36th annual

MISSISSIPPI Water Resources CONFERENCE

2006

April 25-26

Hilton Hotel
Jackson, MS

Conference Sponsors:

Mississippi Water Resources Research Institute, MSU
U.S. Geological Survey
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Executive Summary

It has been said that water is the rarest commodity in the world. We often don't think of water as a rare commodity, especially in the U.S. where we always have clean drinking water at our fingertips. However, water is, without question, one of the most important issues facing our state and nation. When I arrived in Mississippi a year ago, I made many visits across the State introducing myself and meeting natural resources leaders. I asked each of them the same question, "What are the most important issues facing Mississippi in the next 5-10 years?" Five people, including the Director of the Mississippi Department of Wildlife, Fisheries and Parks; the Mississippi Forestry Commission's State Forester; the Director of the Mississippi Wildlife Federation; the Director of the Mississippi Forestry Association; and the Director of the Delta Council; all gave the same response—WATER. They all had different reasons but the same answer.

When asked to accept the role as the Water Resources Research Institute Director, I was thrilled at the prospect of adding this element to the strong water research program within the Forest and Wildlife Research Center (FWRC) at Mississippi State University (MSU). The FWRC has been conducting water related projects for many years. In forestry, we have been assessing silviculture practices and Best Management Practices on the impact of water quality in our streams and creeks. We have been studying numerous wildlife and fisheries issues such as reservoir and river system management, waterfowl management, catfish farm pond management, and fisheries research on paddlefish, mussels, catfish, and others. It seemed like a natural fit to add water resources to the other natural resource elements that we study in MSU's FWRC.

As Director, it is my desire that the Water Resources Research Institute at Mississippi State University be a premier entity in the state and region for information and expertise on water resources issues. I want the Institute to be in the forefront on water resource issues and be proactive in public outreach and education.

Since March, faculty and staff in the FWRC have been building and refreshing the Water Resources Research Institute. The WRRI web site (www.wrri.msstate.edu) is a one-stop shop for information on water-related projects. Soon, an online database of publications, including these conference proceedings will be added to the web site. We anticipate the publication of an annual report and a newsletter this year. Additionally, we have been working with our sponsors and cohosts in planning the 2007 conference. We anticipate a bigger and better conference next year.

Later this year we will host the Water Resources Research Institute Advisory Council. The purpose of the Council is to improve communication throughout water leadership, identify needs and future opportunities, and accomplish the WRRI mission of serving public and private interests related to conservation and development of Mississippi water resources. We will soon send out our annual requests for proposals with the USGS contract and call for papers for the 2007 conference.

Plentiful supplies of clean water represent a critical natural resource to sustain economic success in our towns and cities, our fields and forests, and among our industries. Mississippi faces serious issues to ensure a plentiful supply of clean water while sustaining the vital ecological functions of our landscapes. We must provide clear understanding of those activities that impact our water quantity and quality into the future. Thank you for your participation in these endeavors.

George Hopper

George M. Hopper
Director, Mississippi Water Resources Research Institute

36th annual Mississippi Water Resources Conference

2006 Program and Abstracts

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Program

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9:20 am	Kimberly Caviness *Modeling The Big Black River: Evaluation of a Simplistic Water Quality Model*	
9:40 am	Alison Faulkner *Greenhouse Modeling of Nitrogen Use Efficiency in Two Wetland Cyperus Species at the University of Mississippi Field Station*	
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11:00 am	Chris Bowen (Invited Speaker)	
11:20 am	Charles Chisolm (Invited Speaker)	
12:20 pm	**Luncheon –** Dr. Charlie Wax, GeoSciences Department, MSU	**Madison Hall**

NOTICE:
A meeting of the Mississippi Water Resources Advisory Council
will be held at 1:30 pm on Wednesday, April 26, 2006,
in the Auditorium of the Mississippi Farm Bureau Federation offices located near the Hilton Hotel.
All are invited to attend.
For more info, contact Sam Mabry, 601–961–5200.

