thalia was collected by Travis Salley in 1989 from a homeowner's yard in Bolivar County, Mississippi. According to the homeowner, the plants were originally moved from Washington County, Mississippi. It is a herbaceous perennial arising from thick (up to 4 cm in diameter) rhizomes, with a bluish, glaucous coating on leaves, flower stalks, and flowers. The 1- to 1.2-m long leaves arise from the base of the plant and have a shape reminiscent of a *Canna* L., with a stout petiole and a large, ovate to lanceolate leaf blade. Flowers are produced from late May to September with fruit maturing throughout the summer. The attractive purple to bluish flower clusters are produced on scapes extending 0.6 to 1 m above the foliage. Fruit type is an urticle, consisting of a bladdery, membranous covering loosely surrounding a single seed. Seeds are approximately 6 mm in diameter, subglobose, dark-brown

[1] Biologist, USDA-NRCS, Jamie L. Whitten Plant Materials Center, 2533 County Road 65, Coffeeville, MS 38922-2652.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

speckled with tan or gray, with a conspicuous tan to brown hilum. There are approximately 3,184 seeds per kilogram.

Leaf River

Woolgrass is a bulrush, in the Cyperaceae or sedge family that is commonly found in shallow wetlands throughout the State of Mississippi. Its native range covers most of the Eastern United States and Canada, extending westward into eastern Texas [USDA-Soil Conservation Service (no date)]. Leaf River source was collected by B.B. Billinglsey, Jr., Harvey Huffstatler, and Jeff Tillman from a site near the Leaf River in Jones County, Mississippi. It is a clump-forming perennial with short rhizomes. The grass-like basal leaves are up to 1.5 m in length and arch outwards from the base of the plant like a fountain. The flowering culms are 1.2 to 1.8 m tall, leafy, somewhat coarse, and obtusely triangular. The dense inflorescence contains numerous pale-green spikelets that become brown and wooly as the seeds mature. The tiny achenes are light tan, with six long, twisting perianth bristles. Flowering begins in June, and seeds mature by September. The achenes were too small to be accurately counted by PMC equipment, but it was estimated that there are upwards of 18 million per kilogram.

Leflore

Creeping burhead can be found in swamps, marshes, and ditches in the Southeast and Lower Midwest, westward into Oklahoma and Texas (Godfrey and Wooten 1979). It is a member of the Alismataceae or water-plantain family. The Leflore source was collected by B.B. Billingsley, Jr., Joe Snider, Joel Douglas, and Janet Grabowski from a flooded lowland in Leflore County, Mississippi. Leflore is an annual or short-lived perennial that often spreads or creeps by rooting scapes. Basal leaves are broadly ovate, cordate at the base, 5 to 18 cm long and almost as wide. The principal veins are conspicuous and impressed on the upper surface of the leaf blade. Petioles are 10 to 50 cm long, enlarged and spongy towards the base. The leafy scapes are upright when young, becoming prostrate, up to 1 m or more in length, producing new plantlets at the tip and nodes. Numerous whorls of 12- to 20-mm wide flowers, with three white petals and rounded, greenish centers are produced at nodes along the scape. Flowering begins in June and continues until frost. The fruiting heads are round, bur-like clusters of small brown, flattened achenes, which have a long beak on one end. There are approximately 5 million achenes per kilogram.

VEGETATIVE PROPAGATION

All of these releases can be vegetatively propagated by division of the parent plant. The large size of the shoots and rhizomes of Indian Bayou dictate that a fairly large planting piece is required. A section about 15 to 24 cm long with a few shoots is ideal. Plantings made by the PMC on a Wetland Reserve Program (WRP) site have indicated that Indian Bayou propagules are capable of withstanding fairly adverse environmental conditions after planting (data not presented); these results can probably be attributed to the large amount of food reserves stored in the rhizomes.

Leaf River clumps can be dug and individual shoots divided somewhat easily. Older clumps tend to die back in the center, so actively growing shoots around the margins of the plant should be selected. It is also a good idea to discard shoots containing an old flowering culm because these either will not regenerate or, if they do, the shoot produced will be very weak. Care should be taken in handling Leaf River plants because the edges of the leaves are very sharp and can cut deeply into flesh.

Established Leflore plants can be dug and divided into a small number of propagules. These plants do not have a very deep root system, which makes them fairly easy to dig. Small plants produced on the flowering scape can also be removed and planted.

SEED GERMINATION STUDY

Methods and Materials

Seeds were harvested in late summer to fall of 1995 and 1996 (harvest dates varied between species and between year of collection). Seeds were cleaned immediately after collection without drying. Leaf River seeds were loosened from fruit clusters using a brush machine (Westrup a/s Slagelse, Denmark) and then hand screened to remove inert matter. Indian Bayou seeds were rubbed over a roughened surface to remove the papery fruit coverings. Leflore seeds required only hand screening to remove small amounts of trash.

Three storage treatments were tested: (1) dry storage in a cooler at 13 °C and 45 percent relative humidity; (2) moist storage in a cooler at 5.5 °C; and, (3) water storage in a cooler at 5.5 °C. Dry storage was tested on all species. Moist storage was used for all seedlots, except the 1996 Leaf River lot. Water storage was tested on the 1995 seedlot of Leaf River and 1996 Leflore only. Small quantities of seeds of each species were divided from the main lot and placed in moist and water storage immediately after cleaning. All seeds in water storage were put in a nylon-mesh bag and then submerged in tap water in a glass jar. Water was not changed during the storage period for the first year of the test; however, it was changed monthly during the second year in an attempt to limit algae growth. Moist, stored seeds of Leaf River and Leflore were placed on a moistened brown paper towel and placed in a self-sealing plastic bag with sufficient additional water to maintain high moisture levels during the storage period. Indian Bayou seeds are too large to allow good seed contact with paper toweling, so seeds were stratified in moist sphagnum moss from which they were easily separated at planting. All remaining seeds were dried thoroughly and placed in a self-sealing plastic bag for the dry-storage treatments.

Preplanting treatments applied to dry-stored seeds were a 3- to 4-month stratification period (Strat.), and for Indian Bayou seed only, scarification (Scar.) using mechanical means, and combinations of stratification and scarification. In 1995, seeds were scarified by placing them in a coffee can lined with sand paper and tumbling with some gravel added to increase abrasion. A concern that seeds were not uniformly scarified using this method required use of an alternate method in the second year where seeds were individually rubbed against sandpaper. All stratification treatments were placed in the cooler during November 1995 and 1996.

Treatments were planted in the greenhouse within a 1-week period in March 1996 and 1997. Stratified, moist-stored, and water-stored Leaf River and Leflore seeds were allowed to surface dry during the planting process to facilitate sample counting. Sample sizes used were 100 seeds of Leaf River and Leflore for each treatment and 20 seeds of the 1995 lot and 25 seeds of the 1996 Indian Bayou lot. Germination containers were 17.8 cm by 13.3 cm by 5.9 cm black plastic bedding-plant liners. Growing medium used was a 3:1 peat moss/sand growing medium amended with commercially recommended quantities of pelletized slow-release fertilizer, dolomitic lime, Micromax micronutrient fertilizer, and Aquagro wetting agent. Seeds were sown on the surface of the growing medium, except Indian Bayou seeds, which were planted approximately 6 mm deep. Seeds were spread as uniformly as possible on the medium.

Two germination conditions were tested: (1) moist conditions, where containers were placed on a normal greenhouse bench and watered regularly (normal bench); and, (2) continuously saturated conditions on an ebb and flow greenhouse bench (flood bench). Water on the ebb and flow bench was maintained at a depth of 6 to 12 mm except for short periods of time when the bench was drained and rinsed to remove alga growth. Experimental design was a factorial experiment in a randomized complete block with three replications arranged as a split plot, with germination condition as the main plot and seed treatment (storage condition and preplanting seed treatment) as the split plot.

An initial seedling count was made when it was deemed that a sufficient number of seedlings were present to justify counting. Two additional counts were made at 3-week intervals following the initial count. A few seedlings of some species died before the initial count, especially on the normal greenhouse bench, where the surface of the growing medium dried for short periods of time. Dead seedlings were counted, and their numbers were included in the initial count because germination had occurred. Later counts included only those plants that had some green tissue. Total germination could not be determined, so results presented are from the count with highest overall number of seedlings. The study was analyzed with years of testing treated separately. Data was subjected to an analysis of variance, and appropriate mean separation was performed using a least significant difference test (LSD) at P = 0.05. All data on seedlots are referenced by year of seed collection (1995 and 1996), not year of testing.

RESULTS AND DISCUSSION

Indian Bayou Germination

Indian Bayou appears to have complex germination and seedling growth requirements. Only a few seedlings have established naturally in PMC ponds, mainly along the margin of the pond. Previous attempts at seedling production in the greenhouse indicated that germination was erratic and slow.

Initial germination percentages in this study were fairly low, and seeds continued to germinate in small numbers throughout the test. For the 1995 seedlot, there was a significant response to seed treatment only (table 1). Overall germination was better on the normal bench than on the flood bench. In this environment, seeds responded positively to moist storage and both stratification treatments. The high

Table 1—Indian Bayou mean germination percentages for seed exposed to two storage conditions, three preplanting treatments, and two germination conditions

Storage and pretreatment	Germination condition	1995	1996
		- - - *Percent* - - -	
Dry/strat.	Normal bench	35	17
Dry		8	1
Dry/scar. + strat.		25	17
Dry/scar.		5	40
Moist		27	15
Dry/strat.	Flood bench	0	3
Dry		0	3
Dry/scar. + strat.		15	8
Dry/scar.		0	9
Moist		3	1
LSD (P = 0.05)		17	13

Strat. = stratification; scar. = scarification; LSD = least significant difference.

germination percentage for scarified plus stratified seeds compared to stratified seeds indicate that this seedlot responded to scarification.

For the 1996 seedlot, there was a significant interaction between seed treatment and germination condition. This interaction was probably due to poor germination of all treatments on the flood bench. Also, germination of dry-stored seeds was poor on both benches. Germination of scarified seeds on the normal bench was significantly higher for this seedlot than any other treatment, which is curious because of the low percentage for this treatment in the previous year. The scarification method used for this seedlot was more vigorous and may have increased response to this treatment. However, this cannot explain the response of scarified plus stratified seeds, where germination was not different than stratified seeds.

In both years, seedlings on the flood bench were more susceptible to infestation by aphids than those on the normal bench, indicating that the plants were probably stressed. This improved germination and growth on the normal bench appears to corroborate observations of seedling establishment in drier conditions along the margins of PMC growing ponds. The response to stratification and moist storage indicates that Indian Bayou has an internal dormancy that is overcome by cold, moist treatment. Germination percentages were slightly higher for dry-stored seeds that were subsequently stratified than for moist-stored seeds regardless of treatment year, which suggests capability for long-term seed storage. Mechanical scarification, especially when accompanied by stratification, also appears to be beneficial, perhaps because seeds have a waxy coating that may prevent water infiltration. A previous attempt at sulfuric acid scarification on a small quantity of seeds showed that even a short duration (< 1 minute) treatment was toxic (Janet Grabowski, unpublished data).

However, because mechanical scarification did not consistently improve germination, it appears that stratification alone is probably sufficient to promote germination.

Leaf River Germination

Results for the 1995 and 1996 seedlots were similar for those treatments common to both years of testing (table 2). The 1995 lot showed a significant response to seed treatment. Dry-stored Leaf River seeds appeared to germinate more quickly after stratification, but germination percentages were not significantly different from the dry-storage treatment alone, which conflicts with Isley (1944) who found that woolgrass seeds require stratification for 6 months or longer to overcome dormancy. This southern Mississippi ecotype is adapted to areas with only short periods of chilling winter temperatures and apparently does not have a dormancy mechanism that requires stratification. Garbisch and McIninch (1992) also reported a lack of seed dormancy. There was a trend towards improved germination of the 1995 seeds stored in water, but the difference was significantly higher than only the dry-stored seeds on the normal bench and the moist-stored seeds on the flood bench.

Leaf River seeds showed a significant response to germination conditions in both years of testing. Highest germination was on the flood bench with saturated conditions. Seedling mortality on the normal bench was higher for this species, probably because of the small size of the seedlings, which made them highly susceptible to desiccation as the surface of the growing medium dried. This problem was more pronounced in 1996, probably due to cooler temperatures and longer periods of cloud cover in 1997. However, later seedling growth in both years was more vigorous for surviving plants on the normal bench. Seedlings growing on the flood bench showed evidence of poor root growth due to a lack of aeration, and there was extensive algae and slime mold growth on the growing medium surface, which may have been toxic to the seedlings. Most of the plants on the flood bench were barely alive by the third evaluation date. Observations made of plants growing in the wild indicate that this species requires fairly wet conditions for germination, but plants become increasingly tolerant of drying substrates as they grow, with mature plants possessing a higher level of drought tolerance than would be anticipated for a wetland plant.

Table 2—Leaf River mean germination percentages for seed exposed to various seed storage conditions, one preplanting treatment, and two germination conditions

Storage and pretreatment	Germination condition	1995	1996
		- - - *Percent* - - -	
Dry/strat.	Normal bench	19	17
Dry		12	9
Moist		15	—
Water		24	—
Dry/strat.	Flood bench	35	61
Dry		41	57
Moist		32	—
Water		44	—
LSD (P = 0.05)		8	23

Strat. = stratification; LSD = least significant difference.

Leflore Germination

Mean germination percentages for all treatments were extremely low (< 5 percent) in both years (data not presented). For 1995 and 1996 seedlots, there was a significant response to germination condition, with best germination on the flood bench. However, these differences were too minuscule to support a recommendation on proper germination and seedling growth environments. Storage treatments yielded the same results in both years. Although water storage was added in the second year to see if germination could be improved over that of moist storage, no improvement was observed. Water- and moist-stored seeds showed a slight trend towards improved germination compared to dry storage. An attempt was made to mechanically scarify some seeds from the 1996 lot, but visual inspection after treatment did not indicate that this treatment had any appreciable effect on seed coverings. Seed germination in the PMC production ponds has been noted, mainly along the margins and in shallow water; however, this test apparently did not provide suitable conditions for germination.

SUMMARY

These three source-identified releases can potentially be used in wetland plantings in Mississippi and perhaps other areas in the Southeast. Because of their specific germination requirements, direct seeding is probably not advisable. However, greenhouse seedling production is practical for Indian Bayou and Leaf River, with stratification recommended for Indian Bayou seeds. Additional research would be necessary to determine methods to successfully propagate Leflore from seed.

REFERENCES

Garbisch, E.W.; McIninch, S. 1992. Seed information for wetland plant species of the Northeast United States. Restoration and Management Notes. 10: 85–86.

Godfrey, R.K.; Wooten, J.W. 1979. Aquatic and wetland plants of Southeastern United States monocotyledons. Athens, GA: The University of Georgia Press. 712 p.

Isley, D. 1944. A study of conditions that affect the germination of *Scirpus* seeds. Memoir 257. Ithaca, NY: Cornell University Agricultural Experiment Station. 28 p.

McNaughton, S.J. 1966. Ecotype function in the *Typha* community-type. Ecological Monographs. 36: 297–325.

U.S. Army Corp of Engineers. 1978. Wetland habitat development with dredged material: engineering and plant propagation. Tech. Rep. DS–78–16. Vicksburg, MS: U.S. Army Corp of Engineers, Waterways Experiment Station. 158 p.

U.S. Department of Agriculture. [n.d.] Midwestern wetland flora field office guide to plant species. Lincoln, NE: U.S. Department of Agriculture, Soil Conservation Service. 615 p.

WINTER BIRD COMMUNITIES IN AFFORESTATION: SHOULD WE SPEED UP OR SLOW DOWN ECOLOGICAL SUCCESSION?

Paul B. Hamel, Daniel J. Twedt, Timothy J. Nuttle, Christopher A. Woodson, Fred Broerman, and Joseph M. Wahome[1]

Abstract—Recent assessments of afforestation on agricultural lands in the Mississippi Alluvial Valley imply the importance of quickly developing vertical forest structure to benefit wildlife. Examining this assumption, we find that mammals and birds occur through the full successional sere as targets of proactive management and control. Different species of animals thrive in structures available at different times during succession. Thus, forest managers' choices of strategies favor species' success differentially. Early successional species, particularly those avian communities occurring during winter, have heretofore been considered only in passing. However, because they occur in areas where herbaceous plants dominate vegetation structure, these communities include species otherwise rare or absent from the landscape. Extensive afforestation in the Mississippi Alluvial Valley provides ephemeral habitat for birds that winter in herbaceous areas. To provide habitat for winter birds, managers may wish to consider maintaining large tracts in herbaceous vegetation similar to that occurring 3 to 7 years after cessation of farming activities.

INTRODUCTION

Recent assessments of afforestation of agricultural lands in the Mississippi Alluvial Valley (MAV), particularly the Delta area of Mississippi, have stressed the importance of quickly attaining the physical structure and stature of forests (Schweitzer and others 1997). Implementation of land management plans designed for neotropical migratory birds in the Lower MAV (LMAV) (Mueller and others 1995) will benefit from dependable, rapid afforestation as well. Benefits of afforestation include provision of habitat for middle and late successional birds, production of pulpwood, production of sawtimber wood products, and erosion control, as well as carbon sequestration and accumulation of soil organic matter. All these benefits are positively associated with the speed with which afforestation occurs. Rapid afforestation implies swift accumulation on the landscape of the physical structure and stature of forest. It is a means of carbon sequestration (Cannell 1999b, Chang 1999) and of land rehabilitation in Amazonia (Bauch and others 1999) and elsewhere (Harrington 1999). Fast development of vertical forest structure is implicit in the environmental (Joslin and Schoenholtz 1997) and economic (Pande and others 1999, Scholtens 1998) analyses of afforestation.

Afforestation, including rapid afforestation, is assumed to be beneficial to wildlife (Boyle 1999, Cannell 1999a, Helmer 1999, Weaver and Pelton 1994, Weaver and others 1990, Willoughby and McDonald 1999). On the other hand, certain native wildlife and grazing animals can hinder afforestation efforts (Anderson and Katz 1993, Houston 1991, Niyaz and others 1999).

Vegetation structure is an important determinant of bird species occurrence and community composition (DeGraaf 1987, DeGraaf and others 1992, James 1971). Hamel (1992) associates birds with combinations of vegetation structure, such as trees, shrubs, and herbaceous vegetation. Afforestation yields unusual elements in secondary succession, such as tall cottonwood (*Populus* L. spp.) trees and herbaceous vegetation with little woody understory. A first hypothesis is that the bird community developing in afforestation will reflect vegetation structure.

Wintering birds use the early successional herbaceous community. During rapid afforestation, the early successional periods are shorter than during natural succession. Our second hypothesis is that early successional species may not benefit from rapid afforestation of agricultural fields as much as from natural succession.

To examine the assumptions that rapid afforestation is beneficial to wildlife and that bird species occurrence is a function of vegetation structure, we consider the presence or absence of particular bird species and their communities rather than the rate of development of ecological function. As hawks and owls prey on mammals, then succession of mammal species is, in turn, dependent on bird community dynamics.[2]

[1] Hamel, Research Wildlife Biologist, USDA Forest Service, Southern Research Station, Stoneville, MS; Twedt, Wildlife Biologist, U.S. Department of the Interior, Geological Survey, Vicksburg, MS; Nuttle, Graduate Research Fellow, Utah State University, Department of Biology and Ecology Center, Logan, UT; Woodson, Assistant Refuge Manager, U.S. Fish and Wildlife Service, Cruger, MS; Broerman, Wildlife Biologist, U.S. Fish and Wildlife Service, Bethel, AK; and Wahome, Associate Professor, Mississippi Valley State University, Department of Natural Sciences & Environmental Health, Itta Bena, MS.

[2] Wahome, Joseph. 1999. Unpublished data. On file with: Department of Natural Sciences & Environmental Health, Mississippi Valley State University, 14000 Highway 82 West, Itta Bena, MS 38941.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

A review of literature provides a summary of expected responses of birds and small mammals to early successional habitats in MAV landscapes. Empirical observations compare responses of early successional wildlife species to four different afforestation treatments. Our primary data set involves winter bird populations in the first few years after abandonment of agricultural lands. Nuttle (1997) investigated bird communities at afforested sites of different ages in the LMAV that were established by planting oak (*Quercus* L. spp.) seedlings. Nuttle (1997) and Nuttle and Burger (1995) summarize breeding bird communities in oak plantations, some of which were 30 years old. However, we have too few data to demonstrate what happens to species during middle successional conditions occurring 20 to 60 years after initial afforestation.

METHODS

In a comparison of bird occurrence among different methods of affortestation, we focus on total abundance, density of individuals, and species richness.

Literature Analysis

The National Council of the Paper Industry for Air and Stream Improvement, Inc. (NCASI) has published annotated bibliographies on wildlife, forestry, and habitat in the East (NCASI 1993, 1999). A keyword search yielded papers dealing with wildlife response to afforestation, reforestation, forest restoration, and old-field succession in bottomland landscapes in the Southeastern United States. However, few publications treat responses to afforestation in the MAV. Studies done in other landscapes and regions can contribute insights as similar species respond to different conditions.

Empirical Observations of Winter Bird Communities

Between late October and late April from 1996 to 2000, Broerman surveyed 8 bird species on 59 afforestation sites in 9 Mississippi counties. The sites were all former agricultural land 2 to 9 years after cessation of farming activities. Some had been planted with tree seedlings, principally Nuttall oak (*Quercus nuttallii* Palmer), green ash (*Fraxinus pennsylvanica* Marshall), willow oak (Q. *phellos* L.), or water oak (*Q. nigra* L.); some had not. Herbaceous vegetation on these sites exceeded the height of woody vegetation. The survey involved using a four-wheeler to flush birds from herb or grass-dominated vegetation. During this survey he made 273 trips to the sites. Of these, 271 trips to 57 sites took place in 7 counties.

In 1999 to 2000, Hamel conducted Winter Bird Populations Studies (WBPS) (Kolb 1965) on the treatment plots in the Sharkey large-scale restoration experiment near Anguilla, Sharkey County, MS (Sharkey Site) (Schweitzer and others 1997). Hamel and Woodson made additional observations on these plots in 1998 and 1999. As a comparison of afforestation methods, the design of the Sharkey Site was randomized complete blocks design, involving three replicates each of three afforestation treatments and a natural succession and regeneration control (NAT). The treatments were: (1) sown Nuttall oak acorns (SOW), 2,500 per hectare; (2) planted Nuttall oak seedlings (PLN), 750 per hectare; and (3) planted cottonwood stem cuttings (NUR), 750 per hectare, followed by underplanting of Nuttall oak seedlings, 375 per hectare, 2 years later. Individual treatment plots were approximately 8 ha in extent. Initial establishment of the treatments was in spring 1995, with the underplanting in the NUR plots conducted in spring 1997.

Hamel conducted a WBPS on each of the 12 plots on the Sharkey Site. His WBPS counts were 30 minutes during the morning hours on 8 days from December 1999 to March 2000. He recorded all birds seen or heard in the vegetation of the plot, or actively foraging over the plot. He visited 1 of 12 plots in a random sequence. He summarized these results as average number of individuals encountered per species per hectare.

We test the hypothesis that early successional species may not benefit from rapid afforestation as much as from natural succession. Our null hypothesis for this test is that bird species dependent on herbaceous vegetation will be found in the faster growing cottonwood plantations as well as in other treatments that are accumulating woody structure more slowly. Our data for this test consist of the abundance and occurrence of the individual species recorded on the WBPS in the different treatment plots of the Sharkey Site.

We test the hypothesis that the bird community that develops in afforestation areas reflects the presence of separate elements of vegetation structure. We compare the occurrence of bird species among the treatments on the Sharkey Site in terms of their foraging preferences from Hamel (1992) for this test. The null hypothesis for this examination is that no association between foraging substrate and occurrence by treatment will be apparent among the birds.

We test the hypothesis that accumulating vegetation structure determines the occurrence of birds in afforestation treatments. Physical vegetation structure accumulates on the experimental treatments at a predictable sequence, measured by the height of the woody vegetation, NUR>PLN>SOW>NAT. Using the data on species richness and density of individual species among treatments from the Sharkey Site, we conduct analysis of variance (ANOVA).

Our analysis uses PROC GLM or PROC ANOVA, as a randomized complete block design with treatment or block and treatment as main effects (SAS Institute Inc. 1989). Significance was accepted at $p < 0.05$; means testing was conducted using Duncan's multiple comparison of means. For tests of the difference of abundance of individual species among treatments, we accepted significance at $p < 0.10$ experiment wide with sequential Bonferroni correction.

RESULTS AND DISCUSSION

Literature Analysis

Of 38 papers in the literature search (citations in table 1), 13 deal exclusively with single species response to habitat restoration, 6 with songbirds in general, 2 with waterfowl, 3 with general vertebrate response to reforestation, 3 with effects of streamside restoration on aquatic communities, 3 with effects of fire on habitat restoration, and 4 with the

Table 1—Literature review of wildlife responses to afforestation, reforestation, and forest restoration

Category	Citation
Single species responses to restoration	
Black bear (*Ursus americanus* Pallas)	Weaver and others 1990, Weaver and Pelton 1994
Red-cockaded woodpecker [*Picoides borealis* (Vieillot)]	Cantrell and others 1995, Conner and Rudolph 1995, Gaines and others 1995, Watson and others 1995, Wilson and others 1995
American woodcock (*Scolopax minor* Gmelin)	Sepik and Blumenstock 1993
Wild turkey (*Meleagris gallopavo* L.)	Dickson 1992
Florida scrub-jay [*Aphelocoma coerulescens* (Bosc)]	Root and others 1995; Schmalzer 1993, 1994; Schmalzer and others 1994
Beaver pond reforestation	Houston 1991
Deer browse affect on reforestation	Anderson and Katz 1993
Songbirds in general	Bielefeldt and Rosenfield 1994, Nuttle and Burger 1995, Ribbeck and Hunter 1994, Tomlinson 1977, Wesley and others 1976
Restoration for songbirds	Ribbeck and Hunter 1994
Reforestation and waterfowl	Kaminski and others 1993, Reinecke 1994
Bird dispersal of fruiting plants into restoration areas	Robinson and Handel 1993
General vertebrate response to reforestation	Askins and Philbrick 1987, Litvaitis 1993
Using vertebrates to assess cumulative impacts	Croonquist and Brooks 1991
Land management planning	McCullough 1994
Forest restoration of streamside zones, effect on aquatic species	Flebbe and Dolloff 1995, O'Brien-White and Thomason 1995, Sweeney 1993
Fire effects in restoration	Fitzgerald and Tanner 1992, Provencher and others 1995, Simberloff 1993
Old-growth forest restoration, biodiversity conservation	Harris and Scheck 1991; Mladenoff and others 1993, 1994; Vora 1994
Clearcuts	Hurst and Bourland 1996, Mitchell 1989

importance of reforestation to the maintenance of old-growth conditions on the landscape. Three papers discuss the role of animals, specifically beaver (*Castor canadensis* Kuhl), white-tailed deer [*Odocoileus virginianus* (Zimmermann)], and fruit-eating birds as agents modifying plant community composition. Predetermined objectives are essential in restoration planning, as in all other land management, a point made by a single paper (McCullough 1994). No paper explicitly treats early secondary succession on abandoned farmland in the MAV. The greater proportion of the papers indicate that restoration that proceeds faster from open ground to closed-canopy forest is more effective than that which proceeds more slowly.

Twelve of the papers treat bottomland hardwood or other lowland forest types. Among these, Ribbeck and Hunter

(1994) note that many bird species of highest conservation priority in the MAV are late successional species; therefore, rapid afforestation will benefit them.

Wesley and others (1976) studied winter birds in 44 cottonwood plantations aged 4, 5, or 6 years old, within primarily forested landscapes in the MAV. In an associated study, Tomlinson (1977) conducted WBPS on five plots, including three on two cottonwood plantations and two mature hardwood controls. Six species preferred the cottonwood plantations, 4 showed no preference, and 15 preferred the natural stands (Tomlinson 1977, Wesley and others 1976). Wesley and others (1976) compared bird communities in plantations to those in nearby mature natural stands, but not in plowed or harvested agricultural fields. The winter bird community of agricultural fields in the MAV is a simple one (table 2).

In 1994, Twedt conducted three WBPS in planted cottonwoods surrounded by agricultural fields on Fitler Managed Plantation, near Fitler, MS.[3] He found 35 species on the plots, 4 of which were not found by Tomlinson (1977) or Wesley and others (1976): red-shouldered hawk, northern bobwhite, chipping sparrow, and Lincoln's sparrow.

Litvaitis (1993) and Sepik and Blumenstock (1993) note that as landscapes change from primarily agricultural to primarily forested settings, species of early successional vegetation benefit in the short term and ultimately decline to the point that specific manipulative action is required to maintain their populations on the landscape. This test does not refute the hypothesis that early successional species will benefit less from afforestation than later successional species.

Empirical Observations

Survey—Table 2 lists the winter bird community Twedt observed in agricultural fields in the MAV. In his surveys of early successional habitats, Broerman tracked occurrence of eight species of birds (table 3). These species are relatively rare in the MAV and of specific conservation interest. Two species—sedge wren and Le Conte's sparrow—occurred on more than half of the surveyed sites (table 3). Each species is associated with grasslands or with herbaceous vegetation in the earliest stages of forest succession (Hamel 1992); neither appears in the cottonwood plantations studied by Wesley and others (1976), Tomlinson (1977), or Twedt.[4]

Experimental test-species occurrence—Vegetation structure on the plots at Sharkey Site differs by treatment at 5 years after establishment. On the NUR treatments, cottonwood trees approach 10 m or more in height. Nuttall oak seedlings are approximately 3 to 4 m tall in the PLN and 1 to 3 m tall in the SOW. On the NAT, few woody stems exceed the 1- to 3-m height of the herbaceous vegetation. These differences in structure are consistent with the

[3] Twedt, Daniel. 1999. Unpublished data. On file with: U.S. Department of the Interior, Geological Survey, Patuxent Wildlife Research Center, 2524 South Frontage Road, Vicksburg, MS.

[4] Twedt, Daniel. 1994. Unpublished data. On file with: U.S. Department of the Interior, Geological Survey, Patuxent Wildlife Research Center, 2524 South Frontage Road, Vicksburg, MS.

Table 2—Bird species commonly found in fallow agricultural fields in the Mississippi Delta in the winter from three Winter Bird Populations Studies conducted in 1994[a]

Common and scientific names	Bird density[b]		
	Per km²		
Great blue heron *Ardea herodias* Linn.	0.1	±	0.1
Northern harrier *Circus cyaneus* (L.)	.4	±	.2
Cooper's hawk *Accipiter cooperii* (Bonaparte)	.4	±	.4
Red-tailed hawk *Buteo jamaicensis* (Gmelin)	.7	±	.3
American kestrel *Falco sparverius* L.	.1	±	.1
Killdeer *Charadrius vociferus* L.	23.6	±	21.0
Common snipe *Gallinago gallinago* (L.)	.1	±	.1
Rock dove *Columba livia* Gmelin	9.7	±	9.7
Mourning dove *Zenaida macroura* (L.)	17.1	±	11.1
Red-bellied woodpecker *Melanerpes carolinus* (L.)	.2	±	.1
Loggerhead shrike *Lanius ludovicianus* L.	1.9	±	1.3
Blue jay *Cyanocitta cristata* (L.)	.9	±	.7
Horned lark *Eremophila alpestris* (L.)	17.3	±	12.1
European starling *Sturnus vulgaris* L.	189.0	±	189.0
Vesper sparrow *Pooecetes gramineus* (Gmelin)	.4	±	.4
Savannah sparrow *Passerculus sandwichensis* (Gmelin)	61.9	±	57.7
Song sparrow *Melospiza melodia* (Wilson)	3.8	±	2.7
White-throated sparrow *Zonotrichia albicollis* (Gmelin)	10.2	±	9.5
White-crowned sparrow *Z. leucophrys* (Forster)	.7	±	.7
Dark-eyed junco *Junco hyemalis* (L.)	.1	±	.1
Northern cardinal *Cardinalis cardinalis* (L.)	8.2	±	7.1
Red-winged blackbird *Agelaius phoeniceus* (L.)	2.3	±	1.4
Eastern meadowlark *Sturnella magna* (L.)	1.2	±	1.2
Mean density	350.4	±	210.9
Mean species richness	13.3	±	2.8

[a] Twedt, Daniel. 1994. Unpublished field notes. On file with: U.S. Department of Interior, Geological Survey, Patuxent Wildlife Research Center, 2524 South Frontage Road, Vicksburg, MS.
[b] Plus or minus standard error.

Table 3—Bird species observed on 273 trips to 59 afforestation sites in 9 counties in Mississippi from October 17, 1996 to April 24, 2000, by F. Broerman

Common and scientific names [a]	Sites	Birds/trip
	- - - - *Number* - - - -	
American bittern		
Botaurus lentiginosus (Rackett)	14	2.3
Yellow rail		
Coturnicops noveboracensis (Gmelin)	4	1.0
Sora		
Porzana carolina (L.)	16	5.1
Short-eared owl		
Asio flammeus (Pontoppidan)	21	5.6
Sedge wren		
Cistothorus platensis (Latham)	33	2.5
Marsh wren		
C. palustris (Wilson)	20	2.4
Field sparrow		
Spizella pusilla (Wilson)	10	3.1
Le Conte's sparrow		
Ammodramus leconteii (Audubon)	36	4.5

[a] In addition to the species listed, Broerman did not count individuals of savannah, swamp, and song sparrows, or of eastern meadowlarks, all of which were numerous on the surveyed sites.

intensity of the management of the plots at establishment, with the age of the propagules when planted, and with the growth rates of the different species planted. Vegetation structure has accumulated more rapidly in plots in which more intense effort was made to establish woody vegetation.

Hamel found a total of 51 bird species in the 1999 to 2000 WBPS at the Sharkey Site (table 4). We examine the occurrence of individual species as a function of their association with vegetation structure, as well as with respect to their conservation priority. The ANOVA of species richness among treatments revealed that significantly more species (30 plus or minus 4.6 S.D. vs. 11.7 plus or minus 1.8 S.D.) occurred in the NUR treatment than in the others (F = 54.2, d.f. = 5,6, P<0.0001 (table 4). The ANOVA of total abundance was similar among treatments (F = 1.13, d.f. = 5,6, P<0.43 (table 4). Thus 5 years after establishment, the treatment with the greatest development of vegetation structure (NUR) harbored the greatest number of bird species, although it did not harbor a greater number of individuals than the other treatments. Among the 28 species found only in the NUR treatment, 12 foraged on trees (Hamel 1992). Four of the five species never found in the NUR foraged on the ground or herbaceous vegetation. These results are consistent with the hypothesis that bird species occurrence reflects physical vegetation structure.

As vegetation structure develops, avian species appear that are associated with that structure, such as Eastern phoebe and yellow-rumped warbler. These birds, which forage from trees or at variable heights, were recorded only in the NUR plots in 1998 to 1999. In 1999 to 2000, these birds occurred rarely in the emerging woody vegetation of the other treatments, particularly the PLN. In both years, species associated primarily with open vegetation also occurred beneath the trees in the NUR, notably song sparrow, swamp sparrow, and red-winged blackbird.

Other species associated with open habitats, such as Northern harrier, sedge wren, savannah sparrow, Le Conte's sparrow, and eastern meadowlark rarely occurred in the NUR treatment. For example, sedge wren was found regularly in all open habitats (54 of 72 visits), but in the NUR only once (of 24 visits) in a very open spot near the edge where flooding had killed some of the planted cottonwoods. An association between occurrence and physical vegetation structure does not explain why Le Conte's sparrow occurred primarily in one spot overlapping the border of a NAT and a PLN plot. The distribution of this bird does not seem to be a straightforward response to the afforestation treatments in this experiment.

Experimental test-species conservation priority—The Partners in Flight (PIF) offers a generally accepted system for recording bird species conservation priority (Partners in Flight 2000). The PIF system gives each species a priority ranking score based on several aspects of occurrence, abundance, and threats to its population. These priority scores are recorded for physiographic areas in which a species breeds, but not for areas where it only winters (Colorado Bird Observatory 1999); the maximum possible score is 35 the minimum is 7 (Carter and others 2000). When possible, we used the PIF conservation priority scores from the MAV for this analysis; where scores were unavailable for the MAV, we used the score from a representative physiographic region in the breeding range of the species. Nuttle and others (2000) have used PIF concern scores similarly.

While bird species richness increases with vegetation structure in the Sharkey Site, conservation priority of individual species does not (table 4). Using the PIF priority rankings, several observations are suggestive. First, the average conservation priority of all 51 species recorded on the WBPS is 15.2. The average ranking of all species found in common among all treatments is a similar 15.5. Second, 28 species that were unique to the NUR treatment average 14.6, a slightly lower value. Third, 5 species not found in the NUR average 16.6, a value higher than the average. When the sedge wren is included in this group, average priority ranking of species found in the other treatments increases to 17.3. Fourth, among the six species of highest conservation priority, two never occurred in the NUR and one occurred only in the NUR. Therefore, rapid afforestation provides winter habitat for a number of species quickly, at the expense of a few high-priority species found in early successional habitats.

CONCLUSIONS

Extensive restoration of forests in the MAV may provide demonstrable, albeit unintended, benefits to birds that winter within afforested sites in early successional stages. This data set illustrates that bird species composition in the MAV

Table 4—Foraging site, abundance on different afforestation treatments, and conservation priority of bird species recorded on four afforestation treatments during winter 1999 to 2000 at the Sharkey Large-Scale Demonstration Project, Sharkey County, Mississippi

Common and scientific names	Foraging site[e]	Abundance in treatment[d]				Conservation priority rating[f]
		NAT	SOW	PLN	NUR	
Northern harrier						
Circus cyaneus (L.)	G, H	13	13	13	1	21[aa]
Red-tailed hawk						
Buteo jamaicensis (Gmelin)	G, H	2.1	9.9	2.6	2.1	12
American kestrel						
Falco sparverius L.	G	.5	0	0	.5	12
Northern bobwhite						
Colinus virginianus (L.)	G	1	0	0	0	20
Common snipe						
Gallinago gallinago (L.)	W	2.1	1	.5	0	13[aa]
American woodcock						
Scolopax minor Gmelin	G	0	0	.5	.5	19
Mourning dove						
Zenaida macroura (L.)	G	0	0	0	2.6	14
Great horned owl						
Bubo virginianus (Gmelin)	G	0	0	0	.5	12
Barred owl						
Strix varia Barton	V	0	0	0	2.6	16
Red-bellied woodpecker[a]						
Melanerpes carolinus (L.)	T	0	0	0	2.1	17
Yellow-bellied sapsucker[a]						
Sphyrapicus varius (L.)	T	0	0	0	1	16[aa]
Downy woodpecker[a]						
Picoides pubescens (L.)	T, B	0	0	0	14.6	14
Hairy woodpecker[a]						
P. villosus (L.)	T	0	0	0	1.6	14
Northern flicker[a]						
Colaptes auratus (L.)	G, T	0	0	0	**7.8**	16
Eastern phoebe						
Sayornis phoebe (Latham)	V	.5	0	2.1	**19.3**	15
Loggerhead shrike						
Lanius ludovicianus L.	G	4.7	3.1	8.3	.5	19
Blue jay						
Cyanocitta cristata (L.)	T	0	0	0	2.1	13
American crow						
Corvus brachyrhynchos Brehm	G	0	0	0	.5	10
Carolina chickadee[a]						
Poecile carolinensis (Audubon)	T	0	0	0	13	20
Tufted titmouse[b]						
Baeolophus bicolor (L.)	T	0	0	0	.5	14
Carolina wren[b]						
Thryothorus ludovicianus (Latham)	G, B	0	0	0	**9.9**	17
Winter wren[b]						
Troglodytes troglodytes (L.)	G	0	.5	0	8.8	14[bb]
Sedge wren						
Cistothorus platensis (Latham)	H	23.4	23.7	21.4	.5	21
Golden-crowned kinglet						
Regulus satrapa Lichtenstein	T	0	0	0	3.1	17[bb]

continued

Table 4—Foraging site, abundance on different afforestation treatments, and conservation priority of bird species recorded on four afforestation treatments during winter 1999 to 2000 at the Sharkey Large-Scale Demonstration Project, Sharkey County, Mississippi (continued)

Common and scientific names	Foraging site[e]	Abundance in treatment[d] NAT	SOW	PLN	NUR	Conservation priority rating[f]
Ruby-crowned kinglet[a]						
Regulus calendula (L.)	T, B	0	0	0	7.3	16[bb]
Eastern bluebird[a]						
Sialia sialis (L.)	G	0	0	0	2.1	14
Hermit thrush[a]						
Catharus guttatus (Pallas)	G, B	0	0	0	**3.1**	16[bb]
American robin[a]						
Turdus migratorius	G, B, T	0	0	0	13	9
Northern mockingbird						
Mimus polyglottos (L.)	G, B	0	0	.5	0	14
Brown thrasher						
Toxostoma rufum (L.)	G, B	0	0	0	.5	17
Cedar waxwing						
Bombycilla cedrorum Vieillot	B, T	0	0	0	2.1	12[aa]
Yellow-rumped warbler[a]						
Dendroica coronata (L.)	V, T	0	0	.5	**65.6**	16[bb]
Palm warbler						
Dendroica palmarum (Gmelin)	G, H, B	0	0	0	.5	16[cc]
Common yellowthroat						
Geothlypis trichas (L.)	H, B	.5	2.6	1	.5	16
Eastern towhee						
Pipilo erythrophthalmus (L.)	G, B	0	0	0	.5	15
Field sparrow						
Spizella pusilla (Wilson)	H	0	2.1	0	.5	20
Savannah sparrow						
Passerculus sandwichensis (Gmelin)	G, H	94.8	74.5	140.1	0	13[aa]
Le Conte's sparrow						
Ammodramus leconteii (Audubon)	H	1.6	0	4.7	0	23[aa]
Fox sparrow						
Passerella iliaca (Merrem)	G	0	.5	.5	**15.1**	12[bb]
Song sparrow[c]						
Melospiza melodia (Wilson)	G, H	68.2	78.1	53.1	30.2	12
Swamp sparrow[c]						
M. georgiana (Latham)	H	78.6	86.4	124	55.2	16[aa]
White-throated sparrow[c]						
Zonotrichia albicollis (Gmelin)	G, B	0	0	0	43.2	14[cc]
White-crowned sparrow						
Z. leucophrys (Forster)	G, H	0	0	0	**2.1**	12[bb]
Dark-eyed junco[c]						
Junco hyemalis (L.)	G	0	0	0	10.4	14[cc]
Northern cardinal						
Cardinalis cardinalis (L.)	G, B	0	0	0	5.2	12
Red-winged blackbird[c]						
Agelaius phoeniceus (L.)	G, H	241.7	18.2	63.5	252.6	12
Eastern meadowlark						
Sturnella magna (L.)	G, H	50	47.9	61.5	4.2	17

continued

Table 4—Foraging site, abundance on different afforestation treatments, and conservation priority of bird species recorded on four afforestation treatments during winter 1999 to 2000 at the Sharkey Large-Scale Demonstration Project, Sharkey County, Mississippi (continued)

Common and scientific names	Foraging site[e]	Abundance in treatment[d]				Conservation priority rating[f]
		NAT	SOW	PLN	NUR	
Rusty blackbird						
Euphagus carolinus (Müller)	G	0	0	0	8.3	16[cc]
Brewer's blackbird						
E. cyanocephalus (Wagler)	G	0	0	0	1.6	15[aa]
Common grackle						
Quiscalus quiscula (L.)	G	0	0	.5	**222.4**	16
American goldfinch[c]						
Carduelis tristis (L.)	V	3.6	0	3.1	16.7	12
Unknown species	NA	6.3	5.2	5.2	7.3	NA
Mean total density per treatment	NA	592.6	366.7	507.2	865.3	NA
Mean no. of species per treatment	NA	11	11	13	30	NA
Total species per treatment	NA	17	14	19	47	51
Unique species per treatment	NA	1	0	1	28	NA

NA = not applicable; NAT = natural succession and regeneration control; SOW = sown Nuttall oak acorns, 2500 per hectare; PLN = planted Nuttall oak seedlings, 750 per hectare; NUR = planted cottonwood stem cuttings, 750 per hectare, followed by underplanting of Nuttall oak seedlings, 375 per hectare, 2 years later.

[a] Species associated with cottonwood plantations, which are listed by Wesley and others (1976) as preferring natural stands.

[b] Species associated with cottonwood plantations, which are listed by Wesley and others (1976) as preferring neither plantations nor natural stands.

[c] Species listed by Wesley and others (1976) as preferring cottonwood plantations.

[d] Entries reflect mean abundance as birds per km^2. Boldfaced numbers indicate mean abundance different from other treatments by one-way analysis of variance with treatment as main effect, adjusted for an experiment-wide error rate of 0.10 using sequential Bonferroni correction. Standard error values available on request from Hamel (lead author of publication).

[e] Foraging sites listed as B = bush, G = ground, H = herbaceous vegetation, T = tree, V = various heights, W = water (Hamel 1992).

[f] Conservation priority ratings are the Partners in Flight Concern Scores (Colorado Bird Observatory 1999) for the Mississippi Alluvial Valley, with the exception of [aa] = Drift Prairie, [bb] = Central Rocky Mountains, and [cc] = Great Lakes Transition physiographic regions.

follows vegetation structure and development rate in afforestation of abandoned agricultural lands. However, as woody vegetation develops, some bird species of herbaceous vegetation disappear. Perhaps more importantly, the early successional avian species that specialize on herbaceous vegetation are of higher than average conservation priority among the birds found in afforestation areas.

Different species of mammals and birds respond positively to the structure available at different times during succession. Thus, managers must decide on the species and communities they wish to favor. Winter avian communities of early successional stages have been considered heretofore only in passing. However, because birds occur in areas where vegetation structure is dominated by nonagricultural herbaceous plants, they include species otherwise rare or absent from the MAV landscape. To provide bird habitat through the full successional sere, proactive managers may wish to dedicate certain areas to natural succession or to maintenance of herbaceous vegetation.

Forest land managers and biodiversity preservationists make clear the essential relationship between management practices and management objectives (McCullough 1994, Noss 1999). Management (1) is a conscious, goal-directed activity with goals specified in advance, and (2) employs practices to achieve goals within an acceptable period of time. Managers can apply a variety of afforestation methods under different conditions to achieve different management objectives. Managerial discretion influences the rate of structural development of afforestation efforts. Inclusion of management goals to produce habitats in afforestation for early successional species is certainly possible, as is inclusion of management goals to maximize the rate of development of forest vegetation structure and the birds associated with it.

ACKNOWLEDGMENTS

This work has benefited from the generous field assistance of the staff and residents of Sunflower Landing, Dublin, MS. Mark Woodrey and Jeff Buler of the Mississippi Museum of

Natural Science aided in the fieldwork, as did a number of other collaborators on the Sharkey Large-Scale Restoration Project.

LITERATURE CITED

Anderson, R.C.; Katz, A.J. 1993. Recovery of browse-sensitive tree species following release from white-tailed deer *Odocoileus virginianus* Zimmerman browsing pressure. Biological Conservation. 63(3): 203–208.

Askins, R.A.; Philbrick, M.J. 1987. Effect of changes in regional forest abundance on the decline and recovery of a forest bird community. Wilson Bulletin. 99(1): 7–21.

Bauch, J.; Dunisch, O.; Gasparotto, L., eds. 1999. Investigations on tree species suitable for the recultivation of degraded land areas in central Amazonia: Brazilian-German cooperation within the project 'studies on human impacts on forests and floodplains in the tropics' (SHIFT) supported by the Ministry of Education, Science, Research, and Technology, Bonn, FRG (0339638) and the Conselho Nacional de Desenvolvimento Cientifico e Tecnologica (CNPq), Brasilia, Brazil (ENV 42) and Instituto Brasileiro do Meio Ambiente e Recursos Naturais Renovaveis (IBAMA). No. 193. Hamburg, Germany: Mitteilungen der Bundesforschungsanstalt fur Forst- und Holzwirtschaft. 138 p.

Bielefeldt, J.; Rosenfield, R.N. 1994. Summer birds of conifer plantations in southeastern Wisconsin. Passenger Pigeon. 56(2): 123–135.

Boyle, J.R. 1999. Planted forests: views and viewpoints. New Forests. 17(1/3): 5–9.

Burgess, P.J. 1999. Effects of agroforestry on farm biodiversity in the UK. Scottish Forestry. 53(1): 24–27.

Cannell, M.G.R. 1999a. Environmental impacts of forest monocultures: water use, acidification, wildlife conservation, and carbon storage. New Forests. 17(1/3): 239–262.

Cannell, M.G.R. 1999b. Forests, Kyoto and climate. Outlook on agriculture. 28(3): 171–177.

Cantrell, M.A.; Britcher, J.J.; Hoffman, E.L. 1995. Red-cockaded woodpecker management initiatives at Fort Bragg Military Installation. In: Kulhavy, D.L.; Hooper, R.G.; Costa, R., eds. Red-cockaded woodpecker: recovery, ecology and management. Red-cockaded woodpecker symposium III: species recovery, ecology and management; 1993 January 24–28; North Charleston, SC. Nacogdoches, TX: Stephen F. Austin State University, Center for Applied Studies, College of Forestry: 89–97.

Carter, M.F.; Hunter, W.C.; Pashley, D.N.; Rosenberg, K.V. 2000. Setting conservation priorities for landbirds in the United States: the partners in flight approach. Auk. 117(2): 541–548.

Chang, ChingCheng. 1999. Carbon sequestration cost by afforestation in Taiwan. Environmental Economics and Policy Studies. 2(3): 199–213.

Colorado Bird Observatory. 2000. Colorado Bird Observatory PIF database. Barr Lake State Park, CO: Colorado Bird Observatory. [Available on the Web: http://www.cbobirds.org/pif/index.html].

Conner, R.N.; Rudolph, D.C. 1995. Losses of red-cockaded woodpecker cavity trees to southern pine beetles. Wilson Bulletin. 107(1): 81–92.

Croonquist, M.J.; Brooks, R.P. 1991. Use of avian and mammalian guilds as indicators of cumulative impacts in riparian-wetland areas. Environmental Management. 15(5): 701–714.

DeGraaf, R.M. 1987. Managing northern hardwoods for breeding birds. In: Nyland, R.D., ed. Proceedings of the silvicultural symposium managing northern hardwoods; 1986 June 23–25; Syracuse, NY. Misc. Publ. 13 (ESF 87–002). Syracuse, NY: Syracuse University, Faculty of Forestry: 348–362.

DeGraaf, R.M.; Yamasaki, M.; Leak, W.B.; Lanier, J.B. 1992. New England wildlife: management of forested habitats. Gen. Tech. Rep. NE–144. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 271 p.

Dickson, J.G., ed. 1992. The wild turkey: biology and management. Harrisburg, PA: Stackpole Books. 463 p.

Fitzgerald, S.M.; Tanner, G.W. 1992. Avian community response to fire and mechanical shrub control in south Florida. Journal of Range Management. 45(4): 396–400.

Flebbe, P.A.; Dolloff, C.A. 1995. Trout use of woody debris and habitat in Appalachian wilderness streams of North Carolina. North American Journal of Fisheries Management. 15: 579–590.

Gaines, G.D.; Franzreb, K.E.; Allen, D.H. [and others]. 1995. Red-cockaded woodpecker management on the Savannah River Site: a management/research success story. In: Kulhavy, D.L.; Hooper, R.G.; Costa, R., eds. Red-cockaded woodpecker: recovery, ecology and management. Red-cockaded woodpecker symposium III: species recovery, ecology and management; 1993 January 24–28; North Charleston, SC. Nacogdoches, TX: Stephen F. Austin State University, Center for Applied Studies, College of Forestry: 81–88.

Hamel, P.B. 1992. A land manager's guide to the birds of the South. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region; Chapel Hill, NC: The Nature Conservancy, Southeastern Region. 367 p.

Harrington, C.A. 1999. Forests planted for ecosystem restoration or conservation. New Forests. 17(1/3): 175–190.

Harris, L.D.; Scheck, J. 1991. From implications to applications: the dispersal corridor principle applied to the conservation of biological diversity. In: Saunders, D.A.; Hobbs, R.J., eds. Nature conservation 2: the role of corridors. Chipping Norton, Australia: Surrey Beatty & Sons: 189–220.

Helmer, W. 1999. [Establishment of broadleaved floodplain forest]. Opkomst van het hardhoutooibos. Nederlands Bosbouwtijdschrift. 71(3): 130–134.

Houston, A.E. 1991. Beaver control and reforestation of drained beaver impoundments. Knoxville, TN: University of Tennessee. 143 p. Ph.D. dissertation.

Hurst, G.A.; Bourland, T.R. 1996. Breeding birds on bottomland hardwood regeneration areas on the Delta National Forest. Journal of Field Ornithology. 67: 181–187.

James, F.C. 1971. Ordinations of habitat relationships among breeding birds. Wilson Bulletin. 83: 215–236.

Joslin, J.D.; Schoenholtz, S.H. 1997. Measuring the environmental effects of converting cropland to short-rotation woody crops: a research approach. Biomass and Bioenergy. 13(4/5): 301–311.

Kaminski, R.M.; Alexander, R.W.; Leopold, B.D. 1993. Wood duck and mallard winter microhabitats in Mississippi hardwood bottomlands. Journal of Wildlife Management. 57(3): 562–570.

Kolb, H. 1965. The Audubon winter bird population study. Audubon Field Notes. 19(3): 432–434.

Litvaitis, J.A. 1993. Response of early successional vertebrates to historic changes in land use. Conservation Biology. 7(4): 866–873.

McCullough, D.R. 1994. Importance of population data in forest management planning. Forestry Chronicle. 70(5): 533–537.

Mitchell, L.J. 1989. Effects of clearcutting and reforestation on breeding bird communities of baldcypress-tupelo wetlands. Raleigh, NC: North Carolina State University. 92 p. M.S. thesis.

Mladenoff, D.J.; White, M.A.; Crow, T.R.; Pastor, J. 1994. Applying principles of landscape design and management to integrate old-growth forest enhancement and commodity use. Conservation Biology. 8(3): 752–762.

Mladenoff, D.J.; White, M.A.; Pastor, J.; Crow, T.R. 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. Ecological Applications. 3(2): 294–306.

Mueller, A.J.; Loesch, C.R.; Twedt, D.J. 1995. Development of management objectives for breeding birds in the Mississippi Alluvial Valley. In: Bonney, R.; Pashley, D.N.; Cooper, R.J.; Niles, L., eds. Proceedings of the Partners in Flight international workshop strategies for bird conservation: the Partners in Flight planning process; 1995 October 1–5; Cape May, NJ. Ithaca, NY: Cornell Lab of Ornithology. [Various pagination]. [Available on the Web: http://www.birds.cornell.edu/pifcapemay/mueller.htm].

National Council of the Paper Industry for Air and Stream Improvement, Inc. (NCASI). 1993. Forestry, wildlife, and habitat in the East: an annotated bibliography, 1986–1990. NCASI Tech. Bull. 651. Research Triangle Park, NC: National Council of the Paper Industry for Air and Stream Improvement, Inc. 340 p.

National Council of the Paper Industry for Air and Stream Improvement, Inc. (NCASI). 1999. Forestry, wildlife, and habitat in the East: an annotated bibliography, 1991–1995. NCASI Tech. Bull. 781. Research Triangle Park, NC: National Council of the Paper Industry for Air and Stream Improvement, Inc. 753 p.

Niyaz, S.; Nigi, T.; Koshika, K. 1999. [Growth of planted subsidized trees in natural forests in the central Tianshan Mountains and damage caused by grazing cattle]. Research Bulletin of the Hokkaido University Forests. 56(1): 94–104.

Noss, R.F. 1999. Assessing and monitoring forest biodiversity: a suggested framework and indicators. Forest Ecology and Management. 115(2/3): 135–146.

Nuttle, T.J. 1997. Response of breeding bird communities to afforestation of hardwood bottomland sites in Mississippi. Mississippi State, MS: Mississippi State University. 68 p. M.S. thesis.

Nuttle, T.J.; Burger, L.W. 1995. Breeding bird response to hardwood bottomland restoration in the Mississippi Alluvial Valley. [Abstract]. In: Program of Wildlife Society second annual conference: excellence in wildlife stewardship through science and education; 1995 September 12–17; Portland, OR: 92–93.

Nuttle, T.; Leidolf, A.; Burger, L.W. 2000. Using Partners in Flight species concern scores to compare the conservation value of bird communities [Abstract]. In: One hundred eighteenth stated meeting of the American Ornithologists' Union; 2000 August 14–19; St. John's, Newfoundland, Canada. St. John's, Newfoundland, Canada: American Ornithologists' Union: 26.

O'Brien-White, S.; Thomason, C.S. 1995. Evaluating fish habitat in a South Carolina watershed using GIS. Proceedings of the annual conference of the Southeastern Association of Fish and Wildlife Agencies; 1995 September 23–27; Nashville, TN. Tallahassee, FL: Southeastern Association of Fish and Wildlife Agencies. 49: 153–166.

Pande, V.C.; Nambiar, K.T.N.; Singh, H.B. 1999. Afforestation on community land in the semi-arid tropics of Gujarat—an economic analysis. Indian Forester. 125(2): 212–218.

Partners in Flight. 2000. Partners in Flight – U.S. home page. Patuxent National Wildlife Refuge, MD: U.S. Department of the Interior, Biological Resources Division. [Various pagination]. [Available on the Web: http://www.partnersinflight.org].

Provencher, L.; Demarest, D.; Brennan, L. [and others]. 1995. Avian community responses to herbaceous invertebrates and vegetation in burned old-growth and fire-suppressed longleaf pine habitats: experimental design and pretreatment results [Abstract]. In: Program of Wildlife Society second annual conference: excellence in wildlife stewardship through science and education; 1995 September 12–17; Portland, OR. Portland, OR: The Wildlife Society: 105.

Reinecke, K.J. 1994. Regional waterfowl habitat trends and implications for neotropical migratory birds. In: Smith, W.P.; Pashley, D.N., comps., eds. A workshop to resolve conflicts in the conservation of migratory landbirds in bottomland hardwood forests; 1993 August 9–10; Tallulah, LA. Gen. Tech. Rep. SO–114. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 20–24.

Ribbeck, K.F.; Hunter, W.C. 1994. Is reforestation an adequate restoration of bottomland hardwood functions for the needs of neotropical migratory birds? In: Smith, W.P.; Pashley, D.N., comps., eds. A workshop to resolve conflicts in the conservation of migratory landbirds in bottomland hardwood forests; 1993 August 9–10; Tallulah, LA. Gen. Tech. Rep. SO–114. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 25–26.

Robinson, G.R.; Handel, S.N. 1993. Forest restoration on a closed landfill: rapid addition of new species by bird dispersal. Conservation Biology. 7(2): 271–278.

Root, K.V.; Breininger, D.R.; Bowman, R.; Swain, H.M. 1995. What can population viability analysis (PVA) tell us about the probable success of a spatially explicit reserve design: the Florida scrub jay in Brevard County as a case study [Abstract]. Bulletin of the Ecological Society of America. 76(2 Suppl. Part 2): 229.

SAS Institute Inc. 1989. SAS/STAT user's guide. Version 6. 4th ed. Cary, NC: SAS Institute Inc. 846 p. Vol. 2.

Schmalzer, P.A. 1993. Characteristics of long unburned stands of oak-saw palmetto scrub [Abstract]. Bulletin of the Ecological Society of America. 74(2 Suppl.): 428.

Schmalzer, P.A. 1994. Restoring unburned shrublands: an example using cutting and burning in oak-saw palmetto scrub [Abstract]. Bulletin of the Ecological Society of America. 75(2 Suppl.): 206.

Schmalzer, P.A.; Breininger, D.R.; Adrian, F.W. [and others]. 1994. Development and implementation of a scrub habitat compensation plan for Kennedy Space Center. Tech. Memo. 109202. Kennedy Space Center, FL: National Aeronautics and Space Administration. 54 p.

Scholtens, L.J.R. 1998. Environmental, developmental and financial risks of tropical timber plantation investment funds. Natural Resources Forum. 22(4): 271–277.

Schweitzer, C.J.; Stanturf, J.A.; Shepard, J.P. [and others]. 1997. Large-scale comparison of reforestation techniques commonly used in the Lower Mississippi Alluvial Valley: first year results. In: Pallardy, S.G.; Cecich, R.A.; Garrett, H.; Johnson, P.S., eds. Proceedings of the 11th central hardwood forest conference; 1997 March 23–26; Columbia, MO. Gen. Tech. Rep. NC–188. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station: 313–320.

Sepik, G.F.; Blumenstock, B. 1993. Management and research on the American woodcock at the Moosehorn National Wildlife Refuge [Abstract]. In: Longcore, J.R.; Sepik, G.F., eds. Proceedings of the eighth American woodcock symposium; 1993; West Lafayette, IN. Biol. Rep. 16. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service, National Biological Survey: 136–137.

Simberloff, D. 1993. Effects of fragmentation on some Florida ecosystems, and how to redress them. In: Saunders, D.A.; Hobbs, R.J.; Ehrlich, P.R., eds. Nature conservation 3: reconstruction of fragmented ecosystems. Chipping Norton, Australia: Surrey Beatty & Sons: 179–187.

Sweeney, B.W. 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in Eastern North America. Proceedings of the Academy of Natural Sciences of Philadelphia. 144: 291–340.

Vora, R.S. 1994. Integrating old-growth forest into managed landscapes: a northern Great Lakes perspective. Natural Areas Journal. 14(2): 113–123.

Watson, J.C.; Hooper, R.G.; Carlson, D.L. [and others]. 1995. Restoration of the red-cockaded woodpecker population on the Francis Marion National Forest: three years post Hugo. In: Kulhavy, D.L.; Hooper, R.G.; Costa, R., eds. Red-cockaded woodpecker: recovery, ecology and management. Red-cockaded woodpecker symposium III: species recovery, ecology and management; 1993 January 24–28; North Charleston, SC. Nacogdoches, TX: Stephen F. Austin State University, Center for Applied Studies, College of Forestry: 172–182.

Weaver, K.M.; Pelton, M.R. 1994. Denning ecology of black bears in the Tensas River basin of Louisiana. International Conference on Bear Research and Management. 9(1): 427–433.

Weaver, K.M.; Tabberer, D.K.; Moore, L.U., Jr. [and others]. 1990. Bottomland hardwood forest management for black bears in Louisiana. Proceedings of the annual conference of the Southeastern Association of Fish and Wildlife Agencies; 1990 October 21–24; Richmond, VA. Tallahassee, FL: Southeastern Association of Fish and Wildlife Agencies. 44: 342–350.

Wesley, D.E.; Perkins, C.J.; Sullivan, D.A. 1976. Preliminary observations of cottonwood plantations as wildlife habitat. In: Thielges, B.A.; Land, S.B., Jr. eds. Proceedings of the symposium on eastern cottonwood and related species; 1976 September 28–October 2; Greenville, MS. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 460–476.

Willoughby, I.; McDonald, H.G. 1999. Vegetation management in farm forestry: a comparison of alternative methods of inter-row management. Forestry (Oxford). 72(2): 109–121.

Wilson, C.W.; Masters, R.E.; Bukenhofer, G.A. 1995. Breeding bird response to pine-grassland community restoration for red-cockaded woodpeckers. Journal of Wildlife Management. 59(1): 56–67.

GLOBE, STUDENT INQUIRY, AND LEARNING COMMUNITIES

C.L. Henzel[1]

Abstract—The Global Learning and Observations to Benefit the Environment (GLOBE) database is a web-based archive of environmental data gathered by K through12 students in over 85 countries. The data are gathered under protocols developed by research scientists specializing in various fields of earth science. Students gather information, then enter and visualize the data via the Internet. GLOBE's potential is to provide two major components that affect sustainability and conservation: quality earth system data and science education opportunities for students. First, GLOBE maintains a quality, accessible database of millions of environmental data at a geographic scale never before attempted. Second, GLOBE is a real opportunity for students to learn field study and scientific research methods. GLOBE provides easy access to data and visualization tools via the Internet to help teachers and students understand, interpret, and ask relevant questions about the earth system. Students who have a greater understanding of how the environmental system in their community operates can make more informed decisions on sustainability and conservation issues on a larger scale.

INTRODUCTION

Global Learning and Observations to Benefit the Environment (GLOBE) is an international science and education program for K through 12 students. Initiated in 1994, GLOBE is currently supported by 87 countries and over 100 regional support groups, called GLOBE franchises, across the United States. Scientists involved in the program develop detailed guidelines for the collection of environmental data, monitor data quality, use the data in their own research, and mentor students. The GLOBE students, under the guidance of trained GLOBE teachers, collect data using the science protocols, input the data into a central database via the Internet, and use the data for their own research.

The dual role of GLOBE as a science and education program provides a unique educational advantage for students. The data students collect are of a quality usable by earth scientists and are easily accessible to anyone via the World Wide Web. Dr. Roger Bales, Principal Investigator for GLOBE Hydrology at the University of Arizona, has said, "GLOBE hydrology measurements fill a critical gap in water monitoring and assessment efforts in the United States and worldwide. GLOBE schools sample many smaller streams and lakes that are underrepresented in the professional monitoring programs run by government agencies. We have now reached the point where GLOBE constitutes one of the largest water-quality networks in the United States and is certainly the one with the most readily available data" (GLOBE offline, in press).

The GLOBE scientists along with other scientists who have already discovered and are using this unique database have a real interest in encouraging and mentoring these students. The students are provided with motivation to participate in science when they know the data they collect are 'real' and being used—not just an assignment to be discarded after grading. The value of what the students are doing has been illustrated by the number of scientists who are not directly involved in GLOBE, but are using GLOBE data or who have approached GLOBE for help in collecting special measurements.

David Verbyla of the University of Alaska, Fairbanks, asked GLOBE students to provide information on budburst in their area States." There is no network of on-the-ground plant phenology observations to validate growing season models and estimates derived from satellite data. GLOBE schools' plant phenology observations will be the only source of wide-spread growing season observations for research to better understand climate change" (GLOBE offline, in press).

Students are further motivated by the opportunity to provide valuable input into their own communities through their own research. Visualization tools provided on the GLOBE Web site allow students to graph and map GLOBE data in real time. New educational materials guide students through the process of research. The techniques of data collection and assessment, such as the creation of classified and assessed land-cover maps from Landsat images, are often the most current and accurate tools available for assessment of these regions.

EARTH AS A SYSTEM

GLOBE data students are collecting fall under a variety of environmental fields including atmosphere, soil, land cover, and hydrology. An investigation team or teams that include science and education co-principal investigators lead each of these areas. The GLOBE Hydrology Investigation Team includes three scientists from the University of Arizona: Dr. Roger Bales, Dr. Martha Conklin, and Dr. Katrina Mangin, and an education co-pi, Cyndy Henzel. The teams work together to ensure both the scientific integrity of the data and the educational quality of the learning materials.

[1] Education Development, Scientists Organized to Integrate Learning, Inc., 4920 S. Browning Lane, Tucson, AZ 85746.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

Students are encouraged to make their observations in the context of the earth as a system. Rather than only collecting point data for a water site, students are asked to also collect and analyze soil samples, record daily atmosphere measurements, and create detailed land cover maps. The water data they collect can then be analyzed in the context of their specific site and other inputs to their system. Scientists are finding this wide spatial coverage of integrated data collection an invaluable tool for filling in gaps for earth system research. Students gain a greater appreciation of how the intricate involvement of many systems plays a role in understanding an environment. For instance, how does the character of your soil and type of land cover help explain the amount of input and quality of water at your hydrology site? What happens if the land cover or pattern of precipitation changes?

GLOBE recognized early in the development of the program that the exploration of watersheds was an excellent tool to use to gain a greater understanding of the interplay of the earth system on a local, regional, or even global scale. Students were encouraged to identify their local watershed, supply contextual information about their hydrology measurements, and interpret their findings in light of other measurements and environmental observations. Specific educational activities to achieve these objectives were difficult to develop in the first years of GLOBE, however, due to the lack of long-term data sets to use. But, by the beginning of this year, the GLOBE student archive contained over 4 million data points.

STUDENT INQUIRY

In 1999, GLOBE educators began development of materials to help guide teachers and students in the use of GLOBE data in student research. Through a number of research seminars for teachers, it was found that although most science teachers taught the steps of the scientific method, few felt comfortable leading students to develop appropriate research questions, collect and analyze data, and draw conclusions. The visualization tools provided on the GLOBE Web site were being underutilized: students were collecting and inputting data but never looking at it or using it for their own research.

Development of guiding materials to help students do their own research easily met the dual science and education roles within GLOBE. Scientists recognized that students exploring the database and using the data in their own research were more likely to recognize the value of quality data and consistency in data collection. Educators in schools across the country and around the world were scrambling to meet new standards that included original science research or inquiry.

Dr. Leon Lederman, Nobel Laureate: “GLOBE is the quintessentially ideal program for involving kids in science. GLOBE teaches science content and also the process of science. Facts are important, but the younger students are, the more important it is to learn the process of science. Science isn’t about providing answers as much as it is about asking questions. As a hands-on program, GLOBE provides opportunities for teachers and scientists to talk informally with kids and get them to ask questions” (GLOBE offline, in press).

Data collected by GLOBE students in the area of hydrology include measurements of transparency, water temperature, pH, conductivity or salinity, dissolved oxygen, alkalinity, and nitrate. New measurements for freshwater and saltwater macroinvertebrates are currently being developed and should be implemented by the end of 2000. These measurements, combined with those from the other GLOBE investigation areas, provide a comprehensive set of data from which students can develop research questions and projects. GLOBE, recognizing that any dataset is more valuable if it can be combined with other datasets, has provided reference datasets on the GLOBE Web site from other sources that allow students easy access. In addition, GLOBE data can easily be pulled out in table format to export into a spreadsheet or other analysis tool such as Geographic Information Systems (GIS).

GLOBE-A-THONS AND LONG-TERM MONITORING

The power of GLOBE student inquiry will hopefully be seen in two areas. First, students are being encouraged to collect quality, consistent environmental research data. GLOBE was developed with the idea of creating a large, accessible environmental data archive that would have both a wide spatial scale and long-term monitoring. We are also now encouraging GLOBE-a-thons, or intensive cooperative data collection by GLOBE students and perhaps other environmental organizations within a region. These may provide a snapshot of an entire watershed that may be then revisited over time.

Second, GLOBE student inquiry is filling a need within the schools for original student research. New materials will help guide students through the research process. The GLOBE Web site is constantly being improved with the addition of new tools for student visualization and research, more student data, and more access to other datasets. In addition, the Web site now provides space for publication of student investigations, communication being recognized as an essential component of the research process. Perhaps most important of all, GLOBE can help influence those students to become the mentoring GLOBE scientists for future generations.

THE GLOBE COMMUNITY

In 1995 when GLOBE trained the first teachers in the program, it envisioned having a GLOBE teacher in every school. As the program has grown, so has the vision. Because of the broad, rich content of the program and broad applications at all age levels, GLOBE began to encourage the training of teams of teachers in schools. These schools quickly found the value of local community resources such as parent volunteers, professional volunteers from other environmental groups and Federal, State, and local government agencies. Some of these groups, including such organizations as universities, museums, and parks, developed more formal ties to GLOBE and have become GLOBE franchises. The franchises take on the responsibility of training teachers and providing follow-up to local schools. Franchises have also taken the initiative in taking GLOBE beyond what can be provided at the Federal level. Franchises in Alabama and Montana have hosted GIS conferences for GLOBE franchises and trainers. Franchises in Arkansas, South Carolina, Idaho, and Arizona have worked together to promote the second International GLOBE Student Conference to be held in Arkansas in June of 2000. Internationally, GLOBE countries have also been active in creating the community. For example, Finland hosted the first International student conference in 1998. Croatia recently hosted a land-cover symposium.

What this all means to education and the sustainability of the important data collection being done by the students is that the students are now part of a vast learning community. Students, teachers, parents, governments, universities, and others are all working toward a common goal. Students are not just on the receiving end with facts to learn and homework to turn in. They are an important part of a local effort to understand their own backyards as well as a global effort to understand how the jigsaw of local regions fits together.

CONCLUSION

The study of water has long been used as a key to developing student interest and skills in science. Many programs regionally, nationally, and internationally, have provided and still provide quality materials to help students gain an understanding of key science concepts like the water cycle and the measurement of water quality. Others have concentrated their efforts on the importance of developing an appreciation for the environment and conservation of resources. Although GLOBE obviously has some overlapping objectives with many other environmental programs, the strength of the program is not realized by competing for student time and resources.

GLOBE teaches students to ask questions based on observation, gather data in a scientifically rigorous manner, analyze data, and draw conclusions based on those results, and communicate the results to others in the community. This is the way to make good decisions about the environment. Good decisions will lead to sustainable development.

REFERENCES

GLOBE offline. [In press]. Scientists are talking about GLOBE; spring 2000. Washington, DC: The GLOBE Program.

USING KEPONE TO EXEMPLIFY THE IMPORTANCE OF NATURAL VARIABILITY IN ESTIMATING EXPOSURE TO TOXIC CHEMICALS FROM AQUATIC ENVIRONMENTS

Robert J. Huggett[1]

Abstract—Kepone, decachlorooctahydro-l, 3, 4-metheno-2H-cyclobuta (cd) pentalen-2-one, is a known mammalian carcinogen. From at least 1967 to 1975 when production stopped, it contaminated the Chesapeake Bay. Action levels for kepone in seafood were established by the U.S. Environmental Protection Agency, and various species of finfish, oysters, *Crassostrea virginica*, and crabs, *Callinectes sapidus*, often were found to exceed those levels. Detailed sampling and analyses of biota showed that interspecies variability in concentrations often exceeded an order of magnitude. Further examination of the data showed that much of the variability could be explained by factors such as sex, spawning cycle, and migratory patterns. Estimates of human exposure to kepone-contaminated seafood, and, hence, estimates of risk from consuming it, were quite inaccurate unless natural variability was considered. On the positive side, an understanding of the factors controlling natural variability provided alternative risk-management options to minimize risk by decreasing exposure without totally prohibiting harvest or consumption of the resource.

INTRODUCTION

The major components of a human health-risk assessment are relatively well known. The “Red Book” of 1983 examined how to assess risk to humans from cancer-causing chemicals (NRC 1983). The paradigm set forth in this paper has been expanded and modified over time, but the basic principles have remained. To assess health risk from chemicals, one must determine two things: the potential effects of the chemical and exposure to it.

In order to estimate a chemical’s effect, the hazard and dose/response of the substance must be examined. Hazard is the inherent ability of a substance to cause harm. For instance, a metabolite of benzo(a)pyrene (BaP), a compound formed by the incomplete combustion of fossil fuel, is thought to be a human carcinogen. So, the hazard of the 7-8-dihydroxy-9-10-epoxide of BaP is that it can cause cancer. Dose/response, on the other hand, estimates how much of the substance is required, at the right place and time, to cause a manifestation of the effect.

Once one knows what harm a chemical may inflict and how much is required to do it, an estimate of exposure to that chemical allows a characterization of the risk involved.

Action levels or tolerance levels are often defined as the maximum concentration of a hazardous substance one can have in food without experiencing the ill effects of the substance. In effect, these are regulatory or risk-management tools used by the U.S. Food and Drug Administration (FDA) or the U.S. Environmental Protection Agency (EPA) to limit exposure to hazardous substances from ingestion of contaminated food.

There are numerous difficulties involved in estimating the hazard and dose/response of a chemical to humans. Are the animal models used in toxicity tests appropriate for extrapolating to humans? Is it a zero threshold chemical? Are there sensitive subpopulations, etc.? There are also numerous difficulties in estimating exposure to the chemicals; these are often due to the inhomogeneous nature of the substance in question in the environment. Because a determination of risk can be no more accurate than the most inaccurate number that goes into calculating it (the Significant Figures Paradigm), one must strive to obtain the very best estimates of all components of the assessment process, including exposure.

Natural variability, a particularly difficult aspect of accurately determining exposure to environmental contaminants, is the focus of this manuscript. The much-studied chemical contamination by kepone of the James River system is used to exemplify various aspects of this phenomenon.

BACKGROUND

The James River is a major tributary to the Chesapeake Bay. With its drainage basin of 25 600 km², it delivers 16 percent of the freshwater to the bay. The river is tidal for 160 km, from its mouth near Norfolk, VA, to the fall line at Richmond, VA. The salinity at its mouth is usually near 25 parts per trillion (ppt), and brackish waters extend about 50 km upstream, depending on precipitation within the drainage basin. From this point upstream to Richmond, it is a tidal freshwater river, and from this point to the mouth, it is technically an estuary.

Approximately 110 km upstream, in the tidal freshwater portion of the river, lies Hopewell, VA. From 1966 until 1974,

[1] Vice President, Office for Research and Graduate Studies, Michigan State University, 232 Administration Building, East Lansing, MI 48824-1046.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

the pesticide kepone was manufactured there by Allied Chemical Company and Life Science Products, Inc. Over this period, thousands of kg of kepone entered the river via municipal waste effluents and other point and nonpoint sources. Investigation of the river's bottom sediments led to an estimate of 14 000 kg of kepone being sorbed to the particles. This became the major source of contamination to the biota of the James River and the Chesapeake Bay (Bender and Huggett 1984).

Kepone, (decachlorooctahydro-1, 3, 4-metheno-2H-cyclobuta (cd) pentelen-2-one), once intended to control fire ants and cockroaches and eventually used for the banana root borer in Central and South America, is extremely persistent and bioaccumulative (Huggett and Bender 1980). It is also a suspected human carcinogen.

Because kepone was found to be present in the edible portion of finfish, crabs, clams, and oysters inhabiting the James River and adjacent Chesapeake Bay, the EPA, under the auspices of the Federal Insecticide, Fungicide, Rodenticide Act, established action levels for the compound. Based on NOELS from rodent testing and average food consumption patterns, the following levels were originally established: finfish: 0.1 mg per kilogram wet weight; crabs: 0.4 mg per kilogram wet weight; oysters: 0.3 mg per kilogram wet weight. The action level for finfish was later raised to its present level of 0.3 mg per kilogram.

It should be noted that action levels or tolerance levels are enforceable by the FDA only if the contaminated commodity crosses State lines. If the contaminated seafood, in this case, was harvested and sold in Virginia, the Federal government had no jurisdiction. Even so, the Commonwealth of Virginia adopted the Federal action levels for instate public health protection.

The problem that faced the State health regulators was to determine accurately the concentrations that existed in the seafood and to permit or ban fishing in certain areas or for certain species in order to control exposure to kepone. This required an accurate assessment of the kepone concentration distributions in various species by location and time. The findings of numerous sampling expeditions and what is commonly called natural variability are described and exemplified in the remainder of this paper.

KEPONE NATURAL VARIABILITY

Many, if not most, of the commercial species of finfish that inhabit the Chesapeake Bay spend part of each year in the ocean. In the spring, they re-enter the bay and spend several weeks near its mouth, and hence, the James River, to equilibrate to the lower salinities. Some organisms enter the James River and remain throughout the season, and others enter and leave again, migrating to other portions of the bay. Concentrations of kepone in 91 blue fish collected in June 1976 from one location in the bay are presented in figure 1 (Huggett and others 1980). There is an obvious biomodal distribution of concentrations with approximately 40 percent of the animals being above the action level of 0.1 mg per kilogram. These more contaminated animals presumably spent some time in the James River before leaving and grouping with fish that had not. Knowledge of

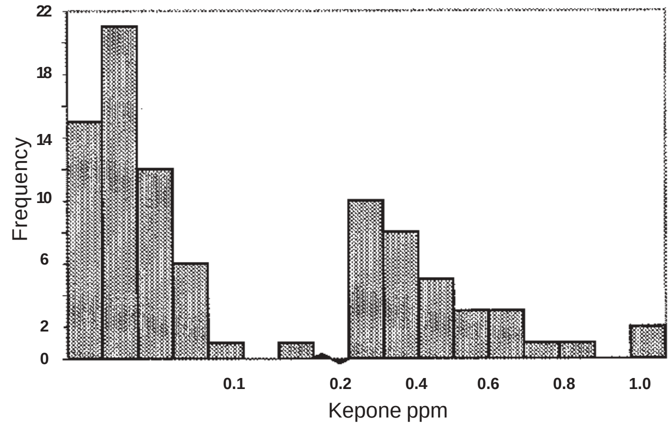


Figure 1—Histogram of kepone concentrations in bluefish from the James River showing bimodal distribution.

the distribution of concentrations greatly increases the accuracy of estimates of exposure from consumption of the fish.

Concentrations of kepone in the edible flesh of aquatic organisms can also be influenced by the sex of the animal, with males often being more contaminated. This is presumably due to the substance being partitioned to the lipid-rich eggs of the females (Huggett 1981). Figure 2 exemplifies this phenomenon. For the American shad, *Alosa sapidissima*, 41 percent of the males contained more than 0.1 mg per kilogram whereas only 6 percent of the females had levels as high.

Blue crabs, *Callenectes sepidus*, from the James River were extensively sampled and analyzed. The difference in concentrations between males and females was striking (fig. 2). As was the case with the American shad, the females continued to show low levels relative to the males. All of the male crabs sampled exceeded the action level of 0.4 mg per kilogram whereas only 11 percent of the females were high. This finding resulted in Virginia allowing commercial harvest and sale of female crabs but not males.

The body burden of kepone is somewhat proportional to the duration that the animal is exposed to the substance. Croakers, *Micropogon undulates*, enter the bay from the ocean in early spring and return in the fall. Croakers collected from the James River throughout the summer of 1976 show that kepone concentrations increase with exposure time with no plateauing noted (fig.3) (Huggett and others 1980).

The physiology of the organism, as influenced by temperature, can also affect tissue concentrations. This is exemplified in figure 4, which shows kepone concentrations in eastern oysters, *Crassostrea virginica*, from 1976 through 1980 (Bender and others 1985). During winter, when water temperatures drop, the organisms have less planktonic food available. Their water-filtering rate is drastically reduced, and they metabolize stored body fat. In doing so, some of the lipidophyllic kepone is liberated. As water temperatures rise, feeding increases, and, concurrently, kepone levels increase, thus giving rise to a yearly cycle of concentrations.

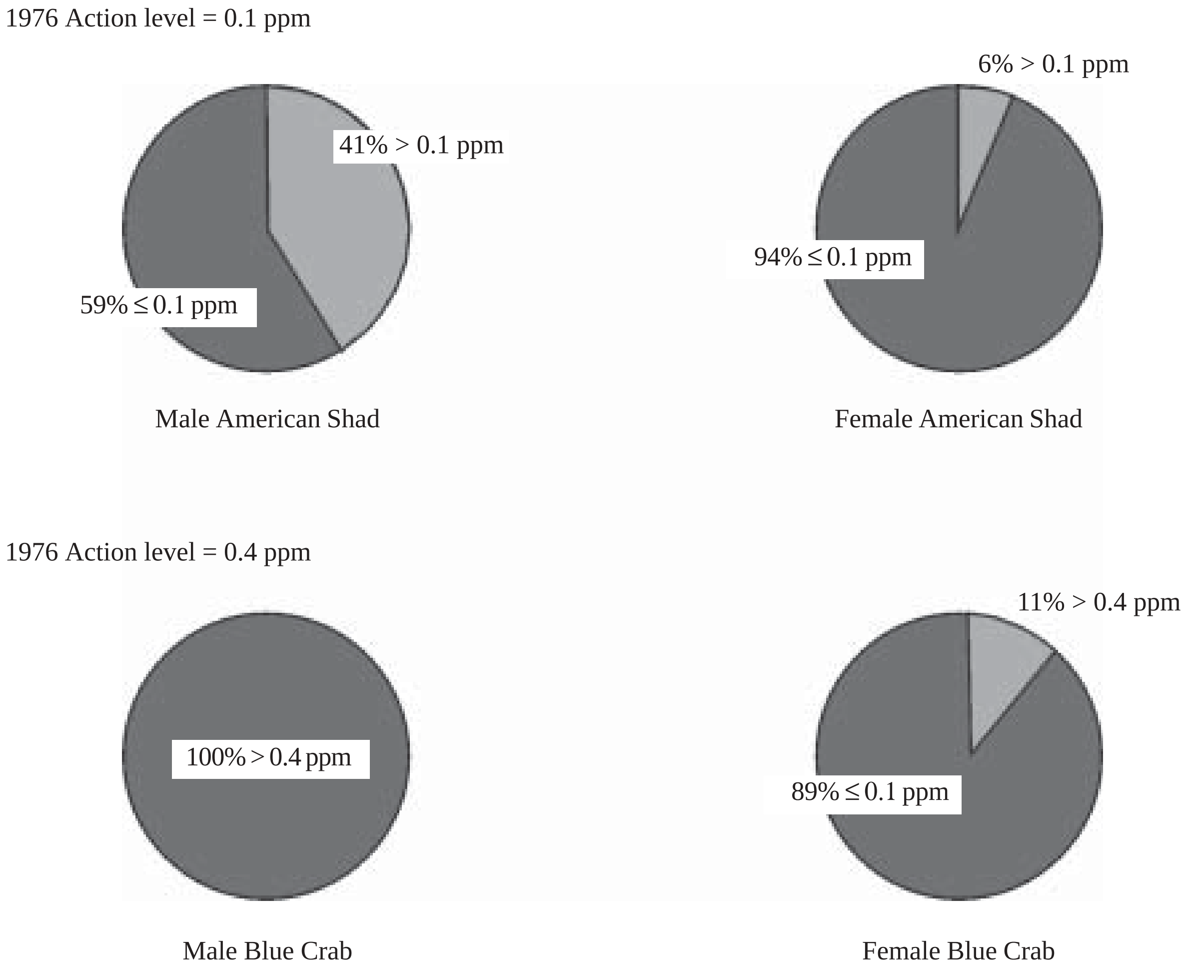


Figure 2—Concentrations of kepone in male and female blue crabs, *Callinectes sapidus*, and American shad, *Alosa sapidissima*.

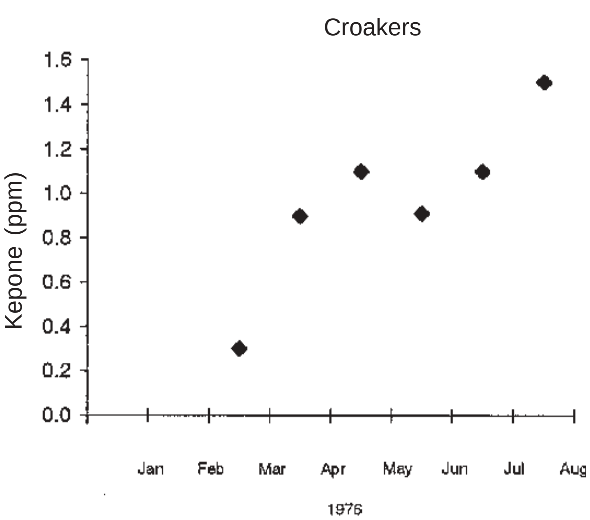


Figure 3—Kepone concentrations in croakers, *Micropogon undulatus*, collected from the James River in 1976.

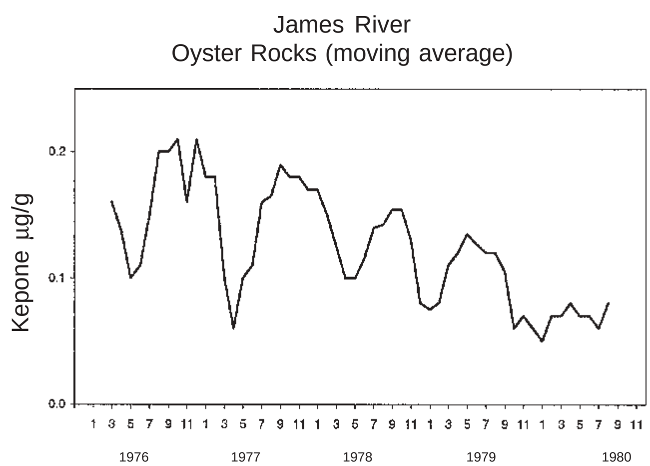


Figure 4—Kepone residues in oysters vs. time.

Figure 4 also shows an overall decline in kepone concentrations over time due to the burial of kepone in the sediments after production of the pesticide was stopped in 1975.

Since the production of kepone stopped in the mid-1970s, the concentrations in seafood have slowly decreased. The harvest of finfish in the James River was totally or partially banned until 1988 when concentrations fell below action levels (Huggett 1989). Table 1 presents data that show that by 1996 the levels were less than one quarter of the action level (Unger, M.A. 2000. Personal communication. Glouster Point, VA: Virginia Institute of Marine Science).

Table 1—Yearly averages of kepone concentrations in striped bass (*Morone saxatilis*) from the James River

Year	Animals analyzed	Mean kepone	Standard deviation
	- - No. - -	*Mg/kg*	
1989	40	0.13	0.03
1990	40	.20	.06
1991	40	.22	.07
1992	40	.20	.07
1993	40	.12	.11
1994	60	.17	.08
1995	60	.16	.07
1996	55	.07	.05

CONCLUSION

Studies designed to determine human exposure to hazardous chemicals from consumption of contaminated foods need to take into account natural variability. Failure to do so may result in inadvertently placing consumers at risk or, on the other hand, imposing harvest restrictions more severe than necessary to protect public health. It is also important to consider natural variability when assessing the effects of hazardous chemicals on the organisms themselves. Partitioning of lipidophyllic compounds to gametes poses a relatively high risk of toxic reproductive effects.

REFERENCES

Bender, M.E.; Huggett, R.J. 1984. Fate and effects of kepone in the James River estuary. In: E. Hodgson, ed. Reviews in environmental toxicology. New York: Elsevier Science Publishers: 373–393.

Bender, M.E.; Huggett, R.J.; Slone, H.D. 1985. Kepone uptake: a comparison of field and laboratory data. In: White, H.H., ed. Concepts in Maine pollution measurements. College Park, MD: Maryland Sea Grant Publications: 279–290.

Huggett, R.J. 1981. The importance of natural variability in the total analytical scheme. Biomedical Mass Spectrometry. 8(9): 416–418.

Huggett, R.J. 1989. Kepone and the James River. In: Contaminated marine sediments—assessment and remediation. Washington, DC: National Research Council. National Academy Press: 417–424.

Huggett, R.J.; Bender, M.E. 1980. Kepone in the James River. Environmental Science and Technology. 14(8): 918–923.

Huggett, R.J.; Nichols, M.M.; Bender, M.E. 1980. Kepone contamination of the James River estuary. In: Baker, R.A., ed. Contaminants and sediments. Ann Arbor, MI: Ann Arbor Science Publishers. 1: 33–52.

National Research Council. 1983. Risk assessment in the Federal government: managing the process. Washington, DC: National Research Council. National Academy Press. 191 p.

POTENTIAL EFFECTS OF RESTORATION ON BIOGEOCHEMICAL FUNCTIONS OF BOTTOMLAND HARDWOOD ECOSYSTEMS

Graeme Lockaby and John Stanturf [1]

Abstract—The concept of wetland restoration carries multiple meanings and implications. The scientific usage of the term connotes re-establishment of wetland functions, and often it is the functions, which society deems most valuable, that receive highest focus. Arguably, among key wetland functions, the highest societal value may be linked with the biogeochemical transformation or filtration function, a key contributor to maintenance of water quality. This function requires flow-through hydrology such as that associated with unimpeded riverine forests, and, consequently, its re-establishment is negated by the absence of such hydrology. Consequently, afforestation of former agricultural areas, which were protected from flow-through hydrology, i.e., flooding, by dikes, ditches, etc., cannot be considered restoration in a complete sense unless some semblance of flow-through hydrology is also restored. The term quasi-depressional wetland is suggested as being appropriate for afforested areas where hydrologic restoration is unfeasible.

Rising awareness of the important functions associated with wetlands as well as the extent of historic losses have stimulated societal perceptions of wetlands as rapidly diminishing, yet, highly useful natural resources. Consequently, public support (in principle) for restoration of wetlands is quite high. However, a complex array of socioeconomic factors as well as ecological uncertainties continues to cause confusion in regard to implementation of restoration efforts.

The term ecosystem restoration may vary in connotation depending on the legal, political, scientific, or aesthetic context in which it is used. We suggest that, regardless of context, wetland restoration implies re-establishment of key traits or functions that are valued by individuals and society or both. Thus, the most common way to evaluate particular restoration efforts is to compare functionality between restored systems and a reference, which hopefully represents the predisturbance state.

On former floodplains, afforestation of land previously used for agriculture might be considered as wetland restoration. Regeneration of tree seedlings on areas that were once intensively cultivated may produce several effects that are beneficial to society. These include reduction of sediment export, improvement of wildlife habitat, production of wood and fiber, and enhanced sequestration of carbon. However, despite the value placed on these benefits by society, they are not unique to wetlands.

It is arguable that the most important biogeochemical function that some forest wetlands perform is filtration or removal from waters of sediment, inorganic forms of nutrients, and other substances considered impurities by society. Obviously, this function is predominantly associated with riverine systems, which develop and operate under a lateral or flow-through type of hydrology so that water continually passes through the wetland filter.

The biogeochemical filtration function of riverine forests is closely linked with the inherently open nature of their geochemical cycles. Nonriverine forest ecosystems as well as those in which contact with their aquatic components has been restricted tend to develop more closed geochemical cycles as they aggrade. The degree of closure is associated with the magnitude of outputs in relation to inputs. Whereas an upland forest may theoretically aggrade to a point where outputs or losses are negligible, i.e., a tight or closed cycle, unaltered riverine forests are somewhat unique in that their geochemical cycles do not approach this degree of closure.

The open-ended geochemical cycles of the riverine forests allow exportation of significant quantities of dissolved organic carbon (DOC), a critical source of energy for aquatic food webs. This biogeochemical function is probably secondary to filtration in terms of societal value but, nonetheless, is quite important.

Consequently, many would agree that riverine forests, apart from those physically separated from their river systems, display two critically important biogeochemical functions: filtration and DOC export. It is also apparent that both functions are intrinsically linked with the flow-through hydrology (and open geochemical cycles) that typifies riverine systems. We suggest that these very simple ideas can be used to examine the degree to which wetlands such as bottomland hardwoods are being restored by particular types of activities.

Afforestation of agricultural fields located on sites historically occupied by riverine forests is currently a widespread activity at the national as well as regional level (Clewel and Lea

[1] Lockaby, Professor, School of Forestry and Wildlife Sciences, Auburn University, Auburn, AL 36849; Stanturf, Soil Scientist, Center for Bottomland Hardwoods Research, Southern Research Station, P.O. Box 227, Stoneville, MS 38776, Present address: Forestry Sciences Laboratory, 320 Green St., Athens, GA 30602-2044.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

1990; Sharitz 1992; Stanturf and others 1998, 2000). In the Mississippi Delta region of the Southern United States, such afforestation is often referred to as bottomland hardwood restoration (King and Keeland 1999, Stanturf and others 2000b). As previously mentioned, there are several excellent reasons for pursuing such activities. However, the topic at hand dictates that we examine this practice from a biogeochemical standpoint.

In the vast majority of these cases, the practice of agriculture on floodplains necessitated protection from annual flooding in addition to removal of forests. Most often, dikes, levees, or ditches or all were used to physically separate the aquatic and terrestrial components. Obviously, these structures were intended to modify or eliminate the flow-through hydrology that typifies riverine forests. In most cases following agricultural abandonment, socioeconomic factors cause removal of the protective structures to be unfeasible.

An exception is the major restoration effort associated with the Kissimmee River and adjoining lands in Florida (http://www.saj.usace.army.mil/ dp/kissimmee.html). In that case, approximately 100,000 acres of floodplain are being restored to a state subject to natural flooding. However, as would be expected, restorations at this scale and degree of complexity are very costly, i.e., $372 million (http://www.fcn.state.fl.us/eog/govdocs/opbenv/ saveglades/everglades/html/kissimee.htm).

It should be noted that some afforestation areas retain flow through hydrology. These include areas within the batture (unprotected floodplain of the Mississippi River) or those tributaries such as the Yazoo subject to backwater flooding. However, as previously mentioned, restoration of natural, riverine flooding regimes is rarely feasible. As a result, the aggrading forests on afforested sites cannot exhibit open-ended biogeochemistry and, consequently, will not function as biogeochemical filters or exporters of DOC. This limitation of afforestation activities has been previously recognized (Allen 1997, King and Keeland 1999). Suggested remedies have included plugging drainage ditches or building water control structures on portions of the afforested sites so that controlled flooding can be induced in much the same way that it is applied within greentree reservoirs. On public land such as national wildlife refuges and national forests, relatively large areas have been restored in this fashion as greentree reservoirs, moist soil management units, or permanent water bodies. In addition, it is not uncommon for some flooding to occur on lower lying portions from accumulation of precipitation.

However, it should be recognized that these types of flooding reflect those more commonly associated with depressional wetlands than with riverine. Since it is commonly acknowledged that hydrology represents the dominant controlling process within a wetland (Mitsch and Gosselink 1993), the nature of the hydrology in the restored system is a critical factor in our evaluation of restoration success. Thus, the significance of restoring a former riverine system to quasi-depressional hydrology lies in biogeochemical differences between depressional vs. riverine wetlands.

The chief contrast between riverine vs. depressional hydrology is a predominance of lateral vs. vertical flows respectively (Brown 1990). Consequently, biogeochemical functions are aligned primarily with precipitation, evapotranspiration, and infiltration in basins (fig. 1) as opposed to sheet flow in riverine systems (fig. 2). Lugo and

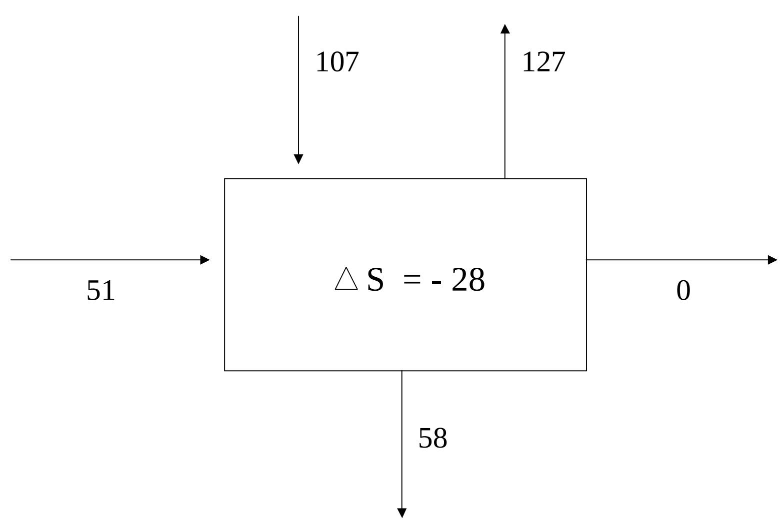


Figure 1—Hydrologic budget for a typical depressional wetland, a cypress dome in Florida. Units are in centimeters per year; depressional wetlands typically have greater inflow than outflow of surface water (Brown 1990).

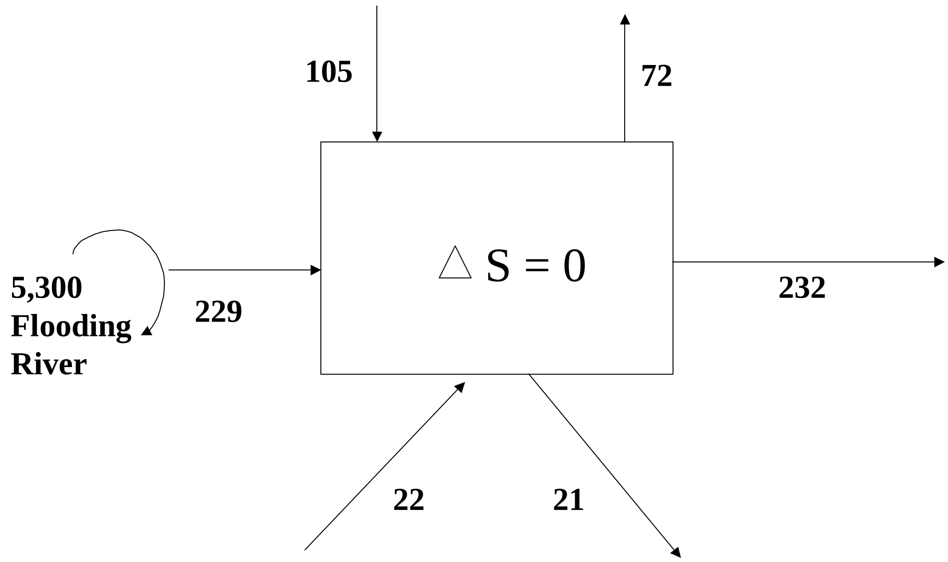


Figure 2—Hydrologic budget for a typical riverine wetland, an alluvial cypress swamp in Illinois. Units are in centimeters per year; riverine wetlands typically exhibit flow-through hydrology (Mitsch and Gosselink 1993).

others (1990) felt that this hydrologic distinction drives differences in terms of how elements are accumulated in the two wetland types as well as which elements would be limiting to NPP; e.g., theoretically P would be more limiting in basins due to the scarcity of vertical input mechanisms.

However, we should also be cautious in assuming that this type of afforestation will result in the formation of typical basin wetlands. There likely will be functional distinctions between a basin wetland, which formed and functions under true basin hydrology, vs. an afforested field, which may occasionally accumulate water. We suggest use of the term quasi-depressional system to indicate this distinction. Because these quasi-depressional systems remain isolated from riverine influences, they contribute little to biogeochemical filtering or to DOC export to aquatic systems.

It is apparent that some basin wetland forests are capable of accumulating nutrients when subjected to elevated inputs (Ewel and Odum 1984). There is also evidence that infiltration export is generally low (Brown 1990), and, consequently, some basins are regarded as nutrient accumulation zones rather than filters or exporters of DOC. However, their lack of hydrologic linkages to major landscapes minimizes their biogeochemical value to society. In particular, the quasi-depressional systems created behind levees or dikes may have even less opportunity to accumulate nutrient inputs because they may exist in a more hydrologically isolated state than true basins.

The U.S. Environmental Protection Agency (EPA) has identified the Yazoo-Mississippi Basin as an area of significant concern for surface and ground water quality. Although surface water runoff in the Lower Mississippi Alluvial Valley (LMAV) contributes only 20 percent of the nitrate loading implicated in the expansion of the hypoxic zone in the Gulf of Mexico, the EPA is expected to focus significant resources on the LMAV to improve water quality. Policy alternatives under consideration include reducing nitrogen use by 20 to 40 percent and converting agricultural land to forests in an effort to restore and enhance natural denitrification processes (EPA 1999). The assumption is made that restoration (afforestation) of bottomland hardwood forests will reduce nutrient export into the Gulf. This will be true to the extent that a potential source of nutrients will be reduced by changing land use from row crop agriculture to forests (Thornton and others 1998). But the restored system will play at most a small role as a nutrient filter unless it is hydrologically linked to a riverine system. Thus, the greater benefit, in terms of nutrient filtration, would come from conversions within the active floodplain of small rivers throughout the basin of nutrient origin and from buffer strips planted along drainage ways (Castelle and Johnson 2000, Castelle and others 1994). Whereas forested buffer strips may provide advantages to the landowner over grass or herbaceous strips, the relative effectiveness of forest vs. grass buffers on nutrient filtration remains uncertain.

Whereas public support for wetland restoration has traditionally been strong, that support may weaken when local populations are made aware of the implications of true hydrologic restoration within a former floodplain. Such restoration will almost certainly necessitate elimination of land uses that are incompatible with significant flooding, e.g., farming, habitation, etc. Thus, added to the engineering cost of levee removal, etc., are significant compensation expenditures to landowners. In addition, the unpopular nature of a major taking of private property will also weaken political support for State and Federal appropriations to support the projects. While we neither advocate nor support this approach, the obstacles associated with such an endeavor appear to be significant, and, thus, will make implementation of true riverine restorations difficult to achieve.

In summary, it is recognized that there are many very worthwhile reasons for conversion of former farmland to hardwood forests. However, continued growth in population levels dictates that utilization of landscapes will grow increasingly complex in the years to come. Similarly, it will become critically important that professionals as well as the general public be aware of the specific nature of the changes that are induced (or not induced) by farmland conversion. Only the development and widespread dissemination of that knowledge will protect society from unrealistic expectations, which lead to landscape-level mistakes in natural resource management.

REFERENCES

Allen, J.A. 1997. Reforestation of bottomland hardwoods and the issue of woody species diversity. Restoration Ecology. 5(2): 125–134.

Brown, S. 1990. Structure and dynamics of basin forested wetlands in North America. In: Lugo, A.E.; Brinson, M.; Brown, S., eds. Ecosystems of the world 15: forested wetlands. New York: Elsevier: 171–199.

Castelle, A.J.; Johnson, A.W. 2000. Riparian vegetation effectiveness. NCASI Tech. Bull. 799. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc. 26 p.

Castelle, A.J.; Johnson, A.W.; Conolly, C. 1994. Wetland and stream buffer size requirements—a review. Journal of Environmental Quality. 23(5): 878–882.

Clewell, A.F.; Lea, R. 1990. Creation and restoration of forested wetland vegetation in the Southeastern U.S. In: Kusler, J.A.; Kentula, M.E., eds. Wetland creation and restoration. Wetland creation and restoration: the status of the science. Washington, DC: Island Press: 195–231.

Dahm, C.N.; Cummins, K.W.; Valett, H.M.; Coleman, R.L. 1995. An ecosystem view of the restoration of the Kissimmee River. Restoration Ecology. 3: 225–238.

Ewel, K.C.; Odum, H.T. 1984. Cypress swamps. Gainesville, FL: University Presses of Florida. 472 p.

King, S.L.; Keeland, B.D. 1999. Evaluation of reforestation in the Lower Mississippi River Alluvial Valley. Restoration Ecology. 7: 348–359.

Lugo, A.E.; Brinson, M.M.; Brown, S. 1990. Synthesis and search for paradigms in wetland ecology. In: Lugo, A.E.; Brinson, M.; Brown, S., eds. Ecosystems of the world 15: forested wetlands. New York: Elsevier: 447–460.

Mitsch, W.J.; Gosselink, J.G. 1993. Wetlands. 2d ed. New York: Van Nostrand Reinhold. 722 p.

Sharitz, R.R. 1992. Bottomland hardwood wetland restoration in the Mississippi drainage. In: Restoration of aquatic systems: science, technology, and public policy. Washington, DC: National Academy Press: 496–605.

Stanturf, J.A.; Schoenholtz, S.; Schweitzer, C.J.; Shepard, J.P. 2001. Achieving restoration success: myths in bottomland hardwood forests. Restoration Ecology. 9(2): 189–200.

Thornton, F.C.; Joslin, J.D.; Bock, B.R. [and others]. 1998. Environmental effects of growing woody crops on agricultural land: first-year effects on erosion and water quality. Biomass and Bioenergy. 15: 57–69.

Toth, L.A. 1995. Principles and guidelines for restoration of river/floodplain ecosystems—Kissimmee River, Florida. In: Cairns, J., ed. Rehabilitating damaged ecosystems. 2d ed. Boca Raton, FL: Lewis Publishers: 49–73.

Toth, L.A.; Anderson, D.H. 1998. Developing expectations for ecosystem restoration. In: Wadsworth, K.G., ed. Transactions of the 63rd North American wildlife and natural resources conference; 1998 March 20–24; Orlando, FL. Washington, DC: Wildlife Management Institute: 122–134.

U.S. Environmental Protection Agency. 1999. Integrated assessment of hypoxia in the northern Gulf of Mexico. Stennis Space Center, MS: U.S. Environmental Protection Agency, Gulf of Mexico Program, Stennis Space Flight Center. 54 p. [Draft for public comment].

FOREST RESTORATION IN THE NORDIC COUNTRIES

Palle Madsen, Ása L. Arad•ttir, Emile Gardiner, Pelle Gemmel, Kåre Lund Høie, Magnus Löf, John Stanturf, Peter Tigerstedt, Hardi Tullus, Sauli Valkonen, and Veiko Uri[1]

Abstract—The Nordic countries include Iceland, Norway, Sweden, Finland, and Denmark, which range from lat. 54° in southern Denmark to lat. 72° at North Cape, Norway. This region is dominated by the boreal coniferous vegetational zone. Denmark and southern Sweden are, however, located in the deciduous (nemoral) forest zone, whereas the interior part of Iceland and the high altitudes of Norway and Sweden are in the mountainous zone. Forests cover 1, 11, 28, 60, and 66 percent of the land area in Iceland, Denmark, Norway, Sweden, and Finland, respectively. Traditional forestry has mainly concentrated on conifers in the boreal and the nemoral zones. Increased concern about nature conservation and sustainable land use along with economic constraints and reduced softwood timber prices have led to increased focus on growing and regenerating broadleaves. Planting nursery-grown seedlings is the common and preferred method for forest restoration and afforestation. However, this technique is expensive unless wide spacing is used. Direct seeding is a less expensive alternative that may still give acceptable results. This regeneration practice has been examined for a number of broadleaved species within a collaborative project including forest researchers from Mississippi, Estonia, and the five Nordic countries. Results from this work are presented, and conclusions are drawn for forest management and future research and development in this field.

INTRODUCTION

This paper summarizes the context of forest restoration in the Nordic countries and Estonia including the debate on changing aims of future forestry and silviculture. Furthermore, we present the objectives and the main conclusions of a cooperative project between the Nordic countries, Estonia, and the United States on direct seeding of broadleaves. The cooperative project was supported from both national funds from each of the participating countries as well as funds from the Nordic Forest Research Cooperation Committee (SNS).

CONTEXT OF FOREST RESTORATION

The Nordic countries include Iceland, Norway, Sweden, Finland, and Denmark, which range in latitude from 54° N in southern Denmark to 72° N at North Cape, Norway. Estonia is one of the Baltic countries and is located south of Finland (fig. 1). Forest cover ranges from a very limited part of the land area in Iceland to a dominant part in Sweden and Finland (table 1).

The boreal coniferous vegetational zone dominates the area. Denmark and southern Sweden are, however, located in the deciduous (nemoral) forest zone, whereas the interior part of Iceland and the high altitudes of Norway and Sweden are in the mountainous zone (Walter 1985). Forest industry plays an important role for the Swedish, Finnish, Norwegian, and Estonian economy, but the economical importance of wood production is rather marginal for Denmark and Iceland. In general, most of the forestland is private property, which is a relatively new ownership status in Estonia.