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Opening Plenary Session

Mickey Plunkett, Moderator

Robert Hirsch, USGS
USGS Science in Support of Water Resources Management

Peter Howd, USGS
The Response of Mississippi Beaches and Barrier Islands to Hurricane Katrina

Tom Doyle, USGS
Hurricanes on the Gulf, 2005: Biological Impacts and Landscape Changes

Opening Plenary Session

Robert Hirsch

Robert M. Hirsch serves as Associate Director for Water. In this capacity, Dr. Hirsch is responsible for all U.S. Geological Survey (USGS) water science programs. These programs encompass research and monitoring of the nation's ground water and surface water resources including issues of water quantity as well as quality. He has served as the leader of USGS water programs since 1994. As Associate Director he represents the interests of the USGS in scientific, technical, and leadership aspects of hydrology and serves as the Director's principal advisor on water-related issues. In his capacity as spokesperson for the USGS and its water resources mission, Hirsch holds the title of Chief Hydrologist. He also serves as the co-chair of the Subcommittee on Water Availability and Quality of the Committee on Environment and Natural Resources of the National Science and Technology Council. Hirsch was born in Highland Park, Illinois. He received degrees from Earlham College (bachelor of arts degree in geology, 1971), University of Washington (master of sciences degree in geology, 1974), and The Johns Hopkins University (doctorate in geography and environmental engineering, 1976). Dr. Hirsch is a Fellow of the American Association for the Advancement of Science and an active member of the American Geophysical Union and the American Water Resources Association.

Peter Howd

Peter Howd received his PhD in Oceanography from Oregon State University in 1991. Since his professional career began in 1978 as a lowly GS-4 physical science technician, he has been employed by the US Geological Survey, the US Army Corps of Engineers Field Research Facility, Duke University and the University of South Florida. He rejoined the USGS Extreme Storm Studies Group in 2005, just in time for a remarkable hurricane season. His research interests include the impact of storm events on beach and surf zone morphology, low-frequency wave motions in the nearshore zone, and sediment transport in combined wave-current environments.

Tom Doyle

Thomas W. Doyle is a research ecologist with the U.S. Geological Survey National Wetlands Research Center in Lafayette, Louisiana. He received his M.S. and Ph.D. in systems ecology and environmental science from the University of Tennessee in Knoxville. He has 20+ years of field and modeling experience in temperate and tropical forest ecosystems of the southeastern U.S. and Caribbean relating hurricane effects and history. His research disciplines focus on wetland ecosystem analysis and modeling, forest stand and landscape simulation, hurricane and climate change science, tree ring analysis, plant competition and growth modeling, and disturbance ecology. Dr. Doyle has developed several landscape simulation models for various federal parks and refuges across the Southeast to forecast potential threats of habitat loss or conversion by natural and man-induced disturbances including hurricanes, sea-level rise, and future climate change.

Poster Session

Charles Cooper
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Jim Weston
Analysis of Mississippi and Alabama Water and Sediment Samples after Hurricane Katrina with H4IIE and YES Assays

The Yalobusha River–Grenada Reservoir Watershed: Sediment Movement, Accumulation and Quality in a Mississippi Intensive Agricultural Landscape