The term forest restoration covers very different silvicultural challenges in the Nordic countries. In Iceland, afforestation on totally barren and degraded land, practically deserts, is one important approach to forest restoration. Special attention is paid to restoration of the birch woodlands, which covered more than 25 percent of the land area at the time of settlement in the 10th century (Sigurdsson 1977). In contrast to the Icelandic situation, afforestation efforts in the other Nordic countries and Estonia occur on fertile farmland. Aims of afforestation are rather different within and between the countries. In Finland, Sweden, and Norway the expected extent of afforestation is rather limited (table 1) and serves mainly as an alternative land use to small scale, inefficient agriculture. In Estonia, many small farms have been turned over to the descendants of former owners. They have no experience and knowledge of farming practices, so forestry may be of interest to these landowners as a low-cost, land-use alternative. Consequently, a significant increase in forestland on abandoned farmland is expected in Estonia. In Denmark, the goal of the afforestation program is to double the forested area within one tree rotation (about 100 years). There are several aims of this program including:

- increased concern for sustainability, nature conservation, and biodiversity;
- protection of ground water resources;

[1] Madsen, Senior Researcher, Danish Centre for Forest, Landscape, and Planning (FSL), Kvak Møllevej 31, 7100 Vejle, Denmark; Arad•ttir, Division Leader, Soil Conservation Service, Gunnarsholt, 850, Hella, Iceland; Gardiner, Research Forester, and Stanturf, Project Leader, Center for Bottomland Hardwoods Research, Southern Research Station, P.O. Box 227, Stoneville, MS 38766, Present address: Forestry Sciences Laboratory, 320 Green St., Athens, GA 30602-2044; Gemmel, Professor, and Löf, Researcher, Swedish University of Agricultural Sciences, S-230-53, Alnarp, Sweden; Høie, Senior Researcher, The Norwegian Crop Research Institute, Plant Protection Centre, N-1432 Ås, Norway; Tigerstedt, Professor, University of Helsinki, FIN-00014 Helsinki, Finland; Tullus, Professor, and Uri, Researcher, Estonian Agricultural University, Tartu, Estonia; Valkonen, Senior Researcher, Finnish Forest Research Institute, FIN-01301 Vantaa, Finland.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

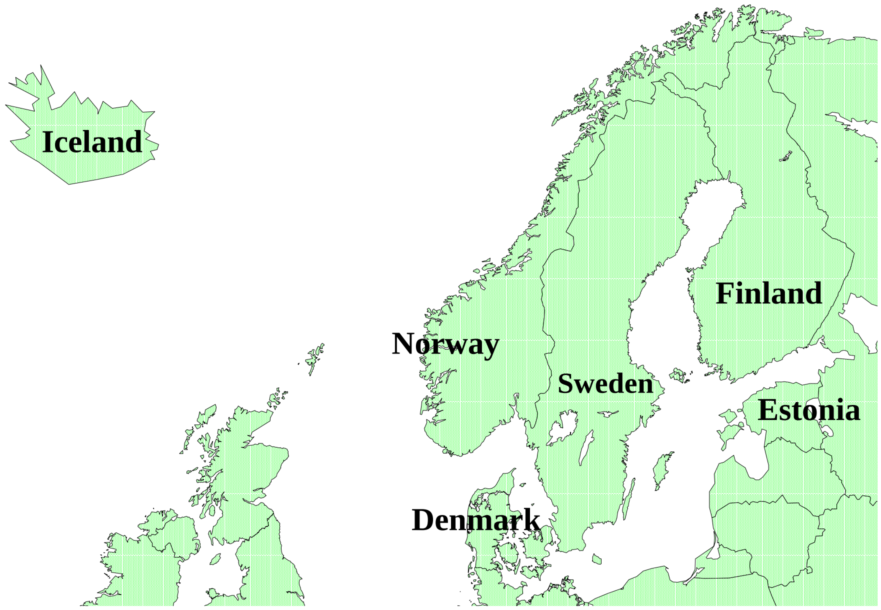


Figure 1—The Nordic countries and Estonia.

Table 1—Forestland and woodlands in the Nordic countries and Estonia[a]

Country	Forestland and woodlands	Forestland, % of total land area	Expected afforestation, % of total land area	Private- (includ. companies) owned forestland
	- - 1,000 km² - -			*- - - - - - - Percent - - - - - - -*
Finland	201	66	1	71
Sweden	244	60	1	78
Estonia	21	50	7	35
Norway	87	28	< 1	85
Denmark	5	11	11	69
Iceland	1	1	2–5	70

[a] Woodlands are roughly nonproductive in terms of wood production. In Iceland, in particular, they cover a significant part of the forested land. Expected afforestation is estimated based on personal judgements for the next 50 to 100 years (Finland, Sweden, Norway, Iceland, and Estonia), or it is based on a political decision (Denmark). In Estonia, the government aims at a 50-percent private forestland rate.

- improvement of recreational values of the landscape; and
- reduction of subsidized agricultural production.

Moreover, at present in the deciduous zone (southern Sweden and Denmark), conifer plantations are being transformed into broadleaf stands, particularly on better soils. Today, conifer plantations cover about 65 to 80 percent of the forestland in this area, and the main species is Norway spruce [*Picea abies* (L.) Karst]. Norway spruce is outside or on the edge of its natural range in Denmark and southern Sweden. The conifers were initially planted due to their high productivity and low cultivation costs. However, they have shown in many cases poor wind stability and health depending on site and species. Such catastrophes not only destroy the existing forests but also leave an area with considerable regeneration problems. The forest climate is lost, and weeds, frost, drought, and wind may cause problems for regeneration.

As illustrated above, the background for forest restoration with respect to ecological conditions and forest industry is very diverse in the Nordic countries. However, cultural, political, and economical similarities of the countries form a common platform for changes and aims in the forestry of these countries.

CHALLENGES IN SILVICULTURE

Traditional forestry in the Nordic countries has concentrated mainly on growing conifers for timber and pulp in the boreal and the nemoral zones. During the past two decades, increased concern over ecological sustainability, nature conservation, and sustainable land use in conjunction with economical constraints and reduced softwood timber prices has led to an increased focus on the use of broadleaf species and close-to-nature forest management. The more diverse and multifunctional aims of forestry have emphasized the need for the forests to be flexible with respect to future outputs as wood and nonwood products and values. Additionally, the importance of such flexibility is stressed by the long production periods in Nordic forestry. The rotation length usually ranges between 50 to 120 years, depending on site, species, silvicultural practices, etc. The problem is that the main role of the future forests cannot be predicted precisely.

The stability of forests can be expressed in terms of resistance and resilience of the forest ecosystem (Larsen 1995). Poor resistance may express a considerable susceptibility of the forest to be damaged or destroyed by strong winds, drought, fire, or a complex of factors. Poor resilience may entail considerable regeneration problems because of the difficulties regaining the forested condition after a catastrophe. Active forest management is crucial for maintaining a stable forest ecosystem. Otherwise, there is considerable risk that the forest will not fulfil the aims of its establishment.

Traditional and close-to-nature forest management is probably best described as relative degrees of forest management, which may overlap. Stands resulting from traditional forest management are usually homogeneous with respect to species and age. In traditional forest management, stands are harvested when the average tree has reached maturity, and planting is the dominant regeneration method with natural regeneration utilized to a lesser extent. Simple administration, apparently cost-effective methods, and convincing economical models are probably the main reasons for the widespread use of this forest-management approach.

In close-to-nature forest management, natural forest ecosystem processes are used and supported by the silviculturist to achieve inexpensive natural regeneration and optimise production value of each tree during the later part of the rotation period. The latter is performed by harvesting single trees as they reach a target diameter; e.g., 57 cm d.b.h. for beech (*Fagus sylvatica* L.) in Denmark and southern Sweden. Close-to-nature forestry is also characterized by the high priority given to the use of site-adapted species in heterogeneous forest structures, which will gradually develop as a consequence of the regeneration system. It should be stressed that stand heterogeneity is not the goal itself but a way to allocate the species to various soil conditions and improve forest-floor conditions for natural regeneration.

Proponents of the principles of close-to-nature forest management regard it as a silvicultural system that may support the multifunctionality of a forest. This may include economic benefits from the extensive use of natural regeneration and single-tree management.

WHY IS DIRECT SEEDING INTERESTING?

In close-to-nature forestry, how is the strong focus on minimizing cultivation costs and promoting site-adapted species relevant to forest restoration? Low cultivation costs may be essential for the long-term economic success of close-to-nature silviculture. Natural regeneration is, in many cases, not a reliable and realistic option on sites that lack relevant seed sources or appropriate germination environments. Wind-dispersed species like birch (*Betula* spp. L.) may, however, have a potential for natural regeneration on bare land. Examples of successful natural regeneration of birch on former farmland or on clear-felled areas are well known. It was common practice in traditional forestry to remove regenerated birch to reduce competition on planted conifers. In Iceland, it is proposed to plant birch at very wide spacings or in clusters to distribute seed sources in the landscape and thereby support the restoration of the birch woodlands (Arad•ttir 1991).

Planting seedlings is the most common practice of today's forest restoration programs. It is a reliable but expensive method, particularly with respect to establishment of broadleaves. It typically costs $0.40 to $0.70 to purchase and plant seedlings. Usually 5,000 broadleaf seedlings are planted per hectare. Direct seeding is an alternative regeneration method that may be less costly. For $1.00 it is possible to buy 20 to 100 acorns or 100 to 200 beechnuts; but, in bumper crop years, the prices may decrease. Consequently, higher seedling densities could potentially be established with direct seeding at lower costs than planting seedlings. High-density stands could also have a positive impact on future stand quality. Table 2 outlines the approximate ranges of costs related to direct seeding and planting of beech and oak. The oak may be either pedunculate oak (*Quercus robur* L.) or sessile oak (*Q. petraea* Liebl.). The question is whether it is possible to

Table 2—Approximate costs for direct seeding of beech and oak in Denmark and southern Sweden[a]

	Direct seeding	Planting
	- - - - *Dollars per hectare*[b] - - - -	
Site preparation	0 – 700	0 – 700
Seeds or transplants	200 – 500	1,200 – 2,400
Sowing or planting	100 – 350	400 – 800
Fence	0 – 1,300	0 – 1,300
Total	300 – 2,850	1,600 – 5,200

[a] The costs depend to a large extent on several factors such as management objectives, economy of the landowner, goals for future wood quality, site quality, deer population, area of the site, and cost of seed/transplants.

[b] United States currency.

achieve the knowledge, skills, and methods that make direct seeding competitive with planting in terms of costs, reliability, and stand quality (Johnson 1981, Kübner and Wickel 1998, Leder and Wagner 1996). Additionally, direct seeding may offer a number of other advantages compared to planting:

- The natural root development of seedlings that developed from direct seeding may be advantageous with respect to future stability against windthrow or drought events.
- The high stock density may reduce the need for deer fences.
- Direct seeding of species mixtures could be a relatively inexpensive way to establish a stand with good potential for structural adaptation to microsite variability.

Governmental subsidies to establish broadleaves are commonly available for foresters and forest authorities in Sweden and Denmark. Subsidies in Denmark (table 3) are supposed to support the fulfilment of the Danish afforestation program and enhance the transformation of conifer plantations into hardwood forests. Subsidies are, however, not necessarily prudent in terms of economical sustainability of forestry, and they call for further research and development that eventually can provide forestry with more inexpensive regeneration methods.

Table 3—Examples of present subsidy programs for afforestation and regeneration in Denmark[a b]

Forest restoration type	Maximum subsidy
Afforestation in areas where it is highly preferred (per ha)	
Planting of broadleaves	$2,900
Planting of conifers	1,900
Direct seeding	1,900
Afforestation in other areas (per ha)	
Planting of broadleaves	1,900
Planting of conifers	1,200
Direct seeding	1,200
Additional subsidies for afforestation	
Pesticide-free afforestation (per ha)	500
Fence (per m)	2
Income compensation (per ha per yr)	350
Regeneration on forest land (per ha)	
Natural regeneration of beech, oak, or ash (*Fraxinus excelsior*)	1,300
Planting beech, oak, ash, or basswood (*Tolia cordata* or *T. platyphyllos*)	3,000

[a] Eight to twelve years after establishment there must be a minimum of 2,500 to 4,000 saplings (depending on species and site type) per hectare with an average height of more than 1 m.
[b] United States currency.

CURRENT PRACTICE

There is limited current knowledge and experience on direct seeding broadleaves in the Nordic countries. However, direct seeding was commonly practiced to establish beech and oak in southern Sweden and Denmark during the late 19th and the early to mid-20th century. The intensity of these former regeneration efforts far exceeded the present level in terms of labor input, and today it seems relevant to draw an analogy to horticulture. Intensive weeding and pest-management practices were applied together with the use of very high seed densities. Stock densities exceeding 100,000 seedlings per hectare were not unusual. Regulations on provenance use reduced-seed availability and consequently led to the increased use of nursery stock.

Today, direct seeding of oak is gaining new popularity for afforestation of farmland in southern Sweden and Denmark. The method is regarded as reliable, and costs are between 30 to 50 percent of costs for planting seedlings. Oak seeding can be reliable on bare farmland because of the absence of small rodents, as they have no access to vegetative cover. Moreover, European oaks typically exhibit pioneer characteristics, which make them well adapted to the site conditions on open fields.

DIRECT SEEDING PROBLEMS

At present, there are a number of practical reasons for the limited use of direct seeding. Insects, slugs, rodents, birds, and deer can consume large amounts of seeds and young seedlings (Nielsson and others 1996). Late-spring frost may damage sprouting seedlings, and germination can fail or be delayed due to drought or incompletely broken seed dormancy. Moreover, seed availability is limited by protocols established for seed-source approval. These limiting factors must be addressed if direct seeding is going to be a planting alternative.

Furthermore, it is difficult to successfully incorporate new methods into silvicultural practice. Increasingly, many foresters face time constraints in their work, which reduces the time available for learning new regeneration methods.

DIRECT SEEDING RESEARCH AND DEVELOPMENT

The ultimate goal of the joint Nordic project on direct seeding of broadleaves was to develop new, reliable, and inexpensive regeneration methods compared to the conventional practice of planting seedlings. Furthermore, it was the aim of the project to test new methods on a range of typical site types for forest restoration.

The main hypothesis was that seedlings could be established through direct seeding with similar success as seen for planted nursery seedlings if:

- seed was of high quality;
- seed dormancy was broken by a relevant pretreatment procedure before sowing; and
- seed and established seedlings were protected against pests.

Additionally, it was hypothesised that short seeding tubes (10 to 25 cm tall) could protect seed and seedlings sufficiently. Light-seeded species like birch and alder (*Alnus* spp. Ehrhart) were, however, not expected to benefit from the protection against rodents. Instead, the tubes were supposed to create a calm and moist microclimate. This would hypothetically prevent light seeds from being blown away, improve seed germination, and improve initial seedling growth.

WHAT WAS INVESTIGATED?

Approximately 50 field experiments were established in the participating countries from spring 1995 until spring 1998. As many as three experiments were carried out at the same site in successive years. The main emphasis in the Nordic countries was generally on pedunculate oak, beech, and birch (*Betula pubescens* Ehrh. or *B. pendula* Roth) (table 4), but other species were also tested. In Mississippi, water oak (*Q. nigra* L.) was investigated, and in Estonia, grey alder (*Alnus incana* Moench), pedunculate oak, and birch were investigated. In Denmark, a number of other species were tested including wild cherry (*Prunus avium* L.), hawthorn (*Crataegus monogyna* Jacq.), ash (*Fraxinus excelsior* L.) and sycamore maple (*Acer pseudoplatanus* L.).

Various types of short seed tubes were tested. The tubes were manufactured in Denmark especially for this project, and their design was changed from year to year after evaluation of preliminary results. Initially (1995), the tubes were 25 cm tall, 28 mm in diameter, and were made out of polyethylene. In 1996 tubes of biodegradable plastics (mainly a mix of cellulose and starch) were used. Tube lengths and diameters tested ranged from 5 to 25 cm and 14 to 38 mm, respectively.

There were generally two types of experiments:

- Some experiments were relatively intensive and included both planted seedlings and sown seed; the latter seeded with and without seed-tube protection. These experiments were intensively monitored. Some of these intensive studies also included treatments with different densities of a lupine (*Lupinus nootkatensis* Donn ex Sims) cover crop (Iceland), weed control treatments (Norway, Denmark, Sweden, Estonia), or soil preparation (Sweden).
- Other experiments were nonintensive and were often established in close collaboration with forest managers (Denmark, Finland). These low-budget trials served as a supplement to the more intensive experiments allowing direct seeding and the seed tubes to be tested at a range of sites in different years.

CONCLUSIONS OF THE DIRECT SEEDING PROJECT

Results are not presented here since some of the work is still in the process of being published. However, some of the main research conclusions, implications for forest management, and directions for future research are summarized.

SEED TUBES

Seed tubes showed promising preliminary results with both light-seeded and heavy-seeded species. In many cases, germination was good in the tubes, and they protected the seed and seedlings from rodents and weevils. However, it was not complete protection, and, in some cases, the tubes even increased rodent problems because the animals learned that the tubes contained food. Additionally, some

Table 4—Main species and site types of the experiments in the participating countries

Country	Main species	Site types
Iceland	Birch (*Betula pubescens*)	Severely disturbed or partially reclaimed soils, which are being colonized by lupines
Norway	Birch (B. *pendula*)	Agricultural fields
Sweden	Beech (*Fagus sylvatica*) and pedunculate oak (*Quercus robur*)	Clearcut following a conifer stand, conifer shelterwood, or agricultural fields
Denmark	Beech (*F. sylvatica*) and pedunculate oak (*Q. robur*)	Clearcuts following conifer stands, conifer or broadleaved shelterwoods, or agricultural fields
Finland	Pedunculate oak (*Q. robur*)	Agricultural fields
Estonia	Birch (*B. pendula*)	Agricultural fields or a clearcut following a conifer stand
United States		
Mississippi	Water oak (*Q. nigra*)	Agricultural fields

seedlings suffered from winter frost damage. The warm and moist microclimate inside the seed tubes apparently prevented some seedlings from developing frost hardiness. Other problems encountered included waterlogging in the tubes, poor light conditions caused by soil sticking to the tubes, and frost heaving of the tubes. Moreover, some of the biodegradable materials used for tube construction apparently decreased germination of birch, alder, and beech. In summary, the seed tubes tested in these experiments did not lead to new, reliable, and inexpensive regeneration methods.

FOREST MANAGEMENT

In spite of the shortcomings of the seed tubes, we did obtain relevant knowledge that can be applied towards future forest management, research, and development. On the degraded sites of Iceland, planting gave higher rates of birch establishment than did direct seeding, but the seeding success varied between sites and appeared promising on some sites. Furthermore, a properly managed lupine cover crop improved birch establishment. On farmland in Norway and Estonia, no reliable methods were found for direct seeding birch and alder. In these countries, weed competition was identified as a main problem for the small seedlings.

The research indicated that heavy seeded species, like beech and oak, may be successfully established by direct seeding on sites and in years with low rodent populations. However, competitive weeds may still need to be controlled, particularly on fertile soils. Additionally, site suitability of each species must be considered for inexpensive and successful regeneration. For example, beech favors the environmental conditions under a shelterwood, whereas oak is better adapted to the microclimate of clearcuts or farmland.

FUTURE RESEARCH AND DEVELOPMENT

The Nordic project on direct seeding broadleaves has identified the need for further development of silvicultural skills, methods, and direct-seeding techniques. Additionally, there is a need for monitoring methods and rodent population control in concurrence with seeding of heavy seeded tree species. Likewise, the development of pretreatment methods aimed particularly at sites to be direct seeded is regarded as an important field of research.

REFERENCES

Arad•ttir, A.L. 1991. Population biology and stand development of birch (*Betula pubescens* Ehrh.) on disturbed sites in Iceland [Abstract]. College Station, TX: Department of Range Science, Texas A&M University. Ph.D. dissertation. [In: Dissertation abstracts international. 52–06B: 2,869].

Johnson, R.L. 1981. Oak seeding—it can work. Southern Journal of Applied Forestry. 5(1): 28–33.

Kübner, R.; Wickel, A. 1998. Entwicklung einer Buchensaat (*Fagus sylvatica* L.) unter Fichte (*Picea abies* (L. Karst.)) im Osterzgebirge. Forstarchiv. 69: 191–198.

Larsen, J.B. 1995. Ecological stability of forests and sustainable silviculture. Forest Ecology and Management. 73: 85–96.

Leder, B.; Wagner, S. 1996. Bucheckern/Streu- Voraussaat als Alternative beim Umbau von Nadelholzreinbeständen in Mischbestände. Forstarchiv. 67: 7–13.

Nielsson, U.; Gemmel, P.; Löf, M.; Welander, T. 1996. Germination and early growth of sown *Quercus robur* L. in relation to soil preparation, sowing depths, and prevention against predation. New Forests. 12: 69–86.

Sigurdsson, S. 1977. Birki á Íslandi (Birch in Iceland). In: Gudmundsson and others, eds. Skógarmá. Reykjavik, Iceland: Edda Print: 146–172. In Icelandic.

Walter, H. 1985. Vegetation of the earth and ecological systems of the geo-biosphere. 3rd ed. New York: Springer-Verlag. 318 p.

APPLICATION OF THE SOIL PERTURBATION INDEX TO EVALUATE CREATED AND RESTORED WETLANDS

Rebecca Smith Maul and Marjorie M. Holland[1]

Abstract—Biogeochemical properties of wetlands have recently been investigated to assess recovery of wetland ecosystems following human alteration. Analyses of soil samples have shown that the natural regeneration of timber-harvested wetlands exhibits predictable trends for soil organic matter, total organic carbon, total Kjeldahl nitrogen, and total phosphorus. Incorporating these four nutrients, a Soil Perturbation Index (SPI) was previously developed to aid in biogeochemical comparisons of altered wetlands at different successional stages to mature reference wetlands. The current study explores whether reforested sites previously in agriculture exhibit similar biogeochemical trends to forested wetlands previously timber harvested. Results indicate that reforested sites previously in agriculture exhibit biogeochemical trends similar to timber-harvested wetlands, although perhaps at slower rates. Trends for the SPI developed from the agriculturally based sites were very similar to that of the original SPI developed from naturally regenerated sites.

INTRODUCTION

The Nature Conservancy (1992) estimates that about 4.9 million ha of forested wetlands remain in the Lower Mississippi River Alluvial Valley. In fact, between 1883 and 1991, 77 percent of southern bottomland hardwood forests are estimated to have been lost due to timber harvesting, conversion to agriculture, and other human uses (Gosselink and Lee 1987, King and Keeland 1999, Mitsch and Gosselink 2000, The Nature Conservancy 1992). The combined efforts of the Wetlands Reserve Program (WRP) and the Conservation Reserve Program (CRP) within the U.S. Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS) and other Federal programs have led to the planting and contracts for planting of about 126 000 ha of bottomland hardwoods in the Southern United States since 1985 (Deavers 2000). We need sound methods to evaluate these and various other wetland creation and restoration efforts.

It has been shown that the natural regeneration of timbered wetlands exhibits similar trends for soil organic matter (SOM), total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) (Smith 1997). The Soil Perturbation Index (SPI) evaluates how the different successional stages compare to mature reference wetlands from a biogeochemical standpoint. The SPI uses the means of SOM, TOC, TKN, and TP data collectively for each wetland to compare with the biogeochemical reference, which is established from mature wetlands of the same type and ecoregion. The SPI consists of data that were transformed to percentage data by the following equation using the parameter TP as an example:

$$[(u - c)/ u] \times 100 = \text{perturbation number},$$

where

u = TP mean value for the 0 (mature) successional stage wetlands, and
c = TP mean for the cut wetland in question.

Percentages are calculated for all four parameters and then plotted according to successional stage (fig. 1). Our assumption is that the greater the change from the reference conditions, the lower the ecological integrity of the system (Maul and others 1999, Smith 1997). The SPI (Smith 1997) indicates a shift in biogeochemical conditions after timber harvest, with the greatest change from reference conditions reached at approximately 8 to 9 years after alteration (fig. 1). As calculated previously, this index predicts that it would take 16 to 17 years for concentrations of the SPI components (SOM, TOC, TKN, and TP) to return to preharvest conditions (Maul and others 1999). We suggest that the SPI be one component of an index of biotic integrity for freshwater wetlands.

Our hypothesis was that if agriculturally based sites exhibit a 2[d] order polynomial curve as do logged and naturally regenerated wetlands, one could use the SPI to evaluate how the created and restored sites are developing. Thus, the objectives of this study are to determine if reforested sites previously in agriculture exhibit biogeochemical trends similar to forested wetlands previously timber harvested (Smith 1997), and, if so, to determine if the SPI is useful in evaluating the restoration of created wetlands and restored wetlands or both previously in agriculture. Soils from WRP, CRP, and cottonwood sites within Mississippi, U.S.A., (fig. 2) were analyzed to determine if reforested sites previously in agriculture exhibit biogeochemical trends similar to those exhibited by forested wetlands previously timber harvested

[1] Maul, Graduate Student, Department of Geology and Geological Engineering, The University of Mississippi, University, MS; Holland, Associate Professor, Department of Biology, The University of Mississippi, University, MS 38677.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

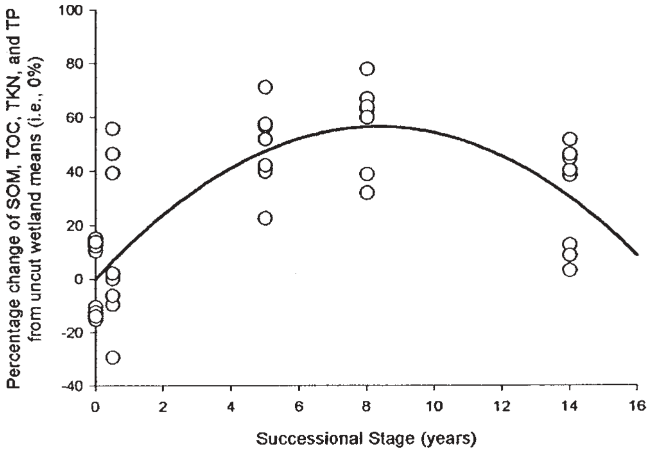


Figure 1—The Soil Perturbation Index (SPI = percentage change of SOM, TOC, TKN, and TP from uncut wetland means) from Maul and others (1999) shows percent change from a biogeochemical reference determined from mature wetlands prior to harvest (0 years = highest ecological integrity exhibited in mature reference wetlands, and 100 percent indicates greatest change from reference condition). The SPI shows a decrease in biogeochemical function (55 percent) that is greatest 8 to 9 years after timber harvesting. This index predicts that it would take 16 to 17 years for the SOM, TOC, TKN, and TP to return to preharvest conditions.

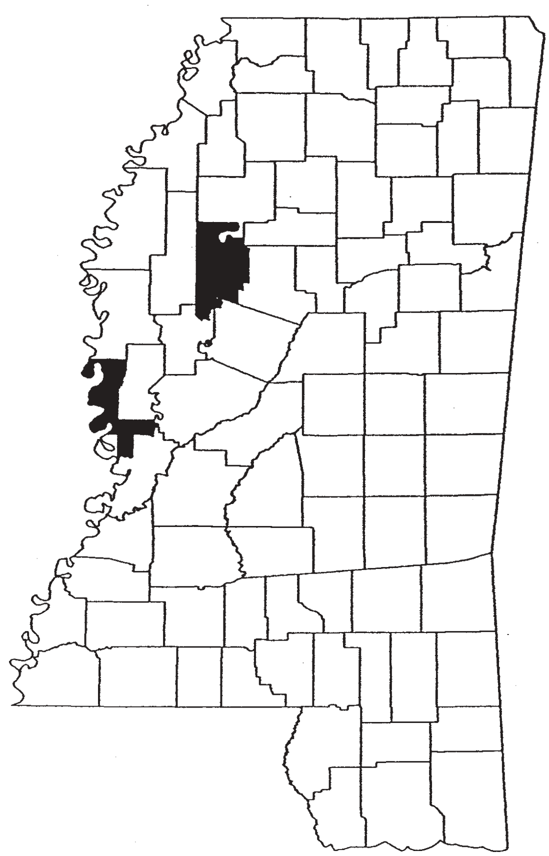


Figure 2—Map of Mississippi, U.S.A., showing Leflore County (uppermost designated county) and Issaquena County. Mature and WRP sites in Leflore County were sampled by Balducci (1998) during 1997. CRP and all cottonwood sites in Issaquena County were sampled during 1998. Map created by William A. Tedesco, Department of Geology and Geological Engineering, The University of Mississippi.

(as described by Smith 1997 and Smith and others 1996). Because NRCS was interested in differences between WRP and CRP, data are reported separately throughout our paper. However, we were interested in overall SPI trends, so we have shown WRP and CRP results on a single AG-SPI curve (see fig. 6).

SITE DESCRIPTIONS

WRP sites with stands 3 and 5 years old and mature natural forested wetlands with stands greater than 60 years old were sampled in August and September of 1997 by Balducci (1998) (table 1). Hereafter, the mature sites are referred to as 0 successional stage because no creation or restoration procedures were performed on those sites. These sites are located in Leflore County, MS (fig. 2). In 1998, we focused on sites reforested on agricultural land of various successional stages ranging from 1 to 22 years since cultivation. To encompass this range of successional stages, CRP and cottonwood sites in Issaquena County, MS (fig. 2) were sampled in August 1998 (table 1). The cottonwood sites consist of two types: the term cottonwood refers to sites that were planted on previous agricultural land and have not yet been harvested; and coppice cottonwoods have been through one harvest and resprouting cycle. The created sites of table 1 include all WRP and CRP sites.

METHODS

Soil

Ten soil cores were randomly sampled within each site. While in the field, cores were placed on ice and, once in the lab, were frozen until processed. The top 5 cm were chosen to represent the most recent influx of nutrients into the soil (Smith 1997). The SOM was measured by loss on ignition at 550 ^{o}C for 5 hours (Craft and others 1991). The TOC was analyzed by a Leco CN 2000 Carbon and Nitrogen Analyzer (Leco Corporation 1996). The TKN and TP were measured by a TRAACS 800 Autoanalyzer (Technicon Industrial Systems Corporation 1987). Soil particle size analyses were performed on a Horiba LA-910 Particle Size Analyzer (Horiba Ltd. 1997).

Statistical Analyses

Individual parameter response—Each individual parameter was analyzed separately using a one-way analysis of variance (ANOVA) for each site. In the first analysis, responses of SOM, TOC, TKN, and TP of the cottonwood sites and mature sites (0 successional stage) were observed. The treatment was successional stage consisting of five groups: 0, 5, 8, 16, and 22 years (n = 3, 2, 2, 2, and 1, respectively). The second analysis compared SOM, TOC, TKN, and TP among the WRP, CRP, and mature sites. The treatment was successional stage consisting of four groups: 0, 3, 5, and 9 years (n = 3, 3, 2, and 2, respectively). The third analysis compared SOM, TOC, TKN, and TP between 9-year coppice cottonwood sites and 9-year CRP sites (n = 2). All variables were normally distributed based on Shapiro-Wilk statistic for normality (SAS Institute, Inc. 1988). All ANOVAs and Student-Newman-Kuels (SNK) tests ($p < 0.05$) were performed with PC/SAS software (SAS Institute, Inc. 1988).

Table 1—Study sites, successional stage (years since alteration), treatment (coppice, cottonwood, created, or natural), and soil nutrient parameters

Site	Stage	Treatment	SOM	TOC	TKN	TP
	Year		*- - - - - - - - - - - -*	*mg/g*	*- - - - - - - - - - - -*	
WRP3A	3	Created[a]	66.4	20.88	3.150	1.653
WRP3B	3	Created[a]	86.5	27.55	3.637	1.029
WRP3C	3	Created[a]	40.0	12.53	2.175	.475
5A	5	Cottonwood[b]	88.8	29.01	3.380	1.206
5B	5	Cottonwood[b]	52.1	14.40	1.863	.996
WRP5A	5	Created[a]	82.9	35.22	4.792	.745
WRP5B	5	Created[a]	30.6	5.24	1.213	.369
8A	8	Cottonwood[b]	48.8	20.09	2.168	1.053
8B	8	Cottonwood[b]	40.9	15.83	1.949	.845
C9A	9	Coppice[c]	101.6	36.71	4.972	1.339
C9B	9	Coppice[c]	95.3	32.79	4.216	1.171
CRPA	9	Created[d]	76.2	20.22	2.535	.805
CRPB	9	Created[d]	91.6	27.34	3.252	1.245
16A	16	Cottonwood[b]	105.9	39.67	5.323	1.205
16B	16	Cottonwood[b]	77.8	29.98	3.919	1.133
22A	22	Cottonwood[b]	88.4	34.35	5.175	1.300
Mature	> 60	Natural[a]	189.3	86.85	13.400	1.200
Mature	> 60	Natural[a]	170.9	72.96	15.380	1.760
Mature	> 60	Natural[a]	172.7	80.27	11.260	1.520

SOM = soil organic matter; TOC = total organic carbon; TKN = total Kjeldahl nitrogen; TP = total phosphorus.
[a] Sites sampled by Balducci in August and September 1997.
[b] Cottonwood sites were planted on land previously used for agriculture and have not been harvested (sampled August 1998).
[c] Coppice sites were allowed to sprout from trunks after harvesting (sampled August 1998).
[d] Conservation Reserve Program sites sampled in August 1998.

Soil Perturbation Index—The SPI (Maul and others 1999, Smith 1997) uses the means of SOM, TOC, TKN, and TP data for each wetland to calculate the percent change from the biogeochemical reference determined from the mature wetland soils. Because there are replicate wetlands per successional stage (n) and four parameters analyzed per wetland, 4 by (n) replicate points for each successional stage are plotted. The cottonwood and mature sites were used to develop a single agriculturally based SPI (AG-SPI) for reforested sites previously in agriculture. As in Maul and others (1999), the perturbation numbers were plotted for each parameter (TP, SOM, TOC, and TKN) and for the different successional stages. A 2[d] order polynomial equation for AG-SPI [(Y = M0 + M1(x) + M2(x)) where M0 = 5.424, M1 = 10.660, M2 = -0.434, and x = successional stage (years)] provided the best fit line for the index with r = 0.77.

RESULTS

Individual Parameter Response

The majority of all the sites were classified as having soil textural classes of silt or silt loam (table 2). The cottonwood sites exhibited similar observable trends for all four parameters (TP, SOM, TOC, and TKN). Concentrations of soil parameters changed until the 8-year stage, then moved back toward mature conditions (fig. 3). The SOM, TOC, TKN,

Table 2—Soil particle size analysis and textural class data of Balducci (1998) and this study

Site	Sand	Silt	Clay	Textural class
	- - - - - *Percent* - - - - - -			
WRP3A	1.7	88.1	10.2	Silt
WRP3B	2.5	84.9	12.6	Silt loam
WRP3C	22.6	69.9	7.4	Silt loam
5A	14.8	80.1	5.1	Silt loam
5B	9.1	85.5	5.4	Silt
WRP5A	1.9	91.5	6.7	Silt
WRP5B	48.2	46.2	5.6	Sandy loam
8A	40.6	58.1	1.3	Silt loam
8B	39.0	59.6	1.4	Silt loam
C9A	7.1	87.6	5.3	Silt
C9B	8.0	87.1	4.9	Silt
CRPA	4.0	88.7	7.3	Silt
CRPB	8.1	83.7	8.2	Silt
16A	20.6	75.0	4.4	Silt loam
16B	29.7	67.0	3.3	Silt loam
22A	18.6	77.7	3.6	Silt loam
Mature	5.0	86.6	8.4	Silt
Mature	25.7	65.9	8.4	Silt loam
Mature	16.6	79.1	4.3	Silt

and TP trends exhibited in this study (fig. 4) are similar to trends found in the timber-harvested wetland study sites of Maul and others (1999). Coppice sites contained higher concentrations of SOM, TOC, TKN, or TP than the created sites (fig. 5).

Soil Perturbation Index

The AG-SPI combines data from the cottonwood and mature sites (fig. 6). According to this study, reforested sites previously in agriculture exhibit similar biogeochemical trends as natural forested sites previously timber harvested. The best-fit curve for both SPIs is a 2[d] order polynomial curve. The AG-SPI indicates that reforested sites previously in agriculture take longer to recover biogeochemically than timber-harvested wetlands. The point indicating greatest change from reference conditions for the AG-SPI is about 12 years whereas the original SPI indicates the greatest change at 8 to 9 years after alteration. The AG-SPI predicts it would take 24 years for SOM, TOC, TKN, and TP to return to mature conditions compared to the 16 to 17 years suggested by the original SPI. The AG-SPI biogeochemical levels deviate approximately 70 percent from the mature sites where the original SPI only changes about 55 percent from the mature sites.

The WRP and CRP sites were plotted on the AG-SPI to provide a biogeochemical comparison (fig. 6). The AG-SPI indicates that WRP sites may exhibit a greater change from mature biogeochemical conditions compared to other wetlands of similar age and background. According to index calculations run for the current study, the 9-year CRP sites

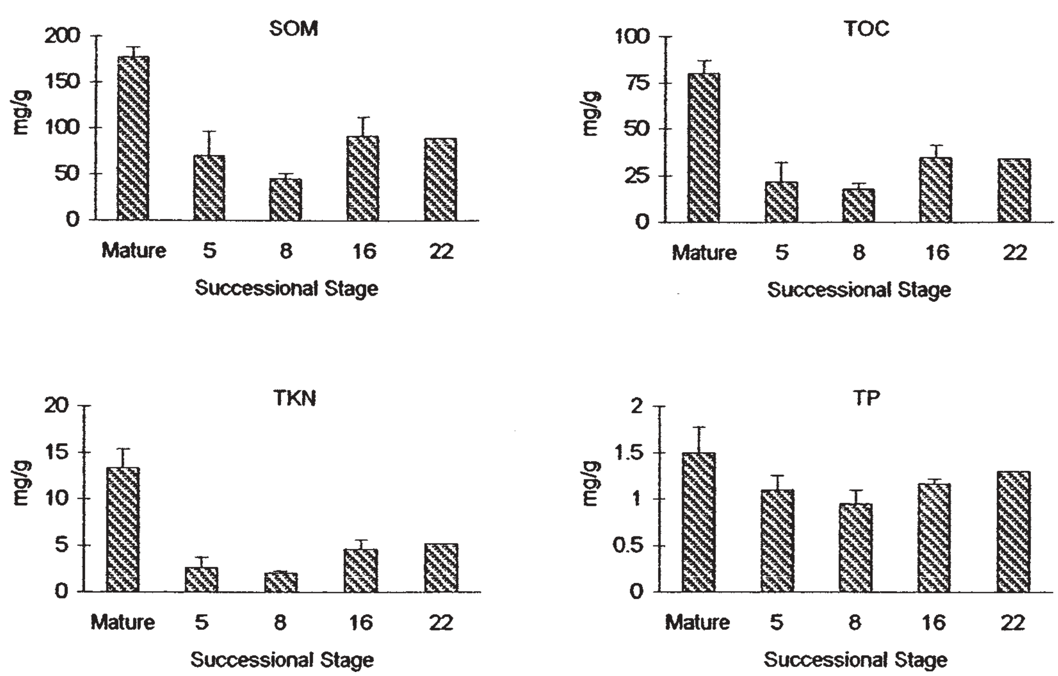


Figure 3—Trends of soil organic matter (SOM), total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) among cottonwood sites of different successional stages and mature sites. Error bars represent one standard error. Data for mature sites are from Balducci (1998). Cottonwood sites were sampled in August 1998.

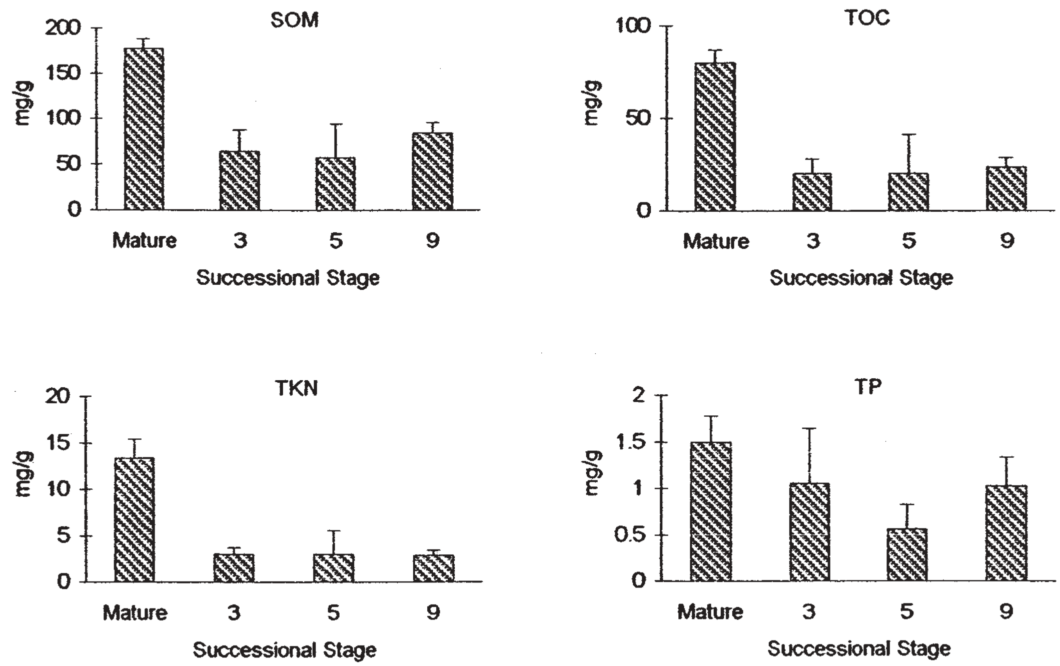


Figure 4—Trends of soil organic matter (SOM), total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) among different successional stages of WRP (3 and 5 year), CRP (9 year) and mature sites. Error bars represent one standard error. Mature and WRP sites were sampled in August and September 1997 (Balducci 1998). The CRP sites were sampled in August 1998.

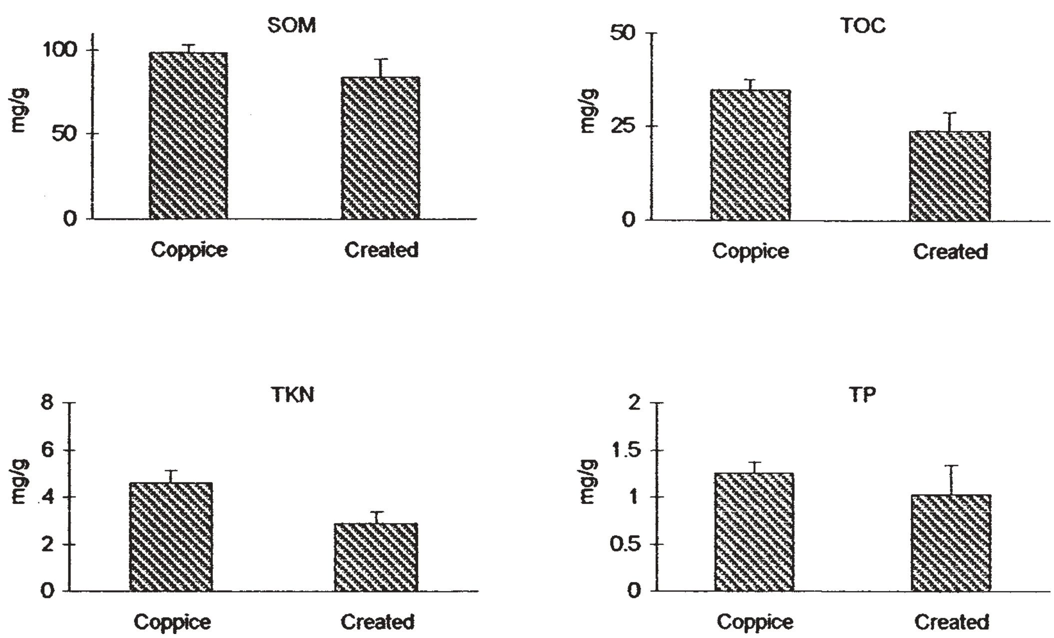


Figure 5—Comparisons of soil organic matter (SOM), total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) between 9-year coppice and 9-year CRP sites. Error bars represent one standard error. Sites sampled in August 1998.

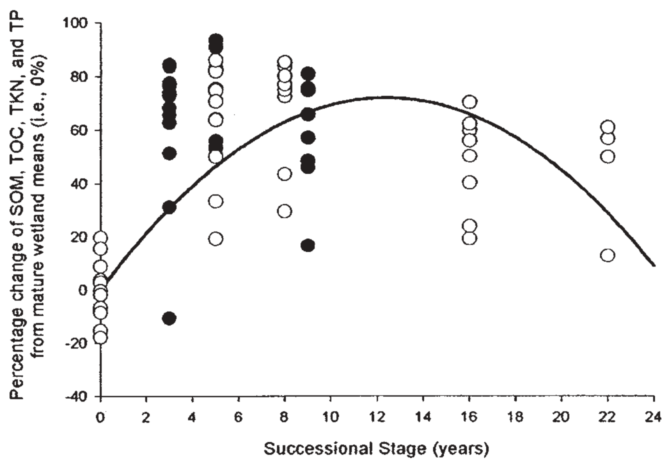


Figure 6—The agriculturally based Soil Perturbation Index (AG-SPI) shows percent change from a biogeochemical reference determined from mature wetlands (0 years before alteration). The AG-SPI shows a change in biogeochemical function (70 percent) that is greatest 12 years after planting. This index predicts that it would take 24 years for the SOM, TOC, TKN, and TP to return to mature condition. The white circles represent mature and cottonwood sites and the black circles represent WRP- and CRP-created sites. The WRP and CRP sites were plotted for comparison. Mature and WRP sites were sampled by Balducci (1998). Cottonwood and CRP sites were sampled in August 1998.

are functioning in the same range as other sites (cottonwood) previously in agriculture.

DISCUSSION

Individual Parameter Response

It is indicated in this study that reforested sites (cottonwood) previously in agriculture exhibit similar biogeochemical trends, although at lower levels, as wetlands allowed to naturally regenerate following timber harvest. The SOM, TOC, TKN, and TP concentrations decrease until the 8-year stage then increase toward mature conditions. The pattern observed in the current study is the same as that exhibited by timber-harvested wetlands found in the Maul and others (1999) study. Several factors promote such a change from the biogeochemical parameters of mature wetlands and subsequent rebound of soil nutrient concentrations. Young aggrading stands have higher rates of nutrient uptake than mature systems (Lockaby and Walbridge 1998). The young successional stages produce less foliage and other organic components contributing to organic matter than mature systems. As the sites mature, the vegetation produces more leaf litter, which upon recycling of nutrients, increases the productivity of the wetland (Smith and others 1996). Changes in the SOM concentrations or in rates of carbon cycling may have important effects on nutrient cycling, vegetative composition, and productivity (Trettin and others 1996). Thus, time is required in order for the system to reach the biogeochemical conditions of mature reference wetlands.

Soil Perturbation Index

The SPI can aid as one component in evaluating created or restored forested wetlands, or both, previously in agriculture. The agriculturally based cottonwood sites produce an AG-SPI very similar to that of the Maul and others (1999) SPI developed from previously timber-harvested wetlands. The recovery time of the AG-SPI is longer, and there is a greater change in biogeochemical conditions than seen in the SPI reported by Maul and others (1999). This longer recovery time is expected due to the lack of organic matter accumulation in the soil resulting from agricultural practices. The disturbance of the soil and drier conditions of these sites may contribute to the lower concentrations of SOM, TOC, TKN, and TP retained within soils of the current study.

The WRP and CRP sites were plotted on the AG-SPI to provide a biogeochemical comparison of how the created wetlands are progressing. From a biogeochemical standpoint, the 3- and 5-year WRP sites are functioning at lower biogeochemical levels than other wetlands of the same stage and background. Visual observation (nonquantitative) indicated very few tree saplings were present in any of the WRP sites. The 9-year CRP sites were functioning within the range of other agriculturally based sites (cottonwood) of the same stage. Visual observation indicated trees were present and hardy in both CRP sites. The SPI should be combined with other quantitative data such as for vegetation and hydrology to provide a more comprehensive evaluation of these created or restored wetlands, or both. Further studies should focus on other human alterations as well as natural disturbances. Wetlands created for mitigation purposes might also be studied to determine if the SPI is useful as one component in the evaluation of these wetlands.

CONCLUSION

The SPI is a useful component and tool to evaluate created or restored wetlands, or both, previously in agriculture. The SPI is applicable to reforested wetland systems previously in agriculture and can be one useful component of an index of biotic integrity for wetland ecosystems. This study and that of Maul and others (1999) indicate that wetlands previously altered by human activity (agriculture and timber harvesting) exhibit similar biogeochemical trends following alterations. The SPI can be used to evaluate the biogeochemical trends of created or restored wetlands, or both, (WRP and CRP) compared to other wetlands of similar land use history, type, and ecoregion.

ACKNOWLEDGMENTS

The authors thank Paul Rodrigue for valuable comments and field assistance, Jonathan Maul for assistance with statistical and graphical applications, and Tara Wildlife Inc. for lodging. We are grateful for the use of equipment and assistance from the USDA, ARS National Sedimentation Laboratory in Oxford, MS. As with earlier studies, we are grateful to Charlie Cooper and Al Mikell for thoughtful discussions and suggestions. In addition, we appreciate the helpful comments of Sammy King and Joy Zedler on an earlier draft of this manuscript. This project was funded in part by USDA, NRCS Wetland Science Institute (Agreement 68–7482–7–234 to The University of Mississippi, Department of Biology, Marjorie M. Holland, Principal Investigator).

REFERENCES

Balducci, J.A. 1998. Ecological integrity of wetland soils: testing of a perturbation index in northern Mississippi wetlands. University, MS: Department of Biology, The University of Mississippi. 63 p. M.S. thesis.

Craft, C.B.; Seneca, E.D.; Broome, S.W. 1991. Loss on ignition and Kjeldahl digestion for estimating organic carbon and total nitrogen in estuarine marsh soils: calibration with dry combustion. Estuaries. 14: 175–179.

Deavers, L. 2000. Total acres enrolled in wetland reserve program. USDA, Natural Resources Conservation Service. http://www.wl.fb-net.org/wrp-acres.gif. [Accessed January 20, 2001.]

Gosselink, J.G.; Lee, L.C. 1987. Cumulative impact assessment in bottomland hardwood forests. LSU–CEI–86–09. Center for Wetland Resources. Baton Rouge, LA: Louisiana State University. 109 p.

Horiba, Ltd. 1997. Laser scattering particle size distribution analyzer LA-910. Kyoto, Japan. 72 p.

King, S.L.; Keeland, B.D. 1999. Evaluation of reforestation in the Lower Mississippi River Alluvial Valley. Restoration Ecology. 7(4): 1–12.

Leco Corporation. 1981. Leco instruction manual for CR12 carbon analyzer No. 200-195. St. Joseph, MI: Leco Corporation.

Lockaby, B.G.; Walbridge, M.R. 1998. Biogeochemistry. In: Messina, M.G.; Conner, W.H., eds. Southern forested wetlands. Boca Raton, FL: CRC Press LLC: 149–172.

Maul, R.S.; Holland, M.M.; Mikell, A.T.; Cooper, C.M. 1999. Resilience of forested wetlands located in the Southeastern United States: demonstration of a soil perturbation index. Wetlands. 19: 288–295.

Mitsch, W.J.; Gosselink, J.G. 2000. Wetlands. New York: John Wiley & Sons, Inc. 920 p.

SAS Institute, Inc. 1988. SAS/STAT user's guide. Release 6. 3rd ed. Cary, NC: SAS Institute Inc.

Smith, R.L. 1997. The resilience of bottomland hardwood wetlands soils following timber harvest. University, MS: The University of Mississippi. 99 p. M.S. thesis.

Smith, R.L.; Holland, M.M.; Cooper, C.M. 1996. The resilience of bottomland hardwood wetland soils following timber harvest. In: Heard, L.P.; Stewart, R.E., eds. Proceedings of a conference; the Delta: connecting points of view for sustainable natural resources. Memphis, TN: U.S. Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute: 739–745.

Technicon Industrial Systems Corporation. 1987. TRAACS 800 method. Industrial method 787–86T; Elmford, NY: Braun and Luebbe Analyzing Technologies, Inc.

The Nature Conservancy. 1992. The forested wetlands of the Mississippi River: an ecosystem in crisis. Baton Rouge, LA: The Nature Conservancy. 53 p.

Trettin, C.C.; Davidian, M.; Jurgenson, M.G.; Lea, R. 1996. Organic matter decomposition following harvesting and site preparation of a forested wetland. Soil Science Society of America Journal. 60: 1994–2003.

PESTICIDE MITIGATION CAPACITIES OF CONSTRUCTED WETLANDS

Matthew T. Moore, Charles M. Cooper, Sammie Smith, Jr., and John H. Rodgers, Jr.[1]

Abstract—This research focused on using constructed wetlands along field perimeters to buffer receiving water against potential effects of pesticides associated with storm runoff. The current study incorporated wetland mesocosm sampling following simulated runoff events using chlorpyrifos, atrazine, and metolachlor. Through this data collection and simple model analysis, researchers conservatively predicted wetland buffer travel distances necessary to mitigate potential pesticide effects on receiving systems, which ranged from 100 to 400 m for the pesticides. This research provides fundamental answers concerning the use of constructed wetlands for pesticide mitigation in agricultural watersheds.

INTRODUCTION

In response to growing concern over agricultural pesticide runoff effects on flora and fauna of receiving water bodies, recent research has focused on potential best management practices (BMPs) to minimize such effects. One such suggestion is the use of constructed wetlands along field perimeters to buffer receiving lakes, rivers, and streams against potential effects of agricultural pesticides associated with stormwater runoff. The U.S. Department of Agriculture (USDA) established the new National Conservation Buffer Initiative in 1997, which is aimed at installing 3.2 million km of conservation buffers by the year 2002. In the Mississippi Delta, areas containing wetlands have been drained for agriculture and silviculture purposes. According to Mitsch and Gosselink (1993), draining and filling of wetlands since the mid-1800s has resulted in the loss of more than 50 percent of the Nation's original wetlands. Perhaps by placing constructed wetlands in areas where natural wetlands once thrived, agriculture could regain the original wetland area function of water-quality enhancement while minimizing potential impacts to aquatic-receiving systems.

MATERIALS AND METHODS

A series of constructed wetland mesocosms specifically designed to evaluate the fate of pesticides in wetlands was used for this research (Rodgers and Dunn 1992). Darby (1995) previously characterized wetland mesocosm sediments. Four wetland mesocosms were chosen as test cells, with one additional mesocosm serving as an unamended control. Three remaining mesocosms were used as water sources for the simulated storm event. Constructed wetland mesocosms (59 to 73 m by 14 m) were amended on two different occasions (one for the insecticide chlropyrifos, the other for a mixture of the herbicides atrazine and metolachlor). Following each pesticide amendment, a simulated cropland runoff and rainfall event equal to three volume additions was imposed on each wetland. Targeted concentrations of chlorpyrifos were 73 µg/L, 147 µg/L, and 733 µg/L, whereas targeted concentrations for both atrazine and metolachlor were 73 µg/L and 147 µg/L in addition to an unamended control (0 µg/L). Water, sediment, and plant samples were collected weekly for the duration of the experiment (chlorpyrifos, 84 days; atrazine and metolachlor, 35 days). Samples were collected from sites longitudinally distributed within each wetland and analyzed for the respective pesticides using gas chromatography.

RESULTS AND DISCUSSION

Chlorpyrifos rapidly sorbed to sediment and plant material, and the half-life in water, in this research, ranged from 5 to 13 days. Approximately 47 to 65 percent of measured chlorpyrifos was within the first 30 to 36 m (from inflow) of wetland mesocosms. Approximately 55 percent and 25 percent of measured chlorpyrifos mass were retained by sediments and plant material, respectively. Conservative mathematical models were used to derive adequate wetland design parameters for mitigation of chlorpyrifos-associated stormwater runoff. Recommended wetland travel distances were 184 m for wetlands receiving 147 µg/L chlorpyrifos runoff and 230 m for wetlands receiving either 73 or 733 µg/L chlorpyrifos runoff. It is imperative to remember that these calculations were made from conservative models.

Between 17 and 42 percent of the measured atrazine mass was within the first 30 to 36 m (from inflow) of wetlands. Atrazine concentrations were below analytical limits of quantification (0.05 µg/L) in all sediment and plant samples collected for this research. Atrazine aqueous half-lives ranged from 16 to 48 days. According to conservative design models, for atrazine concentrations of 73 µg/L, wetland travel distances for effective mitigation ranged from 101 to 164 m. For wetlands receiving initial atrazine concentrations of 147 µg/L, effective constructed wetland travel distances ranged from 103 to 281 m.

Between 7 and 25 percent of measured metolachlor mass was in the first 30 to 36 m (from inflow) of wetlands immediately following application and simulated rainfall.

[1] Moore, Cooperator, Cooper, Supervisory Ecologist and Research Leader, and Smith, Research Chemist, USDA-ARS National Sedimentation Laboratory, P.O. Box 1157, Oxford, MS 38655; Rodgers, Professor and Director, Clemson Institute of Environmental Toxicology, 509 Westinghouse Road, Pendleton, SC 29670.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

Approximately 10 percent of measured metolachlor mass was in plant samples. Based on models, adequate wetland travel distances receiving 73 µg/L metolachlor would range from 102 to 170 m. For those constructed wetlands receiving initial concentrations of 147 µg/L metolachlor, wetland travel distances necessary for effective mitigation would range from 100 to 400 m.

Based on presented wetland designs, it may not be economically feasible for a farmer to implement constructed wetlands as a sole BMP if their fields were small, e.g., < 4 ha. Constructed wetlands are not panaceas for environmental problems. Given proper situations, however, they can greatly enhance water quality in a variety of municipal, industrial, and agricultural settings. This research has offered valuable data concerning effectiveness for using constructed wetlands as buffers for pesticide-associated stormwater runoff.

ACKNOWLEDGMENTS

Authors thank P.B. Rodrigue and the USDA NRCS Wetland Science Institute for partial funding. Special thanks to G.M. Huddleston; W.B. Gillespie, Jr.; R.W. Steinriede, Jr.; D.B. Huggett; M. Baker; and D. Mathis for technical assistance and advice.

REFERENCES

Darby, M.A. 1995. Modeling the fate of chlorpyrifos in constructed wetlands. University, MS: The University of Mississippi. M.S. thesis.

Mitsch, W.J.; Gosselink, J.G. 1993. Wetlands. New York: VanNostrand Reinhold. 722 p.

Rodgers, J.H., Jr.; Dunn, A.W. 1992. Developing design guidelines for constructed wetlands to remove pesticides from agricultural runoff. Ecological Engineering. 1: 83–95.

NUTRIENT STORAGE RATES IN A NATURAL MARSH RECEIVING WASTE WATER

J. A. Nyman[1]

Abstract—Artificial wetlands are commonly used to improve water quality in rivers and the coastal zone. In most wetlands associated with rivers, denitrification is probably the primary process that reduces nutrient loading. Where rivers meet oceans, however, significant amounts of nutrients might be permanently buried in wetlands because of global sea-level rise and regional subsidence. We determined nutrient storage rates in the marshes adjacent to a minor stream to test the hypothesis that a natural wetland adjacent to Lake Pontchartrain (Louisiana) does not affect nutrient inputs to this estuary. Data from soil cores indicated that marshes store 240 t of N and 11 t of P each year. These data demonstrate that wetlands associated with rivers substantially modify nutrient transfer from terrestrial to aquatic habitats. Hydrologic data are being collected that can be used to determine what portion of the nutrient load of the river is buried in the marsh, and what portion of the nutrient load is discharged into the receiving basin.

INTRODUCTION

Artificial wetlands are now commonly used to improve water quality; long-term operational information exists for many full-scale projects (Kadlec and Knight 1986). Despite a growing body of data relating artificial wetlands and waste water, there have been few studies of the effects of natural wetlands on waste water (see Kadlec and Knight 1986 for a historical perspective). Those studies generally focused on denitrification, which converts nitrates and ammonia to nitrogen gas rather than on burial because, in most regions, burial is too slow to remove significant nutrients. In subsiding environments, however, burial might permanently store significant amounts of nutrients. Nutrient burial rates have been studied in natural wetlands in the Florida Everglades, North Carolina estuarine marshes, and in southwestern Louisiana (Craft and Richardson 1993, Craft and others 1993, Foret 1997). Nutrient storage rates in soil have rarely been studied in natural wetlands receiving waste water and never in wetlands of the Lake Pontchartrain Basin, Louisiana. This is surprising given the large public concern regarding eutrophication in the lake and the relatively rapid subsidence of coastal Louisiana.

A lack of information on nutrient storage rates in wetlands underlies public and professional controversy regarding the benefits of water-quality regulations, wetland protection and restoration activities, and the benefits of introducing nutrient-rich water to estuarine marshes. Relationships among nutrient sources and rates and nutrient sinks and rates need to be better understood in wetland nutrient dynamics in general, and in Lake Pontchartrain, in particular. We tested the hypothesis that a natural wetland at the mouth of a freshwater stream emptying into Lake Pontchartrain does not affect nutrient inputs into the lake.

STUDY AREA

The study was conducted on the northern shore of Lake Pontchartrain in a naturally occurring marsh that predates the earliest aerial photographs. Lake Pontchartrain is a large (the diameter is approximately 60 km) estuary in southeastern Louisiana that receives freshwater from numerous rivers and streams on its northern shore and sea water from a tidal pass at its eastern end. The marsh studied was located at the mouth of Salt Bayou, which carries freshwater from the Pearl River to Lake Pontchartrain. The marsh, which lies on a tributary (Salt Bayou) to Lake Pontchartrain, was classified as brackish and intermediate (Chabreck and Linscombe 1978). Construction of railways and highways across Salt Bayou reduced freshwater input from the Pearl River into the marsh in the mid-1950s. That reduction in freshwater inflow is believed to have contributed to subsequent conversion of marsh to shallow open water. Since 1956, 2,485 ac of marsh have converted to shallow, open water; 3,438 ac of marsh remained in 1990 (NRCS 1997).

A canal adjacent to Fritchie Marsh, the W-14 Canal, carries urban runoff and tertiary-treated domestic sewage from the community of Slidell to Lake Pontchartrain. Most water in the W-14 Canal discharges directly into Lake Pontchartrain, but some leaves the canal and flows through the Fritchie Marsh before rejoining the W-14 Canal and then discharging into Lake Pontchartrain. A Coastal Wetland Planning, Protection, and Restoration Act (CWPPRA) project is scheduled for construction in the Fritchie Marsh by the end of 1999. That CWPPRA project will increase the amount of water flowing from the W-14 Canal into the Fritchie Marsh. The intent of that freshwater introduction is offset by the loss of freshwater inputs from the Pearl River caused by the railways and highways crossing Salt Bayou that were constructed earlier this century. The Louisiana Department of Natural Resources (DNR) will collect data that can be used to estimate the effect of the W-14 Canal water on vegetation and marsh loss in the Fritchie Marsh (Department of Natural Resources 1996).

[1] Department of Biology, University of Louisiana at Lafayette, P.O. Box 42451, Lafayette, LA 70504-2451.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

METHODS

Marsh Soil

One data set was collected to quantify long-term nutrient storage rates in Fritchie Marsh. Nutrient storage rates were determined from the accretion rate of soil and the nutrient content of soil in the marsh. This is possible because new soil continually forms on the surface of estuarine marshes; this process is called vertical accretion and is essential because global sea-level rise and subsidence would otherwise drown estuarine marshes (Mitsch and Gosselink 1986: 178–181). A pair of cores was collected from DNR vegetative monitoring station 22; another pair of cores was collected from DNR vegetative monitoring station 1. Station 1 was at the northern, upstream end of the study area; station 22 was at the southern, downstream end. The cores were returned to the lab, sectioned, weighed, ovendried, weighed, and crushed. Bulk density of each section was calculated. Vertical accretion since 1963 was determined with the ^{137}Cs dating technique (DeLaune and others 1978). Activity of ^{137}Cs was determined with an Ortec GMX series Gamma-x-high purity, N-type germanium coaxial photon detector system. Vertical accretion rates were compared to regional estimates of subsidence to determine if there was a vertical accretion deficit. Vertical accretion deficits are often the mechanism by which sediment starvation and rapid subsidence cause wetland loss in coastal Louisiana, e.g., Nyman and others 1993.

The gross material accumulation rate, i.e., the accumulation of mineral sediments as well as organic matter, was calculated for each core from the bulk density of soil samples overlying the 1963 marsh surface. The accumulation rate of mineral sediments and organic matter was similarly calculated from the bulk density and ash content of each section overlying the 1963 marsh surface. The accumulation rate of nitrogen (N), carbon (C), and phosphorus (P) was similarly calculated from the bulk density and the nutrient content of soil overlying the 1963 marsh surface. Carbon and nitrogen content were determined by ignition in a Carlo-Erba Elemental Analyzer equipped with a thermoconductivity detector. Phosphorus content was determined following acid digestion using Parson's colormetric analysis (Parsons 1984).

Water Quality

A second data set was collected to compare changes in nutrient concentration in waters that empty directly into the lake to changes in nutrient concentration in waters that flow through the Fritchie Marsh before emptying into the lake. Data were collected on two transects that originate at a common point in the W-14 Canal and terminate at a common point in Lake Pontchartrain. However, one transect runs through the W-14 Canal, and the other runs through the Fritchie Marsh. Selecting sites for collecting water samples was more complicated than selecting sites for soil samples. We had planned to collect water samples at DNR sites simultaneously with DNR personnel. However, it was impossible to sample along the W-14 Canal from the DNR airboat because of numerous low bridges that cross the canal. Alternate transportation was arranged after ULL purchased a boat for research. Using a small boat rather than an airboat reduced the area accessible for collecting water samples and increased time required to collect samples because shallow water areas cannot be crossed. Collecting water samples was further delayed because the landowners forbade sampling during waterfowl hunting season.

Both transects had 10 sampling stations. At each station, a surface water sample was collected. Salinity and conductivity were measured with a YSI model 6920 salinometer (Yellow Springs, Ohio) simultaneously with the collection of each water sample. Nutrient concentration was determined on the unfiltered water samples. Total N and P content were determined using Parson's colormetric analysis (Parsons 1984). Changes in nutrient concentration resulting from dilution with lake water were differentiated from nutrient uptake with mixing diagrams, i.e., ionic ratios (Day and others 1989: 81–85).

RESULTS

Marsh Soil

Average bulk density was greater at the northern site than at the southern site. The greater bulk density at the northern site incorrectly suggested that more material is introduced into Fritchie Marsh from the W-14 Canal than from Lake Pontchartrain. However, examination of density profiles with depth suggests that recently formed, surface soils were similar at the two sites. Thus, the northern and southern sites currently appear to be storing more similar amounts of material than the average bulk density suggests.

Accretion in Fritchie Marsh was slightly less than the average for brackish marshes in the Mississippi River Deltaic Plain (0.72 cm per year) (Nyman and others 1990). The more rapid average accretion in the entire Mississippi River Deltaic Plain than in the Fritchie Marsh results from the more rapid subsidence rate elsewhere in the Mississippi River Deltaic Plain (approximately 1.0 cm per year) (Penland and others 1990).

Accretion rates within sites were almost identical, but the two sites differed greatly. Accretion at the northern site was 40 percent slower than at the southern site. Accretion at the northern site is similar to estimates of subsidence on the northern shore of Lake Pontchartrain, which is estimated at 0.45 cm per year (Penland and Ramsey 1990). However, it is difficult to reconcile the rapid accretion at the southern site with the slow subsidence reported by Penland and Ramsey (1990). It is also unusual for accretion rates to vary so much within such a small area, e.g., Nyman and others 1990, Nyman and others 1993. The large difference between northern and southern sites suggests that a shallow, active fault runs through Fritchie Marsh. Such faults are common in coastal Louisiana in general and around Lake Pontchartrain. Lopez (1991) examined seismic data and reinterpreted the location of the Baton Rouge/Denham Springs fault system and concluded that the fault lay farther south than previously believed and positioned it such that it would bisect the Fritchie Marsh. Our findings support Lopez's (1991) conclusions regarding the position of that fault system. Lopez (1991) also concluded that the fault system was active and responsible for a 6-inch offset on the State Highway Bridge 11 crossing eastern Lake Pontchartrain. The tremendous difference in accretion rates between the northern and southern sites that we observed support Lopez's (1991) conclusion that the fault system is active, although different faults within the fault system would be required to produce offset in accretion in the Fritchie Marsh and the offset in the highway bridge. Whereas this active fault may contribute greatly to wetland loss in the Fritchie Marsh, it also increases the potential for burial of nutrients in the marsh.

Gross material accumulation was 20 percent slower at the northern site than at the southern site. The faster accumulation rate at the southern site results from the more rapid vertical accretion induced at the southern site by the fault. Extrapolated to the entire area, it appears that the 1,040-ac Fritchie Marsh restoration site stores slightly over 13,660 t of material annually. Some of this material is organic carbon that is produced in the marsh, but the associated nutrients and sediments otherwise would be discharged into Lake Pontchartrain.

The material being stored in wetland soil at the Fritchie Marsh includes mineral sediments, organic matter, and the ecologically important elements C, N, and P. A significant amount of the nutrients N and P is being stored in the marsh soils, but the amount stored has decreased 48 percent since the 1950s because of the conversion of wetlands to shallow open water areas, which are assumed to be stable rather than accreting.

The N:P ratio of soil at the northern site was slightly higher than that at the southern site. Higher ratios indicate a greater potential for P availability to limit plant growth at the northern site, but the small difference between the sites may not be ecologically meaningful. These ratios appeared typical; they were similar to those reported for unmanaged, *S. patens* dominated marsh at Rockefeller Refuge (Foret 1997).

Water Quality Data

Nutrients were more concentrated in water in the W-14 Canal than in Lake Pontchartrain water. This situation is typical of estuaries in general (Liss 1976). For example, N concentrations in the Mississippi River generally average 3 to 4 parts per million (ppm) but drop rapidly to approximately 1.0 ppm soon after entering estuarine marshes (Lane and others 1995). Phosphorus in the Mississippi River also drops rapidly soon after entering estuarine marshes from approximately 0.4 ppm P in the river to approximately 0.2 ppm P after entering estuarine marshes. Nutrient concentrations in the W-14 Canal were less than those reported for the Mississippi River (Antweiler and others 1995, Lane and others 1999) and did not exceed 1.5 ppm N (108 µmole N/l) or 0.5 ppm P (16 µmole P/l).

A simple comparison of nutrient concentrations between water in the W-14 Canal and the marsh incorrectly suggests that the marsh was removing nutrients when the water samples were collected. Proper evaluation of the water-quality data indicates that the low-nutrient water in the marsh was actually low-nutrient, high-salinity water introduced from Lake Pontchartrain at an earlier date and, subsequently, stored in the marsh. Thus, the low-nutrient, high-salinity water in the marsh interior indicates at least periodic hydrologic isolation of the marsh interior. The isolation apparently results from a lack of inflow from the W-14 Canal at the northern (upstream) end of the marsh, and possibly from high-water levels in the W-14 Canal that inhibit drainage of the marsh at the southern (downstream) end. The proposed restoration project should introduce more water through the marsh and thereby reduce the hydrologic isolation that the marsh currently experiences. It could not be determined if the failure to detect nutrient uptake with the water-quality data resulted from a true lack of uptake or from an inability of the technique to detect real differences.

Water in the W-14 Canal had a very low N:P ratio relative to N:P ratio in the marsh interior. The difference in N:P ratio resulted from 7-fold lower P concentrations rather than greater N concentrations. The N:P ratio of the W-14 Canal was similar to ratios in P-rich waters entering the Florida Everglades; the N:P ratio of the marsh interior was similar to ratios in unimpacted portions of the Florida Everglades (Reddy and others 1999). The N:P ratio in water samples suggests that P availability limits algae production in the marsh interior but not the W-14 Canal. Increasing input of P-rich water to the marsh should increase algae production, which might increase the formation of recalcitrant, organic-P compounds.

CONCLUSIONS

Data from four cores indicated that the marshes at the Fritchie Marsh store significant amounts of N and P each year. Collectively, the marshes remove 240 t of N and 10.9 t of P from the Lake Pontchartrain estuary each year. While significant, these rates are only 42 percent of the removal rates that existed before extensive wetland loss that occurred since 1956.

Nutrient storage in wetland soils was faster at the southern end of the marshes despite nutrient concentrations being lower because soil formation there is more rapid. More rapid soil formation at the southern end of the marshes appears to result from an active fault system previously reported in the area. Additional data are needed to characterize nutrient storage given the great difference between the northern and southern ends of the study area.

Water-quality data indicated limited water exchange between the marsh interior and the adjacent water channels. Water-quality data failed to indicate nutrient uptake when mixing diagrams were used, but N:P ratios in water samples indicated that algae production in the marsh interior was P-limited when samples were collected.

The imminent wetland restoration project should increase nutrient storage rates in marsh soil and may increase the production of recalcitrant, organic forms of P that can be stored in bottom sediments of shallow, open-water areas common in the marsh interior. The imminent wetland restoration project should also reduce the hydrologic isolation of the marsh interior and may thereby reduce plant stress, increase plant production, and increase nutrient burial in soil.

REFERENCES

Antweiler, R.C.; Goolsby, D.A.; Taylor, H.E. 1995. Nutrients in the Mississippi River. In: Meade, R.H., ed. Contaminants in the Mississippi River, 1987–92. U.S. Geol. Sur. Circ. 1133. Denver: U.S. Geological Survey: 73–86.

Chabreck, R.H.; Linscombe, G. 1978. Vegetative type map of the Louisiana coastal marshes. Baton Rouge, LA: Louisiana Department of Wildlife and Fisheries.

Craft, C.B.; Richardson, C.J. 1993. Peat accumulation and N, P, and organic C accumulation in nutrient-enriched and natural everglades peatlands. Ecological Applications. 3: 446–458.

Craft, C.B.; Seneca, E.D.; Broome, S.W. 1993. Vertical accretion in microtidal regularly and irregularly flooded estuarine marshes. Estuarine, Coastal and Shelf Science. 37: 371–386.

Day, J.W.; Hall, C.A.S.; Kemp, W.M.; Yanez-Arancibia, A. 1989. Estuarine ecology. New York: John Wiley. 576 p.

DeLaune, R.D.; Patrick, W.H., Jr.; Buresh, R.J. 1978. Sedimentation rates determined by ^{137}Cs dating in a rapidly accreting salt marsh. Nature. 275: 532–533.

Department of Natural Resources. 1996. Monitoring plan. Proj. PO–06; Fritchie Marsh. Baton Rouge, LA: Louisiana Department of Natural Resources, Coastal Restoration Division.

Foret, J.D. 1997. Accretion, sedimentation, and nutrient accumulation rates as influenced by manipulations in marsh hydrology in the Chenier Plain, Louisiana. University of Southwestern Louisiana. 56 p. M.S. thesis.

Kadlec, R.H.; Knight, R.L. 1996. Treatment wetlands. Boca Raton, FL: CRC Press, Inc. 893 p.

Lane, R.R.; Day, J.W., Jr.; Thibodeaux, B. 1999. Water quality analyses of a freshwater diversion at Caernarvon, Louisiana. Estuaries. 22: 327–336.

Liss, P.S. 1976. Conservative and non-conservative behaviour of dissolved constituents during estuarine mixing. In: Burton, J.D.; Liss, P.S., eds. Estuarine chemistry. New York: Academic Press: 93–130.

Lopez, J.A. 1991. Origin of Lake Pontchartrain and the 1987 Irish Bayou earthquake. In: Coastal depositional systems of the Gulf of Mexico: quaternary framework and environmental issues. Gulf Coast Section Society of Economic Paleontologists and Mineralogists Foundation, 12th annual research conference; Austin, TX: Earth Enterprises: 103–110.

Mitsch, W.J.; Gosselink, J.G. 1986. Wetlands. Van Nostrand Reinhold Co. 539 p.

Natural Resources Conservation Service. 1997. Project plan and environmental assessment, Fritchie Marsh hydrologic restoration; PO–6; St. Tammany Parish. Alexandria, LA: U.S. Department of Agriculture, Natural Resources Conservation Service.

Nyman, J.A.; DeLaune, R.D.; Patrick, W.H., Jr. 1990. Wetland soil formation in the rapidly subsiding Mississippi River Deltaic Plain: mineral and organic matter relationships. Estuarine, Coastal and Shelf Science. 31: 57–69.

Nyman, J.A.; DeLaune, R.D.; Patrick, W.H., Jr. 1993. Relationship between vegetation and soil formation in a rapidly submerging coastal marsh. Marine Ecology Progress Series. 96: 269–279.

Parsons, T.R. 1984. A manual of chemical and biological methods for seawater analysis. New York: Pergamon Press. 173 p.

Penland, S.; Ramsey, K.E. 1990. Relative sea-level rise in Louisiana and the Gulf of Mexico: 1908–1988. Journal of Coastal Research. 6: 323–342.

Reddy, K.R.; White, J.R.; Wright, A.; Chua, T. 1999. Influence of phosphorus loading on microbial processes in the soil and water column of wetlands. In: Reddy, K.R.; O'Conner, G.A.; Schelske, C.L., eds. Phosphorus biogeochemistry in subtropical ecosystems. Boca Raton, FL: CRC Press: 249–273.

EFFECTS OF SOIL OXIDATION-REDUCTION CONDITIONS ON INTERNAL OXYGEN TRANSPORT, ROOT AERATION, AND GROWTH OF WETLAND PLANTS

S.R. Pezeshki and R.D. DeLaune[1]

Abstract—Characterization of hydric soils and the relationship between soil oxidation-reduction processes and wetland plant distribution are critical to the identification and delineation of wetlands and to our understanding of soil processes and plant functioning in wetland ecosystems. However, the information on the relationship between flood response of wetland plants and reducing soil conditions is limited. We have examined the influence of intensity and capacity of soil reduction on internal oxygen transport, rhizosphere oxygenation, nutrient uptake, root and shoot growth, and survival of several wetland species. Whereas the study species displayed a wide range of responses, intense soil reduction below -200 mV adversely affected growth and biomass accumulation in the majority of these species. It is clear that high oxygen demand in soil resulting from intense reduction influences oxygen transport and release to the rhizosphere. In addition, root elongation and shoot growth are profoundly influenced by the intensity and capacity of soil reduction.

INTRODUCTION

Numerous reviews and book chapters on plant responses to soil waterlogging are available that provide detailed literature synthesis on this topic (Armstrong and others 1994; Drew 1990, 1997; Hook and Crawford 1978; Jackson and others 1991; Kozlowski 1984a, 1984b, 1997; Pezeshki 1994; Vartapetian and Jackson 1997). Despite this wealth of literature, close examinations reveal that in most studies the status of soil oxidation-reduction conditions has not been reported. Although some researchers have reported oxygen concentrations in the root medium, such measure in wetland soil does not provide adequate information to allow evaluation of the intensity of soil reduction (DeLaune and Pezeshki 1991). This point is important because in wetland systems most plants are well adapted to endure soil-oxygen deficiency but may differ in ability to withstand certain levels of intense soil-reducing conditions.

In a typical flooded wetland soil, plants respond to the soil physicochemical changes. These responses may lead to a wide range of plant-stress symptoms.

Although various plant responses to flooded soil conditions have been addressed in numerous publications, little information can be found on the relationship between wetland plant functions and the two aspects of soil redox potential—the intensity and the capacity of reduction. The reduction of the inorganic redox systems in a flooded soil may be characterized in intensity or capacity terms. The intensity factor determines the relative ease of reduction whereas the capacity factor refers to the amount of the redox system undergoing reduction (DeLaune and Pezeshki 1991). From a plant physiological-ecology standpoint, there are many uses of interpretations of redox processes in soils; one example is that the knowledge of the soil redox potential represents an indication of the oxidation-reduction status of various soil compounds. For example, a redox potential of zero mV indicates that oxygen and nitrate are not likely to be present and that the bioreducible iron and manganese compounds are in a reduced state. At this same potential, however, sulfate is stable in the soil with no sulfide being formed, which is toxic to plants. A redox potential of +400 mV indicates that oxygen may be present even though there may be excess water in the soil (DeLaune and Pezeshki 1991). Thus, the primary objective of the present paper is to summarize and assess evidences on the significance of the intensity and capacity of reduction in soils to wetland plant functioning. We will emphasize the relationships between soil flooding, reduced soil conditions (low soil redox potential, Eh), the components of the soil oxidation-reduction system, namely the intensity and the capacity of reduction, and their influence on internal oxygen transport, rhizosphere oxygenation, root and shoot growth, and survival of wetland species.

Soil Oxidation-Reduction Potential (Eh)

Soil flooding initiates a chain of reactions leading to reduced soil conditions (low soil redox potential, Eh). These reactions include physical, chemical, and biological processes that have significant implications for wetland plant functioning, survival, and productivity (see Gambrell and others 1991, Gambrell and Patrick 1978, Ponnamperuma 1984). Physical processes include restriction of soil-atmospheric gas exchange and depletion of soil oxygen needed for root respiration. Once flooded, the limited supply of oxygen in floodwater is depleted by roots, soil microorganisms, and soil reductants rapidly (Ponnamperuma 1972). The depletion of oxygen results in a series of chemical changes in soil including accumulation of CO_2, methane, N_2, and H_2 (Ponnamperuma 1984). The processes that follow include denitrification, reduction of iron, manganese and sulfate, and

[1] Pezeshki, Professor of Biology, Department of Biology, University of Memphis, Memphis 38152, TN; DeLaune, Professor, Wetland Biogeochemistry Institute, Louisiana State University, Baton Rouge, LA 70803.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

changing soil pH and Eh (Gambrell and others 1991). In a typical series of reduction, NO_3^- is reduced to NO_2^-, followed by reductions of Mn^{+4} to Mn^{+2}, Fe^{+3} to Fe^{+2}, SO_4^{2-} to S^{2-} and accumulations of acetic and butyric acids produced by microbial metabolism (Gambrell and Patrick 1978, Ponnamperuma 1984). Thus, soil redox potential decreases (become more negative) in response to flooding (Gambrell and Patrick 1978; Patrick and DeLaune 1972, 1977; Ponnamperuma 1984). Aerated soils have characteristic redox potentials in the range of +400 to +700 mV, whereas waterlogged soils may exhibit redox potentials as low as -300 mV.

In wetland soils, plants are faced with not only the lack of oxygen but also a substantial demand for oxygen in root medium (DeLaune and others 1990, van Wijck and others 1992). Such conditions create the potential for root oxygen loss to soil; thus, additional root stress (Brix and Sorrell 1996, DeLaune and others 1990).

Quantifying Soil Redox Conditions

Many of the terms used in the literature to define flooded soil conditions are difficult to quantify. For example, “flooded”, “saturated”, or “waterlogged” are frequently used to describe low oxygen supply conditions in soils. However, these terms hardly define root-zone conditions (DeLaune and Pezeshki 1991). In addition, due to the absence of oxygen in most “waterlogged” soils, methods used for the measurement of oxygen content and oxygen diffusion rate in well-drained soils cannot be used effectively (Gambrell and others 1991).

In contrast to the mostly nonquantifiable terms mentioned above, soil Eh is a useful term because it can be measured easily in the laboratory and in the field. The electrodes are relatively easy to build and handle under most conditions (for details see Faulkner and others 1989; Patrick and DeLaune 1972, 1977). Furthermore, quantifying soil Eh is particularly advantageous in periodically flooded soils because the range of Eh is much wider, ranging between approximately -300 to +700 mV, than either aerated or permanently waterlogged soils (DeLaune and Pezeshki 1991). Thus, soil Eh measurement is perhaps the best available quantifying tool of defining soil chemical status in wetland systems (Patrick and DeLaune 1977).

Intensity and Capacity of Reduction

The reduction of the inorganic redox systems including oxygen in hydric soils following flooding can be described in “intensity” and “capacity” terms. The intensity factor determines the relative ease of the reduction, whereas the capacity factor denotes the amount of the redox system undergoing reduction, e.g., oxygen consumption at root interface. It is represented by the free energy of the reduction, or, more commonly, by the equivalent electromotive force (EMF) of the reactions (Patrick and others 1986). In natural systems such as soils where there is biological activity and where many redox systems function, the oxidation-reduction or redox potential is ordinarily used to denote the intensity of reduction. The capacity factor of a redox system probably can be best described in terms of its oxygen equivalent (Kludze and DeLaune 1995, Reddy and others 1980). The capacity factor of the various redox systems can vary from one soil to another.

Proper evaluation of wetland plant responses to soil flooding requires evaluation of both intensity and capacity of soil reduction because these two components influence oxygen demand (Kludze and DeLaune 1995). DeLaune and others (1990) were the first to demonstrate that oxygen demand (capacity of reduction) in conjunction with intensity of reduction (as determined from soil Eh measurements) in the root medium was important to predicting wetland plant functioning. Furthermore, they noted that using oxygen-depleted solutions by nitrogen gas introduction did not represent a high root-oxygen demand environment; thus, they were a poor analogue of wetland soils. DeLaune and others (1990), Kludze and others (1993), and Sorrel and others (1993), using titanium citrate solution to create a high oxygen-demand root environment, reported that root oxygen transport and release were affected by such conditions in several wetland species. However, such a solution, while a significant improvement over de-oxygenated solution, at best mimics wet soil conditions (Kludze and DeLaune 1995); but, it does not represent other important characteristics of wet soils including the soil’s capacity for phytotoxin production that has a significant effect on most wetland species. There is a critical need for development of reliable methods for quantifying capacity of reduction in soils to complement the more easily measured reduction intensity (Eh).

Plant Responses to Reducing Soil Conditions: The Internal Oxygen Transport System

In wetland plants, usually an extensive oxygen transport system of aerenchyma tissue facilitates oxygen diffusion from aerial parts to the roots. Such a system may exist in roots, stems, and leaves but is found primarily in roots (Armstrong and others 1994). This system allows a plant to transport the needed oxygen to the roots for maintaining aerobic respiration and to oxidize reducing compounds in the rhizosphere. In addition, the internal system of large gas spaces also reduces internal volume of respiring tissues and oxygen consumption, thus, enhancing the potential for oxygen reaching the distant underground portions of the plant (Armstrong and others 1994, 1996). Due to such advantages, the oxygen transport system has been considered as a major mechanism critical to a plant’s ability to cope with soil anaerobiosis (Armstrong and others 1991, 1994, 1996; Drew 1990, 1992, 1997; Hook and Crawford 1978; Kozlowski 1982, 1984a, 1984b, 1997; Pezeshki 1994, Teal and Kanwisher 1966).

Root oxygenation helps plants overcome intense anaerobiosis (Armstrong and others 1994, 1996) and has important ecological implications in wetlands. For instance, the vigor and productivity of *Spartina alterniflora* were found to be positively correlated with substrate redox potentials because of the interaction with root aeration (DeLaune and others 1983). Two related factors contribute to limit plant growth under highly reduced soil conditions. First, the low Eh levels represent an oxygen-deficient system. Second, soil phytotoxins, the byproducts of low soil Eh, may continue to accumulate to concentrations that the roots’ oxidizing power can no longer adequately ameliorate them (Mendelssohn and others 1981, Ponnamperuma 1984, Teal and Kanwisher 1966). Under such conditions, the roots must rely more heavily on anaerobic respiration or transport sufficient oxygen to roots to maintain aerobic respiration, lessening its capacity to oxidize the rhizosphere.

Internal plant air space results from aerenchyma tissue formation that is enhanced in many wetland species in response to flooding and is critical to wetland plant functioning in flooded soils (Drew and others 1985, Kozlowski 1997, Schat 1984). The effectiveness of the gas transport is primarily dependent on two factors: first, the resistance to diffusion that is proportional to root length and inversely proportional to root porosity; second, the oxygen demand along the diffusion path resulting from respiratory needs as well as oxygen leakage from the roots into the rhizosphere (Armstrong 1979, Luxmoore and others 1972). It is known that oxygen demands of roots and rhizosphere are competitive because, in flooded soils, these systems compete for the plant pool of oxygen simultaneously (Armstrong and Beckett 1987; Armstrong and others 1991, 1994). As soil reduction continues, there is a progressively greater demand imposed upon roots for oxygen (Delaune and others 1990).

Soil Oxidation-Reduction, Plant Oxygen Transport, Functioning, and Growth

There is a limited body of data on the relationship between functional aspects of gas transport within plants and soil oxidation-reduction conditions. In a few studies that evaluated the relationship between plant responses and the intensity of soil reduction, it became evident that intense soil reduction (low soil Eh) promoted oxygen loss from root to the rhizosphere (fig. 1). For instance, in some wetland species, high correlation (r = 0.96) was found between radial oxygen loss (ROL) from roots and soil Eh intensity; i.e., there was an increasingly higher oxygen loss rate as soil Eh became more reduced (Kludze and DeLaune 1995). In addition, low soil Eh led to decreased leaf carbon assimilation (fig. 2) and substantial inhibition of root elongation (fig. 3).

In *S. patens*, a dominant U.S. gulf coastal marsh species, root porosity increased as soil Eh decreased, resulting in root porosity of 22 percent in plants grown at +200 mV whereas porosity was 45 percent in plants grown at –300 mV. Also, ROL was significantly greater for plants in –300 mV Eh treatment as compared to +200 mV Eh (Kludze and DeLaune 1994). Other studies have shown similar responses for root porosity-soil Eh intensity relationship in other wetland plants including swamp and bottomland woody species (Kludze and others 1994, Pezeshki and Anderson 1997). In contrast, Brix and Sorrell (1996) reported that root porosity in two wetland species, *Phalaris arundinacea* and *Glyceria maxima*, did not change in response to reducing Eh in root medium to as low as –250 mV. However, their experiment lasted only 12 days, of which the Eh in the root medium was at –250 mV for the last 7 days; thus, plants may not have had adequate time for response. Alternatively, the two species may have already had full-scale porosity development prior to treatment initiation. However, this is an unlikely scenario because porosity ranged from 5.3 to 11.6 percent, well below the range reported for many wetland plants. The range of porosity in many wetland species is much higher, particularly if grown under low soil Eh conditions (Kludze and DeLaune

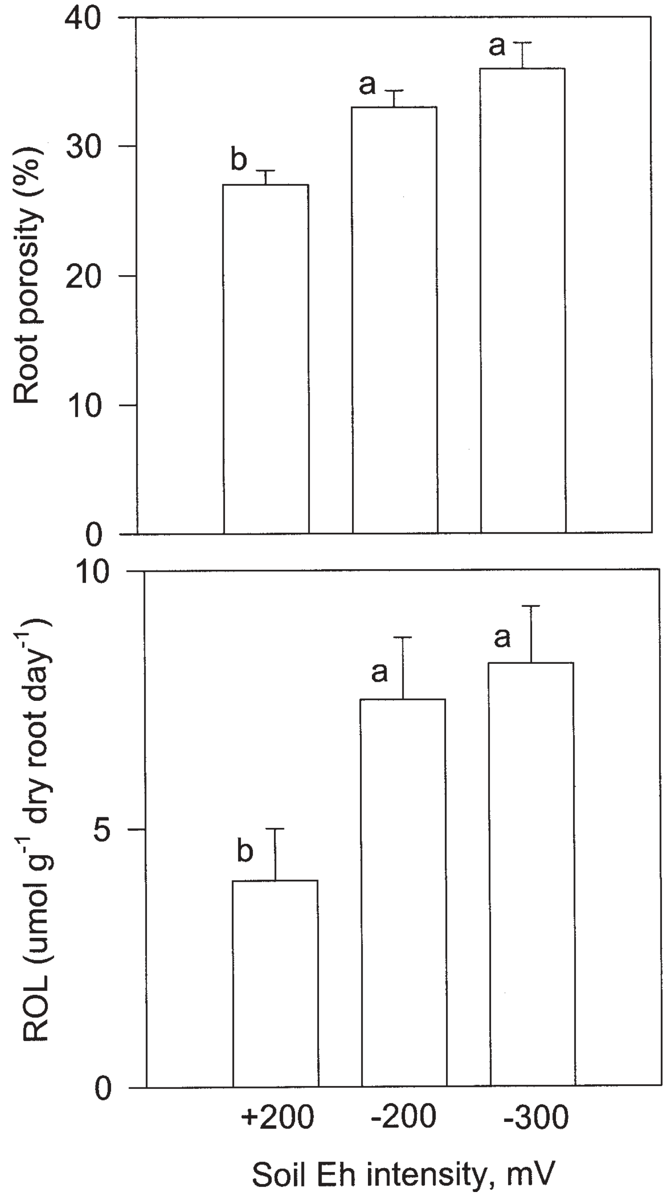


Figure 1—Root porosity (POR) and radial oxygen loss (ROL) in *Spartina patens* grown under various soil redox intensity for 50 days. Bars represent SE. (Redrawn from Kludze 1994.)

1994, Kludze and others 1994). In a study conducted on *Taxodium distichum*, significant increase in root porosity and ROL was noted at Eh intensity of –240 to –260 mV in root medium (Kludze and others 1994). ROL increased from 12.7 in control to 42.3 mmol O_2 per gram per day. Similarly, root air space increased from 13.3 to 41.4 percent in response to the intensity of reduction. In *Oryza sativa* (rice), ROL increased in response to a drop in soil Eh concomitant with root porosity that increased from 26.8 to 35 percent when Eh dropped from +200 to –300 mV (Kludze and others 1993).

Despite the reported increase in aerenchyma tissue formation in many wetland species and, thus, the increase in porosity in response to reducing soil conditions, such increase may not be sufficient to satisfy the root respiratory need for oxygen, perhaps due to the greater ROL in response to high intensity of reduction. Pezeshki and others

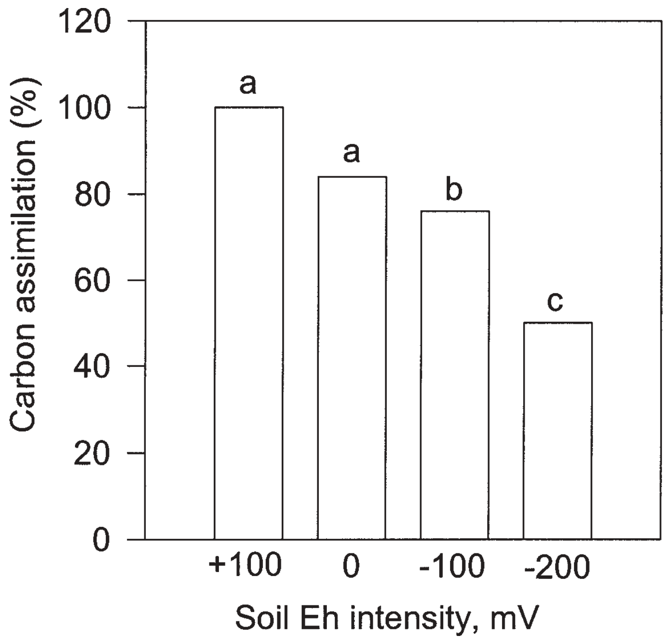


Figure 2—Carbon assimilation in *Spartina patens* grown under various soil reduction intensities. Data were collected at 17 days after growth in incubated soils. Values followed by the same letter are not significantly different at the 0.05 level. (Redrawn from Kludze 1994.)

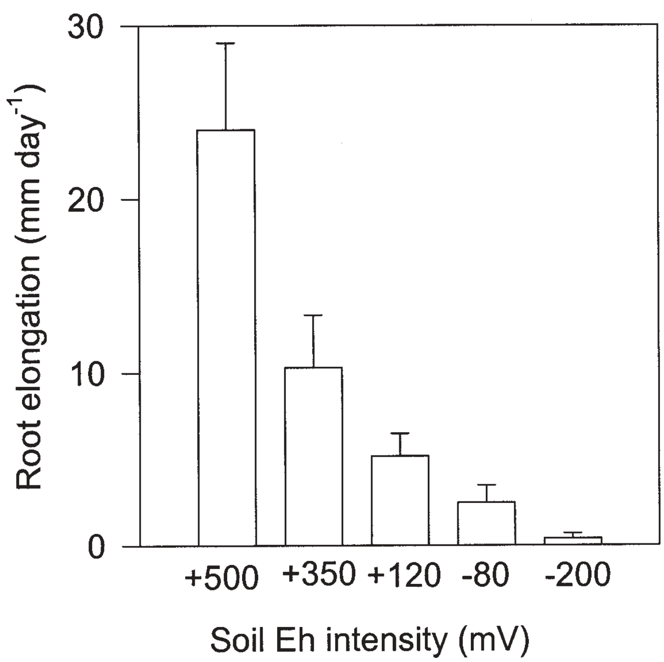


Figure 3—Root elongation in *Spartina patens* grown under various soil reduction intensities (Eh, mV). Data were collected on plants grown in different rhizotrons in which soil Eh was manipulated using various air/nitrogen gas mixtures. (Redrawn from Pezeshki and DeLaune 1990.)

(1991, 1993) concluded that despite a substantial enhancement of aerenchyma tissue formation in *Spartina patens*, alcohol dehydrogenase activity continued to be higher in flooded than control plants indicating continued oxygen stress in the roots of flooded plants. In addition, the increase in ROL reported under intense soil Eh may explain the reported reductions in root growth of several wetland species under low soil Eh conditions. For instance, in *Spartina patens* root and shoot dry weights decreased by 40 percent and 25 percent as soil Eh dropped from +200 mV to –300 mV, respectively. Results clearly indicated the influence of soil Eh intensity on growth of this marsh species. It also was demonstrated that roots were more sensitive to Eh intensity than shoots (Kludze and DeLaune 1994). Pezeshki and DeLaune (1990) reported cessation of root growth in *S. patens* at soil Eh below –100 mV (fig. 3). In addition, Pezeshki and others (1991) noted smaller root systems in *S. patens* under reducing conditions and concluded that such reduction in sink size may, in part, be responsible for a negative feed-back inhibition of photosynthesis resulting in a reduction in productivity of this species. DeLaune and others (1990) studied plant responses to the intensity of soil reduction using a titanium-citrate solution. They demonstrated that the intensity of the reduction in growth medium and the resulting demand for oxygen in the root zone exerted significant influence on plant physiological functioning.

The redox capacity factor is also important although much less is know about its effects on wetland plants than is known about the effects of the intensity factor. In fact, two different soils with the same level of intensity of reduction may differ substantially in the capacity for reduction. Soil reduction capacity can be determined using measurements of soil respiration CO_2 and calculating oxygen equivalent by stoichiometry (Kludze and Delaune 1995). Levels of soil redox capacity may be created or manipulated, or both, by providing extra carbon and energy source (organic matter) to the soil while maintaining the same redox intensity level. In an experimental set-up, reduction capacity may be controlled by adding different amounts of granular D-glucose to the root medium, which is also maintained under preset reducing conditions (Eh< +350 mV) (Kludze 1994).

As was the case for the intensity of Eh reduction, differences in Eh capacity among wetland soils may influence many plant functions (figs. 4, 5) including oxygen transport, rhizosphere oxygenation, photosynthetic rates, and, thus, many aspects of plant functioning (Kludze and DeLaune 1995). Studies showed that increased Eh capacity under a constant Eh intensity of –200 mV did not have any significant effect on root porosity in *S. patens,* but oxygen release was increased in response to the increasing Eh capacity (Kludze and DeLaune 1995). However, the authors reported that there was a threshold of Eh capacity beyond which oxygen release remained constant or decreased, or both, in this species. The response was attributed to the potential effects of several factors such as soil phytotoxins as well as plant physiological responses including stomatal closure. However, the reasons for such a response remain unknown. Plant carbon fixation, root, and shoot growth were significantly inhibited in *S. patens* under increasing soil-reduction capacity. Root and shoot dry weights decreased

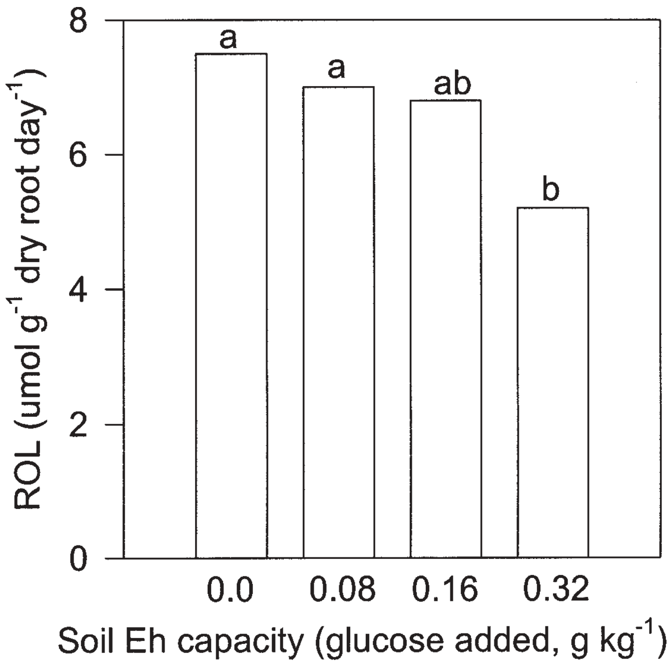


Figure 4—Radial oxygen loss (ROL) in *Spartina patens* grown under various soil reduction capacities while the reduction intensity was maintained at –200 mV. Values followed by the same letter are not significantly different at the 0.05 level. (Redrawn from Kludze and DeLaune 1995.)

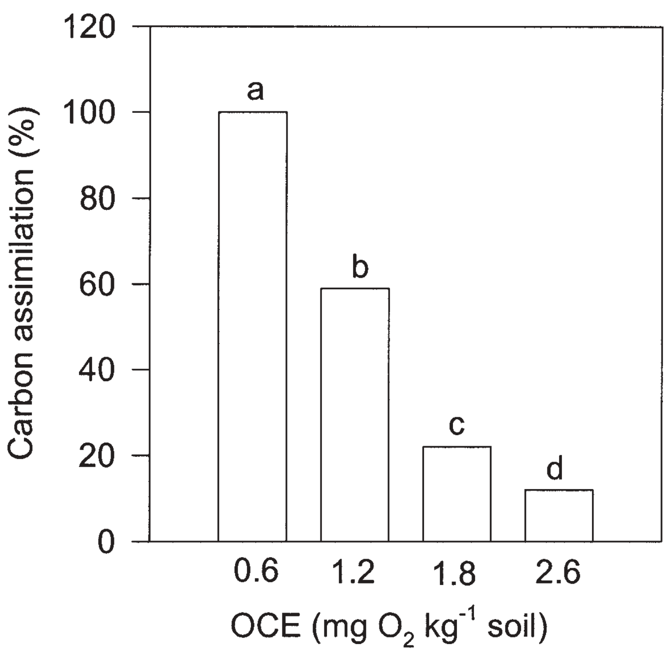


Figure 5—Carbon assimilation in *Spartina patens* grown under various soil reduction capacities. The soil redox intensity was maintained at -200 mV across all treatments. Data were collected at 17 days after growth in incubated soils. Values followed by the same letter are not significantly different at the 0.05 level. OCE: Oxygen Consumption Equivalent, mg O_2 per kilogram soil. (Redrawn from Kludze and DeLaune 1995.)

by 70 and 37 percent in high reduction capacity conditions compared to control plants, respectively (Kludze and DeLaune 1995).

In addition to the effects on growth, limited data suggest that both the intensity and capacity of reduction may govern nutrient uptake in wetland plants. In a study of seedlings of two bottomland woody species grown in soil suspension maintained at three Eh levels, +560, +340, and +175 mV, fertilizer ^{15}N uptake decreased with decreasing soil redox potential, a response to the intensity of reduction (DeLaune and others 1998). Phosphorus uptake by *Typha domingensis* was inhibited in response to decreases in soil redox potential (intensity of reduction) in rooting medium. Increasing capacity of reduction (using a titanium citrate solution) resulted in a further decrease in phosphorus uptake (DeLaune and others 1999), further confirming the effects of reduction capacity on nutrient uptake.

CONCLUDING REMARKS

Based on the limited data discussed above, both intensity and capacity of reduction appear to influence plant functioning in wetland ecosystems. In wetland soils, plants are faced with a substantial demand for oxygen in the rhizosphere and the potential for loss of oxygen to the soil resulting in additional root stress. As soil reduction continues and intensifies, there is a progressively greater demand imposed upon roots for oxygen, and, thus, a greater potential for loss of oxygen to the rhizosphere. The severity of oxygen loss and the effects of reduction intensity and capacity on plant functioning appear to be broad across wetland species. The need for additional data on various aspects of plant functioning and growth in wetland ecosystems in response to soil redox conditions, especially capacity of reduction, is clear. Redox potential measurements collected in soils using platinum electrodes provide information only on intensity of reduction. In studying plant response to a reducing environment, methods must be developed to evaluate the influence of capacity of soil reduction in addition to the intensity of soil reduction on wetland plant growth and functioning.

ACKNOWLEDGMENTS

The authors are thankful to Drs. I. Devai and A. Jugsujinda for helpful comments made on an earlier version of this manuscript.

REFERENCES

Armstrong, J.; Armstrong, W.; Van Der Putten, W.H. 1996. Phragmites die-back: bud and root death, blockage within the aeration and vascular systems and the possible role of phytotoxins. New Phytologist. 133: 399–414.

Armstrong, W. 1979. Aeration in higher plants. Advances in Botanical Research. 7: 225–332.

Armstrong, W.; Beckett, P.M. 1987. Internal aeration and the development of stelar anoxia in submerged roots: a multi-shelled mathematical model combining axial diffusion of oxygen in the cortex with radial losses to the stele, the wall layers, and the rhizosphere. New Phytologist. 105: 221–245.

Armstrong, W.; Brandle, R.; Jackson, M.B. 1994. Mechanisms of flood tolerance in plants. Acta Botanica Neerlandica. 43: 307–358.

Armstrong, W.; Justin, S.H.F.W.; Beckett, P.M.; Lythe, S. 1991. Root adaptation to soil waterlogging. Aquatic Botany. 39: 57–73.

Brix, H.; Sorrell, B.K. 1996. Oxygen stress in wetland plants: comparison of de-oxygenated and reducing root environments. Functional Ecology. 10: 521–526.

DeLaune, R.D.; Jugsujnda, A.; Reddy, K.R. 1999. Effect of root oxygen stress on phosphorus uptake by cattail. Journal of Plant Nutrition. 22: 459–466.

DeLaune, R.D.; Pezeshki; S.R. 1991. Role of soil chemistry in vegetative ecology of wetlands. Trends in Soil Science. 1: 101–113.

DeLaune, R.D.; Pezeshki, S.R.; Lindau, C.W. 1998. Influence of soil redox potential on nitrogen uptake and growth of wetland oak seedlings. Journal of Plant Nutrition. 21: 757–768.

DeLaune, R.D.; Pezeshki, S.R.; Pardue, J.H. 1990. An oxidation-reduction buffer for evaluating physiological response of plants to root oxygen stress. Environmental & Experimental Botany. 30(2): 243–247.

DeLaune, R.D.; Smith, C.J.; Patrick, W.H. 1983. Relationship of marsh elevation, redox potential and sulfide to *Spartina alterniflora* productivity. Soil Science Society of America Journal. 47: 930–935.

Drew, M.C. 1990. Sensing soil oxygen. Plant Cell and Environment. 13: 681–693.

Drew, M.C. 1992. Soil aeration and plant root metabolism. Soil Science. 154: 259–268.

Drew, M.C. 1997. Oxygen deficiency and root metabolism: injury and acclimation under hypoxia and anoxia. Annual Review of Plant Physiology and Plant Molecular Biology. 48: 223–250.

Drew, M.C.; Saglio, P.H.; Pradet, A. 1985. Larger adenylate energy charge and ATP/ADP ratios in aerenchymatous roots of *Zea mays* in anaerobic media as a consequence of improved internal oxygen transport. Planta. 165: 51–58.

Faulkner, S.P.; Patrick, W.H.; Gambrell, R.P. 1989. Field techniques for measuring wetland soil parameters. Soil Science Society of America Journal. 53: 883–890.

Gambrell, R.P.; DeLaune, R.D.; Patrick, W.H. 1991. Redox processes in soils following oxygen depletion. In: Jackson, M.B.; Davies, D.D.; Lambers, H., eds. Plant life under oxygen deprivation: ecology, physiology, and biochemistry. The Hague, The Netherlands: SPB Academic Publishing: 101–117.

Gambrell, R.P.; Patrick, W.H. 1978. Chemical and microbiological properties of anaerobic soils and sediments. In: Hook, D.D.; Crawford, R.M.M., eds. Plant life in anaerobic environments. Ann Arbor, MI: Ann Arbor Science: 375–423.

Hook, D.D.; Crawford, R.M.M. 1978. Plant Life in anaerobic environments. Woburn, MA: Ann Arbor Science. 564 p.

Jackson, M.B.; Davies, D.D.; Lambers, H. 1991. Plant life under oxygen deprivation: ecology, physiology, and biochemistry. The Hague: SPB Publishing. 326 p.

Kludze, H.K. 1994. Gaseous exchange and wetland plant response to soil redox conditions. Baton Rouge, LA: Louisiana State University. 155 p. Ph.D. dissertation.

Kludze, H.K.; DeLaune, R.D. 1994. Methane emission and growth of *Spartina patens* in response to soil redox intensity. Science Society of America Journal. 58: 1838–1845.

Kludze, H.K.; DeLaune, R.D. 1995. Straw application effects on methane and oxygen exchange and growth in rice. Science Society of America Journal. 59: 824–830.

Kludze, H.K.; DeLaune, R.D.; Patrick, W.H. 1993. Aerenchyma formation and methane and oxygen exchange in rice. Science Society of America Journal. 51: 386–391.

Kludze, H.K.; Pezeshki, S.R.; DeLaune, R.D. 1994. Evaluation of root oxygenation and growth in baldcypress in response to short-term soil hypoxia. Canadian Journal of Forest Research. 24: 804–809.

Kozlowski, T.T. 1982. Water supply and tree growth. II: flooding. Forestry Abstracts. 43: 145–161.

Kozlowski, T.T., ed. 1984a. Flooding and plant growth. New York: Academic Press: 129–163.

Kozlowski, T.T. 1984b. Plant responses to flooding of soil. Bioscience. 34: 162–167.

Kozlowski, T.T. 1997. Responses of woody plants to flooding and salinity. Tree Physiology Monographs. 1: 1–29.

Luxmoore, R.J.; Sojka, R.E.; Stolzy, L.H. 1972. Root porosity and growth responses of wheat to aeration and light intensity. Soil Science. 113: 354–357.

Mendelssohn, I.A.; McKee, K.L.; Patrick, W.H. 1981. Oxygen deficiency in *Spartina alterniflora* roots: metabolic adaptation to anoxia. Science. 214: 439–441.

Patrick, W.H.; DeLaune, R.D. 1972. Characterization of the oxidized and reduced zones on flooded soil. Soil Science Society American Louisiana. Estuarine Coastal & Marine Science. 4: 59–64.

Patrick, W.H.; DeLaune, R.D. 1977. Chemical and biological redox systems affecting nutrient availability in the coastal wetlands. Geoscience and Man: 131–137. Vol. 18.

Patrick, W.H.; Gambrell, R.P.; Faulkner, S.P. 1986. Redox measurements of soils. In: Methods of soil analysis. 3: chemical methods. Book Ser. 5. Soil Science Society of America: 1255–1273.

Pezeshki, S.R. 1994. Plant responses to flooding. In: Wilkinson, R.E., ed. Plant-environment interactions. New York: Marcel Dekker, Inc.: 289–321.

Pezeshki, S.R.; Anderson, P.A. 1997. Responses of three bottomland woody species with different flood-tolerance capabilities to various flooding regimes. Wetland Ecology and Management. 4: 245–256.

Pezeshki, S.R.; DeLaune, R.D. 1990. Influence of sediment oxidation-reduction potential on root elongation in *Spartina patens*. Acta Oecologia. 11: 377–383.

Pezeshki, S.R.; Matthews, S.W.; DeLaune, R.D. 1991. Root cortex structure and metabolic response of *Spartina patens* to soil redox conditions. Environmental & Experimental Botany. 31(1): 91–97.

Pezeshki, S.R.; Pardue, J.H.; DeLaune, R.D. 1993. The influence of oxygen deficiency and redox potential on alcohol dehydrogenase activity, root porosity, ehtylene production, and photosynthesis in *Spartina patens*. Environmental & Experimental Botany. 33: 565–573.

Pezeshki, S.R.; Pardue, J.H.; DeLaune, R.D. 1996. Leaf gas exchange and growth of flood-tolerant and flood-sensitive tree species to soil oxygen deficiency. Tree Physiology. 16: 453–458.

Ponnamperuma, F.N. 1972. The chemistry of submerged soil. Advances in Agronomy. 24: 29–96.

Ponnamperuma, F.N. 1984. Effects of flooding on soils. In: Kozlowski, T.T., ed. Flooding and plant growth. Orlando, FL: Academic Press Inc.: 1–44.

Reddy, K.R.; Ras, P.S.C.; Patrick, W.H. 1980. Factors influencing oxygen consumption rates in flooded soil. Science Society of America Journal. 44: 741–744.

Schat, H. 1984. A comparative ecophysiological study on the effects of waterlogging and submergence on dune slack plants: growth, survival, and mineral nutrition in sand culture experimens. Oecologia. 62: 279–286.

Sorrell, B.K.; Brix, H.; Orr, P.T. 1993. Oxygen exchange by entire root systems of *Cyperus involucratus* and *Eleocharis sphacelata*. Journal of Aquatic Plant Management. 31: 24–28.

Teal, J.M.; Kanwisher, J.W. 1966. Gas transport in the marsh grass *Spartina alterniflora*. Journal of Experimental Botany. 17: 355–361.