Charles M. Cooper, Sammie Smith Jr., Jerry Ritchie

We examined sedimentation rates, current watershed contamination contributions and potential impacts of long-term row cropping (cotton, corn, soybeans, and sweet potato) on a small river and a large downstream flood control reservoir in the loess hills of Mississippi, USA. Grenada Reservoir (impounded in 1954) has a total watershed drainage area of ~3,419 square kilometers. Although reservoir life expectancy was originally estimated at 25 years because of high erosion rates in the watershed, our study revealed that the reservoir continues to function with only slightly reduced storage capacity. Sediment delivery to the reservoir by the Yalobusha River at the most downstream measured site from 1996 to 2002, averaged 126 mg/L (range 12 to 767 mg/L, S.D.=136). Long-term sediment accumulation within the permanent pool adjacent to the dam was <1cm yr-1 except for a depositional area near tributary inflow that accumulated sediment at about 5 cm yr-1. The central area of the permanent pool experienced sediment accumulation rates that averaged <1.5 cm yr-1. Sites within the two reservoir arms fed by the two river inflows (Yalobusha and Skuna rivers) showed little or no sedimentation. Sedimentation rates near the two river inflows were also generally low. A large debris jam which formed a river plug southwest of Calhoun City accumulated sediment from the upper portion of the watershed. From 1996 to 2004 analyses were conducted in water and sediment for 8 metals and 48 pesticides/contaminants at 26 stream/river locations and 9 locations within the reservoir. In spite of long-term historical use of residual pesticides in the watershed and widespread use of currently applied agricultural compounds, concentrations in stream or reservoir sediments and overlying water were generally low and sporadic or below detection. Conversely, several metals (arsenic, lead, copper, iron, aluminum and zinc) were abundant in stream and reservoir sediments. Atrazine, a widely used triazine herbicide, was routinely found in stream water and sediment. Atrazine was also detected in reservoir water samples but at only one fifth of contributing stream concentrations. Naturally-occurring aluminum and iron were found in high concentrations. Residual pesticides were generally not detected in water but were detected in stream and reservoir sediments. Sediments within the debris jam contained concentrations of arsenic and mercury lower than watershed and reservoir locations. Debris jam sediments also held highest observed concentrations of beta-BHC but did not contain detectable amounts of several legacy pesticides that were found in both the watershed and reservoir samples. A dredged channel through the debris jam was completed in late 2003. It may affect future sediment and contaminant accumulation in the natural river channel, its floodplain, and Grenada Reservoir. Because of processes associated with transitioning from a channelized stream to a natural one, it is likely that the plug phenomenon will reoccur.

Keywords: Agriculture; contamination; sedimentation; pesticides; non-point source pollution

Introduction

The upper Yalobusha River watershed and Grenada Reservoir flood-control lake are located in north central Mississippi, USA. Flow within the watershed originates near Houston, MS, and is from east to west, with controlled outflow from the reservoir into the Yalobusha River channel below the dam just northeast of Grenada. Flow ultimately joins the Mississippi River along the western border of Mississippi via the Yazoo River that drains most of the northwestern region of the State. The reservoir was impounded in 1954. Total watershed drainage area entering Grenada Reservoir from two rivers (the upper Yalobusha to the south, and the Skuna to the north) and other direct tributaries is approximately 3,419 square kilometers. The contributing watershed associated with the upper Yalobusha River has a floodplain area of intensive agriculture, including large-scale production of sweet potatoes [Ipomoea batatus (L.) Lam.] rotated with cotton (Gossypium hirsutum L.), soybeans [Glycine max (L.) Merr.] and corn (Zea mays L.) centered around the towns of Calhoun City and Vardaman.

Channelization projects and associated channel incision that began in the early 1900s have impacted the entire study watershed. With the exception of approximately 21 km (13 miles) in the Yalobusha River upstream of Grenada Reservoir, all of the river and major tributaries of the watershed have been channelized. Original channelization projects were conducted during the 1910s and 1920s, and additional works were conducted in the late 1930s to 1950s when the Yalobusha River and Topashaw Creek, the major river tributary, became plugged with debris and sediment. Late in the 1960s the U.S. Department of Agriculture Soil Conservation Service began a major series of watershed modifications above the reservoir, including extensive clearing and dredging of many channels and installation of numerous gully erosion control structures. Also during the 1960s some dredging was done in the upper reservoir, but the extent is unknown. A major cycle of channel incision, a response to previous channelization efforts, is currently migrating up watershed streams (Simon and Thomas 2002).

During the 1980s, an occlusive debris jam formed in the Yalobusha River upstream of the non-channelized portion of the river east of the reservoir. This debris jam, in excess of 2 km long and formed from eroded upstream materials, forced river flow into adjacent riparian floodplain bottomland forest and, occasionally, agricultural fields and homes. The US Army Corps of Engineers, under direction from Congress, is currently addressing this and other problems in the watershed through a system-wide approach. Tributary stabilization projects and downstream river stabilization and debris jam clearing have already been enacted. Following completion of these works, the Yalobusha River watershed is expected to become more stable over the next several decades, and movement of sediment and associated contaminants from field soil, urban areas, and streambeds should be minimized. In order to allow for future comparisons, we herein assess the sediment quality and quantity moving in the upper Yalobusha River watershed and Grenada Reservoir. This publication provides additional information and updates a previous publication by these authors (Cooper et al. 2002).