Van Wijck, C.; de Groot, C.J.; Grillas, P. 1992. The effect of anaerobic sediment on the growth of *Potamogeton pectinatus L*.: the role of organic matter, sulphide, and ferrous iron. Aquatic Botany. 44: 31–49.

Vartapetian, B.B.; Jackson, M.B. 1997. Plant adaptations to anaerobic stress. Annals of Botany. 79 (Supp. A): 3–20.

Williams, W.T.; Barber, D.A. 1961. The functional significance of aerenchyma in plants. S. Exper. Bot. Sym. 15: 132–144.

RIVERBANK RESTORATION IN THE SOUTHERN UNITED STATES: THE EFFECTS OF SOIL TEXTURE AND MOISTURE REGIME ON SURVIVAL AND GROWTH OF WILLOW POSTS

S. Reza Pezeshki, Steven D. Schaff, and F. Douglas Shields, Jr.[1]

Abstract—Field studies were conducted to quantify the relationship between soil conditions and growth of black willow posts planted for riverbank erosion control along Harland Creek (HC) and Twentymile Creek (TC) sites in Mississippi. Both sites had a wide range of soil texture and moisture regimes. Soil texture, water level, redox potential (Eh), and willow survival and growth were monitored. At the HC site, growth was lower in posts located at elevational extremes than for posts planted approximately 0.5 m above creek elevation at baseflow. Optimum conditions for growth were provided at moderate elevations characterized by groundwater levels that fluctuated around 50 cm beneath the soil surface. Data from the TC site also indicated a close correlation between soil texture, moisture, and survival of willows. Posts grown in silty-clay soils displayed low survival and growth in comparison to those grown in sandy soils. Locations along the creek characterized by sandy soils, adequate soil moisture, and well-drained conditions resulted in high survival and growth.

INTRODUCTION

Riverbanks and stream reaches in the Southern United States are often subject to accelerated erosion and associated habitat deterioration. However, there are opportunities for both erosion control and improvement of natural habitat. Many techniques are presently used for riverbank restoration including planting vegetation primarily composed of woody species on eroding banks. Willow posts (Salix spp.) have been extensively utilized for such restoration (Roseboom 1993; Shields and others 1995a, 1995b; Watson and others 1997). Large-diameter (> 7.5 cm) willow cuttings with lengths up to 4 m are sometimes used to stabilize rapidly eroding banks. The deeply planted posts provide mechanical control of erosion during the period of initial growth and establishment. A dense array of growing willow posts is intended to modify riparian habitats, fostering natural succession to a diverse riparian plant community and relatively stable banks. Despite reported success on many restoration sites, low survival rates have been reported in certain areas. For instance, survival rates of > 40 percent by the end of the first growing season have been reported for northern Mississippi (Shields and others 1995b). This has been attributed to many factors including post location on the bank, parasites, flooding, drought, and soil texture (Shields and others 1998). Field observations suggest that regimes of excess and deficit soil moisture may be primary factors. Experimental evidence on adverse extreme soil moisture effects have come primarily from a greenhouse study conducted on black willow (Salix nigra Marsh.) cuttings and have shown flooding or drought have significant adverse effects on physiological functioning, growth, and biomass production (Pezeshki and others 1998).

Field observations also suggest the potential effects of soil moisture on the growth of willow cuttings. Willow posts planted on steep banks appeared to do better in middle elevation as compared to those planted close to the river or at higher elevations further from the river (Shields and others 1998). At the two elevational extremes, willows experience quite different situations. At high elevations, periodic soil drought may impose severe water stress to which the posts are likely to be sensitive. In contrast, at low elevations close to the permanent water table, willows are likely to experience frequent root inundation due to soil flooding. In such conditions, willows will be subjected to soil and, thus, root oxygen deficiency as well as to soil chemical changes that result from flooding (Pezeshki 1994). The effects of soil oxidation-reduction conditions (redox potential, Eh) could be significant because reducing soil conditions (low soil Eh) may impose substantial stress on plants including willow, a species considered to be flood tolerant (Hook 1984). Depending on the level of soil Eh in the root zone, significant reductions in root growth can occur in many flood-tolerant woody species (Pezeshki 1991). In addition, substantial alterations in normal root metabolic activities may occur leading to the disruption of various root processes critical to a plant's functioning. Such processes include water and nutrient uptake, production of metabolites, and oxidation of the immediate rhizosphere under flooded conditions (Pezeshki 1994). Partial or complete death of roots may occur leading to insufficient masses of functional roots, and this may create root/shoot imbalances. Such imbalances may initiate a chain of events leading to shoot-water deficits and massive root and shoot death (Kozlowski 1982, 1997; Pezeshki 1991, 1994).

Soil texture is also critical because soil-pore space, moisture-holding capacity, and oxidation-reduction capacity are closely correlated with texture. Poor performance of willow posts planted on streambanks composed of fine soils has been observed in the field (Abt and others 1996). Posts

[1] Pezeshki, Professor of Biology, and Schaff, Graduate Student, Department of Biology, The University of Memphis, Memphis, TN 38152; Shields, Research Hydraulic Engineer, National Sedimentation Laboratory, USDA-ARS, Oxford, MS 38655.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

also could experience severe drought in coarse soils (sand-gravel) on steep banks during the low-precipitation periods due to low moisture-holding capacity of coarse soils. Thus, both soil-moisture regime and texture are inter-related and may play a critical role in the success of willow posts planted on a restoration site.

Clearly, the poor performance of willow posts involves a complex set of environmental factors to which the posts are responding. The environmental conditions and the associated plant responses change substantially over the course of a typical growing season in the highly dynamic riverbank systems. Therefore, the present studies were designed to quantify the relationship between soil texture and moisture regime and physiological functions, survival, and growth of willow posts under field conditions.

MATERIALS AND METHODS

Field work was conducted on black willow (*Salix nigra* Marsh.) posts planted for streambank restoration at two restoration sites in northern Mississippi, the Harland Creek (HC) restoration site, and the Twentymile Creek (TC) restoration site (fig. 1).

Harland Creek Restoration Site

The HC restoration site was located in Holmes County, MS (fig.1). Black willow posts were harvested from local populations during the dormant season in 1994. The posts were approximately 3 m in length with a minimum diameter of 7.5 cm at the base. Posts were placed in holes 2.4 m deep, thus, approximately 0.6 m of the post remained aboveground. Willow posts were spaced at 0.9 m-centers for four rows of posts. The first row was placed 0.6 m from where the stream channel met the bank at the time of planting. Posts were planted along the bank in February 1994. In 1996, a total of 13 posts were selected along an elevational gradient extending from the upstream and ending at the last downstream post. These posts were located at various distances from the creek. PVC piezometers with a diameter of 3.1 cm were established in proximity of each post. Measurement locations were also established at each piezometer to monitor soil Eh. Water, soil, and plant measurements (described in Methodologies) were continued through the end of the 1996 growing season (November).

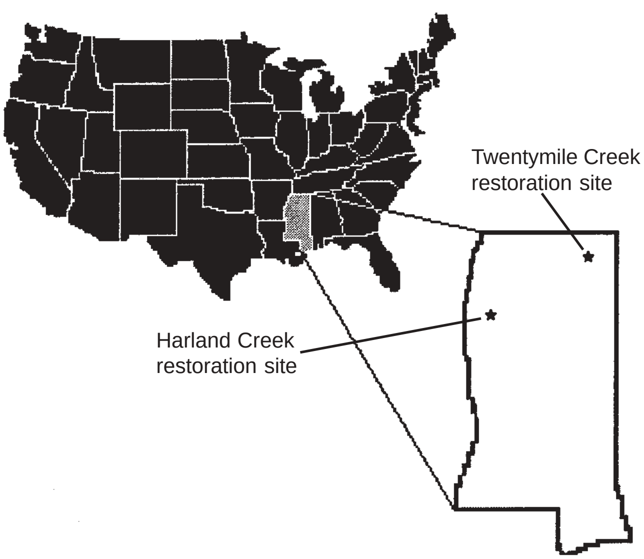


Figure 1—Map of the United States showing locations of the Harland Creek and Twentymile Creek restoration sites in northern Mississippi, USA.

Twentymile Creek Restoration Site

The TC restoration site, located in Lee County, MS (fig. 1), was the subject of intensive data collection. Black willow posts were planted according to the methodology described above in February 1998. At this site, four transects were established along the creek. Each transect was placed perpendicular to the creek and extended from the toe of the creek to the edge of the planting zone, approximately half way up the bank. Three plots were located within each transect: one at each end of the elevational extremes and one in the middle elevation. Piezometers were installed in the middle of each plot to monitor the water level below the soil surface. Each plot consisted of 12 posts. The posts were grouped into 2 rows each consisting of 6 plants, parallel to the stream for a total of 12 posts per plot. Soil and plant measurements were initiated in the spring of 1998 and continued through the end of the growing season.

METHODOLOGIES

The frequency and duration of flooding of both sites was analyzed using stage data collected at 15-minute intervals from nearby U.S. Geological Survey gauging stations. The TC transects were about 0.4 to 1.7 km downstream from the gauge, whereas the HC site was about 6.3 km upstream from the gauge. The applicability of the HC gauge data to our site was verified by comparing the signal from the U.S. Geological Survey gauge with a temporary gauge located about 1 km upstream from our site. Stages were transformed into height above baseflow elevation for assessing impacts at each site.

At both sites, soil samples were collected from 15, 30, 60, and 90 cm below the soil surface during piezometer installation. These samples were used to characterize the soil texture for each post location using standard methods (Brady 1974). For the HC site, the percentage of sand, silt, and clay are approximate as determined by the “feel method.” For the TC site, soil particle size distribution was determined by the “feel method” and by the USDA Agricultural Research Service, Midsouth Area, using a laser scattering particle size distribution analyzer (Horiba, Model LA-910).

At each site, soil redox potential (Eh) was measured at a distance of 15 cm from each well at 15-, 30-, 60-, and 90-cm depths below the soil surface using platinum-tipped redox electrodes, a millivoltmeter (Orion, Model 250A), and a calomel reference electrode (Corning, Model 476350). Measurements were recorded after the redox probes had been in the ground at least for 2 hours to allow for equilibration of the electrodes. Measurements of Eh were replicated (two measurements at each depth) per sampling location per sampling date. The water level at each monitoring well was quantified using a measuring tape.

Plant measurements at each site included selected physiological measurements and growth responses. Gas exchange measurements (leaf conductance and net carbon fixation) were conducted approximately twice each month using a portable photosynthesis system (CIRAS1, PP Systems, England). Measurements were conducted on mature, intact, attached leaves located on the upper one-third of the branches on each post between the hours of 11:00 and 14:00 during each site visit. The environmental factors of air temperature, leaf temperature, and photosynthetic photon flux density (PPFD) were recorded followed by measurements of net photosynthesis (Pn) and stomatal conductance (gw) on each sample leaf. Replicated sample leaves were used for the measurements during each field visit.

The growth of posts at the HC site was quantified by biomass sampling at the end of the growing season. Posts were destructively sampled and dry weight of biomass components was determined (Shields and others 1998). Growth of posts at the TC site was determined by measurement of height on study posts at the end of the 1998 growing season. No destructive samples were conducted on these posts because the posts were to be utilized in a longer study. The general linear models (GLM) procedure of the Statistical Analysis System was used to test for differences in means among different posts planted at various elevations relative to the creek. Regressions for the relationships between post survival, height growth, and soil texture were calculated using the SAS System (Statistical Analysis Systems 1994).

RESULTS

Harland Creek Restoration Site

Posts were planted at the HC site in February 1994, and we conducted measurements during the 1996 growing season. Rainfall and runoff patterns during the period 1994–96 were within limits typical of the period of record. However, in 1994 precipitation and streamflow were well above average during late summer. During our study, inundation of the posts was brief and infrequent. The median duration of flooding ranged from about 26 hours for the lowest posts to 15 hours for the highest posts. Median times between flooding ranged from more than 8 to more than 13 days, depending on elevation.

At the HC site, study posts were grouped into four distinct groups based on soil characteristics, relative elevation from the creek, and the distance to ground water (table 1). Posts in the upper part of the bank (group I) where soil texture was sandy gravel experienced little or no soil flooding as was evidenced by soil Eh data (> 350 mV). However, due to location and soil texture, the posts were likely to have encountered drought. Net carbon fixation indicated a significantly lower mean as compared to the posts that were located in the middle slopes (group II, percent III). No biomass sampling was done for this group, but gas exchange data suggest that these posts were stressed. Groups II and III consisted of posts located in the middle-bank elevation as verified by elevation and depth to water table (table 1). These posts had the highest measured net carbon assimilation rates among the study groups. The two groups showed no significant differences in gas exchange responses, but the mean (root and shoot) biomass per post was significantly greater in group II than group III (table 1). The difference in the observed responses between the two groups was attributed to the difference in soil texture that was more favorable for group II (less cohesive, higher Eh) than group III (more cohesive, lower Eh). Group IV posts were located at low elevations in sand-gravel soil. The water table was higher for this group as compared for the middle-bank groups; thus, soil Eh was in the reducing range. However, it was not significantly greater in intensity compared to the middle-bank groups. This finding was primarily attributed to the sand-gravel soil texture, which had little capacity for intense reduction. Examination of root

Table 1—Field data of post location, soil texture, depth to water table, soil Eh, net carbon fixation, root dry biomass, and shoot dry biomass for posts studied at Harland Creek restoration site, Mississippi

	Post location on the streambank			
Variable	Lower bank group IV	Medium group III	Medium group II	Upper bank group I
Soil texture	Sand-gravel	Sand-silt-clay	Sand-silt	Sand-gravel
Water depth (m)	0.33 c	0.58 b	0.54 b	1.13 a
Soil Eh (mV)	202 b	107 b	172 b	384 a
Net carbon fixation (μmol CO_2 m^{-2} leaf s^{-1})	8 b	10 a	11 a	7 b
Root dry mass (g)	100	29 b	192 a	NA
Shoot dry mass (g)	24	173 b	1029 a	NA

NA = missing data.
Different letters within each column indicate statistically significant differences ($p< 0.05$).

distribution patterns showed that for posts located at low elevations characterized by low soil Eh conditions, 100 percent of roots excavated were located between a depth of 0 to 15 cm. Posts at medium elevation showed substantially different root-growth patterns depending on the soil texture. Group II located on sand-silt soil had significantly higher root biomass than group III located on soils that contained some clay (fig. 2). In addition, roots were more uniformly distributed at various depths in soil profile in group II whereas roots were confined to the top 15 cm of the soil in group III. Overall, conditions for growth of willow posts at HC site were best at moderate elevations characterized by sand-silt soil texture and water levels that fluctuated close to the soil surface allowing for adequate soil moisture but frequent drainage. In contrast, posts located at higher elevations relative to the baseflow stage suffered from soil moisture deficits whereas posts located at the bank toe were hampered by flooded soils and low soil Eh conditions.

Twentymile Creek Restoration Site

At the TC site, the study plots on each transect were located along an elevational gradient. At each transect, the low-elevation plot was flooded frequently due to proximity to the creek, which rose whenever light to moderate rains fell. The middle plot flooded at a moderate frequency but required moderate to heavy rainfall throughout the watershed whereas the high-elevation plot was located at the top of the planting zone and was never flooded. Rainfall and runoff patterns during the 1998 growing season were typical of the period of record. Plots located 0.3 m above baseflow were flooded about 10 percent of the time during March through July (later data not available at this writing), whereas those at intermediate elevations (+ 1.0 to 1.5 m above base stage) were flooded only about 1 percent of the time, and those at highest elevations (>3 m above base stage) were not flooded during this period. Flooding episodes lasted only 2 to 39 hours, and median times between flooding ranged from 8 to 26 days, depending on elevation.

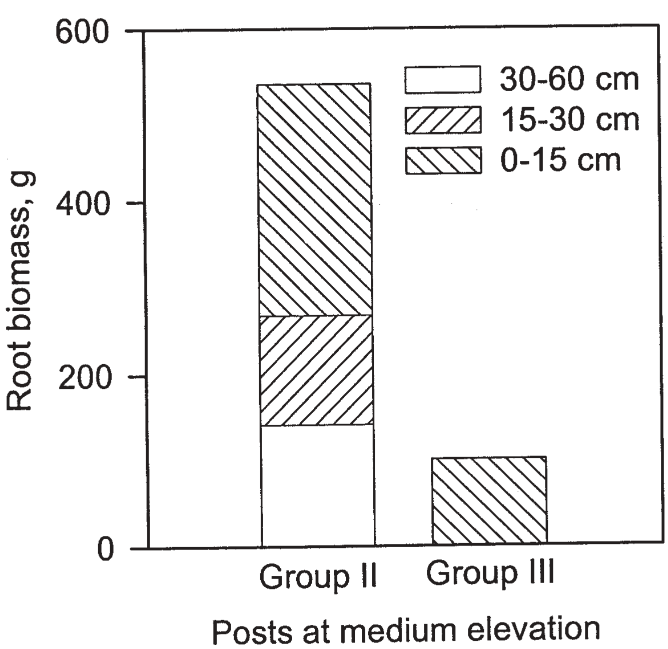


Figure 2—Total root biomass distribution for willow posts in soil profile at three depths: 0 to 15 cm, 15 to 30 cm, 30 to 60 cm, for posts harvested at Harland Creek site, Mississippi. Data for group II represent the mean for two posts while data of group III represent the mean for four posts.

The soils in each of the elevational categories were fairly similar in texture ranging between 76 to 62 percent coarse sandy materials (table 2). The low-elevation plots were closest to the water table (0.3 m) and had moderately reduced soils due to the saturated conditions (Eh +230 mV). In contrast, the medium- and high-elevation plots had aerated soils with Eh values of +480 and +520 mV, respectively (table 2).

Height growth, survival rates, and gas-exchange measurements showed no significant differences across the elevational groups. However, posts in the medium-elevation plots grew 98.4 cm on average, 196 percent more than the 50.2-cm average for low-elevation plots and 154 percent more than the 63.8 cm recorded for the high-elevation plots. The high-elevation plots had the highest survival rate at 67 percent followed by the middle plots at 50 percent and the low plots at 46 percent (table 2). However, differences in growth and survival were not statistically significant.

The soil texture effects were apparent when plots of the same elevation were compared between reach 1 and reach 3. The amount of fine-grained soil (silt and clay) in reach 3 plots was double that of reach 1 plots at all elevations. The depth to the water table showed no significant difference across the lowest plots; however, there were slight differences (0.5 and 0.3 m, respectively) between the middle- and high-elevation plots (table 3). This difference may be biologically insignificant, however, because this is well below the root establishment zone. Similarly, the only

Table 2—Field data of post location, soil texture, depth to water table, soil Eh, height growth, survival, and net carbon fixation for willow posts studied at the Twentymile Creek restoration site, Mississippi

	Post location on the streambank		
Variable	Lower bank	Medium	Upper bank
Soil texture (percent coarse)	70	62	76
Water depth (m)	0.3 a	1.4 b	1.7 c
Soil Eh (mV)	230 a	480 b	520 c
Height growth (cm)	50.2 a	98.4 a	63.8 a
Survival (percent)	46 a	50 a	67 a
Net carbon fixation (μmol CO_2 m^{-2} leaf s^{-1})	11.7 a	12.3 a	10.3 a

Means in each row followed by the same letter are not significantly different across groups at the $p < 0.05$ level.

Table 3—Field data comparing the effects of soil texture on net carbon fixation when plot elevation (thus, depth to water table and soil redox potential) is held constant. Data represents mean values taken over the first growing season for depth to water table, soil Eh, and net carbon fixation for willow posts studied at Twentymile Creek restoration site, Mississippi

		Silt + clay	Sand	H_2O depth	Soil Eh	Pn
		- - - - Percent - - -		*m*	*mV*	*μmol CO_2 m^{-2} leaf s^{-1}*
Low plots	Reach 1	23	77	0.26 a	146 a	12.4 a
	Reach 3	48	52	.31 a	259 b	11.0 a
Middle plots	Reach 1	29	71	1.2 a	378 a	13.9 a
	Reach 3	67	33	1.7 b	376 a	9.7 a
High plots	Reach 1	11	89	1.8 a	553 a	11.7 a
	Reach 3	30	70	1.5 b	491 b	7.0 b

Means in each column for each elevation followed by the same letter are not significantly different across groups at the $p < 0.05$ level.

plots to have biologically significant differences in soil Eh were the low plots. The low plot in reach 1 had a significantly lower Eh value than in reach 3, even though there was only a 3-cm difference in the depth to the water table, and the majority of the soil was sand, which characteristically has a lower capacity for reduction. Nevertheless, both plots were in the anaerobic range (below + 350 mV). All of the middle and high plots had Eh values indicative of aerated soils (above + 350 mV) (table 3).

Gas exchange data showed a consistent pattern of response to soil texture. Net photosynthetic rates were consistently higher in sandy soils; however, rates were significantly higher only in high-elevation plots. In these plots, average carbon fixation was 40 percent higher in coarse-grained sandy soils than in fine soils (table 3). The middle plots showed moderately significant results ($p = 0.0545$) with an average increase in carbon fixation of over 30 percent. Average height growth per post followed similar trends (fig. 3). The low-elevation plots failed to show a significant difference in growth; however, posts grown in sandy soils were considerably taller than posts grown in silt and clay soils (139 cm and 77 cm, respectively). Posts grown in coarse soils showed a significantly greater amount of growth at both the middle- and high-elevation plots. Posts grown in sandy soils in the middle-elevation plots showed a threefold increase in growth (246 cm compared to 74 cm) and a fourfold increase at the high-elevation plots (131 cm compared to 31 cm) (fig. 3). Post survival showed the most consistent significant effects of soil texture as posts grown in sandy soils had greater survival in all plots. Posts planted in predominantly sandy soils had mean survival rates of 67 percent, 75 percent, and 92 percent, for low-, middle-, and high-elevation plots, respectively. In contrast, posts in fine-grained soil (silty/clay soils) had significantly lower survival rates of 25 percent, 25 percent, and 42 percent, for the same respective plot elevations (fig. 3). The relationship between willow survival and soil conditions indicated that

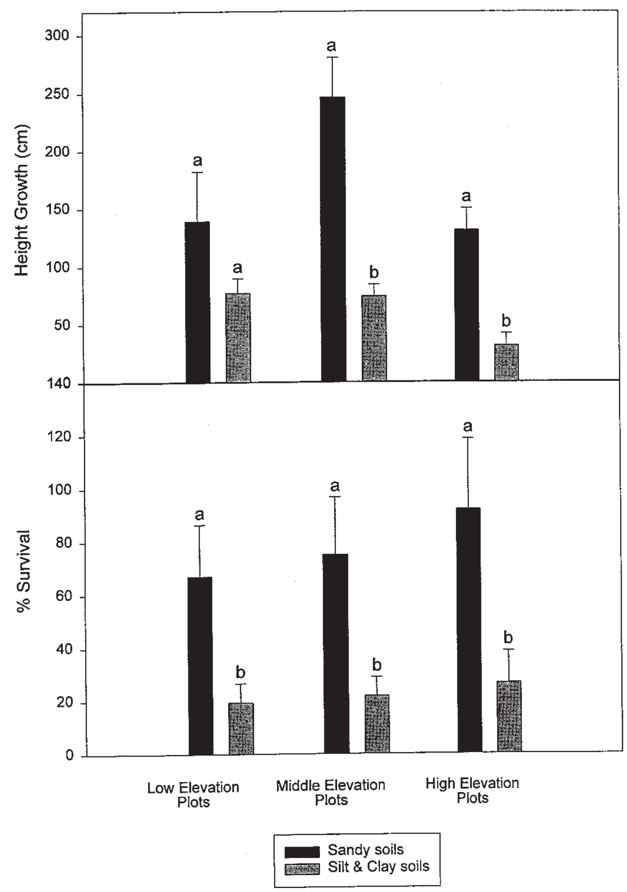


Figure 3—Survival and height growth of willow posts planted on fine vs. coarse soils at the Twentymile Creek site, Mississippi. Data represents the mean height growth per post (cm) and percent survival per plot for posts grown at similar elevation with varying soil textures. Sandy soils were consistently observed at reach 1 in comparison to the silt and clay soils observed at reach 3 (for specific proportions of each soil texture see table 3). Means for each elevation with similar letters represent no significant difference at the $p < 0.05$ level.

posts appear to have low-survival rates in soils containing a high percentage of fine particles (silt plus clay) within the upper soil layer of 0 to 30 cm in depth. Height growth was also sensitive to a high percentage of fine particles.

DISCUSSION

The results from both restoration sites demonstrated the complexity of soil-plant interactions in highly dynamic creek banks such as the HC and the TC restoration sites that are characterized by steep slopes, a wide range of soil moisture conditions, and various soil textures. Willow posts displayed a great level of sensitivity to drought and flooding (low soil Eh conditions). For instance, on the HC site, soil Eh for posts in group I was above + 350 mV, thus, in the aerated range, suggesting that these posts were not exposed to flooding or soil saturation during any of the measurement days (table 1). Nevertheless, this group had the lowest gas exchange rates relative to other groups. This finding was attributed to periodic drought encountered by this group due to elevation and the poor water-holding capacity of the soil (sand-gravel). Similar results were obtained in a greenhouse study where posts under drought treatment had low photosynthetic rates (Pezeshki and others 1998). In contrast, posts in group IV were at the lowest elevations to the creek relative to other groups. Here, water levels were high (close to the soil surface) creating reducing soil conditions indicated by low Eh values (+202 mV). This group had significantly lower gas exchange rates relative to other groups (II and III). In a greenhouse study, willow posts subjected to a continuously flooded condition had significantly lower net photosynthesis compared to control (Pezeshki and others 1998). Reducing soil Eh can adversely affect photosynthetic rates in species considered to be flood tolerant (Pezeshki 1994). Field root biomass data indicated that reductions in soil Eh had a negative impact on root biomass. Root development is slowed in areas of low soil Eh conditions because of the lack of oxygen, which is needed to carry out normal root respiration. In a greenhouse study, little or no root growth was found for black willow posts at locations that remained subjected to continuous flooding and soil Eh around - 200 mV (Pezeshki and others 1998). This lack of roots indicated that willow posts either failed to initiate roots or that such roots, if initiated, died in response to the continued, intense soil-reducing conditions. Other studies have shown that root biomass in black willow was significantly decreased as flooding depth and duration increased (Donovan and others 1988).

Black willow is a flood-tolerant species; thus, it possesses physiological/morphological adaptations allowing oxygen transport to the root zone. However, as soil Eh decreases, the chemical and biological demand for oxygen within the soil rises. Thus, the oxygen-delivery system may be overwhelmed by such increasing demand that is concomitant with increased internal demand for oxygen (Pezeshki 1994). The lower gas exchange rates found in posts under drought and flooding on HC site partially explain the observations on changes in vigor and survival of willow posts due to elevation reported by Abt and others (1996). These observations agreed that posts planted along an elevational gradient from the creek differed in survival rates. Posts located close to the creek at similar elevations to the creek and exposed to prolonged, frequent flooding had mortality rates of 80 percent. The posts at the highest elevations and greatest distance from the creek, which were likely to experience droughty conditions, had mortality rates of about 91 percent. However, higher survival rates of 39 to 58 percent were associated with the posts located in the area between these zones (Abt and others 1996).

The TC data further indicated the significant influence of soil texture on willow performance. Net carbon fixation substantially decreased for posts grown at middle to high elevations in coarse-grained sandy soils (table 3). Soil texture also had a substantial impact on height growth and survival as they were both significantly greater in coarse sandy soils than in fine silt and clay soils (fig. 3). Regression analysis indicated a significant negative correlation ($r = -0.68$ $p = 0.0438$) between the percentage of fine particles (silt + clay at 15- and 30-cm depths) and survival. Height growth was also significantly correlated ($r = -0.69$ $p = 0.0412$) to the average percentage of fine particles (at 15- and 30-cm depths) showing low growth at soils that contained a high percentage of fine particles (silt + clay).

CONCLUSIONS

At both restoration sites, soil texture and moisture regime were important factors affecting survival, physiological functions, and growth of willow posts. For instance, much of the variations in survival and growth were due to soil texture represented by the percentage of fine particles (silt + clay) in the upper 30 cm of the soil. High survival and growth in willow posts required ample soil moisture (but nonwaterlogging conditions) and adequate drainage in the upper soil layer, approximately the top 45 to 60 cm of the soil. In addition, under favorable soil-moisture conditions, posts perform better in soils characterized by coarse-textured rather than fine-grained soils. From a restoration perspective, use of willow posts for riverbank restoration projects is intended to help stabilize the banks while improving habitat quality. Thus, the use of willow posts remains a low-cost restoration strategy; however, a proposed planting site must be evaluated prior to the undertaking of large-scale restoration. Many factors must be considered including soil texture, moisture regime, slope, distance to the creek, and soil oxidation-reduction conditions in order to improve the prospect for high survival and growth of willow posts.

ACKNOWLEDGMENTS

This work was supported in part by a grant from the U.S. Department of Agriculture, National Sedimentation Laboratory, Cooperative Agreement #58–6408–6–029. Graduate students Christopher Brown, Mitch Elcan, and Jason Farmer helped with field data collection. Sara Davis and Michael Eubanks read an earlier version of this paper and made many helpful comments.

REFERENCES

Abt, S.R.; Watson, C.C.; Burgi, J.P. [and others]. 1996. Willow posts construction and evaluation on Harland Creek, Mississippi. In: Proceedings of the sixth Federal interagency sedimentation conference: II14–II23.

Brady, N.C. 1974. The nature and properties of soils. New York: McMillan Publication Co. 639 p.

Donavan, L.A.; McLeod, K.W.; Sherrod, K.C.; Stumpff, N.J. 1988. Response of woody swamp seedlings to flooding and increased water temperatures. I: growth, biomass, and survivorship. American Journal of Botany. 75: 1181–1190.

Hook, D.D. 1984. Waterlogging tolerance of lowland tree species of the South. Southern Journal of Applied Forestry. 8: 136–149.

Kozlowski, T.T. 1982. Water supply and tree growth. II: flooding. Forestry Abstracts. 43: 145–161.

Kozlowski, T.T. 1997. Responses of woody plants to flooding and salinity. Tree Physiology Monographs. 1: 1–29.

Pezeshki, S.R. 1991. Root responses of flood-tolerant and flood-sensitive tree species to soil redox conditions. Trees. 5: 180–186.

Pezeshki, S.R. 1994. Plant responses to flooding. In: Wilkinson, R.E., ed. Plant environment interactions. New York: Marcel Dekker, Inc.: 289–321.

Pezeshki, S.R.; Anderson, P.H.; Shields, F.D. 1998. Effects of soil moisture regimes on growth and survival of black willow (Salix nigra) posts (cuttings). Wetlands. 18: 460–470.

Pezeshki, S.R.; Shields; F.D., Jr.; Anderson, P.H. 1998. The relationships between soil conditions and growth of willow posts on streambanks. In: Abt, S.R.; Young-Pezeshk, J.; Watson, C.C., eds. Proceedings of the international conference on water resources engineering. New York: American Society of Civil Engineers: 447–452.

Roseboom, D. 1993. Breakwater installation and vegetative stabilization in Illinois. In: Proceedings, U.S. Army Corps of Engineers workshop on reservoir shoreline erosion. Publ. W–93–1. Vicksburg, MS: Army Corps of Engineers, Waterways Experiment Station: 166–173.

Statistical Analysis Systems. 1994. Procedures guide. Version. 6. 3rd ed. Cary, NC: SAS Institute Inc. 705 p.

Shields, F.D., Jr.; Cooper, C.M.; Knight, S.S. 1995a. Experiments in stream restoration. Journal of Hydraulic Engineering. 121: 494–502.

Shields, F.D., Jr.; Knight, S.S.; Cooper, C.M. 1995b. Streambank protection and habitat restoration. In: Espey, W.H., Jr.; Combs, P.G., eds. Water resources engineering; Proceedings of the first international conference on water resources engineering. New York: American Society of Civil Engineers: 721–725.

Watson, C.C.; Abt, S.R.; Derrick, D. 1997. Willow post for bank stabilization. Journal of American Water Research Association. 33: 293–3.

DIFFERENCES IN NET PRIMARY PRODUCTION AND BIOGEOCHEMISTRY BETWEEN CONTRASTING FLOODPLAIN FORESTS

Erik B. Schilling and B. Graeme Lockaby[1]

A firm understanding of the driving forces controlling variation among wetland forests continues to elude scientists and land managers—specifically the biogeochemical processes controlling vegetation production. Within contrasting wetland forests, insight into the biogeochemical processes driving productivity levels may be found by examining the degree to which nitrogen and phosphorus are balanced within the wetland vegetation. Lockaby and Conner (1999) suggest that there exists a biogeochemical continuum for wetland forests based on the relationship between N:P ratios. Koerselman and Meuleman (1996) have indicated that N:P ratios in forest vegetation may serve as a measure of the biogeochemical constraints on vegetation net primary production (NPP). Thus, the position of a particular wetland forest on this N:P continuum reflects the integration of its geomorphic position and biogeochemical history and may have a predictive value with regard to levels of NPP (Lockay and Conner 1999). Ultimately, the synchrony of nutrient availability and plant uptake influences the levels of NPP within wetland forests.

In the Southeastern United States, riverine systems are often broadly characterized as either red-water or black-water systems. Red-water rivers have their origins in the Piedmont and are characterized hydrochemically as rich in nutrients and suspended organic matter. Conversely, black-water rivers, having their origins within the Coastal Plain, are poorer in both. Within these systems, the contrasting quality of detrital inputs influences the mineralization and immobilization patterns governing nutrient availability.

The purpose of this study is to examine and characterize the biogeochemical processes influencing nutrient availability and levels of NPP within two wetland forest types. The river systems being used for this study are the Altamaha River, a red-water river system, and the Satilla River, a black-water river system. In 1999, annual litterfall mass for the Altamaha and Satilla Rivers totaled 8.3 and 6.1 t per hectare per year, respectively. Other data to be presented from this study include: aboveground woody biomass, litter decomposition, microbial biomass C, N, and P, retranslocation efficiency, and soil nutrient availability.

Data to be presented for each of the aforementioned foci are the year-one results of an ongoing study.

REFERENCES

Koerselman, W.; Meuleman, A.F.M. 1996. The vegetation N:P ratio: a new tool to detect the nature of nutrient limitation. Journal of Applied Ecology. 33: 1441–1450.

Lockaby, B.G.; Conner, W.H. 1999. N:P balance in wetland forests: productivity across a biogeochemical continuum. The Botanical Review. 65(2): 171–185.

[1] Schilling, Research Associate, and Lockaby, Professor, Auburn University, School of Forestry and Wildlife Sciences, Auburn, AL 36849-5418.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

USE OF A CONSTRUCTED WETLAND TO REDUCE NONPOINT-SOURCE PESTICIDE CONTAMINATION OF THE LOURENS RIVER, SOUTH AFRICA

Ralf Schulz[1]

Abstract—The Lourens River, Western Cape, South Africa, and its tributaries situated in an intensively cultivated orchard area receive pesticide contamination during rainfall-induced runoff and during spraydrift. A 0.44-ha constructed wetland, built in 1991 in one of the tributaries (summer flow 0.03 m^3 per second), was studied in order to assess its effectiveness in reducing nonpoint-source agricultural pesticide contamination. Even high levels of particle-associated azinphos-methyl (43 µg per kilogram), chlorpyrifos (31 µg per kilogram) and prothiofos (6 µg per kilogram) introduced via runoff were not detectable in the outlet suspended-particle samples. Recovery of water-diluted azinphos-methyl contamination at inlet concentrations of about 0.55 µg per liter during spraydrift indicated much less retention, approximately 51 percent. Comparison of *in situ* bioassays of bloodworms (*Chironomus* spec.) above and below the wetland revealed a reduction of contamination in terms of toxicity from 41 to 2.5 percent. TSS, ortho-phosphate, and nitrate were retained in the wetland with trapping efficiencies of 78, 75, and 84 percent, respectively.

INTRODUCTION

Agriculture is a source of sediment, nutrient, and pesticide input into surface waters (Cooper 1993). Rainfall-related runoff and spraydrift are two major sources for the agricultural nonpoint-source pollution of surface waters (Groenendijk and others 1994). The fact that nonpoint-source pollutants are recognized as the single greatest threat to surface waters (Loague and others 1998) has heightened concerns about sustainable agriculture and has highlighted the need for measures to minimize input and risk (Mainstone and Schofield 1996).

Constructed wetlands have been suggested and used as a potential risk-reduction strategy for nonpoint-source pollution (Hammer 1992, Mitsch 1992, Van der Valk and Jolly 1993). Whereas the fate and retention of nutrients and sediments in wetlands is understood quite well (Hammer 1992, Van der Valk and Jolly 1993), the same cannot be said of agrochemicals (Baker 1993). Only very few studies refer to the potential of wetlands for removal of herbicides and some other organic chemicals (Kadlec and Hey 1994, Lewis and others 1999, Wolverton 1987). Since wetlands have a high ability to retain and process material, it seems reasonable that constructed wetlands as buffer strips between agricultural areas and receiving surface waters could mitigate the impact of pesticides in this runoff (Rodgers and Dunn 1993).

During the last decades, a shift to lower water quality in Western Cape rivers has been observed. This shift has also occurred in the middle and lower reaches of the Lourens River and is attributed to intensified agriculture, sediment input, and loss of indigenous vegetation (Tharme and others 1997). No information is available on the extent to which toxic substances are responsible for the degradation of the Lourens River. In order to minimize the input of sediment into the Lourens River, a small flow-through constructed wetland was built in 1991 into one of the tributaries. Constructed wetlands have been used so far in South Africa mainly for wastewater treatment (Wood 1990).

The aim of the present study is to assess the effectiveness of a small constructed wetland for control of nonpoint-source pesticide input into the Lourens River. Retention is assessed for particle-associated pesticide input via runoff and for water-diluted pesticide input following drift during spray application. Sediment- and nutrient-trapping efficiencies are evaluated, and the effects of the wetland in terms of toxicity alteration are addressed using an *in situ* exposure bioassay.

MATERIALS AND METHODS

Study Region and Climate Conditions

The Lourens River rises at an altitude of 1080 m in the Hottentots Holland Nature Reserve and flows in a southwesterly direction for about 20 km before discharging into False Bay at Strand (34°06´ S., 18°48´ E.). The river is surrounded by intensive farming activities (orchards and vineyards) in its middle reaches after leaving a naturally vegetated fynbos area and before flowing through the town of Somerset West. The Lourens River has a total catchment area of approximately 92 km^2 and receives a mean annual rainfall of 915 mm.

The orchards mainly consist of pears, plums, and apples (total growing area: 4 km^2). The pesticide application period starts in the study area´s orchards in early August and ends at the end of January (pears) or end of March (apples). Organophosphorous (OP) insecticides like azinphos-methyl and chlorpyrifos are used between October and February quite frequently on pears and plums. Endosulfan is applied mainly to apple orchards.

[1] Supervisory Ecologist, Department of Zoology, Private Bag X1, University of Stellenbosch, Matieland 7602, South Africa. Present address: Zoological Institute, Technical University, Fasanenstrasse 3, D-38092 Braunschweig, Germany, R.Schulz@tu-bs.de

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

The constructed wetland studied in the present investigation is located in one of the tributaries shortly before its entry into the Lourens River. This tributary has an average width and depth of 0.89 m by 0.30 m and a current velocity of about 0.1 m per second. Average discharge in the tributary is 0.03 m^3 per second in January and 0.32 m^3 per second in July.

The study period lasted from December 1998 until June 1999. A heavy rainfall event resulting in edge-of-field runoff during the pesticide application period occured on December 14, 1998. The period between January and the middle of April was characterized by low rainfall (dry period), whereas the following months were again coupled with high rainfall (wet period).

Separate studies revealed that the Lourens River receives contamination from the surrounding farming areas via rainfall-related runoff and via spraydrift during spray application. Runoff produced concentrations of particle-associated chemicals as high as 244 µg per kilogram azinphos-methyl, 69 µg per kilogram chlorpyrifos and 245 µg per kilogram total endosulfan. Spraydrift-related pesticide input elevated the levels of water-diluted pesticide in the Lourens River to about 0.04 µg per liter azinphos-methyl, with peak concentrations at about 1.7 µg per liter in those tributaries receiving spraydrift directly. Effectiveness of pesticide retention in the constructed wetland was studied for both scenarios—runoff and spraydrift.

Description of the Constructed Wetland

The constructed wetland was built in 1991 into one of the southeastern tributaries draining the surrounding farmland to prevent nonpoint-source pollution with suspended particles from entering the Lourens River. The catchment area of the wetland comprises about 15 ha of orchard, 10 ha of pasture land, and 18 ha of forest. The tributary flows longitudinally through the wetland, which has a length of 134 m and a maximum width of 36 m giving a total area of 0.44 ha. The actual water depth is between 0.3 and 1 m. The first 30 m of the wetland are free of vegetation, and the remaining area is covered mainly with *Typha capensis* Rohrb. (60 percent coverage), *Juncus kraussii* Hochst (10 percent coverage), and *Cyperus dives* Delile (5 percent coverage).

Sampling Program and Analysis

Two sampling stations were used: inlet sampling was done in the tributary about 5 m before its entry into the wetland; outlet sampling took place in the tributary about 7 m below the wetland. Standard sampling procedure included measurement of discharge, suspended particles, and nutrients as well as taking of water and suspended sediments for pesticide analysis. Discharge was calculated on the basis of standard formulas (Maniak 1992) using velocity measurements along cross-section profiles. Turbidity was measured with a turbidity meter (Dr. Lange) allowing measurements between 1 and 200 NTU. To calibrate the turbidity measurements as described by Gippel (1995), certain samples were filtered through preweighed Whatman GF/F (0.45 mm pore-size) glass microfibre filters and dried at 60 °C for 48 hours. The filter paper was then reweighed to determine total suspended solids (TSS). Ortho-phosphate and nitrate were measured with photometric test kits from Merck, Ingelheim. Discharge, TSS, and nutrients were measured during periods with high-flow conditions between December 1998 and June 1999 (n = 17 measurements).

Suspended sediments were taken to describe particle-associated pesticide contamination following runoff events. Time-integrating samples were obtained from continuously operating suspended particle samplers (Liess and others 1996) installed in the stream bottom. Suspended sediments were taken out about every 2 weeks between December 14, 1998, (10:00 h; before rainfall event) and May 17, 1999, and were analyzed for pesticides.

Samples of water for pesticide analysis were collected at the inlet and outlet site in 3-L glass jars during spray application on January 11, 28, and 29, 1999. A previous experiment with the introduction of a tracer (Rhodamin B) 200 m above the wetland (spraydrift deposition site) was undertaken on January 7 during the same hydrological conditions (discharge: 0.026 m^3 per second) to determine the starting point and duration of sampling intervals at the inlet and outlet station of the wetland used for the pesticide sampling. As a result, a 1-hour composite sampling starting 15 minutes after introduction of spraydrift (200 m above the wetland) was done at the inlet station, whereas 5-hour composite sampling was performed at the outlet station starting 4.5 hours after introduction of spraydrift. This sampling design has been found to enable an integrated sample of more than 85 percent of the pesticide load to be obtained at both stations. Water samples (500 to 900 ml) were solid-phase extracted (SPE) within 10 hours after sampling using C18 columns (Chromabond) that had been previously prepared with 6 ml methanol and then 6 ml water. The columns were air-dried for 30 minutes and kept at –18 °C until analyzed.

Analyses were performed at the Forensic Chemistry Laboratory of the Department of National Health, Cape Town. Pesticide residues were extracted from suspended sediment samples using methanol. Extracts were passed through a C18 column (Chromabond) and eluted with 2 ml hexane and then 2 ml dichloromethane. Measurements were done by means of gas chromatographs (HP 5890) with electron-capture, nitrogen-phosphorus, and flame-photometric detectors. Water samples were eluted from SPE columns and measured as well in a GC. The following detection limits were obtained for water and suspended sediments: 0.01 µg/l and 0.1 µg per kilogram dw.

Exposure Bioassays

Bloodworms (*Chironomus* spec.) were used as a test organism. Animals were obtained from a clean water pond at Somerset West water treatment plants. The organisms were collected 1 hour before the exposure started. At each site, four plastic beakers (9 cm in diameter by 13 cm in height) containing 20 4^{th}-instar larvae were installed in the stream. The front and rear walls of the beakers were made of netting (0.5 mm mesh) to allow water to flow through (current: 0.11±0.02 m per second). In each box, 5 g of mud provided material for constructing the larval tubes. Mortality was measured after a 24-hour exposure period. Two trials were performed, one during a day without any spray application from December 12, 10:00 hour, to December 13, 10:00 hour, and one during azinphos-methyl application resulting in spraydrift into the tributary (December 28, 10:00 hour, to December 29, 10:00 hour).

RESULTS

Sediments and Nutrients

Results obtained for inlet-outlet-measurements of suspended sediments and nutrients are summarized for high-discharge conditions in table 1. Trapping efficiency was calculated by dividing the concentration difference (inlet minus outlet) by the inlet concentration and expressing it as a percentage. TSS, ortho-phosphate, and nitrate levels were reduced by the constructed wetland by 78, 75, and 84 percent. Despite the high inlet concentrations, absolute values for all three parameters at the outlet station were quite low, indicating the effectiveness of the wetland.

Particle-Associated Pesticides

The concentrations of azinphos-methyl, chlorpyrifos, and prothiofos in suspended particles sampled at the inlet station during periods with rainfall-induced runoff events are summarized in table 2. Azinphos-methyl concentration in the inlet sample was 43.3 µg per kilogram during an interval with a high-rainfall event in December. During the same interval, prothiofos was present in the inlet sample at a level of 6 µg per kilogram. Chlorpyrifos occurred at a relatively low concentration of 0.6 µg per kilogram following a small rainfall event in February but at elevated concentrations up to 31.4 µg per kilogram during April and May rainfall events. None of the three OP insecticides were detectable in the outlet samples at any time, indicating a 100-percent retention of particle-associated pesticides that were introduced into the tributary via runoff.

Water-Diluted Pesticides

The effectiveness of retention of water-diluted pesticide was estimated during spraydrift events following application of azinphos-methyl (table 3). Inlet concentrations varied during the three trials between 0.36 and 0.69 µg per liter in the 1-hour composite samples. Five-hour composite samples taken at the outlet contained levels between 0.02 and 0.03 µg per liter. Average concentrations detected in the composite samples and discharge (0.026 m^3 per second) were used to calculate chemical loads for both stations. Retention was calculated by dividing the load difference (inlet minus outlet) by the inlet load and expressing it as a percentage. Retention ranged between 44 and 60 percent and was, on average, 51 percent.

Exposure Bioassay

Mortality of bloodworms exposed for 24 hours *in situ* at the inlet and outlet of the constructed wetland is shown in table 4. During periods without spraying activity, the average mortality was = 1.25 percent. In the 24-hour period during spraydrift trial 1, the mortality at the inlet station was elevated considerably to 41.3 percent, whereas the mortality at the outlet station stayed low at 2.5 percent.

DISCUSSION

Sediment and Nutrient Trapping Efficiency

As expected, considerable reductions of sediment and nutrient loads were observed in the wetland. Retention of TSS was 78 percent during the wet period, when, on average, 105 mg per liter TSS entered the wetland. Cooper and Knight (1991) reported a 78-percent trapping efficiency of a 1.1-ha detention reservoir during storms with inflow TSS concentrations of 800 mg per liter or greater. A 2.1-ha

Table 1—Mean (n = 17 measurements, ± SE) total suspended solids and nutrient concentrations of water at the inlet and outlet of the constructed wetland as well as trapping efficiency during high discharge conditions (0.175 ± 0.03 m^3 per second) typical of rainfall conditions between 2 and 35 mm per day

	Inlet	Outlet	Trapping efficiency
	- - - - - - - - *mg/L*	- - - - - - -	*Percent*
TSS	105 ±14	23.0 ±1.9	78.1
Ortho-phosphate	.88 ± .15	.22 ± .02	75
Nitrate	1.84 ± .37	.3 ± .03	83.7

TSS = total suspended solids.

Table 2—Rainfall and concentrations of different OP insecticides associated with suspended particles that were continously sampled above the constructed wetland during intervals with rainfall-induced runoff events (none of the suspended particles sampled during the same intervals at the outlet station contained any detectable pesticide levels)

Rainfall characteristics		Suspended particles-sampling interval	Contamination of inlet station samples	
Date	Intensity		Substance	Concentration
	mm/d			*µg/kg*
12/14/98	18.4	12/14–12/29/98	Azinphos-methyl	43.3
02/26/99	2	02/15–03/01/99	Chlorpyrifos	.6
04/19/99	18.8	03/31–04/19/99	Chlorpyrifos	31.4
04/21/99	29.8	04/19–05/17/99	Chlorpyrifos	23.9
12/14/98	18.4	12/14–12/29/98	Prothiofos	6

Table 3—Concentration of azinphos-methyl in composite samples taken during spraydrift at the inlet and outlet of the constructed wetland. Inlet samples represent 1-hour composite samples, whereas outlet samples represent 5-hour composite samples (retention was calculated based on calculation of chemical load)

Event	Inlet	Outlet	Retention
	- - - - - - - *µg/L* - - - - - - -		*Percent*
1	0.69	0.07	49
2	.36	.04	44
3	.62	.05	60

Table 4—Mortality (± SE; n = 4) of *Chironomus* spec. exposed for 24 hours at the inlet and outlet of the constructed wetland during a period without any spraying activity and during an azinphos-methyl spraydrift event

	Inlet	Outlet
Without spraying (%)	1.25 ± 1.25	0
During spraydrift (%)	41.3 ± 2.4	2.5 ± 1.4

constructed wetland receiving constant volumes of nonpoint-source agricultural and urban pollution reduced suspended solids with inlet concentrations of 50 mg per liter by 78 percent and those of 123 mg per liter by 95 percent (Kadlec and Hey 1994). Lower trapping efficiency observed in the present study may be due to the small wetland size and the fact that the wetland is a flow-through wetland without any water storage capacity and with water inflow rates that vary over time between 0.016- and 0.522-m^3 per second.

Trapping efficiencies during wet conditions for ortho-phosphate (0.9 mg per liter) and for nitrate (1.8 mg per liter) were 75 and 84 percent. Similar results (72 percent and 82 percent) have been found in the 1.1-ha Morris Pond, MS (Cooper and Knight 1990) and in Des Plaines Wetlands, IL, where total P reduction averaged 74 percent and nitrate reduction 78 percent (Kadlec and Hey 1994).

The TSS and nutrient-trapping efficiency is still quite high, although the wetland has never been dug out during the 7-year period since it was built. This indicates the sustainability of constructed wetland use as a strategy for the long-term improvement of water quality.

Retention of Particle-Associated Pesticides: Runoff Scenario

The constructed wetland acted as a very effective sink for sediment-associated pesticides during the study period. Azinphos-methyl (43 µg per kilogram), chlorpyrifos (31 µg per kilogram), and prothiofos (6 µg per kilogram) detected in the suspended particles of the inlet water were never present in the outlet suspended particles. Although it was thought that wetlands have a high potential to reduce specifically sorbed chemical load (Rodgers and Dunn 1993), this fact had not previously been demonstrated for insecticide fate in small constructed wetlands. It is likely that one major reason for the effectiveness in retaining particle-associated chemicals is the sedimentation of suspended particles in the wetland areas with reduced-flow conditions. As discussed above, TSS were removed at a rate of 78 percent during wet weather conditions. However, the 100-percent retention detected for all of the three organophosphate pesticides cannot be explained just by sedimentation processes. Dilution, decomposition, and microbial metabolism may account for the remaining chemical retention (Rodgers and others 1999). The effectiveness of pesticide retention in wetlands may differ with season due to changes in water temperature and flow as well as wetland abiotic and biotic conditions (Spongberg and Martin-Hayden 1997). There is still a need for further studies to demonstrate the long-term fate of insecticide runoff in constructed wetlands.

Retention of Water-Diluted Pesticides: Spraydrift Scenario

Levels of azinphos-methyl during spraydrift were between 0.36 and 0.69 µg per liter in the inflow water, whereas maximal concentrations in the outflow were at about 0.07 µg per liter. It follows that the retention for this pesticide entering the wetland in the water phase was as high as 51 percent. There are no other studies dealing with the retention of water-diluted insecticide input in constructed wetlands. However, the implementation of retention ponds in agricultural watersheds was examined as one strategy to reduce the amount and toxicity of runoff-related insecticide pollution discharging into estuaries (Scott and others 1999). For atrazine, a removal rate between 25 and 95 percent was demonstrated in the Des Plaines Wetland cells (Kadlec and Hey 1994). Reduction of atrazine concentration in water was 11 to 14 percent in 230-m flow-through wetland mesocosms (Detenbeck and others 1996). Processes important for removal of nonpoint-source pesticide runoff in wetlands may include adsorption, decomposition, and microbial metabolism (Rodgers and others 1999). The macrophytes present in the wetland may play an important role in providing an enlarged surface area for sorption as well as microbial activity and due to chemical metabolism within the plants (Wetzel 1993).

Toxicological Evaluation

In situ exposure of bloodworms during spray deposition clearly indicated an increased 24-hour mortality at the inlet station in comparison to an only slightly elevated mortality at the outlet station. It follows that the constructed wetland makes an important contribution in that it reduces the toxicity of azinphos-methyl in the tributary water by 94 percent before it enters the Lourens River. That is, I attribute the observed reduction of toxicity to the wetland and conclude that it would be much lower if the tributary water were to flow over the same distance (230 m) via a conventional channel into the Lourens River. Azinphos-methyl concentrations measured in the inlet water were in the range of acutely toxic concentrations for various crustacean species including *Gammarus fasciatus* Say (Sanders 1972) and *Hyalella azteca* Saussure (Ankley and Collyard 1995). They exceeded the 96-hour LC_{50} of *Chironomus tentans* Fabricius, which is 0.37 µg per liter (Ankley and Collyard 1995). According to prevous experiments, the 24-hour LC_{50} for the chironomid species used in the present study is approx. 7.3 µg per liter.

On the basis of the 24-hour LC_{50} and the azinphos-methyl concentration measured in the field, it is not possible to explain the observed mortality of exposed bloodworms. Similar differences between toxicological reactions in the field and those predicted from laboratory toxicity tests at the same contaminant levels have been described by other authors as well. Matthiesen and others (1995) observed 100 percent mortality of caged *Gammarus pulex* L. following exposure to a peak concentration of 27 µg per liter carbofuran, which exceeded the 24 hour LC_{50} of 21 µg per liter only for a period of 3 to 5 hours. Baughman and others (1989) expected differences in measured and real exposure concentrations to be a reason for higher mortalities in *in situ* bioassays than predicted from laboratory data.

CONCLUSION

It was shown that the investigated wetland is capable of reducing the load and toxicity of nonpoint agricultural pesticide pollution. Keeping in mind the relative importance of runoff-related pesticide contamination associated with suspended particles, constructed wetlands can be regarded as a very effective tool for the reduction of nonpoint agricultural pesticide contamination in small streams.

ACKNOWLEDGMENTS

This study is part of a joint research project between the University of Stellenbosch, South Africa (Professor A.J. Reinecke) and the Technical University of Braunschweig, Germany, and is funded by the National Research Foundation, Pretoria. The author thanks Corlie Hugo and Victor Krause for cooperating in the field technical operations, Sue Peall for performing the pesticide analysis, and Beate Helling and Ann Thorson for helpful comments on the manuscript.

REFERENCES

Ankley, G.T.; Collyard, S.A. 1995. Influence of piperonyl butoxide on the toxicity of organophosphate insecticides to three species of freshwater benthic invertebrates. Comparative Biochemistry and Physiology. 110: 149–155.

Baker, L.A. 1993. Introduction to nonpoint source pollution and wetland mitigation. In: Olson, R.K. comps. Created and natural wetlands for controlling nonpoint source pollution. Boca Raton, FL: CRC Press: 12–45.

Baughman, D.S.; Moore, D.W.; Scott, G.I. 1989. A comparison and evaluation of field and laboratory toxicity tests with fenvalerate on an estuarine crustacean. Environmental Toxicology and Chemistry. 8: 417–429.