Methods

Sample collection and storage were done according to suggested American Public Health Association (APHA) methods (Greenberg et al. 1992). Suspended sediment load grab samples from the stream thalweg were collected from 13 sites on at least one hundred and two dates between October 1996 and December 2001. Values were quantified at the USDA National Sedimentation Laboratory as the difference between total solids and total dissolved solids dried at 103 – 105 degrees Celsius.

Metal and pesticide/PCB samples were collected from overlying water and bottom sediment into properly cleaned and solvent-rinsed glass sample containers according to APHA methods (but not EPA trace element methods). Analyses included concentration determinations for up to 8 metals and 48 pesticide/PCB contaminants likely to occur, given the historical and current land use (Tables 3, 4, & 5). Not all samples were tested for each analyte due to changing emphasis and resource availability.

Samples from 26 stream/river locations within the Yalobusha River watershed were taken around November 1 during 1996, 1997 and 1999 from mid water-column and the upper 10 cm of bottom sediments. Additional water column samples were taken from up to 14 of these sites on twenty-seven dates between July 1997 and April 2004. Added bottom sediment samples were taken from up to eight of the sites on nine dates ending in November 2003.

Sediment samples from Grenada Reservoir were taken at 9 locations between December 1998 and May 1999. Nearby samples were taken again at seven of these sites in November 1999, and at five of these sites in December 2002 (some sites were inaccessible because of low water). At each site, sediment cores were taken with manual coring equipment from an anchored boat. Ten-centimeter-diameter sediment cores were driven, lifted into a clean semi-tubular ruled trough, and divided into incremental 10-cm sections by depth from sediment surface for metals, pesticides, contaminants and cesium-dating analyses. One kg of sediment from each 10-cm depth increment was acquired for cesium dating that was conducted at the USDA Hydrology and Remote Sensing Laboratory. Surface sediments at seven of these sites were sampled again in November 1999. Concomitant with sediment sampling in the reservoir, water samples from mid water-column were collected for metals, pesticides and contaminants analyses.

Sediment samples from within the debris jam occluding the Yalobusha River upstream of the reservoir were collected from four transects spaced along the length of the jam on March 22, 2001, and from six additional locations on July 8, 2003. A 5-cm diameter stainless steel hand corer was used to collect samples At each transect, sediment to a maximum depth of 0.3 m was collected at three evenly spaced locations across the width of the channel and composited into a single sample. Woody and other (anthropogenic) debris within the jam prevented collection of deeper sediments. Relative location of all sampling sites is shown in Figure 1.

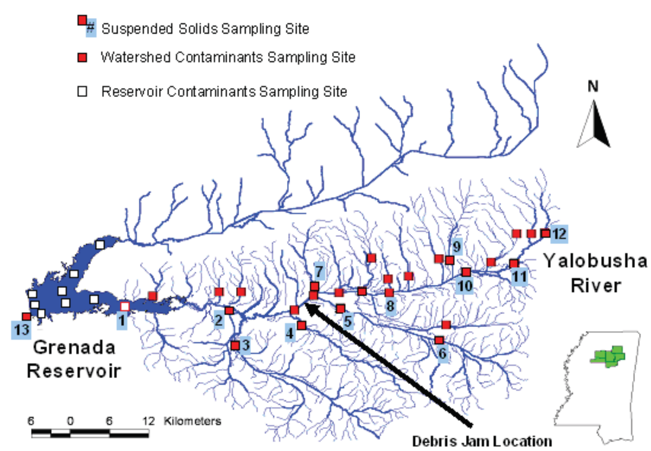

Figure 1. Grenada Reservoir drainage system, including Grenada Reservoir, Yalobusha River Watershed (South) and Skuna River Watershed (North), with sampling locations indicated by rectangles, and suspended solids collections sites by number (#).

Table 1. Methods used and method detection limits (MDL) for quantifying metal concentrations during this study. Information for pesticides is given in the text.

Analyte	Method	MDL ($\mu g\ L^{-1}$ or $\mu g\ kg^{-1}$)
Mercury	EPA 245.1	0.1
Arsenic	EPA 206.2	1
Copper	EPA 200.7	3
Chromium	EPA 200.7	2
Lead	EPA 200.7	15
Zinc	EPA 200.7	3
Aluminum	EPA 200.7	60
Iron	EPA 200.7	2

Analyses for some pesticides were conducted in part (Table 5) at the USDA-ARS National Sedimentation Laboratory using gas chromatographic methods (Bennett et al. 2000) with both method detection and level of quantification limits equal to or less than 0.1 μg kg-1. Other analytes, including a 25 priority pollutant scan and all metals, were quantified at the University of Louisiana Monroe Soil-Plant Analysis Laboratory using ASTM and USEPA approved methods. Priority pollutants in water and sediment were tested according to EPA method SW 846:8140 with a detection limit of 1 μg kg-1. Methods and detection limits for metals analyses were as indicated in Table 1.

Results and Discussion

Suspended Sediment

The overall mean concentration of total suspended solids (TSS) in the Yalobusha River watershed, including all thirteen sites we monitored on the Yalobusha River, its tributaries, and including the reservoir spillway outlet, was 80 mg/L. The overall standard deviation for the thirteen sites was 146, with an overall standard error of 3.73. The median value for all TSS data was 42 mg/L.

Mean concentrations for each site are given in Table 2. Unweighted overall average maximum concentration for all sites was 1061 mg/L, with maxima of about 1500 mg/L for three sites (tributary sites 6 and 3, and river site 10). These concentrations were associated with major storm events. Individual maximum concentrations at the other ten sites were mostly below 1000 mg/L. Individual site minima of zero were observed for nine sites. Seasonal averages for all sites showed summer (62 mg/L), and fall (65 mg/L) to have almost identical concentrations as expected. Winter, with minimum ground cover and maximum rainfall, had higher suspended sediment concentrations (95 mg/L), followed by spring (87 mg/L).

Table 2. Mean concentration of total suspended solids (TSS) observed at 13 sites within the Yalobusha River and Grenada Reservoir Watershed and its outflow, years 1996-2001. (note: asterisk * denotes site downstream of debris jam; caret ^ indicates site influenced seasonally by Grenada Reservoir inundation.)

Site Name	Location	Mean TSS ($mg\ L^{-1}$)
12	Tributary of River	104
9	Tributary of River	53
6	Tributary of River	76
5	Tributary of River	48
4	Tributary of River	50
7	Tributary of River	60
3	Tributary of River	84
11	Yalobusha River	127
10	Yalobusha River	105
8	Yalobusha River	69
2	Yalobusha River *	134
1	Yalobusha River ^	84
13	Grenada Reservoir Outlet	57

Tributary sites had an unweighted average concentration of 68 mg/L, while riverine sites were higher, at 104 mg/L. The reservoir outlet channel mean concentration of suspended solids was 57 mg/L, and three of seven tributary sites had a lower mean concentration for our study period. Site 4 on Shutispear Creek, potentially the least channelized of all the Yalobusha River tributaries, had not only one of the lowest mean TSS concentrations at 50 mg/L but also a standard error of only 7 mg/L.

Mean TSS concentrations exceeded 100 mg/L for only four sites; three being the uppermost of our sampling sites in the watershed (sites 12, 11, and 10) and the lowermost true riverine site, (site 2). Lower observed mean concentrations in the intervening sites are likely due to sediment storage occurring upstream of the debris jam. Once past the impounding influence of the debris jam, the TSS concentration in the river channel was again elevated as erosive tributaries (including Lickup, Savannah and Sabougla Creeks) entered the river.