Cooper, C.M. 1993. Biological effects of agriculturally derived surface-water pollutants on aquatic systems—a review. Journal of Environmental Quality. 22: 402–408.

Cooper, C.M.; Knight, S.S. 1990. Nutrient trapping efficiency of a small detention reservoir. Agricultural Water Management. 18: 149–158.

Cooper, C.M.; Knight, S.S. 1991. Nutrient and sediment trapping efficiency in small farm ponds. Proceedings for the Environmentally Sound Agriculture Conference: 673–680.

Detenbeck, N.E.; Hernanutz, R.; Allen, K.; Swift, M.C. 1996. Fate and effects of the herbicide atrazine in flow-through wetland mesocosms. Environmental Toxicology and Chemistry. 15: 937–946.

Gippel, C.J. 1995. Potential of turbidity monitoring for measuring the transport of suspended solids in streams. Hydrological Processes. 9: 83–97.

Groenendijk, P.; van der Kolk, J.W.H.; Travis, K.Z. 1994. Prediction of exposure concentration in surface waters. In: Hill, I.R.; Heimbach, F.; Leeuwangh, P.; Matthiessen, P., eds. Freshwater field tests for hazard assessment of chemicals. Boca Raton, FL: Lewis Publishers: 105–125.

Hammer, D.A. 1992. Designing constructed wetlands systems to treat agricultural nonpoint source pollution. Journal of Ecological Engineering. 1: 49–81.

Kadlec, R.H.; Hey, D.L. 1994. Constucted wetlands for river water quality improvement. Water Science Technology. 29: 159–168.

Lewis, M.A.; Mayer, F.L.; Powell, R.L. [and others], eds. 1999. Ecotoxicology and risk assessment for wetlands. Pensacola: Society of Environmental Toxicology and Chemistry (SETAC). 392 p.

Liess, M.; Schulz, R.; Neumann, M. 1996. A method for monitoring pesticides bound to suspended particles in small streams. Chemosphere. 32: 1963–1969.

Loague, K.; Corwin, D.L.; Ellsworth, T.R. 1998. The challenge of predicting nonpoint source pollution. Environmental Science & Technology. 32: 130–133.

Mainstone, C.P.; Schofield, K. 1996. Agricultural management for nonpoint pollution control, with particular refernce to the UK. European Water Pollution Control. 6: 21–29.

Maniak, U. 1992. Hydrologie und Wasserwirtschaft. Berlin: Springer Verlag. 568 p.

Matthiesen, P.; Sheahan, D.; Harrison, R. [and others]. 1995. Use of a *Gammarus pulex* bioassay to measure the effects of transient carbofuran runoff from farmland. Ecotoxicology and Environmental Safety. 30: 111–119.

Mitsch, W.J. 1992. Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecological Engineering. 1: 27–47.

Rodgers, J.H., Jr.; Dickson, G.W.; Dillon, T. [and others]. 1999. Workgroup V synopsis: constructed wetlands as a risk mitigation alternative. In: Lewis, M.A.; Mayer, F.L.; Powell, R.L. [and others], eds. Exotoxicology and risk assessment for wetlands. Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC): 315–342.

Rodgers, J.H., Jr.; Dunn, A. 1993. Developing design guidelines for constructed wetlands to remove pesticides from agricultural runoff. In: Olson, R.K., comps. Created and natural wetlands for controlling nonpoint source pollution. Boca Raton, FL: CRC Press: 142–164.

Sanders, H.O. 1972. Toxicity of some insecticides to four species of malacostracan crustaceans. Tech. Pap. 66. Washington, DC: U.S. Department of the Interior, Fishery and Wildlife Service. 19 p.

Scott, G.I.; Fulton, M.H.; Moore, D.W. [and others]. 1999. Assessment of risk reduction strategies for the management of agricultural nonpoint source pesticide runoff in estuarine ecosystem. Toxicology and Industrial Health. 15: 200–213.

Spongberg, A.L.; Martin-Hayden, J.M. 1997. Pesticide stratification in an engineered wetlands delta. Environmental Science and Technology. 31: 3161–3165.

Tharme, R.; Ratcliffe, G.; Day, E. 1997. An assessment of the present ecological condition of the Lourens River, Western Cape, with particular reference to proposals for stormwater management. Cape Town, South Africa: Freshwater Research Unit, University of Capetown. 289 p.

Van der Valk, A.G.; Jolly, R.W. 1993. Recommendation for research to develop guidelines for the use of wetlands to control rural nonpoint source pollution. In: Olson, R.K., comps. Created and natural wetlands for controlling nonpoint source pollution. Boca Raton, FL: CRC Press: 211–234.

Wetzel, R.G. 1993. Constructed wetlands: scientific foundations are critical. In: Moshiri, G.A., comp. Constructed wetlands for water quality improvement. Boca Raton, FL: CRC Press: 3–7.

Wolverton, B.C. 1987. Artificial marshes. In: Reedy, K.R.; Smith, W.H., comps. Aquatic plants for water treatment and resource recovery. Orlando, FL: Magnolia Publishing.

Wood, A. 1990. The application of artificial wetlands in South Africa. In: Cooper, P.F.; Findlater, B.C., comps. Constructed wetlands in water pollution control. Oxford: Pergamon Press: 235–244.

FOREST RESTORATION IN A GLOBAL CONTEXT

John A. Stanturf[1]

Abstract—Forest restoration on land cleared for agriculture is occurring around the world. Often land was abandoned because of infertility, frequent flooding, or other site limitations. In some countries, market forces or changing trade policies drive conversion of cleared land to plantations of exotic or native tree species. The objective of this paper is to introduce the special session on restoration of bottomland hardwoods by placing efforts in the Lower Mississippi Alluvial Valley into a global context. The challenges of forest restoration are surprisingly similar: overcoming site degradation, prescribing appropriate species, and applying cost-effective establishment methods. While plantation forestry remains the most effective approach to large-scale restoration, the trend is toward plantations that are more complex. This trend is characterized by more intimate association with other land uses, more diverse goals for species composition and vegetation structure in restoration planting, and more direct involvement by landowners in both the conception and implementation of restoration schemes. Benefits of restoration planting include reduced soil erosion; improved water quality; increased wildlife habitat; and increased supply of wood for fuel, lumber, and fiber. Increasingly, objectives of restoration planting include carbon sequestration.

INTRODUCTION

Forest cover has declined globally, from an estimated 6.1 billion ha of original forest extent to the present 3.45 billion ha (Krishnaswamy and Hanson 1999). The greatest loss in cover has occurred in Asia-Pacific, Africa, and Europe (all more than 60-percent loss of forest cover). Losses in North America are relatively low (25 percent), while Latin America (Central and South) has lost over 30 percent of the original forest cover. Nevertheless, the area in forest plantations is only 135 million ha, although increasing (Kanowski 1997).

Forest restoration on land cleared for agriculture, often termed afforestation, is widespread. Land may have been abandoned because of infertility, frequent flooding, or other site limitations. Today, as in the past, forest cover in populated areas is in dynamic equilibrium with land cleared for agriculture and taken for urban uses. Market forces, changing trade policies, or agricultural reforms drive conversion of cleared land back to trees. In Europe, for example, afforestation is a policy instrument to retire land from agriculture because of attempts by the European Union to reduce agricultural subsidies (Madsen and others 2001).

The objectives of this paper are to place the afforestation efforts in the Lower Mississippi Alluvial Valley (LMAV) into a global context by drawing parallels to work in other countries. The challenges of forest restoration in different countries are surprisingly similar (Kanowski 1997): overcoming site degradation/limitations, prescribing appropriate species, and applying cost-effective establishment methods. Plantation forestry is the most effective approach to restoration of large areas, and plantations that are more complex are recent trends.

TERMINOLOGY

What constitutes restoration can be confusing as the term is used indiscriminately. Changes in land cover and land use influence the dynamic relationship between degrading and restoring processes. If we consider the undisturbed, idealized natural mature forest as a starting point (fig. 1), then conversions to other land uses such as agriculture or pasture are through deforestation. Relatively frequent but moderate disturbance, such as plowing, herbicides, and grazing, maintain the nonforest cover.

Similarly, a change in both land cover and land use occurs when forests are converted to urban uses, flooded by dams, or removed along with topsoil/overburden in mining and extractive activities. Such drastic conversion usually involves severe disturbance and is maintained more or less permanently by structures more than by cultural activities (fig. 1).

Even-aged harvesting of mature forest in a sustainable manner is a change of land cover but not land use. A new, young forest will result from natural regeneration or by reforestation, i.e., planting trees in a cutover. Unsustainable harvesting such as high-grading degrades stand structure or diversity. Pollutant loading, outbreaks of insects or diseases (especially exotics), invasion by aggressive exotic plants, or disasters such as hurricanes or wildfires can also degrade forests. In all these instances, intervention to restore species diversity or stand structure can be termed rehabilitation (fig. 1).

Given sufficient time and the cessation of disturbances, agricultural land, as well as urbanized land, will revert to forest, if that is the potential natural vegetation as set by climate. Abandonment and reversion to forests, albeit

[1] Project Leader, Disturbance and Management of Southern Ecosystems, USDA Forest Service, 320 Green Street, Athens, GA 30602.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

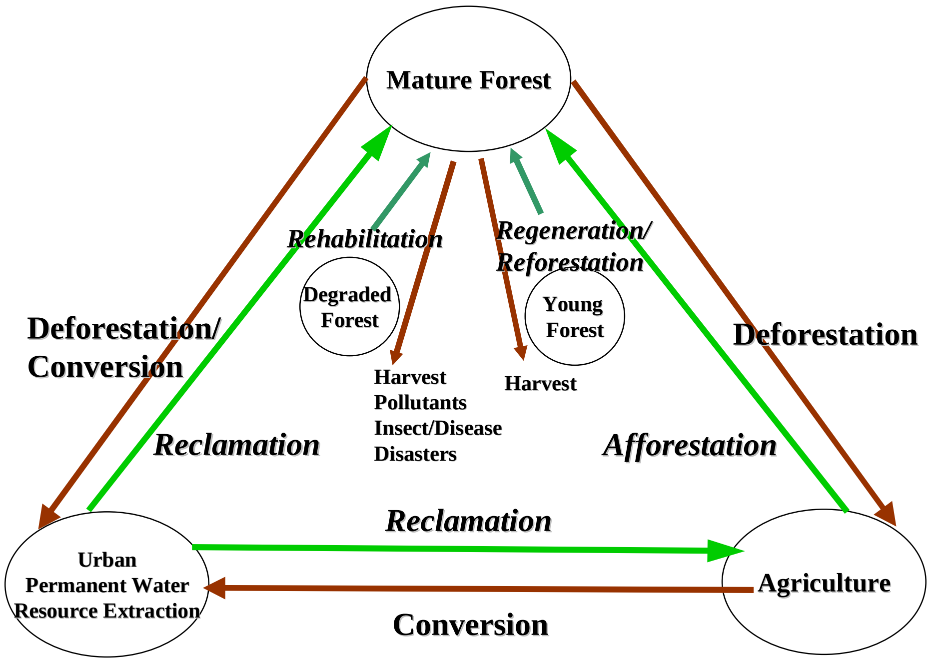


Figure 1—The terminology of forest restoration is best viewed in terms of land use as well as land cover change.

secondary or even degraded forest types, will occur gradually on a time scale of a few decades to centuries. Human intervention, however, can accelerate the reversion process. Afforestation of agricultural land may consist simply of planting trees, although techniques that are more intensive are available. Reclamation of urbanized land usually requires more extensive modification, which may include stabilization of spoil banks or removal of water control structures, followed by tree planting. Because severe degradation may limit the possibilities for reclamation, this process is sometimes called replacement (Bradshaw 1997).

Generally, restoration connotes some transition from a degraded state to a former natural condition. All the restorative activities described (reforestation, rehabilitation, afforestation, and reclamation) have been called forest restoration, although to the purist none would qualify as true restoration (Bradshaw 1997, Harrington 1999). In the narrowest interpretation, restoration requires a return to an ideal ecosystem with the same previous species diversity, composition, and structure (Bradshaw 1997) and, as such, is probably impossible to attain (Cairns 1986). Pragmatically, forest restoration can describe situations where forest land use and land cover are restored through the approaches of afforestation or reclamation.

COMMON CHALLENGES

Three steps are key to planning forest restoration: (1) understanding current conditions (as a starting point), (2) clarifying objectives and identifying an appropriate goal, and (3) defining feasible actions that will move toward the desired condition. In most cases, the silviculturist may choose among multiple silvicultural pathways toward the desired future condition. The choice of intervention affects the financial cost, the nature of intermediate conditions, and the time it takes to achieve the desired condition. It is imperative to make silvicultural decisions with clear objectives and with an understanding of a particular intervention's probable success.

Many examples of forest restoration can be classified as afforestation, reclamation, or rehabilitation (table 1). Three papers in this proceedings discuss in detail: bottomland hardwoods in the Southern United States (Gardiner and others 2001); broadleaves in the Nordic countries (Madsen and others 2001); and mangroves in Southeast Asia (Burbridge and Hallin 2001). The challenges of forest restoration are surprisingly similar: overcoming site degradation, prescribing appropriate species, and applying cost-effective establishment methods.

Overcoming Site Degradation

Previous land use may have degraded site conditions, especially for afforestation and reclamation projects. The specific conditions may vary from soil erosion or salinization, in which physical structure and chemistry of the soil are inhospitable to native trees, to lowered fertility from continuous cropping, e.g., Whalley 1988. In some cases, land becomes available for restoration because it is too infertile for agriculture. Existing forest stands in need of rehabilitation may have become degraded by past mismanagement, such as high-grading (removing all the biggest and best trees, leaving undesirable species or trees of poor form or low vigor), fire suppression, or holding water late into the growing season in green tree reservoirs. In other cases, hydroperiod alterations, hurricanes, severe windstorms, floods, or insect outbreaks may have degraded the stands, but not usually the site.

Table 1—Examples of forest restoration efforts in various parts of the world

Type of restoration	Region	Former condition	Restored condition
Afforestation	Lower Mississippi Alluvial Valley, USA	Agriculture	Bottomland hardwoods
Afforestation	Nordic countries	Agriculture	Hardwoods, sometimes Norway spruce
Afforestation	Tropical countries	Agriculture	Exotic and native hardwoods
Afforestation	Venezuela	Cerrado	Caribbean pine
Afforestation	Iceland	Eroded grazing land	Birch, lupine/birch
Reclamation	Everywhere	Mined land	Various
Reclamation	Asia	Shrimp ponds	Mangrove
Reclamation	Ireland	Mined peatland	Sitka spruce, various hardwoods
Reclamation	India	Saline and sodic soils	Eucalyptus spp., Acacia spp., other native spp.
Rehabilitation	Southeastern United States	Loblolly pine plantations	Longleaf pine woodlands
Rehabilitation	Interior Highlands, Southeastern United States	Shortleaf pine/hardwood forests	Shortleaf pine/bluestem grass woodlands
Rehabilitation	Central Europe	Norway spruce plantations	Oak or beech woodlands
Rehabilitation	England and Scotland	Spruce or pine plantations	Mixed woodlands

Site potential and extent of degradation set limits on what can be achieved by intervention. Site potential refers to the combination of relatively unchanging physical factors that affect species composition and stand vigor. Soil and landform characteristics determine moisture availability, aeration, and fertility. In wetland forests, hydroperiod characteristics are important (flood frequency, seasonality, duration, and depth). Site potential is not immutable, however, and can be influenced positively or negatively by changes in land cover or land use.

The cause and possible continuing problem of site or stand degradation should be identified. For example, alteration of a site by changed hydroperiod poses several questions. Can the hydroperiod be restored or the effects of alteration somehow mitigated? Should the restoration effort target a vegetation assemblage adapted to present hydroperiod and site conditions? Hydroperiod alterations caused by flood control projects, dams, or highway construction tend to be irreversible, at least in the short term. Flooding caused by beaver dams, however, can be reduced by removing the dam, but continued management of beaver population levels will be required to avoid recurring problems. The guiding principle should be to rehabilitate or restore in accordance with existing conditions, unless alteration is feasible, affordable, and within the control of the silviculturist.

Appropriate Species

Most restoration efforts favor native species, although some situations may use exotic species. In the Tropics, population pressures and land scarcity may require that restoration include species that provide early economic returns (Grainger 1988, Islam and others 1999, Parrotta 1992), and native forest species may be unsuited for degraded sites. Fast-growing exotic species alter site conditions enough for native species to thrive (Ohta 1990, Parrotta and others 1997). Nevertheless, lack of knowledge may lead to neglect of native species (Butterfield and Fisher 1994, Fisher 1995, Knowles and Parrotta 1995).

The perception of native species may be contentious. Some fast-growing species may be native but considered undesirable. Public citizens or government agencies may be averse to planting more pine [especially loblolly (*Pinus taeda* L.)] rather than broadleaves in the Southern United States, or eastern cottonwood (*Populus deltoids* Bartr. ex Marsh.) in the LMAV. Species under consideration may be native to the area but not to the site. In the LMAV, for example, extensive hydrologic changes have allowed planting of more oak trees (*Quercus* spp.) than were probably in the forests prior to European settlement (King and Keeland 1999). Even documenting the composition of the predisturbance forested landscape can be difficult and contentious (Hamel and Buckner 1998; Stanturf and others, in press).

An open question is to what extent should the manager today consider the possible effects of global climate change in choosing appropriate species to plant. The different global circulation models used by policymakers yield very different results for the Southern United States at the scale of the forest stand. Nevertheless, managers contemplating long rotations may want to hedge their bets by planting species adapted to drier conditions on upland sites. In bottomlands, the situation is more complicated. Rising sea level will not only inundate coastal forests but also cause a rise in the base level of rivers in the region, changing the hydroperiod of many sites. In some bottomlands, species tolerant of drier conditions would be warranted, but in those likely to be inundated later in the growing season, it is better to plant species that are adapted to prolonged soil saturation.

Effective Establishment Methods

Choosing species appropriate to the site and management objectives of the landowner is an important first step. Choice of stock type and proper handling are important, as well as adequate preparation of the site and postplanting practices, such as weed control. High survival is needed to insure adequate stocking (seedling density) and to minimize costs, especially where seedling costs are high, as in Scandinavia (Madsen and others 2001). Survival rates in industrial plantations, commonly 80 to 90 percent, set the benchmark. However, expecting such high survival in many restoration programs may be unreasonable, as the knowledge base may be insufficient due to limited research, practical experience, or available labor.

Vigorous growth of established seedlings is important to reach target stocking levels. Low-vigor seedlings may survive but are at greater risk of mortality from weed competition, mammal herbivory, or from insects and diseases. Vigorous growth will also speed the development of stand structure and canopy closure, important for achieving conservation and wildlife benefits (Stanturf and others, in press). On the other hand, cultural practices raise establishment costs and may not have a lasting effect on vigor (Stanturf and others 1998).

Planting density is an important decision because of the effect it has on meeting landowner objectives and minimizing costs. In order to determine an adequate stocking level given seedling survival, a simple approach is to calculate the initial density needed to achieve a future density. For example, the Wetlands Reserve Program (WRP) is a Federal incentive program in aid of farmers planting hardwoods on low-lying cropland (Stanturf and others 2000). The WRP target at age 3 is 309 stems per hectare, which is low for timber production and probably inadequate for wildlife (Stanturf and others, in press). Nevertheless, the agency will only share the cost for planting 750 seedlings per hectare with the landowner. Therefore, the initial stocking must allow for intensity of site preparation, planting efficiency, and species survival rates. Nuttall oak (*Q. nuttallii* Palmer) is the most commonly planted oak species in the LMAV and has an average operational survival rate of 60 percent for planted seedlings with minimum site preparation. For other oak species, however, survival is typically lower, 30 to 40 percent; and the target will not be met. In addition, inexperienced crews plant most WRP sites; and survival rates are below the operational benchmark, resulting in significant failures (Stanturf and others, in press).

PLANTATION FORESTRY AS A RESTORATION MECHANISM

The first step in restoring a forest is to establish trees, the dominant vegetation. Although this is not full restoration in the sense of Bradshaw (1997), this necessary step is far from a trivial accomplishment (Stanturf and others 1998; Stanturf and others, in press). Nevertheless, many people object to traditional plantations on the grounds of aesthetics or lack of stand and landscape diversity. The correct ecological comparison, however, is between plantations and intensive agriculture rather than between plantations and a mature natural forest (Stanturf and others, in press). All forest alternatives provide at least some vertical structure, increased plant diversity, and some wildlife and environmental benefits. Kanowski (1997) argued for a dichotomy in concepts between the traditional plantations organized for fiber production and more complex plantation forests that seek to maximize social benefits other than wood. Complex plantations that retain the economic and logistic advantages of simple plantations can meet restoration goals.

Advantages of Simple Plantations

Simple plantations are single purpose, usually even-aged monocultures that can produce as much as 10 times greater wood volume than natural forests (Kanowski 1997). Simple plantations, nevertheless, provide multiple benefits when compared to alternatives such as continuous agriculture; if managed well, they satisfy sustainability criteria. Significant advantages of simple plantations are that they easily can be established using proven technology, their management is straightforward, and they benefit from considerable economies of scale. If financial return is the primary objective of a landowner, simple plantations may be preferred and some restoration goals will be attained (Stanturf and others, in press). Nevertheless, complex plantations can be established that provide greater social benefit but at lower rates of return from timber production, possibly as little as 10 percent less (Kanowski 1997), or even at a net financial gain to the landowner, e.g., Stanturf and Portwood 1999).

Characteristics of Complex Plantations

Association with other land uses—Objections to plantations are often cast in terms of aesthetics. The sharp boundary between a plantation and other land uses is objectionable, as is the uniformity of trees planted in rows. In order to integrate the plantation with other land uses, the sharp edges of plantations can be softened by fuzzy or curved boundaries. Where plantations are on small farm holdings, agroforestry systems of intercropping can blend land uses. Forested riparian buffers are established in agricultural fields to protect water quality by filtering sediment, nutrients, and farm chemicals; and they bar easy access by livestock to stream banks. Riparian buffers add diversity to the landscape and serve as wildlife corridors between patches of fragmented forests. In floodplain landscapes such as bottomland hardwoods, areas of permanently saturated or inundated soil (respectively, moist soil units and open water areas) are common and diversify the interior of plantations.

Several options are available to overcome the uniformity of rows. Perhaps the simplest technique is to offset the rows. Uniform spacing between rows and between seedlings within a row is common, resulting in a square pattern. Rows can be offset to produce a parallelogram instead of a square. Alternatively, plantations can be planned with a recreational viewer in mind so that the view from trails and roads is always oblique to the rows, thereby escaping notice. At any rate, once the canopy reaches sufficient height that ground flora and midstory plants can establish, most plantations take on the appearance of natural stands, at least to the casual observer.

Species composition and vegetation structure—A more serious objection to plantations is the lack of diversity, in terms of species composition and vertical structure. Essentially, simple plantations are not as diverse as natural stands, at least for many years. Foresters have devised several methods to establish multiple species stands. For example, planting several blocks of different species in a stand, or even alternate rows of different species is possible and creates some diversity at the stand level. Distribution, however, remains more clumped than would be typical of a natural stand.

Other methods are available, including nurse crops of faster growing native species (Schweitzer and others 1997) or exotics (Ashton and others 1997, Lamb and Tomlinson 1994). In this approach, there is no intention of retaining the nurse crop species throughout the rotation of the slower growing species. (This could also be termed relay intercropping.) While the nurse-crop method has many advantages and in the short term provides species diversity and probably vertical structure, once the nurse crop is removed the residual stand may lack diversity. The challenge is to develop methods for establishing several species in intimate mixtures, such as would occur in a natural stand, but avoiding excessive mortality during the self-thinning or stem exclusion stage of stand development. Such methods must account for the growth patterns of the species, relative shade tolerances, and competitive ability.

Vertical structure is an important feature of forests for wildlife (DeGraaf 1987; Hamel and others, in press; Twedt and Portwood 1997). Early stages of stand development, whether in natural forests or plantations, are characterized by low light in the understory until crowns differentiate. In most restoration forests, little development of the understory and midstory occurs for many years. Annual disturbance while in agriculture removed buried seed and rootstocks of native plants, and low light levels in the young forest preclude understory development from invaders. To accommodate this deficiency, the manager can intervene to plant understory species; at present, little research affords guidance on methods, planting density, or probable success rates. As indicated above, relay intercropping provides vertical structure for a time. Natural dispersal into gaps can also encourage understory development, whether gaps are created by thinning or left during planting (Allen 1997, Otsamo 2000). The critical factor limiting understory development by natural invasion is whether there are seed sources for the understory plants within dispersal range (Chapman and Chapman 1999, Johnson 1988).

BENEFITS OF RESTORATION

The benefits of restoration are usually identified in terms of agency priorities or social benefits; seldom are the diverse objectives of landowners recognized. In most market economies where rights and obligations of ownership rest with private landowners, what is appropriate for public land may not be the most attractive restoration option for private landowners (Stanturf and others, in press). Nevertheless, there can be considerable overlap in the expected benefits to society and the affected landowner. The array of possible landowner objectives can be illustrated with a limited set of management scenarios from the LMAV (table 2). For simplification, three scenarios are presented: (1) short-rotation management for pulpwood or fuelwood; (2) a longer rotation typical of management for saw log production which is suitable for wildlife species that require complex vertical structure, such as certain neotropical migratory songbirds (Hamel and others, in press); and (3) an option termed green vegetation, which is essentially the no-management scenario. In the green vegetation scenario, species composition and stand structure are secondary concerns to removing land from active agriculture. This option meets the objectives of Federal programs, such as the WRP (Stanturf and others, in press). It may also provide habitat conditions for certain wildlife that otherwise would not occur on the landscape (Hamel and others, in press).

Table 2—Financial, recreational, and environmental benefits expected from three afforestation scenarios common in the Lower Mississippi Alluvial Valley, Southern United States

	Expected benefit level					
	Financial		Recreational		Environmental	
Scenario	Short term	Long term	Hunting	Nonconsumptive	Conservation practices	Land retirement
Short rotation (pulpwood, fuelwood)	High	High	High	Medium	Medium	No
Long rotation (timber, wildlife)	Medium	High	High	High	High	Medium
Green vegetation	Low to no	No	Low	Medium	Medium	High

Benefits are comprised of financial, recreational, and environmental outcomes. Because cash flow is important to many landowners, and the adjustment from annual to periodic income is often cited as a barrier to afforestation, financial benefits are considered as both short term and long term. Recreational benefits are hunting (typically for deer, turkey, and waterfowl) and nonconsumptive benefits, such as bird watching or hiking. Environmental benefits are separated into conservation practices, such as those installed to control soil erosion and protect water quality or enhance wildlife habitat, and land retirement, where there is no on-going management activity.

Financial Benefits

Financial returns from active management are substantial relative to the green vegetation scenario. Saw log rotations of high-value oak and green ash (*Fraxinus pennsylvanica* Marsh.) are expected within 60 to 80 years, with the first commercial thinning beginning in 20 to 30 years. Short-term financial returns from growing pulpwood-sized eastern cottonwood in the LMAV are realized within 10 years of afforestation (Stanturf and Portwood 1999). The short-term financial returns are low from plantations of other species. Nevertheless, other species can be combined with cottonwood in the nurse-crop technique to produce income for one or two pulpwood rotations, hence the medium rating. The green vegetation scenario, typified by the WRP plantings, provides no long-term income, as timber management is unlikely given the understocked stands that develop (Stanturf and others, in press). In the short term, there is income from the one-time easement payment made to the landowner (Stanturf and others 2000).

Other income can be realized by some landowners from hunting leases and potentially from carbon sequestration payments. In the Mississippi portion of the LMAV, hunting rights are leased for $7.50 to $12.35 per hectare per year. There is also a potential for considerable income to landowners from credits from carbon sequestration (Barker and others 1996). While there is considerable uncertainty over the accounting for carbon credits under the Kyoto Protocol, there seems to be agreement that afforestation will be eligible for offset credit (Schlamadinger and Marland 2000). Current projections in the United States for the value of a carbon credit are on the order of $2.72 to $4.54 per megagram of CO_2 sequestered, but the value is much higher in Europe. In Norway, for example, there is already a carbon tax on gasoline equivalent to $49 per megagram of CO_2 (Solberg 1997). Estimates from economic models suggest that a carbon tax of $27 to $109 per megagram of CO_2 would be necessary to stabilize global emissions at the 1990 level (Solberg 1997). Under these conditions, growing biomass for fuel would become an attractive alternative to fossil fuel because biofuels have no net impact on global carbon levels. At some time in the future, landowners in the LMAV may want to optimize carbon sequestration and biofuel benefits by planting willow (*Salix* spp.) on soils too wet for cottonwood.

Recreational Benefits

The primary recreational benefits assumed in the examples are from creating and enhancing wildlife habitat. Not all wildlife species require the same kind of habitat, so for simplicity the expected benefits can be separated into recreational hunting by the landowner (rather than lease fees) and nonconsumptive wildlife activities, such as bird watching or simply the existence value of wildlife to the landowner. Most species hunted in the LMAV benefit from a range of forest conditions and expected benefits are high in stands managed for pulpwood or saw logs. Low expected value is derived from the kind of open stands likely to develop from the green vegetation scenario (Allen 1997, King and Keeland 1999). Neotropical migratory birds and other birds are not uniform in their habitat requirements (Hamel and others, in press) but some will benefit from the kind of early successional habitat typical of short-rotation stands (Twedt and Portwood 1997), as well as the early successional herbaceous fields of the green vegetation scenario. Species of concern are of two kinds, those requiring the early successional herbaceous vegetation and those found in the kind of complex vegetation structure found only in older stands, which the saw log rotation may develop in time (Hamel and others, in press). Birds that use the intermediate conditions of stand development are probably likely to occur in developing stands for which the intended management purpose is sawtimber production.

Environmental Benefits

Water-quality benefits of afforestation accrue from reducing soil erosion (Joslin and Schoenholtz 1998), and filtering, retaining, and assimilating nutrients and farm chemicals from surface runoff and groundwater (Huang and others 1990). As Lockaby and Stanturf (2001) point out, however, typical restoration stands in the LMAV no longer experience the kind of flow-through hydrology of a riverine system; and the filtering action will be limited.

Greater water-quality benefit will be derived from forested riparian buffers. Planted forested buffer strips in an agricultural landscape are uncommon, although several studies have examined the filtering action of natural forested riparian zones (Cooper and Gilliam 1987; Cooper and others 1987; Lowrance and others 1983, 1984a, 1984b, 1986; Peterjohn and Correll 1984; Todd and others 1983). Comerford and others (1992) summarized these studies and concluded that buffer strips are quite effective in removing soluble nitrogen and phosphorus (up to 99 percent) and sediment. The efficiency of pesticide removal by forested buffer strips has been examined in some environmental fate studies that concluded that buffer strips 15 m or wider were generally effective in minimizing pesticide contamination of streams from overland flow (Comerford and others 1992). Recently, forested buffer strips in the LMAV became attractive financially to the landowner by a new incentive program (Continuous Signup/Conservation Reserve Program), which allows use of the cottonwood/red oak nurse-crop system.

CONCLUSION

Forest restoration, in the broad sense that encompasses afforestation, rehabilitation, and reclamation, is occurring throughout the temperate and boreal zones. Site conditions differ, native species are diverse, and the policy context in which restoration occurs varies. Nevertheless, the challenges faced by managers are similar: overcome site degradation, prescribe appropriate species, and apply cost-effective methods. Clarity of objectives is critical to designing a successful restoration program and diagnosis of site conditions and potential should guide intervention.

Knowledge and experience with establishing plantation for timber production can be used to efficiently restore large areas of agricultural land to forest.

While simple plantations have many financial and technological advantages, plantations that are more complex will be required in most countries. Aesthetics, species diversity, and the need to rapidly create vertical vegetation structure are some of the concerns that must be addressed.

The benefits of restoration should be viewed in comparison to the previous conditions of land cover and land use. In most cases, environmental benefits are immediate. Restoration forests can differ in functioning and management, according to the objectives of the landowner. There is usually considerable overlap between social and individual benefits. Even forests primarily managed for timber production provide environmental and recreational benefits to society.

LITERATURE CITED

Allen, J.A. 1997. Reforestation of bottomland hardwoods and the issue of woody species diversity. Restoration Ecology. 5: 125–134.

Ashton, P.M.S.; Gamage, S.; Gunatilleke, I.A.U.N.; Gunatilleke, C.V.C. 1997. Restoration of a Sri Lankan rainforest: using Caribbean pine *Pinus caribaea* as a nurse for establishing late-successional tree species. Journal of Applied Ecology. 34: 915–925.

Barker, J.R.; Baumgardner, G.A.; Turner, D.P.; Lee, J.J. 1996. Carbon dynamics of the conservation and wetland reserve programs. Journal of Soil and Water Conservation. 51: 340–346.

Bradshaw, A.D. 1997. What do we mean by restoration? In: Urbanska, K.M.; Webb, N.R.; Edwards, P.J., eds. Restoration ecology and sustainable development. Cambridge, UK: Cambridge University Press: 8–14.

Burbridge, P.R.; Hallin, D.C. 2002. Rehabilitation of coastal wetland forests degraded through their conversion to shrimp farms. [This proceedings].

Butterfield, R.P.; Fisher, R.F. 1994. Untapped potential: native species for reforestation. Journal of Forestry. 92 (6): 37–40.

Cairns, J., Jr. 1986. Restoration, reclamation, and regeneration of degraded or destroyed ecosystems. In: Soule, M.E., ed. Conservation biology. Ann Arbor, MI: Sinauer Publishers: 465–484.

Chapman, C.A.; Chapman, L.J. 1999. Forest restoration in abandoned agricultural land: a case study from East Africa. Conservation Biology. 13: 1301–1311.

Comerford, N.B.; Neary, D.G.; Mansell, R.S. 1992. The effectiveness of buffer strips for ameliorating offsite transport of sediment, nutrients, and pesticides from silvicultural operations. NCASI Tech. Bull. 631. New York: National Council of the Paper Industry for Air and Stream Improvement. 48 p.

Cooper, J.R.; Gilliam, J.W. 1987. Phosphorus redistribution from cultivated fields into riparian areas. Soil Science Society of America Journal. 51: 1600–1604.

Cooper, J.R.; Gilliam, J.W.; Daniels, R.B. 1987. Riparian areas as filters for agricultural sediment. Soil Science Society of America Journal. 51: 416–420.

DeGraaf, R.M. 1987. Managing northern hardwoods for breeding birds. In: Nyland, R.D., ed. Managing northern hardwoods, Proceedings silvicultural symposium; 1986 June 23–25; Syracuse, NY. Misc. Publ. 13 (ESF 87–002). Syracuse, NY: Syracuse University, Faculty of Forestry: 348–362.

Fisher, R.F. 1995. Amelioration of degraded rain forest soils by plantations of native trees. Soil Science Society of America Journal. 59: 544–549.

Gardiner, E.S.; Russell, D.R.; Oliver, M., Dorris, L.C., Jr. 2001. Bottomland hardwood afforestation: state of the art. In: Holland, M.J.; Warren, M.L., Jr.; Stanturf, J.A.

Grainger, A. 1988. Estimating areas of degraded tropical lands requiring replenishment of forest cover. International Tree Crops Journal. 5: 31–61.

Hamel, P.B.; Buckner, E.R. 1998. How far could a squirrel travel in the treetops? A prehistory of the southern forest. In: Wadsworth, K.G., ed. Transactions of the 63rd North American wildlife and natural resources conference; 1998 March 20–24; Orlando, FL. Washington, DC: Wildlife Management Institute: 309–315.

Hamel, P.B.; Nuttle, T.; Woodson, C.A.; Broerman, F. 2002. Forest restoration as ecological succession: should we speed it up or slow it down? [This proceedings].

Harrington, C.A. 1999. Forests planted for ecosystem restoration or conservation. New Forests. 17 (1–3): 175–190.

Huang, W.-Y.; Algozin, K.; Ervin, D.; Hickenbotham, T. 1990. Using the conservation reserve program to protect groundwater quality. Journal of Soil and Water Conservation. 46: 251–254.

Islam, K.R.; Kamaluddin, M.; Bhuiyan, M.K.; Badruddin, B. 1999. Comparative performance of exotic and indigenous forest species for tropical semievergreen degraded forestland reforestation in Chittagong, Bangladesh: Land Degradation & Development. 10: 241–249.

Johnson, W.C. 1988. Estimating dispersibility of *Acer*, *Fraxinus*, and *Tilia* in fragmented landscapes from patterns of seedling establishment. Landscape Ecology. 1: 175–187.

Joslin, J.D.; Schoenholtz, S.H. 1998. Measuring the environmental effects of converting cropland to short-rotation woody crops: A research approach. Biomass and Bioenergy. 15: 301–311.

Kanowski, P.J. 1997. Afforestation and plantation forestry. In: Proceedings of the XI World Forestry Congress; 1997 October 13–22; Antalya, Turkey. [Available on the Web: http://coombs.anu.edu.au/Depts/RSPAS/RMAP/kanow.htm].

King, S.L.; Keeland, B.D. 1999. Evaluation of reforestation in the Lower Mississippi River Alluvial Valley. Restoration Ecology. 7: 348–359.

Knowles, O.H.; Parrotta, J.A. 1995. Amazonian forest restoration: an innovative system for native species selection based on phenological data and field performance indices. Commonwealth Forestry Review. 74 (3): 230–243.

Krishnaswamy, A.; Hanson, A., ed. 1999. Our forests, our future: summary report, world commission on forests and sustainable development. Cambridge, UK: Cambridge University Press: 37.

Lamb, D.; Tomlinson, M. 1994. Forest rehabilitation in the Asia-Pacific region; past lessons and present uncertainties. Journal Tropical Forest Science. 7 (1): 157–170.

Lockaby, B.G.; Stanturf, J.A. 2002. Potential effects of restoration on biogeochemical functions of bottomland hardwood ecosystems. [This proceedings].

Lowrance, R.R.; Todd, R.L.; Asmussen, L.E. 1983. Waterborne nutrient budgets for the riparian zone of an agricultural watershed. Agriculture Ecosystems and Environment. 10: 371–384.

Lowrance, R.R.; Todd, R.L.; Asmussen, L.E. 1984a. Nutrient cycling in an agricultural watershed: I. Phreatic movement. Journal of Environmental Quality. 13: 22–27.

Lowrance, R.R.; Todd, R.L.; Fail, J. Jr. [and others]. 1984b. Riparian forest as nutrient filters in agricultural watersheds. BioScience. 34: 374–377.

Lowrance, R.R.; Sharpe, J.K.; Sheridan, J.M. 1986. Long-term sediment deposition in the riparian zone of a Coastal Plain watershed. Journal of Soil and Water Conservation. 41: 266–271.

Madsen, P.; Arad•ttir, Á.L.; Gardiner, E. [and others]. 2002. Forest restoration in the Nordic countries. [This proceedings].

Ohta, S. 1990. Initial soil changes associated with afforestation with *Acacia auriculiformis* and *Pinus kesiya* on denuded grasslands of the Pantabangan area, Central Luzon, the Philippines. Soil Science and Plant Nutrition. 36: 633–643.

Otsamo, R. 2000. Early development of three planted indigenous tree species and natural understory vegetation in artificial gaps in an *Acacia mangium* stand on an *Imperata cylindrica* grassland site in South Kalimantan, Indonesia. New Forests. 19: 51–68.

Parrotta, J.A. 1992. The role of plantation forests in rehabilitating tropical ecosystems. Agriculture, Ecosystems, and Environment. 41: 115–133.

Parrotta, J.A.; Turnbull, J.W.; Jones, N. 1997. Catalyzing native forest regeneration on degraded tropical lands. Forest Ecology and Management. 99: 1–7.

Peterjohn, W.T.; Correll, D.L. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of riparian forests. Ecology. 65: 1466–1475.

Schlamadinger, B.; Marland, G. 2000. Land use & global climate change—forests, land management, and the Kyoto Protocol. Arlington, VA: Pew Center on Global Climate Change. 54 p.

Schweitzer, C.J.; Stanturf, J.A.; Shepard, J.P. [and others]. 1997. Large-scale comparison of reforestation techniques commonly used in the Lower Mississippi River Alluvial Valley. In: Pallardy, S.G.; Cecich, R.A.; Garrett, H.G.; Johnson, P.S., eds. Proceedings, 11th central hardwood forest conference; 1997 March 23–26; Columbia, MO. Gen. Tech. Rep. NC–188. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station: 313–320.

Solberg, B. 1997. Forest biomass as carbon sink—economic value and forest management/policy implications. Critical Reviews in Environmental Science and Technology. 27(Special): S323–S333.

Stanturf, J.A.; Gardiner, E.S.; Hamel, P.B. [and others]. 2000. Restoring bottomland hardwood ecosystems in the Lower Mississippi Alluvial Valley. Journal of Forestry. 98(8): 10–16.

Stanturf, J.A.; Portwood, C.J. 1999. Economics of afforestation with Eastern cottonwood (*Populous deltoides*) on agricultural land in the Lower Mississippi Alluvial Valley. In: Haywood, James D. ed. Proceedings of the tenth biennial southern silvicultural research conference; 1999 February 16–18; Shreveport, LA. Gen. Tech. Rep. SRS–30. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 66–72

Stanturf, J.A.; Schoenholtz, S.H.; Schweitzer, C.J.; Shepard, J.P. 2001. Achieving restoration success: myths in bottomland hardwood forests. Restoration Ecology. 9(2): 189–200.

Stanturf, J.A.; Schweitzer, C.J.; Gardiner, E.S. 1998. Afforestation of marginal agricultural land in the Lower Mississippi River Alluvial Valley, USA. Silva Fennica. 32: 281–297.

Todd, R.L.; Lowrance, R.R.; Hendrickson, O. [and others]. 1983. Riparian vegetation as filters of nutrients exported from a Coastal Plain agricultural watershed. In Lowrance, R.R.; Todd, R.L.; Asmussen, L.E.; Leonard, R.A., eds. Nutrient cycling in agricultural ecosystems. Spec. Publ. 23. Athens, GA: University of Georgia, College of Agriculture: 485–496.

Twedt, D.J.; Portwood, C.J. 1997. Bottomland hardwood reforestation for neotropical migratory birds: are we missing the forest for the trees? Wildlife Society Bulletin. 25: 647–652.

Whalley, D.N. 1988. Afforestation of salt-affected soils—an international problem. International Plant Propagator's Society Combined Proceedings. 37: 212–221.

FOREST LINKAGES TO DIVERSITY AND ABUNDANCE IN LOWLAND STREAM FISH COMMUNITIES

Melvin L. Warren, Jr., Wendell R. Haag, and Susan B. Adams[1]

Abstract—In 1999 we sampled fish and fish habitat in 79 stream reaches within watersheds of north-central Mississippi. Despite a program of successful reforestation and soil stabilization (Yazoo-Little Tallahatchie Project, 1949–1985), nearly all streams in the region are channelized or incised. In these sandy, upper Coastal Plain streams, we explored the relationships among in-stream wood, canopy cover, and stream fish assemblages and were particularly interested in how these relationships are affected by extensive channel modification. Minnows, sunfishes, darters, and catfishes, respectively, dominated the fauna. Total fish species, total fish abundance, minnow relative abundance, and canopy cover were related only to watershed size. Stream incision, as indicated by high banks and shallow water depths, showed negative associations with sunfish relative abundance, in-stream wood, and detritus. Flow and large in-stream wood were associated positively with relative abundances of darters and catfishes. Despite scarcity of wood and deep pools in these systems, we detected associations between fish assemblage composition and in-stream wood. Our analysis suggests even modest densities of in-stream wood can shift fish assemblage attributes from colonizing stages to intermediate or stable stages.

INTRODUCTION

Large-scale reforestation emphasizing riparian zones can dramatically affect fish communities of low-gradient Coastal Plain streams. Forested riparian zones provide multiple benefits to stream fishes (Angermeier and Karr 1984, Gregory and others 1991). Indirect benefits include reduction of sediment and nutrient inputs (Lowrance and others 1984), stabilization of stream banks, and moderation of water temperature extremes (Gregory and others 1991). These factors can affect fish productivity, physiology, reproduction, and community composition (Matthews 1987). More directly, organic matter input into streams as leaves and in-stream wood provides the primary energy source for aquatic macroinvertebrates (Wallace and others 1997), which form the food base for most stream fishes. In sandy Coastal Plain streams, debris dams and large wood greatly increase macroinvertebrate production (Benke and others 1984, Smock and others 1989), promote channel stability, and increase habitat complexity for fishes (Shields and Smith 1992).

From 1949 to 1985 the U.S. Department of Agriculture, Forest Service, and other Federal agencies led the Yazoo-Little Tallahatchie Project in the upper Coastal Plain of northern Mississippi (U.S. Department of Agriculture, Forest Service 1988). The program was designed to re-establish forests and stabilize soils in a region subjected to a sequence of massive erosion cycles from 1830 into the 1970s (Schumm and others 1984). Postsettlement alluvium from hillside erosion, reaching 5-m depths in valleys, exacerbated flooding and prompted the dredging and channelization of most streams in the region (Schumm and others 1984). Modification of stream channels induced cycles of stream incision or headcutting, an active geomorphic process on the landscape today (Shields and others 1994, 1998). Although not focused on riparian areas, the reforestation effort successfully stabilized hillslopes and dramatically increased forest cover of the region.

In 1999 we sampled fish and fish habitat in streams within the stabilized, reforested region. In these sandy, upper Coastal Plain streams, we explored the relationships among in-stream wood, canopy cover, and stream fish assemblages. We were particularly interested in how extensive channel modification affected these relationships. We examined whether multivariate relationships among canopy, in-stream wood, and fish community attributes were detectable, and if so, how they covaried across a range of watershed sizes and local habitat configurations.

STUDY REGION

The study region lies primarily within the Holly Springs National Forest located in Benton, Lafayette, Marshall, Tippah, and Union Counties in north-central Mississippi. The proclamation boundaries of potential Federal ownership include 153,000 ha, about 35 percent of which is national forest land. The ecoregion is classified as the Northern Loessial Hills Subsection, Coastal Plain Middle Section, Southeastern Mixed Forest Province (Keys and others 1995).

The study streams drain parts of four major river basins: the Little Tallahatchie and Yocona Rivers (both Yazoo River tributaries) and the Hatchie and Wolf Rivers (both direct Mississippi River tributaries). The topography consists of irregular low hills (200-m maximum relief) dissected by well-developed, dendritic drainage systems with flat, often broad, floodplains adjacent to large streams. Stream substrate is predominantly sand or silty sand with accumulations of finer material in depositional areas.

[1] Warren, Research Biologist, Haag, Fisheries Biologist, and Adams, Research Fisheries Biologist, Center for Bottomland Hardwoods Research, Southern Research Station, 1000 Front Street, Oxford, MS 38655.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

The disturbance history of the area has profoundly modified stream systems through aggradation, channelization, and incision. The soils of the region are highly erodible, being derived from wind-deposited loess and Coastal Plain materials (Dendy and others 1979). Following European settlement, the land was deforested and cultivated. These actions accelerated hillside erosion, led to massive gully formation in uplands, and filled valleys with postsettlement alluvium (Cooper and Knight 1991, Schumm and others 1984, Shields and others 1994). All of these disturbances exacerbated flooding. Between about 1840 and 1930 individuals and local drainage districts attempted ineffectively to reclaim valley lands through channelization of streams and construction of drainage ditches (Shields and others 1994). From 1930 to 1970 in conjunction with reforestation and soil stabilization programs, Federal agencies led a second round of stream channelization and constructed many flood-control impoundments in headwater streams. Streams responded to changes in channel gradients, resulting in large part from channelization by undergoing incision. This process affects entire stream networks upstream to the smallest headwater channels. Incision is rapid, proceeding upstream at rates up to 500 m per year (Schumm and others 1984), often occurs episodically, and is widespread in the region (Shields and others 1998). First, streams eroded through the partially consolidated surface material; then, downcutting proceeded rapidly through underlying strata of unconsolidated sand and clay. Geologic controls in the resulting sand-bed channels are infrequent and mostly consist of outcrops of consolidated clay or cemented sand (Shields and others 1997). Despite successful reforestation and concomitant soil stabilization in the area, few streams have escaped channelization or incision. Massive bank failures from stream channel incision can contribute amounts of sediment to the region's stream systems comparable to amounts produced by deforestation at European settlement (Schumm and others 1984).

METHODS

We sampled fish and fish habitat at 79 wadeable stream reaches in the study region from May to September 1999 (appendix A). These reaches represent a range of watershed sizes, habitat types, and watershed conditions across the study region. When possible, we selected one or more extreme headwater, middle, and downstream reaches in each watershed and covered as many habitat types as possible within selected reaches, e.g., riffle, run, pool. We sampled a predetermined length of stream, such that sampling effort was approximately proportional to stream size (Angermeier and Smogor 1995, Lyons 1992, Paller 1995). Typically, reach lengths were 20 times the average width of the stream. Minimum reach length was 80 m for streams < 4 m in average width, and maximum reach length was 300 m for streams wider than 15 m. We subdivided each reach into four subreaches of equal length. We located the upstream and downstream termini of the reach and subreaches with a hip chain, tape, or range finder before sampling.

Fish Sampling

We employed two methods of fish sampling at each reach: (1) single-pass backpack electrofishing, and (2) seining. Combination of the two methods helped reduce bias and assured capture of most fishes representative of the reach assemblage. We standardized effort for both methods and sampled each subreach separately. For electrofishing, we calculated total sampling time as the reach length times 5 seconds and applied one-fourth the total effort to each subreach. For seining, we conducted 8 seine hauls (2 per subreach) for streams < 5 m in average width and 12 seine hauls (3 per subreach) for streams > 5 m in average width. One haul was either a sustained drag of the seine within a stream macrohabitat, such as a pool or one set-and-kick seine in a riffle (Jenkins and Burkhead 1994). Standardization of fishing times for electrofishing and effort for seining assured that streams of all sizes were sampled with effort proportional to their size. We kept fish samples from each subreach separately for both electrofishing and seining, but pooled all data within each reach for this analysis.

Physical Habitat Sampling

Subsequent to fish sampling, we sampled physical habitat. We established 12 equally spaced transects within each reach so that 3 transects occurred (one-fourth, one-half, and three-fourths of subreach length) within each subreach. At each transect, we recorded wetted width and estimated bank stability (eroding or stable); bank angle (steep, 90 degrees; moderately steep, > 45 degrees; gradual, > 45 degrees); bank height (nearest meter); and dominant bank vegetation. At equally spaced points along each transect, we measured water depth and velocity (at 0.6 depth). We visually categorized dominant substrate type as soft clay, hard clay, silt, sand, fine gravel (2 to 15 mm), coarse gravel (16 to 63 mm), cobble (64 to 256 mm), or bedrock. We also recorded presence or absence of detritus, small wood (< 10 cm in diameter or < 1.5 m in length), large wood (>10 cm in diameter, > 1.5 m in length), and aquatic vegetation within an area approximately 1 m^2 around each point. We estimated canopy cover at each point as 0, 25, 50, or 100 percent cover. We adjusted the number of points per transect for stream width. Transects > 10 m in width had points located at 2-m intervals; transects 5 to 10 m in width, at 1-m intervals; and transects < 5 m had a minimum of 5 equally placed points.

Data Analysis

We calculated the following measures of fish community composition for 66 of 79 reaches: total species, total abundance, and familial relative abundance. We eliminated 13 reaches from the analysis because physical data were incomplete, no fishes were captured, or visits in September 1999 revealed the sampled reach was located in an intermittent stream. Total species and total abundance at a reach were the total recorded number of fish species and individuals, respectively. Familial relative abundance was the proportional abundance of the four most abundant families across reaches: Centrarchidae, sunfishes; Cyprinidae, minnows; Percidae, darters (genera *Etheostoma* and *Percina*); and Ictaluridae, catfishes (mostly bullhead catfishes, *Ameiurus* spp., and madtom catfishes, *Noturus* spp.). Familial responses to physical habitat differences are ecological in origin, and within each family, fish species are reasonably similar in habitat requirements (Shields and others 1998).

For each reach, we derived mean values for a suite of physical and riparian characteristics. We calculated average depth (centimeter), velocity (meter per second), and bank height (meter) from transect points. We determined proportions of detritus, small in-stream wood, and large in-stream wood from the number of points at which each occurred divided by the number of points surveyed. For comparison with other studies, we also were interested in converting the large in-stream wood data into the number of pieces per unit length. Assuming that each transect point with large in-stream wood represented one piece of large wood, we calculated the number of pieces over the length of each reach and averaged across reaches. This rough data transformation to pieces of wood per unit length allows general comparison with other studies. We determined watershed area on 1:100,000-scale topographic maps in a Geographic Information System. We calculated stream order (Strahler 1957) and link magnitude (Osborne and Wiley 1992), including both intermittent and permanent channels on 1:24,000-scale topographic maps.

We used principal components analysis (PCA) as a first approximation to partition covariation of physical variables and fish community composition into one or more component axes and to ordinate reaches in physical variable-fish community space. The PCA contained geomorphic-related variables (watershed area, average velocity, average depth, and average bank height); riparian-related variables (proportions of canopy cover, large wood, small wood, and detritus); and fish community variables (total species, total abundance, and familial proportional abundance). We extracted principal components (SAS 1996) from the correlation matrix of the variables. We transformed watershed area, total species, average depth, and average velocity to logarithms (base 10) and proportional variables to arcsine square roots (Sokal and Rohlf 1995). We used the broken-stick model (Jackson 1993) to evaluate the interpretability of each principal component axis. Using this method, an axis has interpretive value if observed eigenvalues exceed eigenvalues generated by the model.

Table 1—Mean, standard error (SE), minimum, and maximum values for watershed area, physical habitat variables, canopy cover, and in-stream wood variables at 66 reaches in upper Coastal Plain streams of north-central Mississippi

Variable	Mean	SE	Min.	Max.
Watershed area (ha)	2543	457.4	43	15194
Depth (cm)	14.3	1.40	2	49
Velocity (m/s)	.17	.013	< .01	.53
Bank height (m)	2.7	.18	.9	7.8
Canopy cover (%)	71	3.3	6	100
Detritus (%)	16	2.0	0	95
Small wood (%)	27	2.2	0	68
Large wood (%)	7	.9	0	28

Table 2—Mean, standard error (SE), minimum, and maximum values for fish community variables at 66 reaches

Variable	Mean	SE	Min.	Max.
Total species	14.7	0.84	3	33
Total abundance (individuals)	154.2	18.94	7	1009
Minnow relative abundance (%)	41.7	3.41	0	100
Sunfish relative abundance (%)	21.0	2.50	0	100
Darter relative abundance (%)	13.3	1.80	0	73
Catfish relative abundance (%)	6.3	.99	0	32

RESULTS

The sampled reaches represented a range of physical stream conditions (table 1). Watershed area spanned 3 orders of magnitude; however, 55 of 66 reaches drained watersheds of 100 to 10 000 ha. Stream order ranged from first to fifth order and link magnitude from 1 to 314, but 82 percent of reaches were third order. Streams were generally shallow with low velocities. Only seven reaches had mean depths > 30 cm and only eight had velocities > 0.30 m per second. Stream incision as indicated by average bank height and percent shallow depths was a prominent feature across reaches. Most reaches (66 percent) had bank heights > 2 m, notwithstanding inclusion of many small streams. Most depths (61 percent) along transects were < 15 cm. Stream substrate was dominated by sand (71.4 percent of all transect points) with some silt (14.3 percent), hard clay (7.3 percent), and soft clay (5.1 percent); rocky substrates were rarely encountered (< 2.0 percent). Average canopy cover was high despite the range in watershed sizes. Three-fourths of reaches had 50 percent or greater canopy cover. The proportion of large wood was low (table 1), and for half the reaches, large wood occurred in < 5 percent of the sampled points. Large wood averaged 47 pieces per kilometer (SE = 5) and ranged from 0 to 213 pieces per kilometer.