The capacity for up-stream storage occurring in the region above the debris jam, and its potential for remobilization, has recently been studied by Bennett et al. (2005). Their estimates indicate that only about 16% of eroded upstream sediments are stored in Grenada Reservoir, and 8% or less are exported from the reservoir. With an estimated 76% or more of sediments eroded from the upper Yalobusha River tributaries and the river channel (an estimated 10 million tons or more) still stored above the reservoir, contaminants harbored in that sediment are a concern. Not only does sediment accumulate upstream of the debris jam, observations revealed that large amounts of sediment are deposited in the river floodplain where the jam causes high flows to overtop spoil levees.

Bennett et al. (2005) also used multiple methods to estimate point sedimentation rates at several locations within Grenada Reservoir, finding values in general agreement to those obtained earlier by the present authors (Cooper et al. 2002), and from which they calculated that the reservoir has lost only about 3% of its storage capacity during the 50 years it has been in service. During this time, they (Bennett et al. 2005) estimate that 0.036 km^3 of sediment have been impounded with the reservoir.

Metals

Metal concentrations in water samples were often an order of magnitude or even less than concentrations in sediments. Mean water and sediment concentrations for metals in the study area are given in Table 3. Mercury was detectable in watershed water with only 12% frequency, and reservoir water at a 24% frequency; detected concentrations were predominately quite low. This resulted in very low mean concentrations of mercury for both the watershed and reservoir sites. Mean surface water concentrations of other metals of potential concern were near their minimum detection levels, and only iron and aluminum (both elements that naturally occur in high concentration in soils of the watershed) were observed at elevated levels in the water. None of the metals concentrations listed in water exceeded EPA drinking water standards.

Analyses for metal concentrations in sediments revealed moderate arsenic and mercury concentrations. Reservoir mean sediment concentrations of mercury and zinc were somewhat elevated over other watershed levels. Again, iron and aluminum were abundant.

Pesticides

Pesticides were detected in stream water samples at only a 13% frequency (652 of 5,027 potential detections) which was comparable to metal occurrence. Of the 25 priority pollutant scan analytes (Table 4), only six were detected. For those six analytes, detections were rare and at such small concentrations that four had mean concentrations of 0.001 $\mu g\ L^{-1}$ or less. Of current-use and other pesticides for which we sought determination (Table 5), only one compound, atrazine (a herbicide used on corn and cotton), was found to have a mean concentration above 1 $\mu g\ L^{-1}$ in stream waters. All others were below 0.1 $\mu g\ L^{-1}$, and often below 0.01 $\mu g\ L^{-1}$.

In Grenada Reservoir, pesticides in water were also detected in only 13% of analyses (93 of 697 potential detections). Here, none of the 25 priority pollutant compounds were detected in reservoir water. Of current-use and other pesticides we sought, atrazine was again the compound with highest observed mean concentration (0.317 $\mu g\ L^{-1}$) but at approximately one-fifth of the level observed in the watershed samples. All other pesticides had a mean concentration below 0.1 $\mu g\ L^{-1}$ except for the herbicides cyanazine (0.131 $\mu g\ L^{-1}$) and metolachlor (0.118 $\mu g\ L^{-1}$).

In watershed stream sediment samples, only 332 (of 2524 potential; 13% detection rate) results of analyses showed detectable pesticides in measurable quantities. Within the priority pollutant scan, seventeen compounds were detected in measurable quantities; however, none of the PCBs or chlordane were detected. BHC

Table 3. Mean concentration ($\mu g\ L^{-1}$ water or $\mu g\ kg^{-1}$ sediment) of metals in watershed components of the Yalobusha River and Grenada Reservoir.

	Water		**Sediment**		
	Watershed	Reservoir	Watershed	Reservoir	Debris Jam
Mercury	< 0.1	< 0.1	43.1	67.3	34.8
Arsenic	3	4	2686	2553	1834
Copper	10	30	9751	11714	5405
Chromium	3	2	5618	10561	7921
Lead	14	6	21817	21128	5928
Zinc	24	35	16286	44679	24111
Aluminum	3873	2260	3063430	12219000	14437500
Iron	4093	1801	9635468	17329187	12119600

Table 4. Mean concentration (μg L-1 water or μg kg-1 sediment) of 25 priority pollutant insecticides and PCBs in watershed components of the Yalobusha River and Grenada Reservoir. ND = not detected.