The fish fauna was relatively diverse and abundant within most reaches, but highly variable among reaches (tables 2, 3). We captured 65 fish species representing 15 families (table 3), and only 16 reaches had fewer than 10 species. However, variability of total species among reaches was high with a coefficient of variation (CV) of 46.2 percent (table 2). Total abundance also showed high variability with CV of 99.8 percent. Catch per unit effort for electrofishing ranged from 0.02 to 0.71 individuals per second (mean = 0.21, SE = 0.016, CV = 62.4 percent). Capture rate on a reach-length basis ranged from 0.09 to 6.64 individuals per meter (mean = 1.47, SE = 0.138, CV = 76.4 percent). Density of fishes ranged from 0.04 to 1.7 individuals per square meter (mean = O.42, SE = 0.043, CV = 82.5 percent).

Table 3—Fish species frequency and proportional occurrence for 66 stream reaches in the upper Coastal Plain of north-central Mississippi

Family and species	Reaches	
	Proportion	Frequency
Amiidae		
Bowfin		
(*Amia calva* Linnaeus)	.015	1
Aphredoderidae		
Pirate perch		
[*Aphredoderus sayanus* (Gilliams)]	.288	19
Atherinidae		
Brook silverside		
[*Labidesthes sicculus* (Cope)]	.121	8
Catostomidae		
Creek chubsucker		
[*Erimyzon oblongus* (Mitchill)]	.364	24
Northern hog sucker		
[*Hypentelium nigricans* (Lesueur)]	.212	14
Spotted sucker		
[*Minytrema melanops* (Rafinesque)]	.045	3
Blacktail redhorse		
(*Moxostoma poecilurum* Jordan)	.182	12
Centrarchidae		
Green sunfish		
(*Lepomis cyanellus* Rafinesque)	.606	40
Warmouth		
[*L. gulosus* (Cuvier)]	.273	18
Orangespotted sunfish		
[*L. humilis* (Girard)]	.030	2
Bluegill		
(*L. macrochirus* Rafinesque)	.758	50
Dollar sunfish		
[*L. marginatus* (Holbrook)]	.258	17
Longear sunfish		
[*L. megalotis* (Rafinesque)]	.545	36
Redear sunfish		
[*L. microlophus* (Gunther)]	.045	3
Redspotted sunfish		
[*L. miniatus* (Jordan)]	.121	8
Spotted bass		
[*Micropterus punctulatus* (Rafinesque)]	.288	19
Largemouth bass		
[*M. salmoides* (Lacepede)]	.303	20
White crappie		
(*Pomoxis annularis* Rafinesque)	.030	2
Clupeidae		
Gizzard shad		
[*Dorosoma cepedianum* (Lesueur)]	.015	1
Cyprinidae		
Bluntface shiner		
[*Cyprinella camura* (Jordan & Meek)]	.606	40
Blacktail shiner		
(*C. venusta* Girard)	.303	20

continued

Table 3—Fish species frequency and proportional occurrence for 66 stream reaches in the upper Coastal Plain of north-central Mississippi (continued)

Family and species	Reaches: Proportion	Reaches: Frequency
Cyprinidae (continued)		
Common carp (*Cyprinus carpio* Linnaeus)	.015	1
Cypress minnow (*Hybognathus hayi* Jordan)	.015	1
Mississippi silvery minnow (*Hybognathus nuchalis* Agassiz)	.242	16
Striped shiner (*Luxilus chrysocephalus* Rafinesque)	.197	13
Ribbon shiner [*Lythrurus fumeus* (Evermann)]	.136	9
Redfin shiner [*L. umbratilis* (Girard)]	.409	27
Golden shiner [*Notemigonus crysoleucas* (Mitchill)]	.106	7
Orangefin shiner (*Notropis ammophilus* Suttkus & Boschung)	.030	2
Emerald shiner (*N. atherinoides* Rafinesque)	.288	19
Yazoo shiner (*N. rafinesquei* Suttkus)	.258	17
Mimic shiner [*Notropis volucellus* (Cope)]	.121	8
Pugnose minnow (*Opsopoeodus emiliae* Hay)	.045	3
Bluntnose minnow [*Pimephales notatus* (Rafinesque)]	.288	19
Bullhead minnow [*P. vigilax* (Baird & Girard)]	.061	4
Creek chub [*Semotilus atromaculatus* (Mitchill)]	.530	35
Esocidae		
Grass pickerel (*Esox americanus* Gmelin)	.121	8
Fundulidae		
Blackstripe topminnow [*Fundulus notatus* (Rafinesque)]	.561	37
Blackspotted topminnow [*F. olivaceous* (Storer)]	.845	56
Ictaluridae		
Yellow bullhead [*Ameiurus natalis* (Lesueur)]	.333	22
Brown bullhead [*A. nebulosus* (Lesueur)]	.030	2
Channel catfish [*Ictalurus punctatus* (Rafinesque)]	.159	11
Least madtom [*Noturus hildebrandi* (Bailey & Taylor)]	.015	1
Brindled madtom (*N. miurus* Jordan)	.121	8

continued

Table 3—Fish species frequency and proportional occurrence for 66 stream reaches in the upper Coastal Plain of north-central Mississippi (continued)

Family and species	Reaches	
	Proportion	Frequency
Ictaluridae (continued)		
Brown madtom (*N. phaeus* Taylor)	.667	44
Flathead catfish [*Pylodictis olivaris* (Rafinesque)]	.015	1
Lepisosteidae		
Spotted gar [*Lepisosteus oculatus* (Winchell)]	.076	5
Longnose gar [*L. osseus* (Linnaeus)]	.015	1
Percidae		
Bluntnose darter [*Etheostoma chlorosoma* (Hay)]	.045	3
Slough darter [*E. gracile* (Girard)]	.167	11
Harlequin darter (*E. histrio* Jordan & Gilbert)	.045	3
Brighteye darter [*E. lynceum* (Hay)]	.333	22
Johnny darter (*E. nigrum* Rafinesque)	.258	17
Goldstripe darter (*E. parvipinne* Gilbert & Swain)	.424	28
Cypress darter [*E. proeliare* (Hay)]	.106	7
Yazoo darter (*E. raneyi* Suttkus & Bart)	.409	27
Speckled darter [*E. stigmaeum* (Jordan)]	.303	2
Gulf darter [*E. swaini* (Jordan)]	.212	14
Redfin darter [*E. whipplei* (Girard)]	.242	16
Bandfin darter (*E. zonistium* Bailey & Etnier)	.015	1
Dusky darter [*Percina sciera* (Swain)]	.515	34
River darter [*P. shumardi* (Girard)]	.030	2
Petromyzontidae		
Ichthyomyzon sp.	.424	28
Poeciliidae		
Western mosquitofish [*Gambusia affinis* (Baird & Girard)]	.273	18
Sciaenidae		
Freshwater drum (*Aplodinotus grunniens* Rafinesque)	.061	4

Minnows, sunfishes, darters, and catfishes, respectively, dominated the fauna and on average accounted for about 82 percent of individuals captured (table 2). These four families were represented by at least one species in most reaches (table 3). Minnow and sunfish species each occurred in 65 reaches, darter species in 63, and catfish species in 55. Average relative abundance and number of species showed similar ranks among families. Minnows had the highest average relative abundance and the most species (17) and catfishes, the lowest abundance and 7 species (table 3). Sunfishes and darters were represented by 11 and 14 species, respectively. The 11 other families accounted for 16 species that ranged from relatively rare, e.g. bowfin, gizzard shad, to widespread, e.g., blackspotted and blackstripe topminnows (table 3).

Principal components analysis of physical and fish community variables across reaches produced eigenvalues exceeding those of the broken-stick model, providing support for interpretation of the first three axes (table 4). The first three axes accounted for 60 percent of the total variance (table 4). We considered variables with loadings > 0.20 (absolute value) as being associated with an axis; 52 percent of loadings across axes did not exceed this value. Because we were interested in interaction of stream geomorphic variables with in-stream wood and fish community variables, we named axes based on the geomorphic variable(s) with the highest absolute loading.

We interpreted the first principal component axis (PC–I) as a watershed size axis. Magnitudes and polarities of loadings (table 4) contrasted total species (fig. 1A), total abundance (fig. 1B), minnow relative abundance, watershed area, depth, and velocity, which all increased with watershed size, with canopy cover and detritus accumulations, which decreased with watershed size. Total species, total abundance, minnow relative abundance, and canopy cover were only associated with PC–I; absolute loadings on other axes were low. We detected no strong associations between canopy cover and these fish community variables except in the context of watershed size. The watershed size axis was only weakly associated with bank height, an indicator of stream incision.

Table 4—Loadings on the first three principal component axes of habitat, canopy, in-stream wood, and fish community variables for 66 stream reaches in the upper Coastal Plain of north-central Mississippi

Variable	PC–I	PC–II	PC–III
Watershed area	0.421	0.147	0.049
Depth	.304	.391	- .052
Velocity	.214	.158	.280
Bank height	.165	- .368	.066
Canopy cover	- .285	- .053	.066
Detritus	- .227	.282	- .243
Small wood	- .183	.484	.021
Large wood	- .030	.480	.222
Total species	.402	.173	- .024
Total abundance	.396	.037	- .062
Minnow relative abundance	.382	- .141	- .030
Sunfish relative abundance	- .075	.235	- .525
Darter relative abundance	- .068	.061	.543
Catfish relative abundance	- .122	.086	.471
Total variance (%)	29	17	14

We interpreted PC–II as a stream incision-depth axis. Magnitudes and polarities of loadings contrasted sunfish abundance, large and small in-stream wood, detritus, and depth (pool development) with bank height (table 4). The axis arrayed reaches along a gradient from shallow, deeply incised streams with low proportions of wood and sunfishes to deep streams with low banks and relatively high proportions of wood and sunfishes.

We interpreted PC–III as a stream velocity axis. The loadings indicate positive covariation between darter and catfish abundance, velocity, and large wood. Sunfish abundance and detritus covaried negatively with these variables. The axis arrayed reaches along a gradient from flowing, stream habitats having high proportions of large wood, darters, and catfishes and low proportions of detritus and sunfishes to slow or nonflowing habitats with low proportions of large wood, darters, and catfishes and high proportions of sunfishes. Importantly, the pattern of covariation indicates that the relative abundance of two largely rheophilic families, darters and catfishes, covaried positively with large wood in the context of flowing habitats.

DISCUSSION

We detected covariation among fish community measures, canopy, in-stream wood, detritus, and physical habitat configuration across a variety of watershed sizes and conditions. The watershed size axis essentially removed species-area effects from the analysis, such that remaining interpretable axes were independent of watershed size effects. Total species, minnow relative abundance, and total abundance were related positively, and percent canopy cover, negatively to watershed size. Surprisingly, little or no variability in minnow relative abundance, total species, or fish abundance was related to either the stream incision-depth or stream velocity axes, both of which were associated with in-stream wood and/or detritus. Large wood increases habitat heterogeneity (Shields and Smith 1992), and stable, deep, heterogeneous habitats typically yield increased fish species richness (Angermeier and Smogor 1995, Schlosser 1987) and show decreased dominance by minnows (Schlosser 1987). We expected that a high proportion of in-stream wood, especially large wood in pools, would be associated with increased diversity of fishes and decreased dominance of minnows after accounting for watershed size on PC–I. Given the increases in stream productivity associated with debris dams and large wood (Benke and others 1984, Smock and others 1989), we also expected total fish abundance to covary positively with in-stream wood. In contrast, we found no relationship between in-stream wood and total species, minnow relative abundance, or total abundance.

Influences of pool development and flow regimes on in-stream wood abundance might explain why total species, minnow relative abundance, and total abundance were not

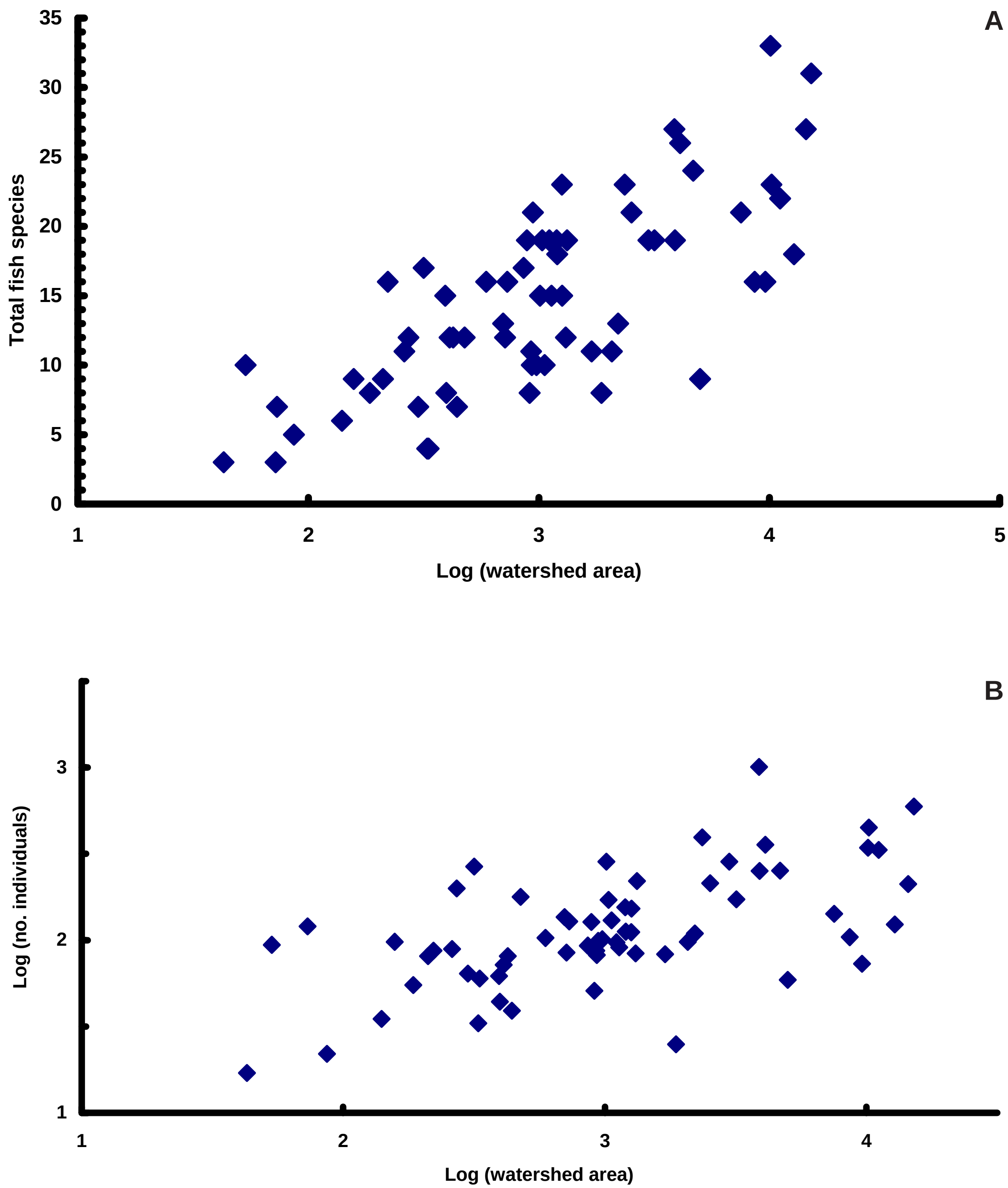


Figure 1—Scatterplots of watershed area and total number of fish species (A) and total number of fishes (B) in 66 stream reaches of the upper Coastal Plain of northern Mississippi.

associated strongly with in-stream wood. The overall amount of large wood in the sampled reaches appeared to be low, relative to conditions prevailing in streams within late successional forests. Large wood in Coastal Plain streams draining late successional forests is expected to approach about 200 pieces per kilometer (Smock and others 1989), and in eastern upland forests it can reach 400 to 500 pieces per kilometer (Hedman and others 1996, Hildebrand and others 1997). Our estimate of large wood across the study region is < 25 percent of the Coastal Plain expectation and only 6 reaches (9 percent) exceeded 100 pieces per kilometer. Shields and others (1995) also indicated scarcity of in-stream large wood in the study area. Maximum depth, an indicator of pool development, also was limited in the study reaches. Only 19 percent of our transect points exceeded 30 cm in depth, and only 6 percent exceeded 60 cm. Wide channels, shallow depths, and flashy flows in streams of the study region (Shields and others 1995, Wallerstein and others 1997) likely limit wood accumulation even in systems with relatively high wood recruitment. We believe the paucity of deep pools and large wood precluded detection of strong associations among total species, fish abundance, and in-stream wood after accounting for watershed size. Obversely, many of the minnow species represented in study reaches are adapted to the shallow, uniform, sandy habitats where they often occur in schools of hundreds to thousands of individuals. Ubiquity and extreme abundance of minnows, when treated as a single group, may explain their strong association with watershed size and lack of association with other axes. We believe analysis by species or habitat guilds within minnows may reveal associations beyond watershed size that were masked by examining the family as a group.

Stream incision strongly affects the interaction of in-stream wood with the quality of fish habitats and the composition of fish assemblages. Pairing Schlosser's (1987) conceptual stream framework for fishes and incised channel models (Schumm and others 1984, Simon 1989), three stages in fish community development for incising channels were hypothesized (Shields and others 1998). Colonizing fish assemblages are associated with shallow, uniform habitats with little debris. These are dominated by small species, such as minnows. In relatively undisturbed watersheds, these assemblages are found in small, headwater streams, but channelization and incision produce colonizing assemblages even in large streams. Intermediate assemblages typify streams with some increase in pool volume and begin to be comprised of larger fishes, such as sunfishes (Shields and others 1998). In both colonizing and intermediate stages, in-stream wood density may be depressed in response to changes in channel geometry (Shields and others 1998). As pool depth and volume increase further, stable assemblages develop with fewer, but larger, piscivores. Abundance of small invertivores and omnivores decreases as predation and resulting competition for refugia among prey species increases (Schlosser 1987, Shields and other 1998). At this stage, shallow riffle areas between pools provide important habitat, e.g., refuge from predators, for benthic invertivores, e.g., darters, madtom catfishes. Channelization and cycles of channel incision produce colonizing to intermediate fish assemblages (Shields and others 1998), but stable assemblage attributes may develop between disturbance events because of channel and adjoining floodplain dynamics.

We detected positive associations between the relative abundance of a dominant family, the sunfishes, and woody, deep habitats, a response attenuated by stream incision. These results are congruent with Schlosser's (1987) conceptual framework for small stream fish communities, in which deep, woody habitat was a key factor, and with fish assemblage changes observed after habitat manipulation in incised streams (Shields and others 1998). Addition of relatively deep, stable pools to incised streams shifted fish assemblages from colonizing stages to those more representative of intermediate stages, a change principally involving sunfishes (Shields and others 1998). Deep, woody pools increase habitat (pool volume), temporal stability, and forage availability, which influence fish community structure through shifts in fish age and size structure, species composition, and trophic composition. Wood in the deepest habitats in a stream may be especially critical to sunfishes during summer low flows, when total habitat area is reduced, and pools become relatively accessible to bird and mammal predators (Angermeier and Karr 1984). In Coastal Plain rivers sunfishes, especially *Lepomis* spp., obtain a large proportion of their food intake from invertebrates produced on wood, forming a distinct trophic pathway from in-stream wood to sunfishes (Benke and others 1985). Despite incision, poor pool development, and scarcity of large wood, the results from the stream incision-depth axis indicate that even modest increases in in-stream wood and pool development can play a strong role in structuring fish assemblages.

On the stream velocity axis, we discovered further evidence of the association of large wood and fish assemblage composition but in the context of water velocity, not pool development. The fish assemblage showed increased darter and catfish relative abundances and decreased sunfish relative abundance as large wood and velocities increased. Notably, this assemblage pattern emerged after watershed size and incision-related variation was removed, suggesting relatively small changes in habitat configuration improved conditions for species in both families. Although pool development is important in fish assemblage composition (Schlosser 1987), we suggest this pattern reflects the importance of large wood in providing cover for darters and catfishes irrespective of watershed size, incision, or the relatively shallow depths and low flows observed in study reaches. We attribute the trend in sunfish relative abundance on this axis, at least in part, to their affinity for low velocities, but realize other more complex factors, e.g., predation, may be involved. During field sampling we noticed that the majority of darters and catfishes (particularly madtoms) showed high affinity for cover, rarely occurring in open, uniformly sand-bottom habitats. Large wood generally formed primary cover in flowing habitats and is likely critical to the persistence of many darters and madtom catfishes (Chan and Parsons 2000, Monzyk and others 1997) in the study area. In sum, our results indicated large wood increases habitat quality for these two dominant and diverse families.

Fish species occurring in the region are those that have persisted through extensive historical and ongoing modifications to stream channels. We believe for most stream reaches surveyed that the number of species was at or near saturation because the available local species pool lacks potential new colonists regardless of the availability of stable, heterogeneous habitats. We did discover fish assemblage patterns associated with in-stream wood, but the legacy of channelization and incision limits the amount and thus the influence of wood on fish habitats. The results on the watershed size axis suggest that most reaches, regardless of watershed size, showed attributes of colonizing assemblages, particularly dominance by minnows. Minnows dominated 61 percent of our reaches. Minnows predictably dominate uniform shallow reaches (Schlosser 1987) and are conspicuously abundant in incised or channelized streams (Shields and others 1998). In contrast, covariation on the stream incision-depth axis indicates some reaches show characteristics of intermediate assemblages with increased representation of sunfishes. Incision apparently limits sunfishes by creating unstable shallow habitats that do not retain wood for cover or pool formation. The higher relative abundances of darters and catfishes on the stream velocity axis suggests that colonizing assemblages, predominated by minnows, may develop stable attributes if large wood and flow are present in sufficient quantity.

We did not examine some important aspects of stream fish assemblages, such as size structure or trophic relationships. Other assemblage attributes might show stronger associations with amounts of in-stream wood, canopy cover, or other aspects of habitat configuration. We plan to examine the data set to discern further assemblage patterns in the context of ecological guilds and watershed conditions.

ACKNOWLEDGMENTS

We appreciate the reviews of an earlier draft provided by A. Dolloff, G. McWhirter, D. Shields, and S. Zarnoch. Discussions with A. Clingenpeel, D. Marion, S. Ross, and D. Shields were valuable in guiding data analysis and interpretation. We thank T. Fletcher, C. Harwell, D. Martinovic, G. McWhirter, A. Pabst, R. Reekstin, and L. Staton for conducting the field survey. H. Halverson, C. Harwell, G. McWhirter, and L. Staton were instrumental in entering, proofing, and summarizing data. G. Henderson provided assistance in data management and summaries, and C. Jenkins and F. McEwen provided administrative assistance. S. Krieger, C. Kilcrease (both Holly Springs National Forest), and A. Clingenpeel (Ouachita National Forest) graciously provided maps, global positioning tools, and other geographic information on streams in the area. G. Yeck (Holly Springs National Forest) provided staff time, vehicles, and other courtesies. We are extremely grateful to numerous private landowners that granted permission to survey streams on their property. The study was partially supported by a U.S. Department of Agriculture Forest Service Chief's Grant to the Holly Springs National Forest.

LITERATURE CITED

Angermeier, P.L.; Karr, J.R. 1984. Relationship between woody debris and fish habitat in a small warmwater stream. Transactions of the American Fisheries Society. 113: 716–726.

Angermeier, P.L.; Smogor, R.A. 1995. Estimating number of species and relative abundances in stream-fish communities: effects of sampling effort and discontinuous distributions. Canadian Journal of Fisheries and Aquatic Sciences. 52: 939–949.

Benke, A.C.; Henry, R.L., III; Gillespie, D.M.; Hunter, R.J. 1985. Importance of snag habitat for animal production in southeastern streams. Fisheries. 10(5): 8–13.

Benke, A.C.; Van Arsdall, T.C., Jr.; Gillespie, D.M.; Parrish, F.K. 1984. Invertebrate productivity in a subtropical blackwater river: the importance of snag habitat and life history. Ecological Monographs. 54: 25–63.

Chan, M.D.; Parsons, G.R. 2000. Aspects of brown madtom, *Noturus phaeus*, life history in northern Mississippi. Copeia. 2000: 757–762.

Cooper, C.M.; Knight, S.S. 1991. Water quality cycles in two hill land streams subjected to natural, municipal, and non-point agricultural stresses in the Yazoo Basin of Mississippi, USA (1985–1987). Organ der Internationalen Vereinigung fur Theoretische und Angewandte Limnologie. 24: 1654–1663.

Dendy, F.E.; Ursic, S.J.; Bowie, A.J. 1979. Sediment sources and yields from upland watersheds in north Mississippi. In: Proceedings of the Mississippi water resources conference; 1979; Mississippi State, MS. Mississippi State, MS: Mississippi State University, Water Resources Research Institute: 49–54.

Gregory, S.V.; Swanson, F.J.; McKee, W.A.; Cummins, K.W. 1991. An ecosystem perspective of riparian zones. BioScience. 41: 540–551.

Hedman, C.; Van Lear, D.H.; Swank, W.T. 1996. In-stream large woody debris loading and riparian forest seral stage associations in the Southern Appalachian Mountains. Canadian Journal of Forest Research. 26: 1218–1227.

Hilderbrand, R.H.; Lemly, A.D.; Dolloff, C.A.; Harpster, K.L. 1997. Effects of large woody debris placement on stream channels and benthic macroinvertebrates. Canadian Journal of Fisheries and Aquatic Sciences. 54: 931–939.

Jackson, D.A. 1993. Stopping rules in principal components analysis: a comparison of heuristic and statistical approaches. Ecology. 74: 2204–2214.

Jenkins, R.E.; Burkhead, N.M. 1994. Freshwater fishes of Virginia. Bethesda, MD: American Fisheries Society. 1,079 p.

Keys, J.E., Jr.; Carpenter, C.A.; Hooks, S. [and others]. 1995. Ecological units of the Eastern United States: first approximation. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region. 1: 3.500,000; presentation scale; color. [Map].

Lowrance, T.; Todd, R.; Fail, J., Jr. [and others]. 1984. Riparian forests as nutrient filters in agricultural watersheds. BioScience. 34: 374–377.

Lyons, J. 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. North American Journal of Fisheries Management. 12: 198–203.

Matthews, W.J. 1987. Physicochemical tolerance and selectivity of stream fishes as related to their geographic ranges and local distributions. In: Matthews, W.J.; Heins, D.C., eds. Community and evolutionary ecology of North American stream fishes. Norman, OK: University of Oklahoma Press: 111–120.

Monzyk, F.R.; Kelso, W.E.; Rutherford, D.A. 1997. Characteristics of woody cover used by brown madtoms and pirate perch in Coastal Plain streams. Transactions of the American Fisheries Society. 126: 665–675.

Osborne, L.L.; Wiley, M.J. 1992. Influence of tributary spatial position on the structure of warmwater fish communities. Canadian Journal of Fisheries and Aquatic Sciences. 49: 671–681.

Paller, M.H. 1995. Relationships among number of fish species sampled, reach length surveyed, and sampling effort in South Carolina Coastal Plain streams. North American Journal of Fisheries Management. 15: 110–120.

Schlosser, I.J. 1987. A conceptual framework for fish communities in small warmwater streams. In: Matthews, W.J.; Heins, D.C., eds. Community and evolutionary ecology of North American stream fishes. Norman, OK: University of Oklahoma Press: 17–24.

Schumm, S.A.; Harvey, M.D.; Watson, C.C. 1984. Incised channels: morphology, dynamics and control. Littleton, CO: Water Resources Publications. 200 p.

Shields, F.D., Jr.; Knight, S.S.; Cooper, C.M. 1994. Effects of channel incision on base flow stream habitats and fishes. Environmental Management. 18: 43–57.

Shields, F.D., Jr.; Knight, S.S.; Cooper, C.M. 1995. Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. Hydrobiologia. 312: 191–208.

Shields, F.D., Jr.; Knight, S.S.; Cooper, C.M. 1997. Rehabilitation of warmwater stream ecosystems following channel incision. Ecological Engineering. 8: 93–116.

Shields, F.D., Jr.; Knight, S.S.; Cooper, C.M. 1998. Rehabilitation of aquatic habitats in warmwater streams damaged by channel incision in Mississippi. Hydrobiologia. 382: 63–86.

Shields, F.D., Jr.; Smith, R.H. 1992. Effects of large woody debris removal on physical characteristics of a sand-bed river. Aquatic Conservation: Marine and Freshwater Ecosystems. 2: 145–163.

Simon, A. 1989. A model of channel response in disturbed alluvial streams. Water Resources Bulletin. 25: 1171–1188.

Smock, L.A.; Metzler, G.M.; Gladden, J.E. 1989. Role of debris dams in the structure and functioning of low-gradient headwater streams. Ecology. 70: 764–765.

Sokal, R.R.; Rohlf, R.J. 1995. Biometry. New York: W.H. Freeman and Co. 887 p.

Strahler, A.N. 1957. Quantitative analysis of watershed geomorphology. Transactions of the American Geophysical Union. 38: 913–920.

SAS Institute Inc. 1996. Statistical analysis software for windows. Version 6.12. Cary, NC: SAS Institute Inc.

U.S. Department of Agriculture, Forest Service. 1988. The Yazoo-Little Tallahatchie flood prevention project: a history of the Forest Service's role. For. Rep. R8–FR8. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region. 63 p.

Wallace, J.B.; Eggert, S.L.; Meyer, J.L.; Webster, J.B. 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. Science. 277: 102–104.

Wallerstein, N.; Thorne, C.R.; Doyle, M.W. 1997. Spatial distribution and impact of large woody debris in northern Mississippi. In: Wang, S.S.Y.; Langenoen, E.J.; Shields, F.D., Jr., eds. Proceedings of the conference on management of landscapes disturbed by channel incision; 1997 May 19–23; Oxford, MS. Oxford, MS: University of Mississippi, Center for Computational Hydroscience and Engineering: 145–150.

APPENDIX A

Localities for 79 stream reaches surveyed for fish and fish habitat in north-central Mississippi. An asterisk indicates the site was not used in the analyses because physical data were incomplete, no fishes were captured, or visits in September 1999 revealed the sampled reach was located in an intermittent stream. Abbreviations: trib. = tributary; CR = county road; FS = Forest Service road; quad. = U.S. Geological Survey 1:24,000 scale topographic map; DMS = degrees, minutes, and seconds; Lat. = latitude; Long. = longitude; DD = decimal degrees; T = township; R = range; S = section; and Q = quarter.

HATCHIE RIVER DRAINAGE

Unnamed trib. N. Br. Hurricane Creek, Muddy Creek trib., Tippah Co. Date: 7/26/99. CR 324 end, 3 km SE Camp Hill, 21 km NE Ashland. Walnut quad. Lat. (DMS): N34-56-14.2 Long. (DMS): W88-59-41.0 Lat. (DD): 34.9357 Long. (DD): -88.990167 T-2S, R-3E, S-4, Q-SW. Stream order: 2. Link magnitude: 4.

Unnamed trib. N. Br. Hurricane Creek, Muddy Creek trib., Tippah Co. Date: 7/26/99. FS 644A end, 3 km SE Camp Hill, 20 km NE Ashland. Walnut quad. Lat. (DMS): N34-55-41.8 Long. (DMS): W88-59-35.1 Lat. (DD): 34.923633 Long. (DD): -88.989183 T-2S, R-3E, S-9, Q-NW. Stream order: 3. Link magnitude: 14.

S. Br. Hurricane Creek, Muddy Creek trib., Tippah Co. Date: 7/27/99. CR 300 bridge, 8 km NNW Falkner, 7 km SE Camp Hill. Walnut quad. Lat. (DMS): N34-54-18.5 Long. (DMS): W88-57-51.4 Lat. (DD): 34.903083 Long. (DD): -88.958567 T-2S, R-3E, S-14, Q-SW. Stream order: 4. Link magnitude: 81.

West Prong Muddy Creek, Muddy Creek trib., Tippah Co. Date: 7/16/99. CR 414 bridge, 17.6 km E Ashland, 8.6 km NNW Ripley. Falkner quad. Lat. (DMS): N34-49-28.0 Long. (DMS): W88-58-59.0 Lat. (DD): 34.821333 Long. (DD): -88.9765 T-3S, R-3E, S-16, Q-NE. Stream order: 4. Link magnitude: 27.

*Unnamed trib. West Prong Muddy Creek, Muddy Creek trib., Tippah Co. Date: 7/16/99. CR 346, 15 km E Ashland, 14 km NNW Ripley. Whitten Town quad. Lat. (DMS): N34-50-55.6 Long. (DMS): W89-01-08.3 Lat. (DD): 34.8426 Long. (DD): -89.01805 T-3S, R-3E, S-6, Q-SE. Stream order: 1. Link magnitude: 1.

LITTLE TALLAHATCHIE RIVER DRAINAGE

*Bagley Creek, Lafayette Co. Date: 6/7/99. CR 244 (Riverside Rd.), 8 km E Abbeville, 19 km SW Potts Camp. Malone quad. Lat. (DMS): N34-30-08.3 Long. (DMS): W89-24-48.1 Lat. (DD): 34.501383 Long. (DD): -89.408017 T-7S, R-2W, S-4, Q-S. Stream order: 4. Link magnitude: 17.

Bagley Creek, Lafayette Co. Date: 6/7/99. Bagley Lake outflow, 8.7 km ESE Abbeville, 20.7 km SSW Potts Camp. Bagley quad. Lat. (DMS): N34-28-50.7 Long. (DMS): W89-24-19.5 Lat. (DD): 34.475117 Long. (DD): -89.40325 T-7S, R-2W, S-9, Q-SE. Stream order: 3. Link magnitude: 10.

Cypress Creek, Lafayette Co. Date: 6/1/99. CR 251 bridge, 6 km WNW Pleasantdale Church, 4 km SE Union Hill Church. Puskus quad. Lat. (DMS): N34-23-36.7 Long. (DMS): W89-17-12.9 Lat. (DD): 34.38945 Long. (DD): -89.285483 T-7S, R-1W, S-15, Q-NE. Stream order: 3. Link magnitude: 11.

Cypress Creek, Lafayette Co. Date: 6/4/99. 1.6 km upstream Goolsby Lake, 20.7 km E Oxford, 5.8 km NNW Lafayette Springs. Denmark quad. Lat. (DMS): N34-21-35.4 Long. (DMS): W89-18-15.3 Lat. (DD): 34.3559 Long. (DD): -89.30255 T-8S, R-1W, S-28, Q-N. Stream order: 2. Link magnitude: 2.

Cypress Creek, Lafayette Co. Date: 6/4/99. CR 252 bridge, 8 km NNW Lafayette Springs, 23.2 km E Oxford. Puskus quad. Lat. (DMS): N34-22-56.4 Long. (DMS): W89-17-54.3 Lat. (DD): 34.376067 Long. (DD): -89.292383 T-8S, R-1W, S-15, Q-SW. Stream order: 2. Link magnitude: 4.

Cypress Creek, Lafayette Co. Date: 5/28/99. CR 244 bridge, 3.6 km WNW Etta, 6.8 km S Cornersville. Puskus quad. Lat. (DMS): N34-28-37.7 Long. (DMS): W89-15-45.1 Lat. (DD): 34.47295 Long. (DD): -89.257517 T-6S, R-1W, S-13, Q-E. Stream order: 5. Link magnitude: 108.

West Cypress Creek, Cypress Creek trib., Lafayette Co. Date: 6/4/99. CR 251, 21.9 ENE Oxford, 12.1 km NNE Denmark. Puskus quad. Lat. (DMS): N34-24-31.5 Long. (DMS): W89-17-34.1 Lat. (DD): 34.40525 Long. (DD): -89.289017 T-8S, R-1W, S-3/10, Q-SW/NW. Stream order: 3. Link magnitude: 16.

West Cypress Creek, Cypress Creek trib., Lafayette Co. Date: 6/1/99. FS 844C, 18.4 km ENE Oxford, 9.2 km N Denmark. Puskus quad. Lat. (DMS): N34-23-29.2 Long. (DMS): W89-19-36.8 Lat. (DD): 34.3882 Long. (DD): -89.3228 T-8S, R-1W, S-17, Q-NW. Stream order: 3. Link magnitude: 8.

*West Cypress Creek, Cypress Creek trib., Lafayette Co. Date: 6/1/99. CR 252, 7.7 km N Denmark, 14.8 km E Oxford. Puskus quad. Lat. (DMS): N34-23-04.8 Long. (DMS): W89-20-06.7 Lat. (DD): 34.384133 Long. (DD): -89.33445 T-8S, R-1W, S-18, Q-SE. Stream order: 2. Link magnitude: 12.

*Unnamed trib. West Cypress Creek, Cypress Creek trib., Lafayette Co. Date: 6/3/99. FS 837E, 18.4 km ENE Oxford, 10.4 km N Denmark. Puskus quad. Lat. (DMS): N34-24-29.1 Long. (DMS): W89-20-11.3 Lat. (DD): 34.40485 Long. (DD): -89.335217 T-8S, R-1W, S-7, Q-NE. Stream order: 2. Link magnitude: 3.

Unnamed northern trib. W. Cypress Creek, Cypress Creek trib., Lafayette Co. Date: 6/3/99. FS 837E, 2.3 km E Keel, 10.9 km N Denmark. Puskus quad. Lat. (DMS): N34-24-27.4 Long. (DMS): W89-20-13.7 Lat. (DD): 34.404567 Long. (DD): -89.335617 T-8S, R-1W, S-6, Q-SE. Stream order: 1. Link magnitude: 1.

Puskus Creek, Cypress Creek trib., Lafayette Co. Date: 6/2/99. CR 237 (FS 830), 14.4 km ENE Oxford, 9.8 km NNW Denmark. Puskus quad. Lat. (DMS): N34-23-43.2 Long. (DMS): W89-22-20.7 Lat. (DD): 34.390533 Long. (DD): -89.370117 T-8S, R-2W, S-11, Q-SE. Stream order: 2. Link magnitude: 3.

Puskus Creek, Cypress Creek trib., Lafayette Co. Date: 6/2/99. CR 237 (FS 830), 14.4 km ENE Oxford, 9.8 km NNW Denmark. Puskus quad. Lat. (DMS): N34-23-43.7 Long. (DMS): W89-22-18.3 Lat. (DD): 34.390617 Long. (DD): -89.369717 T-8S, R-2W, S-11, Q-SE. Stream order: 3. Link magnitude: 7.

Puskus Creek, Cypress Creek trib., Lafayette Co. Date: 7/28/99. CR 245, 17 km NE Oxford, 15 km N Denmark. Puskus quad. Lat. (DMS): N34-26-46.7 Long. (DMS): W89-19-58.3 Lat. (DD): 34.441117 Long. (DD): -89.326383 T-7S, R-1W, S-29, Q-NW. Stream order: 4. Link magnitude: 37.

Puskus Creek, Cypress Creek trib., Lafayette Co. Date: 8/3/99. FS 832-C, 0.8 km downstream Puskus Lake, 16.2 km NNE Oxford, 14 km SSW Cornersville. Puskus quad. Lat. (DMS): N34-26-41.0 Long. (DMS): W89-20-14.8 Lat. (DD): 34.440167 Long. (DD): -89.3358 T-7S, R-1W, S-30, Q-SE. Stream order: 4. Link magnitude: 34.

Bay Springs Branch, Puskus Creek trib., Lafayette Co. Date: 9/14/99. University Mississippi Field Station, 10.2 km NE Oxford, 12 km SE Abbeville. Bagley quad. Lat. (DMS): N34-25-34.1 Long. (DMS): W89-23-44.3 Lat. (DD): 34.42235 Long. (DD): -89.390717 T-7S, R-2W, S-34, Q-SW. Stream order: 1. Link magnitude: 1.

Unnamed trib. Puskus Creek, Cypress Creek trib., Lafayette Co. Date: 7/28/99. CR 832, 1 km N Puskus Lake, 16 km N Denmark. Puskus quad. Lat. (DMS): N34-27-02.6 Long. (DMS): W89-21-00.2 Lat. (DD): 34.450433 Long. (DD): -89.350033 T-7S, R-1W, S-30, Q-NW. Stream order: 2. Link magnitude: 2.

*Little Tallahatchie Canal, Marshall-Lafayette Co. Date: 8/10/99. Riverside recreation site, 5.8 km SSW Bethlehem, 10.7 km SW Cornersville, Bethlehem quad. Lat. (DMS): N34-31-45.6 Long. (DMS): W89-21-58.7 Lat. (DD): 34.524267 Long. (DD): -89.359783 T-6S, R-2W, S-25, Q-SW. Stream order: ND. Link magnitude: ND.

*Little Tallahatchie Canal, Union Co. Date: 8/9/99. downstream Hwy 30 bridge at Etta. Etta quad. Lat. (DMS):

N34-28-55.2 Long. (DMS): W89-13-29.5 Lat. (DD): 34.475867 Long. (DD): -89.221583 T-7S, R-1E, S-8, Q-SW. Stream order: ND. Link magnitude: ND.

*Little Tallahatchie R., Lafayette Co. Date: 8/12/99. Old channel at Riverside Recreation Area, 6.5 km SSW Bethlehem, 10.7 km SW Cornersville. Bethlehem quad. Lat. (DMS): N34-31-11.3 Long. (DMS): W89-21-48.3 Lat. (DD): 34.51855 Long. (DD): -89.35805 T-6S, R-2W, S-36, Q-NW. Stream order: ND. Link magnitude: ND.

*Little Tallahatchie R., Lafayette Co. Date: 9/15/99. 1.2 km downstream Bakers Field boat ramp, 10.8 km ESE Malone, 10.8 km ENE Abbeville. Malone quad. Lat. (DMS): N34-31-11.2 Long. (DMS): W89-22-56.2 Lat. (DD): 34.518533 Long. (DD): -89.376033 T-6S, R-2W, S-35/34, Q-NW/NE. Stream order: ND. Link magnitude: ND.

*Little Tallahatchie R., Lafayette Co. Date: 9/17/99. Old channel, north 'Hudson's Deer Camp', 7 m downstream trail crossing, 8.8 km SSE Bethlehem, 4 km SSW Cornersville. Bethlehem quad. Lat. (DMS): N34-30-20.5 Long. (DMS): W89-16-16.7 Lat. (DD): 34.503417 Long. (DD): -89.26945 T-7S, R-1W, S-2, Q-N. Stream order: ND. Link magnitude: ND.

Lee Creek, Lafayette Co. Date: 8/5/99. 40 m upstream and downstream CR 210 bridge, 14.5 km SSW Bethlehem, 6 km SE Abbeville. Bagley quad. Lat. (DMS): N34-28-25.2 Long. (DMS): W89-26-46.1 Lat. (DD): 34.470867 Long. (DD): -89.441017 T-7S, R-2W, S-18, Q-E. Stream order: 4. Link magnitude: 34.

Lee Creek, Lafayette Co. Date: 8/5/99. 10 m upstream CR 291 bridge, 2.8 km E Abbeville, 14 km SSW Bethlehem. Bagley quad. Lat. (DMS): N34-29-52.1 Long. (DMS): W89-27-26.0 Lat. (DD): 34.492017 Long. (DD): -89.454333 T-7S, R-2W, S-6, Q-SW. Stream order: 4. Link magnitude: 48.

Mitchell Creek, Union Co. Date: 8/2/99. 5 m upstream Hwy 30 bridge, 7.9 km SSW Cornersville, 6.1 km NW Enterprise. Etta quad. Lat. (DMS): N34-28/56.9 Long. (DMS): W89-12-09.7 Lat. (DD): 34.47615 Long. (DD): -89.201617 T-7S, R-1E, S-9, Q-SE. Stream order: 4. Link magnitude: 87.

Mitchell Creek, Union Co. Date: 8/2/99. 5 m upstream CR 12 bridge, 27 km ENE Abbeville, 9.1 km SSW Myrtle. Hickory Flat quad. Lat. (DMS): N34-31-14.1 Long. (DMS): W89-12-09.7 Lat. (DD): 34.519017 Long. (DD): -89.201617 T-6S, R-1E, S-33, Q-Center. Stream order: 3. Link magnitude: 8.

TIPPAH RIVER DRAINAGE

Big Snow Creek, Benton Co. Date: 7/13/99. 10 m upstream CR 607 bridge, 10 km SW Ashland, 17.8 km E Holly Springs. Holly Springs SE quad. Lat. (DMS): N34-46-22.4 Long. (DMS): W89-15-08.3 Lat. (DD): 34.7704 Long. (DD): -89.251383 T-3S, R-1W, S-36, Q-SE. Stream order: 5. Link magnitude: 67.

Big Snow Creek, Benton Co. Date: 7/19/99. End FS Rd. 657-D, 20.3 km ESE Holly Springs, 13.7 km NNW Hickory Flat. Chilli Creek quad. Lat. (DMS): N34-43-51.6 Long. (DMS): W89-14-13.3 Lat. (DD): 34.725267 Long. (DD): -89.23555 T-4S, R-1E, S-18, Q-SE. Stream order: 5. Link magnitude: 95.

Wagner Creek, Big Snow Creek trib., Benton Co. Date: 7/12/99. 5 m downstream CR 652 bridge, 6.4 km SSW Ashland, 21.6 km ENE Holly Springs. Ashland quad. Lat. (DMS): N34-47-26.5 Long. (DMS): W89-12-53.6 Lat. (DD): 34.78775 Long. (DD): -89.208933 T-3S, R-1E, S-29, Q-SE. Stream order: 4. Link magnitude: 11.

Wagner Creek, Big Snow Creek trib., Benton Co. Date: 7/12/99. 15 m upstream CR 607 bridge, 8.9 km SSW Ashland, 20.1 km ESE Holly Springs. Ashland quad. Lat. (DMS): N34-46-05.5 Long. (DMS): W89-13-46.1 Lat. (DD): 34.767583 Long. (DD): -89.22435 T-4S, R-1E, S-6/5, Q-NE/NW. Stream order: 4. Link magnitude: 41.

Big Spring Creek, Marshall Co. Date: 7/29/99. 40 m upstream CR 633 bridge, 8.5 km WSW Potts Camp, 14.5 km SSE Holly Springs. Waterford quad. Lat. (DMS): N34-37-58.2 Long. (DMS): W89-23-49.0 Lat. (DD): 34.626367 Long. (DD): -89.3915 T-5S, R-2W, S-22, Q-NW. Stream order: 5. Link magnitude: 206.

Big Spring Creek, Marshall Co. Date: 8/3/99. Upstream CR 694 bridge, channel runs parallel to road, 8.3 km NNE Abbeville, 9.9 km WSW Bethlehem. Malone quad. Lat. (DMS): N34-33-28.3 Long. (DMS): W89-25-49.6 Lat. (DD): 34.554717 Long. (DD): -89.424933 T-6S, R-W, S-17, Q-SE. Stream order: 5. Link magnitude: 314.

Unnamed trib. Big Spring Creek, Marshall Co. Date: 7/30/99. Musgray Rd off CR 633, upstream and downstream bridge, 9.3 km ENE Potts Camp, 11 km S Holly Springs. Waterford quad. Lat. (DMS): N34-39-48.4 Long. (DMS): W89-24-46.1 Lat. (DD): 34.658067 Long. (DD): -89.407683 T-5S, R-2W, S-9, Q-NW. Stream order: 4. Link magnitude: 71.

Chilli Creek, Benton Co. Date: 7/1/99. 10 m upstream CR 625 bridge, 15 km SSW Ashland, 9.7 km NNW Hickory Flat. Chilli Creek quad. Lat. (DMS): N34-42-10.2 Long. (DMS): W89-12-08.3 Lat. (DD): 34.7017 Long. (DD): -89.201383 T-4S, R-1E, S-28, Q-S. Stream order: 5. Link magnitude: 91.

Chewalla Creek, Marshall Co. Date: 6/16/99. 25 m upstream Hwy 4 bridge, 5 km S Hudsonville, 8.9 km NE Holly Springs. Holly Springs SE quad. Lat. (DMS): N34-41-46.8 Long. (DMS): W89-22-03.4 Lat. (DD): 34.691133 Long. (DD): -89.367233 T-3S, R-2W, S-24/13, Q-NW/SW. Stream order: 3. Link magnitude: 10.

Chewalla Creek, Marshall Co. Date: 6/17/99. 15 m upstream bridge Lacy Ivy Rd. bridge, 3.9 km SE Lake Center, 4.5 km NNW Potts Camp. Potts Camp quad. Lat. (DMS): N34-40-44.0 Long. (DMS): W89-19-56.3 Lat. (DD): 34.674 Long. (DD): -89.32605 T-5S, R-1W, S-5, Q-NW. Stream order: 5. Link magnitude: 189.

Chewalla Creek, Marshall Co. Date: 6/21/99. FS Rd. 661-A end, upstream and downstream, 3.8 km W Lake Center, 6.8 km NNW Potts Camp. Potts Camp quad. Lat. (DMS): N34-41-52.5 Long. (DMS): W89-19-49.0 Lat. (DD): 34.692083 Long. (DD): -89.324833 T-4S, R-1W, S-29/32, Q-SW/NW. Stream order: 5. Link magnitude: 150.

Chewalla Creek, Marshall Co. Date: 6/14/99. 100 m downstream Chewalla Lake outflow, 8.8 km NNW Potts

Camp, 10.6 km SE Holly Springs. Potts Camp quad. Lat. (DMS): N34-43-23.3 Long. (DMS): W89-20-27.8 Lat. (DD): 34.72055 Long. (DD): -89.337967 T-4S, R-1W, S-19, Q-NE. Stream order: 4. Link magnitude: 90.

Unnamed trib. Chewalla Creek, Marshall Co. Date: 7/8/99. 10 m upstream CR 611, 13.9 km ESE Holly Springs, 16.8 km NW Hickory Flat. Potts Camp quad. Lat. (DMS): N34-43-58.8 Long. (DMS): W89-18-12.1 Lat. (DD): 34.726467 Long. (DD): -89.302017 T-4S, R-1W, S-16, Q-SE. Stream order: 3. Link magnitude: 11.

Chilli Creek, Benton Co. Date: 6/30/99. 30 m downstream North Chilli Lake outflow. Chilli Creek quad. Lat. (DMS): N34-42-00.5 Long. (DMS): W89-08-23.2 Lat. (DD): 34.700083 Long. (DD): -89.1372 T-4S, R-2E, S-30, Q-SW. Stream order: 3. Link magnitude: 8.

East Chilli Creek, Chilli Creek trib., Benton Co. Date: 6/30/99. CR 626E downstream Chilli Lake, 40 m upstream confluence southern trib. Chilli Creek quad. Lat. (DMS): N34-41-46.0 Long. (DMS): W89-08-52.4 Lat. (DD): 34.691 Long. (DD): -89.142067 T-4S, R-1E, S-36, Q-NE. Stream order: 3. Link magnitude: 12.

Curtis Creek, Benton Co. Date: 7/22/99. FS Rd. 609A, 30 m downstream Curtis Lake outflow, 8.5 km SE Ashland, 7 km S Yellow Rabbit Lake. Whitten Town quad. Lat. (DMS): N34-46-44.5 Long. (DMS): W89-06-38.1 Lat. (DD): 34.774083 Long. (DD): -89.10635 T-3S, R-2E, S-32, Q-NE. Stream order: 3. Link magnitude: 2.

Oaklimeter Creek, Benton Co. Date: 6/29/99. 40 m downstream Wood Duck Lake outflow, 7.8 km SSE Bethel, 18 km E Potts Camp. Blue Mountain quad. Lat. (DMS): N34-39-59.0 Long. (DMS): W89-06-19.4 Lat. (DD): 34.659833 Long. (DD): -89.103233 T-5S, R-2E, S-8/9, Q-NE/NW. Stream order: 3. Link magnitude: 12.

Oaklimeter Creek, Benton Co. Date: 6/29/99. 15 m upstream CR 626 bridge, 7.1 km SSE Bethel, 18 km E Potts Camp. Blue Mountain quad. Lat. (DMS): N34-39-39.8 Long. (DMS): W89-06-28.1 Lat. (DD): 34.656633 Long. (DD): -89.104683 T-5S, R-2E, S-8, Q-SE. Stream order: 4. Link magnitude: 21.

Oaklimeter Creek, Benton Co. Date: 7/6/99. 15 m upstream CR 640 bridge, 8.1 km WNW Hickory Flat, 3.7 km SE Potts Camp. Potts Camp quad. Lat. (DMS): N34-37-42.5 Long. (DMS): W89-16-27.3 Lat. (DD): 34.62375 Long. (DD): -89.271217 T-5S, R-1W, S-23, Q-S. Stream order: 5. Link magnitude: 277.

Pechahallee Creek, Oaklimeter Creek trib., Benton Co. Date: 6/23/99. 12 m upstream FS 692 bridge, 5 km W Hickory Flat, 6.5 km SE Potts Camp. Hickory Flat. quad. Lat. (DMS): N34-37-12.1 Long. (DMS): W89-14-24.8 Lat. (DD): 34.618683 Long. (DD): -89.237467 T-5S, R-1E, S-30, Q-NW. Stream order: 4. Link magnitude: 31.

*Pechahallee Creek, Oaklimeter Creek trib., Benton Co. Date: 6/25/99. 30 m downstream Hwy 5 bridge, 4.8 km N Hickory Flat, 8 km SSE Bethel. Chilli Creek quad. Lat. (DMS): N34-39-39.1 Long. (DMS): W89-11-03.5 Lat. (DD): 34.656517 Long. (DD): -89.183917 T-5S, R-1E, S-10, Q-S. Stream order: 1. Link magnitude: 1.

Pechahallee Creek, Oaklimeter Creek trib., Benton Co. Date: 7/2/99. 15 m upstream CR 625 bridge, 4 km NW Hickory Flat, 8.3 km E Potts Camp. Chilli Creek quad. Lat. (DMS): N34-38-21.2 Long. (DMS): W89-13-05.3 Lat. (DD): 34.636867 Long. (DD): -89.21755 T-5S, R-1E, S-20, Q-NE. Stream order: 4. Link magnitude: 21.

Unnamed trib. Oaklimeter Creek, Benton Co. Date: 6/25/99. 10 m upstream box culvert opening on north side Hwy 78, 9.3 km ESE Potts Camp, 2.5 km NNW Hickory Flat. Chilli Creek quad. Lat. (DMS): N34-37-43.9 Long. (DMS): W89-12-29.4 Lat. (DD): 34.623983 Long. (DD): -89.2049 T-5S, R-1E, S-21, Q-SW. Stream order: 2. Link magnitude: 3.

*Unnamed trib. Oaklimeter Creek, Benton Co. Date: 7/7/99. 75 m upstream CR 662, 4.3 km E Potts Camp, 7.4 km WNW Hickory Flat. Potts Camp quad. Lat. (DMS): N34-38-20.9 Long. (DMS): W89-15-45.6 Lat. (DD): 34.636817 Long. (DD): -89.2576 T-5S, R-1W, S-24, Q-NW. Stream order: 1. Link magnitude: 1.

*Potts Creek, Benton Co. Date: 6/22/99. 3 m downstream CR 638, downstream Brents Lake, 6.4 km NNW Cornersville, 7 km SSE Potts Camp. Bethlehem quad. Lat. (DMS): N34-35-13.2 Long. (DMS): W89-16-46.1 Lat. (DD): 34.585533 Long. (DD): -89.27435 T-6S, R-1W, S-2, Q-SW. Stream order: 4. Link magnitude: 40.

Potts Creek, Marshall Co. Date: 6/23/99. 15 m downstream Mills Rd. bridge, 2.5 km NNW Bethlehem, 7 km SSW Potts Camp. Bethlehem quad. Lat. (DMS): N34-35-30.6 Long. (DMS): W89-20-32.5 Lat. (DD): 34.588433 Long. (DD): -89.33875 T-6S, R-1W, S-6, Q-NW. Stream order: 5. Link magnitude: 142.

Unnamed trib. Potts Creek, Benton Co. Date: 6/22/99. 7 m upstream 347 bridge, 4.4 km NE Bethlehem, 10 km W Hickory Flat. Bethlehem quad. Lat. (DMS): N34-36-13.2 Long. (DMS): W89-18-55.6 Lat. (DD): 34.6022 Long. (DD): -89.309267 T-5S, R-1W, S-33, Q-SW. Stream order: 4. Link magnitude: 21.

Unnamed trib. Tippah R., Benton Co. Date: 6/17/99. 20 m downstream CR 657 bridge, 7.2 km NNE Potts Camp, 10.8 km W Bethel. Potts Camp quad. Lat. (DMS): N34-42-32.7 Long. (DMS): W89-15-16.5 Lat. (DD): 34.70545 Long. (DD): -89.25275 T-4S, R-1W, S-25, Q-NE. Stream order: 4. Link magnitude: 26.

Unnamed trib. Tippah R., Benton Co. Date: 6/17/99. 110 m upstream CR 606, Gandy Property, 4.4 km NNE Potts Camp, 7.3 km E Pine Grove. Potts Camp quad. Lat. (DMS): N34-40-50.6 Long. (DMS): W89-16-48.3 Lat. (DD): 34.6751 Long. (DD): -89.274717 T-5S, R-1W, S-2, Q-NE. Stream order: 4. Link magnitude: 36.

*Unnamed trib. Tippah R., Benton Co. Date: 7/9/99. 8 m upstream trail crossing and upstream confluence, unnumbered FS rd., 1.7 km NE Potts Camp. Potts Camp quad. Lat. (DMS): N34-39-35.4 Long. (DMS): W89-17-08.3

Lat. (DD): 34.6559 Long. (DD): -89.284717 T-5S, R-1W, S-10, Q-SE. Stream order: 3. Link magnitude: 4.

Unnamed trib. Tippah R., Benton Co. Date: 7/9/99. 5 m downstream confluence, 1.7 km NE Potts Camp. Potts Camp quad. Lat. (DMS): N34-39-36.5 Long. (DMS): W89-17-10.9 Lat. (DD): 34.656083 Long. (DD): -89.28515 T-5S, R-1W, S-10, Q-SE. Stream order: 3. Link magnitude: 7.

Unnamed trib. Tippah R., Benton Co. Date: 7/7/99. FS Rd. 606-L, 30 m downstream Cox Lake outflow, 5.3 km NE Potts Camp, 17.1 km SE Holly Springs. Potts Camp quad. Lat. (DMS): N34-41-28.3 Long. (DMS): W89-17-24.5 Lat. (DD): 34.68805 Long. (DD): -89.287417 T-4S, R-1W, S-34, Q-S. Stream order: 3. Link magnitude: 25.

Yellow Rabbit Creek, Benton Co. Date: 7/21/99. 3 m downstream CR 649 bridge, 6.3 km ESE Ashland, 16.4 km WNW Ripley. Whitten Town quad. Lat. (DMS): N34-49.09.3 Long. (DMS): W89-06-19.0 Lat. (DD): 34.818217 Long. (DD): -89.103167 T-3S, R-2E, S-17/16, Q-SE/SW. Stream order: 3. Link magnitude: 8.

Yellow Rabbit Creek, Benton Co. Date: 7/22/99. 80 m upstream CR 648 bridge, 7 km SSE Ashland, 15 km WNW Ripley. Ashland quad. Lat. (DMS): N34-46-25.8 Long. (DMS): W89-08-41.7 Lat. (DD): 34.770967 Long. (DD): -89.140283 T-3S, R-1E, S-36, Q-SE. Stream order: 4. Link magnitude: 38.

WOLF RIVER DRAINAGE

Indian Creek, Benton Co. Date: 7/14/99. 5 m downstream CR 646 (Blackjack Rd.), 9 km NE Sahland, 5.5 km NNE Yellow Rabbit Lake. Whitten Town quad. Lat. (DMS): N34-51-46.2 Long. (DMS): W89-05-08.0 Lat. (DD): 34.8577 Long. (DD): -89.084667 T-2S, R-2E, S-34, Q-SW. Stream order: 3. Link magnitude: 9.

Sourwood Creek, Benton Co. Date: 7/15/99. 10 m upstream CR 647 bridge, 9.3 km SSE Canaan, 17.8 km NW Benton. Camp Hill quad. Lat. (DMS): N34-52-39.4 Long. (DMS): W89-04-11.3 Lat. (DD): 34.873233 Long. (DD): -89.06855 T-2S, R-2E, S-27/26, Q-SE/SW. Stream order: 3. Link magnitude: 7.

Sourwood Creek, Benton Co. Date: 7/15/99. 8 m downstream CR 646 bridge, 7.3 km SE Canaan, 19.3 km NNW Ripley. Camp Hill quad. Lat. (DMS): N34-53-17.1 Long. (DMS): W89-04-49.4 Lat. (DD): 34.886183 Long. (DD): -89.0749 T-2S, R-2E, S-22/27, Q-NE. Stream order: 3. Link magnitude: 17.

Turkey Creek, Benton Co. Date: 8/11/99. Blackburn Rd off CR 647, 2 m upstream bridge, 12.5 km NNE Ashland, 19.2 km NNW Ripley. Camp Hill quad. Lat. (DMS): N34-53-47.0 Long. (DMS): W89-03-36.6 Lat. (DD): 34.891167 Long. (DD): -89.0561 T-2S, R-2E, S-23, Q-NE. Stream order: 3. Link magnitude: 30.

YOCONA RIVER DRAINAGE

Kettle Creek, Lafayette Co. Date: 6/10/99. 80 m downstream Drewery Lake outflow, 17.3 km E Oxford, 4 km N Denmark. Denmark quad. Lat. (DMS): N34-20-41.8 Long. (DMS): W89-20-12.3 Lat. (DD): 34.3403 Long. (DD): -89.335383 T-8S, R-1W, S-31, Q-NE. Stream order: 2. Link magnitude: 2.

Kettle Creek, Lafayette Co. Date: 6/10/99. 120 m upstream CR 277 bridge, 18.1 km ESE Oxford, 4 km ENE Denmark. Denmark quad. Lat. (DMS): N34-20-07.1 Long. (DMS): W89-19-53.5 Lat. (DD): 34.334517 Long. (DD): -89.325583 T-8S, R-1W, S-32, Q-SW. Stream order: 3. Link magnitude: 4.

Kettle Creek, Lafayette Co. Date: 6/8/99. 10 m upstream Hwy 6 bridge, 18.4 km ESE Oxford, 1.7 km ENE Denmark. Denmark quad. Lat. (DMS): N34-18-48.6 Long. (DMS): W89-19-42.1 Lat. (DD): 34.3081 Long. (DD): -89.323683 T-9S, R-1W, S-8, Q-NW. Stream order: 3. Link magnitude: 9.

Pumpkin Creek, Lafayette Co. Date: 6/11/99. 10 m upstream Hwy 6 bridge, 12 km ESE Oxford, 5.7 km NNE Yocona. Yocona quad. Lat. (DMS): N34-19-37.5 Long. (DMS): W89-23-51.3 Lat. (DD): 34.322917 Long. (DD): -89.391883 T-9S, R-2W, S-3, Q-NW. Stream order: 3. Link magnitude: 11.

Pumpkin Creek, Lafayette Co. Date: 6/11/99. Past end CR 266, 2.2 km upstream Hwy 6 bridge, 7.4 km ESE Oxford, 7.4 km N Yocona. Yocona quad. Lat. (DMS): N34-20-20.8 Long. (DMS): W89-23-02.2 Lat. (DD): 34.3368 Long. (DD): -89.3837 T-8S, R-2W, S-35, Q-NW. Stream order: 3. Link magnitude: 6.

Yellow Leaf Creek, Lafayette Co. Date: 6/8/99. 40 m upstream CR 225, 6.7 km ESE Oxford, 10.4 km NNW Yocona. Yocona quad. Lat. (DMS): N34-22-04.4 Long. (DMS): W89-25-44.1 Lat. (DD): 34.3674 Long. (DD): -89.424017 T-8S, R-2W, S-20, Q-SE. Stream order: 3. Link magnitude: 9.

Yellow Leaf Creek, Lafayette Co. Date: 6/9/99. Deloach property, 10 m upstream from trib. confluence, 8.2 km E Oxford, 16.3 km SE Abbeville. Bagley quad. Lat. (DMS): N34-22-32 Long. (DMS): W89-25-17 Lat. (DD): 34.372 Long. (DD): -89.4195 T-8S, R-2W, S-20/21, Q-NE/NW. Stream order: 2. Link magnitude: 2.

Yellow Leaf Creek, Lafayette Co. Date: 6/9/99. Deloach property, 100 m downstream from trib confluence, 8.2 km E Oxford, 16.2 km SE Abbeville. Yocona quad. Lat. (DMS): N34-22-27 Long. (DMS): W89-25-17 Lat. (DD): 34.371167 Long. (DD): -89.4195 T-8S, R-2W, S-20/21, Q-NE/NW. Stream order: 3. Link magnitude: 8.

Yellow Leaf Creek, Lafayette Co. Date: 6/9/99. Deloach property, 1.2 km upstream from trib. confluence, 8.7 km E Oxford, 16.3 km SE Abbeville. Bagley quad. Lat. (DMS): N34-22-48 Long. (DMS): W89-24-38 Lat. (DD): 34.374667 Long. (DD): -89.406333 T-8S, R-2W, S-16, Q-SE. Stream order: 3. Link magnitude: 6.

Yellow Leaf Creek, Lafayette Co. Date: 7/21/99. FS 849E, 5 m downstream confluence, 17.8 km SW Etta, 17 km SE Abbeville. Bagley quad. Lat. (DMS): N34-23-01.1 Long. (DMS): W89-24-06.9 Lat. (DD): 34.383517 Long. (DD): -89.40115 T-8S, R-2W, S-15, Q-W. Stream order: 2. Link magnitude: 2.

ORIGINS, FATES, AND RAMIFICATIONS OF NATURAL ORGANIC COMPOUNDS OF WETLANDS

Robert G. Wetzel[1]

Abstract—Much of the organic carbon for heterotrophic metabolism in aquatic ecosystems is soluble and derived from structural compounds of higher plants of terrestrial and wetland-littoral sources of both lake and river ecosystems. The chemical recalcitrance of this organic matter and its oxidative utilization are fundamentally different from many sources within the aquatic ecosystems. Within the lake or river, complex physical interactions occur that can greatly modify rates of utilization and biochemical reactions. Natural photolysis by photosynthetically active radiation, UV-A and UV-B, can result in the partial degradation of these macromolecules, reactivate complexed enzymes, and generate simple organic compounds and nutrients. A portion of the dissolved organic matter is photolytically degraded completely to CO_2. The chemical recalcitrance of these organic compounds (a) represents a fundamental, often dominant, subsidy of organic matter that drives metabolism in fresh waters and (b) is an essential aspect of metabolic stability in aquatic ecosystems. Climatic changes affect this metabolic stability in both positive and negative ways but, generally, will increase instabilities.

INTRODUCTION

The land-water interface region of aquatic ecosystems is always the most productive per-unit area along the gradient from land to open water. Because most aquatic ecosystems occur in terrain of gentle slopes and are small (mean area < 100 ha) and shallow (mean < 5 m), the wetland-littoral components usually dominate in productivity and the synthesis of organic matter (Wetzel 1990, 1992). Because of the predominance of small, shallow freshwater bodies, most dissolved organic carbon (DOC) of lacustrine and riverine ecosystems is derived from photosynthesis of higher plants and microflora associated with detritus, including sediments, and is only augmented by releases of organic matter produced by phytoplankton.

The dissolved organic compounds generated by the production and partial degradation of organic matter produced in terrestrial soils, wetland, and littoral interface regions move downgradient toward the open-water regions of the lake or river. En route, partial utilization by largely attached microorganisms effects a selective decomposition of more labile organic compounds and an increase in organic substrate recalcitrance.

Dissolved and colloidal organic matter of inland waters can be separated on the basis of many criteria. Although separated from particulate organic matter by filtration at a size of 0.5 mm, for an array of practical reasons (Wetzel and Likens 2000), dissolved organic matter is often further separated by molecular-sized categories (Cabaniss and others 2000, Gustafsson and Gschwend 1997). The molecular weight of humic substances influences their proton and metal-binding characteristics, partitioning behavior, and adsorption properties. Average molecular weight distributions of aquatic fulvic acids vary from 2.7 to 3.0 kD and include low-MW fractions of 1,110 D and a high-MW fraction of ca. 4,890 D. Less that 10 percent of dissolved humic substances were found to be greater than 5 kD. The low-MW fractions are more hydrophilic, exhibit faster diffusion and increased mobility, and are commonly more bioavailable. In contrast, the high-MW components have increased uptake of hydrophobic organic compounds, greater aromatic components, decreased mobility and diffusion, and are usually appreciably less susceptible to biological utilization. Molecular-sized fractionation helps little, however, in relation to bioavailability of different compounds, particularly above the size of effective cellular membrane permeability of ca. 500 D. Bonding structure, particularly of aromatic components, and availability of sites for enzyme hydrolysis are of greater importance. The present study addresses aspects that alter bioavailability of these humic compounds as they are imported to lakes and rivers.

Because of the relatively high utilization and turnover rates of simple, low molecular weight compounds by microbes, as much as 80 percent of dissolved organic matter in inland waters is composed of organic acids. Many of these compounds originate from higher aquatic and terrestrial plants. Of these organic acids, some 30 to 40 percent are composed of aromatic carbon compounds originating from structural plant tissues (Malcolm 1990).

Because of their large size and limited accessibility of large portions of these molecules to enzymatic hydrolysis, degradation rates of many of these commonly acidic macromolecules is slow. As a result, these heterogeneous compounds exhibit long turnover times with relatively long environmental residence times. Dissolved macromolecules are often of considerable age (months) but are mixed with variable and rapidly changing inputs of younger humic and nonhumic substances. Recent studies indicated that humic substances, particularly relatively recalcitrant fulvic acids, are also generated by algae and decomposing

[1] Bishop Professor of Biological Sciences, University of Alabama, Tuscaloosa, AL 35487-0206, Present address: Professor, Environmental Sciences and Engineering, The University of North Carolina, Chapel Hill, NC 27599-7431.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

microorganisms, and these contribute to the multitude of diverse compounds of the composite dissolved organic matter. This latter source is particularly important in wetland and littoral areas, where attached algal productivity is often many orders of magnitude greater than that of phytoplankton, e.g., review of Wetzel 1996.

Early detailed studies of fluxes of organic carbon in lake ecosystems indicated both that dissolved organic matter did not accumulate or precipitate, and that large quantities of CO_2 evaded to the atmosphere, e.g., Cole and others 1994, Kling and others 1992, Otsuki and Wetzel 1974, Wetzel 1995). The CO_2 evasion of respiratory origins within the lakes was always greatly in excess of autochthonous photosynthetic organic carbon production by phytoplankton. In spite of this evidence, for decades in aquatic ecology, the apparent chemical recalcitrance of fulvic and humic hydrophobic organic substances that dominated the instantaneous bulk DOC of standing and running waters was believed to result in poor utilization by microbiota. Loss rates are slow but consistently in the range of 0.5 to 2 percent per day under many different environmental conditions and, often, are faster.

A number of studies have demonstrated that ultraviolet irradiance (UV) can photolyse portions of proteinaceous and humic macromolecules, e.g., Moran 2000, reviews of Moran and Zepp 1997, Wetzel and others 1995. The resulting dissolved organic substances generated by photolysis resulted in immediate stimulation of and sustained bacterial growth, e.g., Lindall and others 1995; Moran and Zepp 1997; Stewart and Wetzel 1981, 1982; Wetzel and others 1995. Chemical analyses of these phytolytic transformations indicated that small organic fractions, particularly numerous small fatty acids that serve as excellent bacterial substrates, were generated by partial photolysis of the humic substances.

In the present study, dissolved organic matter leached from decomposing foliage of several wetland and floodplain plants was examined at intervals during their individual decomposition over an annual period. The dissolved organic matter was exposed to different durations of spectral fractions of photosynthetically active radiation, UV-A, and UV-B of natural sunlight. Photolytic products of organic matter and CO_2 were examined in relation to bacterial growth on these products under standardized growth conditions with natural wetland bacterial consortia.

MATERIALS AND METHODS

Dissolved organic compounds were isolated from whole leachates from decomposing foliage of emergent littoral aquatic plants and floodplain trees. These plants are dominant species in the Talladega Wetland Ecosystem (TWE) of the Talladega National Forest, Hale County, AL (Mann and Wetzel 1995, 2000). Emergent aquatic plants included: (a) the common cattail (*Typha latifolia* L.), which exhibits four or more cohorts per year (Dickerman and Wetzel 1985) and much standing dead tissue that undergoes considerable leaching of DOC; and, (b) the spike rush (*Juncus effusus* L.), a dominant within the TWE that exhibits at least seven cohorts per year at this latitude and contains much standing dead tissue that is constantly leaching DOC (Mann and Wetzel 1996, Wetzel and Howe 1999). Floating leaves of the common water lily (*Nymphaea odorata* Aiton) served as a representative of a rapidly decaying species that exhibits a leaf turnover rate of about every 30 days throughout the active growing season (March through November) at this latitude [Carter, S.; Ward, G.M.; Wetzel, R.G.; Benke, A.C. Growth, production, and senescence of *Nymphaea odorata* Aiton in a southeastern (U.S.A.) wetland. Manuscript in review. Aquatic Botany]. Leaves of the common semiaquatic alder [*Alnus serrulata* (Ait.) Willd.] were also collected during late summer and autumnal abscission. Alder is well known for rapid leaching of DOC and fast rates of decomposition in streams. Leaves of the riverine red oak (*Quercus rubra* L.) are retained for long periods after senescence, and usually abscise in spring (February at this latitude) after lengthy periods of slow leaching of DOC during the autumnal and winter rainy period.

Combined fresh and partly senesced culms and leaves of the two emergent and the floating-leaved aquatic angiosperms and mature leaves of the semiaquatic alder and riparian oak were separately decomposed anaerobically in closed chambers at 20 °C in the laboratory. At different intervals over a period of decomposition of over 2 years, prefiltered (precombusted at 500 °C Whatman GF/F glass filters, 0.6-µm pore size) whole leachate was then sterile filtered (0.2-µm pore size) and held aseptically at 4 °C until use. Final concentrations were in the range of 10 to 20 mg DOC-C l^{-1} to simulate natural concentrations, as determined by a Shimadzu TOC-5000 analyzer.

DOC solutions were transferred aseptically to gas-tight sealed, sterilized UV-transparent quartz chambers (4 cm in diameter by 50 cm in length, curved upward at one end) that contained ports for the periodic sampling of water and of gases in the headspace above the water. The lower one-third of the tubes was incubated in a water bath to maintain temperatures constant at 25 °C but not to interfere with light penetration into the solutions. Four conditions were assayed: (a) full sunlight (photosynthetically-active radiation, 720 to 400 nm (PAR), + UV-A at 400 to 320 nm + UV-B at 320 to 285 nm; (b) PAR + UV-A only; (c) PAR only; and (d) darkness (Al foil double wrapped). Filters used for these separations were Pyrex glass, Mylar D, and Plexiglas (Acrylite OP-2) (fig. 1). Quartz glass absorbed essentially none of the natural UV and PAR light above 240 nm. Samples for organic compounds in the water and CO_2 in the overlying headspace were removed at 30-minute intervals throughout the incubation periods of 6 hours (usually 09.00 to 15.00 hours).

Organic acids were analyzed by high performance liquid chromatography (HPLC) collected through Teflon microtubing from the quartz incubation chambers at time zero and each subsequent interval. HPLC methodology follows that described in Wetzel and others (1995). Of the numerous fatty acids generated by photolysis of the humic macromolecules, calibration with known compounds of highest available quality in identical media allowed quantitative evaluations. All error estimates are standard deviations (± SD) based on separate analyses or incubations for each variable and time interval (minimally n = 3 to 5).

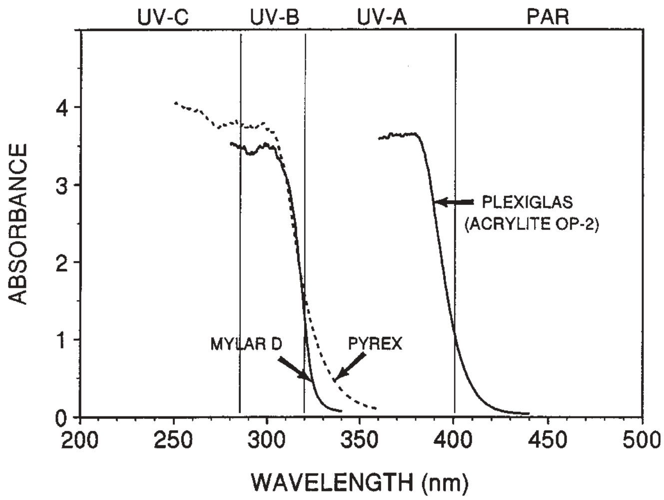


Figure 1—Selective UV absorption of Pyrex glass, Mylar D, and Plexiglas (Acrylite OP–2).

CO_2 gas samples (250 µl) were analyzed with a Varian gas chromatograph (series 3400) equipped with a flame ionization detector (Hayes Sep N column) and an auxiliary oven containing a small nickel catalyst column (column: 60 °C; injector: 150 °C; auxiliary oven: 390 °C; detector: 180 °C; duration: 3 minutes). Calibration was against a standard gas mixture (405 ppm CO_2).

Natural consortia of bacteria were obtained from the TWE, inoculated into modified lakewater medium (inorganic components only, pH 7.0, slightly modified from Veerkamp and others 1980), and grown for 48 hours at 25 °C. Eight hundred µl of the consortium was pipetted into sterile cryogenic (Nunc) vials with 200 µl of sterile glycerol. Tubes were mixed, frozen, and stored in liquid nitrogen (Sambrook and others 1989). These parent cultures were used in all experimentation. For experiments, one frozen tube of the consortium was thawed and added to 150 ml of modified lakewater medium, grown at 25 °C for 24 hours and pelleted (10,000 rpm, 10 minutes). The supernatant was decanted and the pellet washed with 20 mM MOPS (pH 7), a procedure that was repeated twice with MOPS and once with sterile ultrapure water (Millipore Q-pyrogen-free). The final pellet was suspended in 5 ml of sterile Q-water. Dissolved organic C (DOC) content of the Q-water was below detection by high temperature combustion with a Shimadzu TOC-5000 analyzer. Of the final culture suspension, 100 µl was inoculated aseptically into each experimental flask of UV-treated DOC water.

Bacterial production was determined on nonirradiated (time 0) and sunlight-irradiated DOM of leachates of known DOC concentrations from the known time intervals of progressive decomposition over a period of a year. Bacterial protein production (biomass production) was determined with the [^{3}H]leucine uptake and conversion to protein technique with 0.5-hour incubations (dark, 20 °C) on subsamples taken from cultures immediately (day 0) and at 24-hour intervals thereafter for several days. Methods follow Wetzel and Likens (2000) with very high specific activity [^{3}H]leucine (range of many batches 5 to 7 TBq mmole^{-1} or 35 to 47 GBq mg^{-1}) at a final concentration of 10 nM. Calculations of protein production follow Wetzel and Likens (2000).

RESULTS AND DISCUSSION

Photolytic alterations of the recalcitrant dissolved organic compounds were manifested in two important ways, both of which have major significance to the metabolism of organic matter within the ecosystems.

Partial Photolysis of Humic Substances to Simple Organic Compounds

Photolytic changes to DOM of whole leachates, as well as humic and fulvic acid fractions separated by ion chromatography, released from decomposing plant materials, were examined by solid-state ^{13}C nuclear magnetic resonance and low-temperature pyrolytic gas chromatography-mass spectrometry before and after photolysis. These analyses revealed subtle but important qualitative changes to the bulk DOM and small but progressive declines in DOM quantity during photolysis (Wetzel and others 1995). Small organic fractions generated by natural photolysis of DOM leachates showed marked, progressively increasing release of numerous small fatty acids, particularly acetic, levulinic, propionic, pyruvic, formic, and citric tartaric, among others. This common pattern resulted from photolysis by natural sunlight or if controlled UV-A only (Pyrex filtered) light source of leachates from different plant species. An example of the phytolytic generation of fatty acid substrates from humic substance leachates of the aboveground foliage of the rush *Juncus* decomposing over an annual period (fig. 2). Levulinic and formic acids were generated consistently from humic substances over the annual period, whereas acetic, propionic, and pyruvic/malic acids were generated more

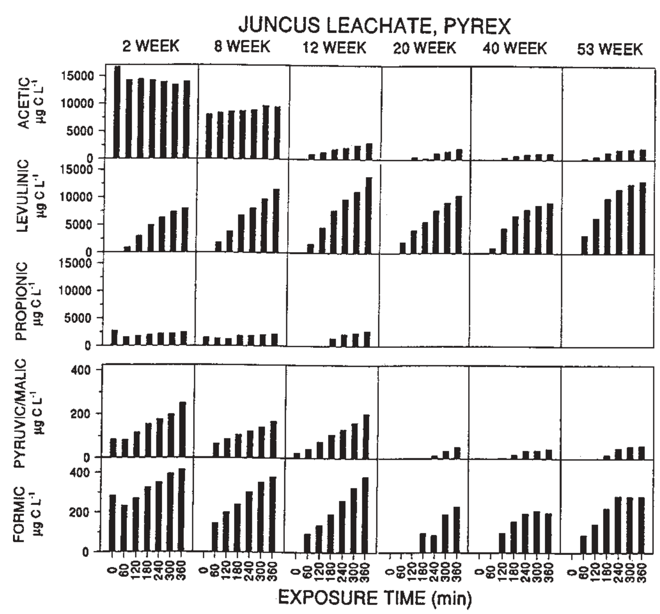


Figure 2—Fatty acid substrates occurring in the solutions of DOM (20 mg C l^{-1}) leachate (0.6-µm pore-size filtrate) from decomposing *Juncus effusus* at intervals over an annual period during exposure (minutes) to UV irradiance of increasing durations (minutes). Pyrex glass shielding simulating UV-A and PAR portions of the spectra.

strongly in earlier stages of decomposition and were less effectively cleaved from the macromolecules after several months of decomposition.

Changes in the photolytic generation of fatty acids from the humic substances leached from other plant species were significantly different from those patterns seen in *Juncus* (Wetzel 2000). Fatty acids generated from photolysis of cattail (*Typha*) leachate exhibited persistent generation of acetic and formic acids over the annual period of decomposition, but other substrates, particularly levulinic, propionic, and pyruvic/malic were of lesser quantitative importance (fig. 3). Levulinic and tartaric acids were generated photolytically early in the decomposition chronology but were not found among the photolytic products of the more recalcitrant DOM.

Dissolved organic matter from leachates of decomposing oak (*Quercus*) leaves was considerably more resistant to photolytic generation of fatty acids than were those of other species evaluated. Acetic and propionic acids were found early in the sequence (fig. 3), but levulinic acid, a dominant product from partial photolysis of other plant DOM sources, was not detected. Citric and succinic acids were generated weakly only from the most recalcitrant DOM after long periods of microbial degradation.

It is important to note that although the UV portion of the spectrum is of major significance in the partial photolysis of DOM with the release of simple substrates from the macromolecules, PAR is also contributing to the photolysis and generation of simple organic compounds. For example, one of the dominant fatty acids, levulinic acid, released from the photolysis of leachates from *Juncus* is quite effectively generated by PAR only (fig. 4). These results indicated that over half of this substrate was generated by PAR irradiance.

Many studies have shown that bacterial production is markedly enhanced when exposed to organic substrates generated by solar insolation and partial photolysis of humic substances from decomposing plant materials (Moran and Zepp 1998, Wetzel and others 1995). Despite the heterogeneous assemblage of simple organic substrates generated photolytically among the many diverse plant sources under many different stages of organic recalcitrance, the common pattern of enhanced bacterial productivity occurred. The photolytic effect of producing substrates that enhanced bacterial productivity increased with increasing age of the DOM. In the example given in figure 5, only the bacterial production is presented in response to photolytically-decomposed DOM of *Juncus* leachates at the beginning and after 6 hours of exposure to sunlight or controlled UV simulations. This same pattern was found with humic macromolecules released in leachates of the other species as well, although the rates at which they occurred differed markedly. For example, this pattern was found among leachate sources from the water lily *Nymphaea* already at week 5, whereas among *Juncus*, *Quercus,* and *Typha* the effects were most pronounced from the more recalcitrant DOM (Wetzel 2000).

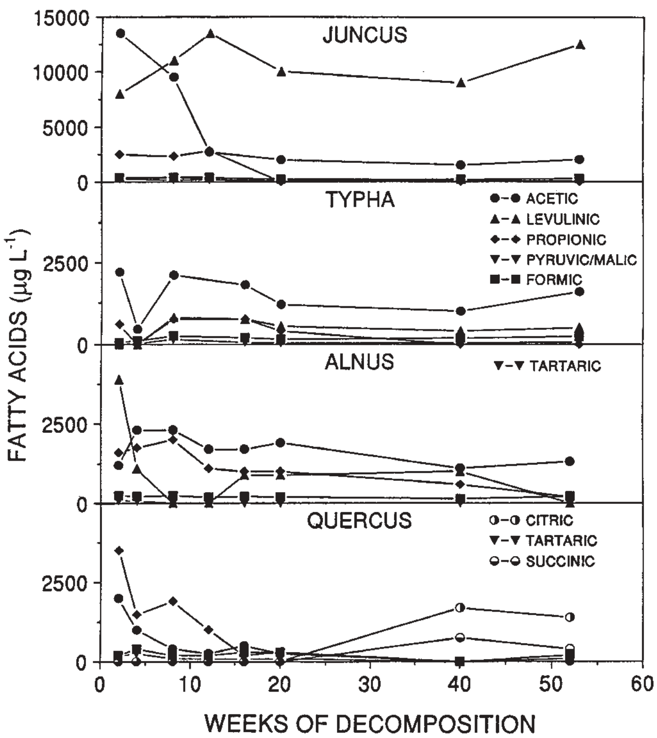


Figure 3—Fatty acid substrates occurring in the solutions of DOM (20 mg C l^{-1}) leachate (0.6-µm pore-size filtrate) from decomposing *Juncus effusus, Typha latifolia, Alnus serrulata,* and *Quercus rubra* at intervals over a 1-year period during exposure to UV-A and PAR portions of the spectra at the maximum exposure duration (360 minutes) (from Wetzel 2000 by permission).

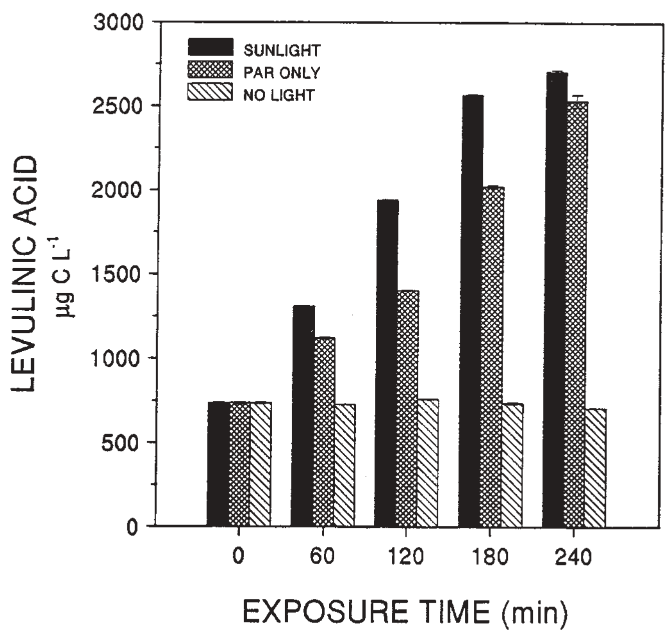


Figure 4—Net generation of levulinic acid from the partial photolysis of sterile whole leachate (0.2-µm pore-size filtrate) of *Juncus effusus* (10 mg C l^{-1}) after 4 weeks of microbial decomposition at 20 °C in the dark. Exposed to full natural sunlight (total insolation over 4-hour period = 13.05 mol m^{-2}), PAR only, or incubated simultaneously in the dark.

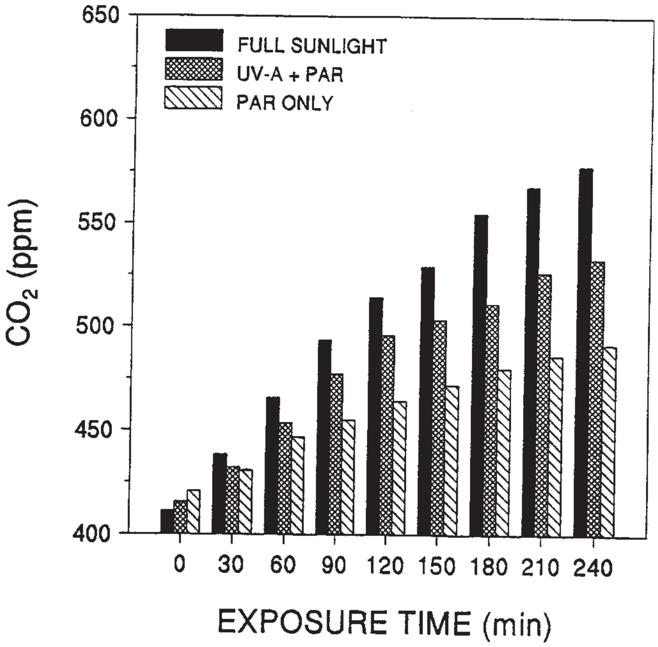


Figure 5—Photolytic degradation of sterile *Juncus* whole leachate (0.2-µm pore-size filtrate) after 4 weeks of microbial decomposition to CO_2 under replicated, aseptic conditions exposed to full sunlight (15.53 mol m^{-2} over the 4-hour period), UV-A + PAR, and PAR only.

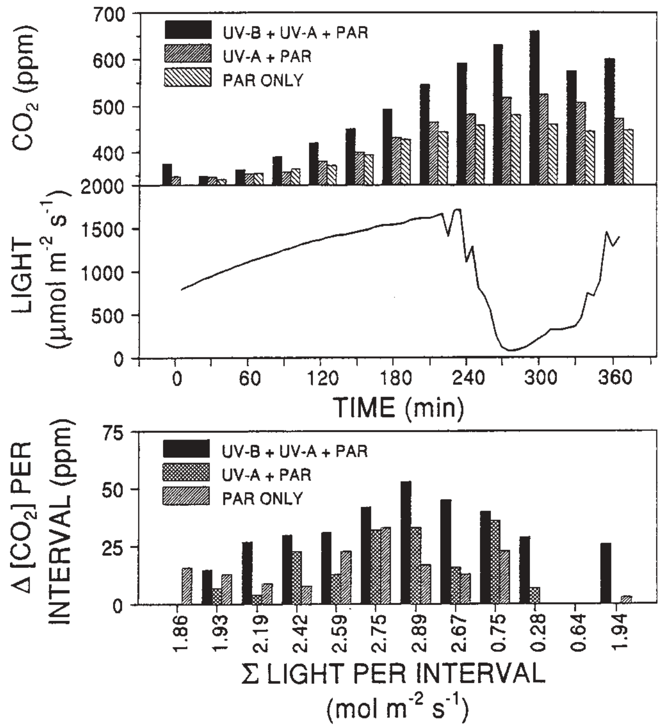


Figure 6—Photolytic degradation of sterile *Juncus* whole leachate (0.2-µm pore-size filtrate) after 20 weeks of microbial decomposition to CO_2 under replicated, aseptic conditions exposed to full sunlight, UV-A + PAR, and PAR only (upper). A severe rainstorm occurred during the incubations, which reduced light severely for an hour (middle). The net change in CO_2 production per amount of light received per interval under these conditions (lower).

Complete Photolysis of DOM to CO_2

Previous studies of the photolytic degradation of dissolved organic matter suggested that the dominant component of solar irradiance was UV-B and UV-A, and that photosynthetically active radiation (PAR) above 400 nm was of little consequence. Many of these studies, however, were not performed under sterile conditions, and, as a result, findings were confounded by microbial contributions to degradation and generation of CO_2. Furthermore, many of the DOM sources of these studies had been exposed to natural sunlight for long and noncomparable periods.

The present studies have employed recalcitrant dissolved organic matter from decaying plant sources over long periods (at intervals to a year or more) of many different types. Natural sunlight was used under many different conditions of light energy seasonally. Using the five terrestrial and aquatic plant species as sources in nearly 200 separate experiments, the results were consistently similar. The UV-B portion of the spectrum was always most effective in complete photodegradation to CO_2, but UV-A was also highly effective with small differences from the photolytic capacities of UV-B, e.g., fig. 5. Contrary to the promulgations of others, PAR is highly effective in photolytic degradation to CO_2.

The relative effectiveness of UV-B, UV-A, and PAR in this photodegradation to CO_2 varies with species and was more effective among leachates from relatively rapidly decomposing plant material such as *Alnus* and *Nymphaea*. Leachates from the cattail (*Typha*), rush (*Juncus*), and oak (*Quercus*) were consistently but less effectively photodegraded by PAR alone, e.g., fig. 6, upper. The effectiveness of relative photolysis by UV-B, UV-A, and PAR was more strongly correlated with species of plant DOM origins and the relative length of time to which the DOM had been degraded microbially before exposed to light rather than to total light intensity received upon exposure. For example, when natural sunlight was attenuated very rapidly, as by a severe thunderstorm, the rate of photolytic degradation of DOC to CO_2 declined precipitously (fig. 6, middle), but the photolytic capacities of PAR declined more rapidly than did the effects of UV-B (fig. 6, lower).

The high concentrations of DOM of inland waters, predominantly humic substances of plant structural tissue origin, serve as massive stores of relatively recalcitrant organic carbon and energy (Wetzel 1990, 1995). Photo-oxidation of DOM by natural sunlight, although recognized long ago, only recently has demonstrated the magnitude of this process of direct utilization and alteration chemically for enhanced microbial utilization. Findings that less energetic spectral components, particularly those of the PAR range, can also alter chemical availability, increase the dimensions of these photochemical processes. PAR penetrates water to a much greater extent than UV-A and especially UV-B and, as a consequence, can influence a large volume of aquatic ecosystems.

Reduction of ozone in the stratosphere and the associated increase in UV irradiance could lead to accelerated photolytic degradation of macromolecules of DOM by both abiotic and biotic pathways to CO_2. In addition, the photolytic enhancement of substrates for bacterial metabolism can result in accelerated rates of biogeochemical cycling of nutrients and stimulated productivity of the ecosystems. Even if UV photoinhibition of microbes occurs in relatively clear surface waters (cf. review of Karentz and others 1994), the photolytic organic substrate products are readily transported by water circulation to photoprotected heterotrophic regions of greater depth. The resulting enhanced microbial respiration will certainly lead to enhanced generation of CO_2 and evasion to the atmosphere.

In addition, as the concentrations of CO_2 in the atmosphere increase, largely from anthropogenic combustion of fossil fuels, plant growth commonly accelerates. Concentrations of CO_2 will double to ca. 720 ppm yet in the 21st century. Experimental enhancement of CO_2 leads to increased rates of photosynthesis and growth by about 135 percent and commonly leads to nitrogen limitations. As a result, C:N ratios increase and commonly structural tissues, particularly lignin content, increase. As these plants grown in enriched CO_2 decay, DOM released from leachates into the surface waters contains greater amounts of dissolved humic substances. Photolysis of this DOM shows a conspicuous increase in CO_2 release from those plants grown under atmospherically CO_2 enriched conditions among both terrestrial plants (fig. 7) and emergent aquatic plants (fig. 8). Clearly, photodegradation of dissolved organic substrates is a major process altering rates of biogeochemical cycling in aquatic ecosystems.

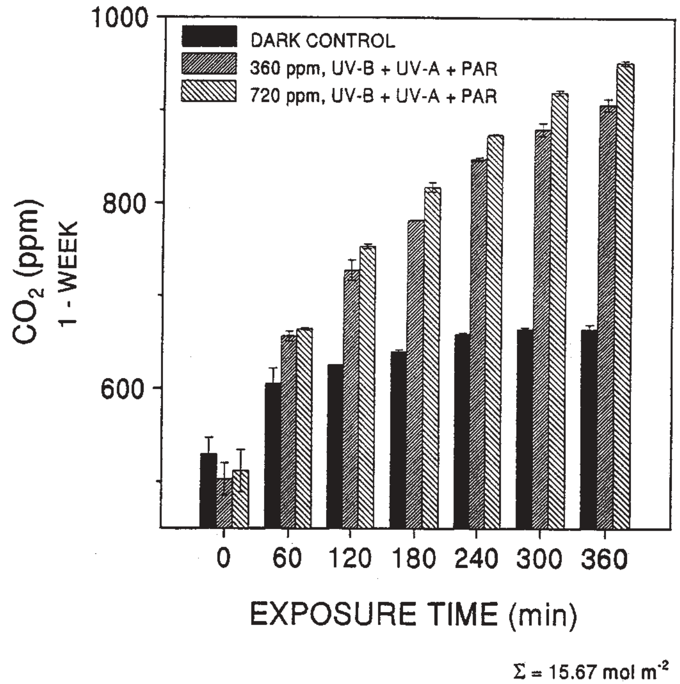


Figure 8—Release of CO_2 by complete photolytic degradation of sterile leachate from leaves of the cattail (*Typha latifolia*), grown under ambient atmospheric CO_2 and doubled CO_2 concentrations, after 1 week of microbial decomposition.

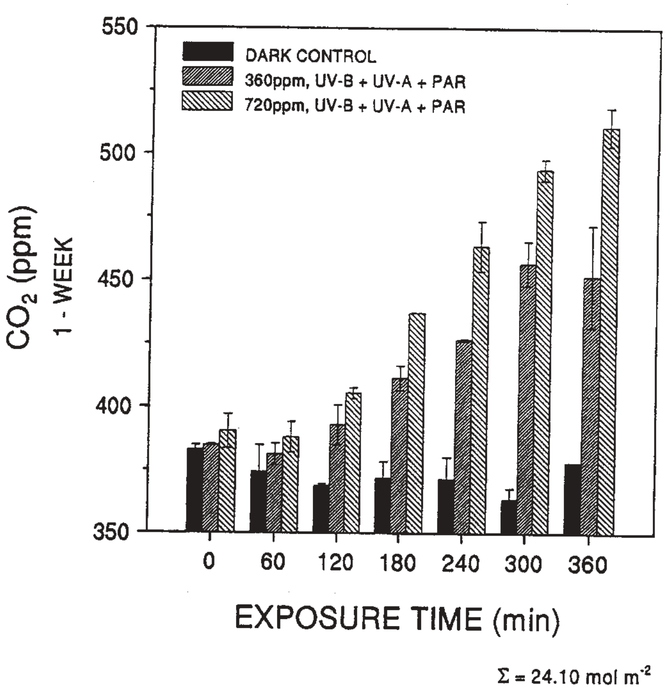


Figure 7—Release of CO_2 by complete photolytic degradation of sterile leachate from leaves of the poplar tree (*Populus tremuloides*), grown under ambient atmospheric CO_2 and doubled CO_2 concentrations, after 1 week of microbial decomposition.

REFERENCES

Cabaniss, S.E.; Zhou, Q.; Maurice, P.A. [and others]. 2000. A log-normal distribution model for the molecular weight of aquatic fulvic acids. Environmental Science and Technology. 34: 1103–1109.

Cole, J.J.; Caraco, N.F.; Kling, G.W.; Kratz, T.K. 1994. Carbon dioxide supersaturation in the surface waters of lakes. Science. 265: 1568–1570.

Dickerman, J.A.; Wetzel, R.G. 1985. Clonal growth in *Typha latifolia*: population dynamics and demography of the ramets. Journal of Ecology. 73: 535–552.

Gustafsson, Ö.; Gschwend, P.M. 1997. Aquatic colloids: concepts, definitions, and current challenges. Limnology & Oceanography. 42: 519–528.

Karentz, D. [and others]. 1994. Impact of UV-B radiation on pelagic freshwater ecosystems. Arch. Hydrobiol. Beih. Ergebn. Limnol. 36: 31–69.

Kling, G.W.; Kipphut, G.W.; Miller, M.C. 1992. The flux of CO_2 and CH_4 from lakes and rivers in arctic Alaska. Hydrobiologia. 240: 23–36.

Lindell, M.J.; Granéli, W.; Tranvik, L.J. 1995. Enhanced bacterial growth in response to photochemical transformation of dissolved organic matter. Limnology & Oceanography. 40: 195–199.

Malcolm, R.L. 1990. The uniqueness of humic substances in each of soil, stream, and marine environments. Anal. Chem. Acta. 232: 19–30.

Mann, C.J.; Wetzel, R.G. 1995. Dissolved organic carbon and its utilization in a riverine wetland ecosystem. Biogeochemistry. 31: 91–120.

Mann, C.G.; Wetzel, R.G. 1996. Loading and bacterial utilization of dissolved organic carbon from emergent macrophytes. Aquatic Botany. 53: 61–72.

Mann, C.J.; Wetzel, R.G. 2000. Hydrology of an impounded lotic wetland - subsurface hydrology. Wetlands. 20: 33–43.

Moran, M.A.; Zepp, R.G. 1997. Role of photoreactions in the formation of biologically labile compounds from dissolved organic matter. Limnology & Oceanography. 42: 1307–1316.

Otsuki, A.; Wetzel, R.G. 1974. Calcium and total alkalinity budgets and calcium carbonate precipitation of a small hard-water lake. Arch. Hydrobiol. 73: 14–30.

Sambrook, J.E.; Fritsch, F.; Maniatis, T. 1989. Storage media. In: Molecular cloning: a laboratory manual. Pt 3. 2d ed. Maine: Cold Spring Harbor Laboratories: A.5.

Stewart, A.J.; Wetzel, R.G. 1981. Dissolved humic materials: photodegradation, sediment effects, and reactivity with phosphate and calcium carbonate precipitation. Arch. Hydrobiol. 92: 265–286.

Stewart, A.J.; Wetzel, R.G. 1982. Influence of dissolved humic materials on carbon assimilation and alkaline phosphatase activity in natural algal-bacterial assemblages. Freshwater Biology. 12: 369–380.

Wetzel, R.G. 1990. Land-water interfaces: metabolic and limnological regulators. Verhand. Internat. Verein. Limnol. 24: 6–24.

Wetzel, R.G. 1992. Gradient-dominated ecosystems: sources and regulatory functions of dissolved organic matter in freshwater ecosystems. Hydrobiologia. 229: 181–198.

Wetzel, R.G. 1995. Death, detritus, and energy flow in aquatic ecosystems. Freshwater Biology. 33: 83–89.

Wetzel, R.G. 1996. Benthic algae and nutrient cycling in standing freshwater ecosystems. In: Stevenson, R.J.; Bothwell, M.; Lowe, R., eds. Algal ecology: benthic algae in freshwater ecosystems. New York: Academic Press: 641–667.

Wetzel, R.G. 2001. Natural photodegradation by UV-B of dissolved organic matter of different decomposing plant sources to readily degradable fatty acids. Verhand. Internat. Verein. Limnol. 27: 2036–2043.

Wetzel, R.G.; Hatcher, P.G.; Bianchi, T.S. 1995. Natural photolysis by ultraviolet irradiance of recalcitrant dissolved organic matter to simple substrates for rapid bacterial metabolism. Limnology & Oceanography. 40: 1369–1380.

Wetzel, R.G.; Likens. G.E. 2000. Limnological analyses. 3rd ed. New York: Springer-Verlag. 429 p.

DEVELOPMENT OF A SHORT-TERM (<12 DAYS), PLANT-BASED SCREENING METHOD TO ASSESS THE BIOAVAILABILITY, BIOCONCENTRATION, AND PHYTOTOXICITY OF HEXAHYDRO-1,3,5- TRINITRO-1,3,5-TRIAZINE (RDX) TO TERRESTRIAL PLANTS

Linda Winfield, Steven D'Surney, and John Rodgers[1]

EXECUTIVE SUMMARY

Limited amounts of information have been published on the environmental impacts of hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) to terrestrial plant communities. RDX is one of the two high-explosive compounds used by the U.S. military (Davis 1998) and classified as a priority pollutant by the U.S. Environmental Protection Agency (USEPA). Millions of acres of land on military installations, as well as manufacturing, storage, and disposal sites, have been contaminated with RDX (Jenkins 1989). Therefore, environmental risk assessments (ERAs) are conducted to determine the potential environmental impacts of RDX on receptors. Research on the environmental impacts of RDX on terrestrial plants is needed to facilitate filling data gaps and decrease the level of uncertainty and costs associated with ERAs on RDX.

Studies were designed and conducted to evaluate the responses of 15 terrestrial plans to short-term (< 12 days) and long-term (2, 4, and 6 weeks) exposures to military grade RDX. Emphasis was placed on cover plants (nontraditional crop plants) because they are more likely to occur in the remote areas near military installations with RDX-contaminated soils. The target plants included species recommended for use in assessing toxic compounds by the USEPA and the Organization for Economic and Cooperative Development (Fletcher and others 1988). Additionally, the target plants naturally occur in the regions around the military installations, are important economically, and are a source of food for wildlife or livestock. The objectives of the short-term (< 12 days) studies were to: (1) identify RDX sensitive plants for use in developing a plant-based, short-term, screening method for assessing RDX bioavailability; and (2) provide information useful for filling data gaps on the environmental impacts of RDX in terrestrial plants. The objectives of the long-term (2, 4, and 6 weeks) studies were to develop RDX uptake coefficients and evaluate the efficacy of experiments in the short-term study for estimating plant responses to long-term RDX exposures. A toxicological approach evaluating a spectrum of plant responses to RDX exposure was used during both studies.

During the short-term studies, 10 cover plants (sanfroin, sunflower, bush grass, Delar small Burnett, red clover, white clover, tall fescue grass, Kentucky blue grass, orchard grass, and rye grass) and 5 crop plants (cucumber, lettuce, rape, corn, and wheat) were exposed to Grenada, Memphis, and Bowdre soils amended with environmentally relevant concentrations of RDX (0 to 4000 ppm). The assessment parameters were percent emergence, root and shoot lengths, and adverse developmental effects. The experiments were concluded 2 days after 50 percent of the control seedlings had emerged, typically < 12 days.

Statistically significant differences were measured in root length, shoot length, or percent emergence in over 50 percent of the plants (sunflower, sanfroin, Delar small Burnett, rape, lettuce, corn, tall fescue, and orchard grass). However, there were no consistent patterns. The most consistent indicators of detrimental impacts following RDX exposures were the observed adverse developmental effects, regardless of soil type. The observed adverse developmental effects included: underdeveloped roots, fused or bifurcated leaves, atypical bilateral symmetry, atypical pigmentation, atypical seedling emergence, curled or irregular leaf margins, necrotic lesions, chlorosis, and yellow spots. Sunflower (a forb), sanfroin (a legume), and corn (a cereal grass) were identified as RDX-sensitive plants. Cover and dicot plants were generally more RDX sensitive than monocot and crop plants. Overall, sunflower was the most RDX sensitive of the test plants and was selected for use during the long-term studies.

Inherent plant characteristics, seed size, and root type were more important than taxonomic status in the development of adverse developmental effects during the short-term studies. The seeds of sunflower, sanfroin, and corn (47.8, 22.3, and 300.5 mg per seed, respectively) were larger than those of the other test plants (0.5 to 6.0 mg per seed for cover plants and 1.4 to 37.3 mg per seed for crop plants). Sunflower, sanfroin, and corn seeds had to imbibe larger volumes of the soil solution containing soluble (bioavailable) RDX in order to germinate as compared to the other test plants. The nitro groups of the RDX molecule are electron deficient (Freeman and others 1976) whereas the cellular biomolecules, i.e., nucleic acids and enzymes, are electron dense. Electron-deficient molecules can participate in electophillic substitution reactions with electron-dense

---

[1] Winfield, Research Biologist, U.S. Engineer Research and Development Center, 3909 Halls Ferry Road, Vicksburg, MS 39180; D'Surney, Genetics Professor, University of Mississippi, University, MS 38677; Rodgers, Director, The Institute of Wildlife and Environmental Toxicology, Clemson University, Pendleton, SC 29670, respectively.

Citation for proceedings: Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

molecules (Gregus and Klaassen 1996). Sunflower, sanfroin, and corn developed branched roots within 8 days following initial RDX exposure, which increased the root surface available for absorption of bioavailable RDX. These factors enhanced the possibilities for interaction between RDX and cellular biomolecules.

The efficacy of the short-term screening experiment for estimating plant responses to long-term RDX exposure was evaluated during the long-term (2, 4, and 6 weeks) studies. The effects of life stage (embryos and 2-week-old seedlings) and exposure duration were also evaluated. The adverse developmental effects observed following short-term RDX exposures were generally less severe when the plants were exposed to RDX-amended Grenada soil. Therefore, Grenada soil was used during the long-term studies to develop RDX uptake coefficients that would be conservative. The RDX concentrations (0, 5.8, 50, and 100 ppm) used were low to enhance the chance for plant survival, yet predetermined to cause adverse developmental effects. The assessment parameters included: adverse developmental effects, growth effects (above- and belowground biomass, maximum shoot and root lengths, maximum stem diameter, number of leaves, root biovolume), and bioconcentrated RDX in leaves and flowers.

The efficacy of the short-term screening experiments for estimating plant responses to long-term RDX exposure was validated. The adverse developmental effects observed during the short-term and long-term exposures were similar, regardless of life stage or exposure duration, and included: underdeveloped roots, fused or bifurcated leaves, atypical bilateral symmetry, curled or irregular leaf margins, necrotic lesions, chlorosis, and yellow spots. Additional adverse developmental effects observed following long-term RDX exposure were wrinkled leaf surfaces and poor seedling development. Statistically significant differences were measured in several of the growth parameters (biomass, number of leaves, shoot length, root length, and root biovolume), but there were no consistent patterns.

Statistically significant differences were measured in the bioconcentrated RDX content of leaves using both life stages during all exposure durations. The amount of flowers produced was not sufficient to provide data for statistical analysis. The highest coefficient of determination (R-square) was calculated using data for embryos following the 6- week exposure (0.80) and following the 4-week exposure using the 2-week-old seedling (0.73).

The short-term screening experiments can be used for site characterizations, risk assessments, and to evaluate remediation activities. Additionally, they are cost effective, simple, and require significantly less time than more traditional approaches. The growth parameters commonly used to evaluate herbicides, inorganic, or organic contaminants did not adequately assess the potential for adverse environmental impacts from RDX exposure. The toxicological approach used during these studies was more appropriate.

REFERENCES

Davis, W.C. 1998. Introduction to explosives. In: Jukas, J.A.; Walters, W.P., eds. Explosive effects and applications. New York, Berlin, Heidelberg: Springer-Verlag Publishers.

Fletcher, J.S.; Johnson, F.L.; McFarlane, J.C. 1988. Database assessment of phytotoxicity and data published on terrestrial vascular plants. Environmental Toxicology & Chemistry. 7: 615–622.

Freeman, D.H.; Angeles, R.M.; Poinescu, I.G. 1976. High performance liquid chromatography separation of RDX and KMX explosives on adsorptive polymers. Journal of Chromatography. 118: 157–166.

Gregus, Z.; Klaassen, C.D. 1996. Mechanisms of toxicity. In: Klaassen, C.D.; Amdur, M.O.; Doull, J.D., eds. Casarett and Doull's toxicology: the science of poisons. New York: McGraw-Hill Publishers: 35–74.

Jenkins, T.F. 1989. Development of an analytical method for determination of extractable nitroaromatics in soils. University of Maryland. U.M.I. Dissertation Services. Bell and Howell Co. Ann Arbor, MI. Ph.D. Dissertation.

Holland, Marjorie M.; Warren, Melvin L.; Stanturf, John A., eds. 2002. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS–50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 191 p.

Abstract: The conference focused on recent work in freshwater wetlands [both natural and constructed] with a view toward understanding wetland processes in a watershed context. Since humans have played important roles in watershed dynamics for years, attention was given to the human dimensions of wetland and watershed uses. Contributed sessions were organized on: biogeochemical cycling in wetlands; human health issues related to water; wetland restoration and reforestation; the role of wetlands in agricultural systems; wetlands and USA environmental law; chemical ecology and natural products from wetlands; water and wetlands in science education; and regional water strategies. The lead paper in the proceedings was prepared by conference plenary speaker Dr. Sandra Postel, director of the Global Water Policy Project in Amherst, MA. Examples and experiences from eight countries were shared during the conference, providing valuable global perspectives.

Keywords: Environmental law, fresh water, restoration, sustainable systems, water resources, watersheds, wetlands.