	Water		Sediment		
	Watershed	Reservoir	Watershed	Reservoir	Debris Jam
Arochlor 1016	ND	ND	ND	ND	ND
Arochlor 1221	ND	ND	ND	ND	ND
Arochlor 1232	ND	ND	ND	ND	ND
Arochlor 1242	ND	ND	ND	ND	ND
Arochlor 1248	ND	ND	ND	ND	ND
Arochlor 1254	ND	ND	ND	ND	ND
Arochlor 1260	ND	ND	ND	ND	ND
BHC-ALPHA	ND	ND	0.641	0.169	ND
BHC-BETA	ND	ND	26.219	27.558	68.622
BHC-DELTA	ND	ND	1.081	ND	0.295
BHC-GAMMA	ND	ND	0.603	ND	1.47
CHLORDANE	ND	ND	ND	ND	ND
TOXAPHENE	ND	ND	0.061	ND	ND
DDD 4,4′	ND	ND	3.253	3.310	7.608
DDE 4,4′	ND	ND	9.626	6.062	4.620
DDT 4,4′	ND	ND	0.607	10.406	4.070
ΣDDT	ND	ND	4.495	6.593	5.433
ALDRIN	< 0.001	ND	15.429	1.506	1.784
DIELDRIN	< 0.001	ND	1.521	ND	ND
ENDRIN	ND	ND	0.191	5.468	ND
ENDRIN ALDEHYDE	ND	ND	0.020	1.018	ND
ENDOSULFAN I	< 0.001	ND	0.016	0.127	0.100
ENDOSULFAN II	0.002	ND	3.879	ND	0.191
ENDOSULFAN SULFATE	ND	ND	0.106	ND	ND
HEPTACHLOR	0.001	ND	0.981	2.563	0.947
HEPTACHLOR EPOXIDE	0.015	ND	3.855	0.163	ND

(beta) (lindane) had the highest concentration of any analyte in watershed sediments, with a mean of 26.2 μg kg^{-1}. Aldrin (an organochlorine banned in 1974) was also present in high enough concentrations to have a mean value of 15.4 μg kg^{-1} in our samples. ΣDDT, endosulfan II, heptachlor epoxide, dieldrin, and BHC (delta) all had mean concentrations in sediment greater than 1 μg kg^{1}. Many of the other current-use pesticides were detected in stream sediments, but only atrazine (3.698 μg kg^{-1}) and fluometuron (1.379 μg kg^{-1}) had mean concentrations above 1.0 μg kg^{-1}.

Sediments of Grenada Reservoir yielded 126 detections of pesticides (of 846 potential; 15% detection rate). Again, no PCBs were detected, nor were seven other analytes in the priority pollutant scan. Ten of the 25 priority pollutants were measurable, with BHC and ΣDDT again having highest observed mean concentrations in our samples. However, four other banned compounds had mean concentrations above 1.0 μg kg^{-1}, including aldrin, endrin, endrin aldehyde, and heptachlor. Not surprisingly, atrazine (322.26 μg kg^{-1}) was observed to have the highest mean concentration of pesticide analytes in reservoir sediments, but two other commonly used herbicides, alachlor and metolachlor, had mean concentrations well above 100 μg kg^{-1}. Additionally, methyl parathion, trifluralin, chlorpyrifos, and cyanazine all had mean concentrations above 1.0 μg kg^{-1}. Only two current-use pesticides were not detected in measurable quantities in reservoir sediments.

Sediments in the debris jam showed similar occurrence of measurable pesticides as samples from other areas, with a 14% detection rate (46 of 328 potential). Here, as in the other sediments, BHC and DDT analytes were found in highest concentrations (Table 4). Aldrin (1.784 μg L^{-1}) was the only other compound found with a mean concentration above 1.0 μg kg^{-1}, though heptachlor was nearly so (0.947 μg L^{-1}. Still, only eleven of the 25 priority pollutants were measured in our samples from the debris jam. Furthermore, only four of the thirteen other pesticides we sought at the jam were detected, with atrazine yet again having the highest observed

Table 5. Mean concentration (μg L-1 or μg kg-1) of current-use and other pesticides in watershed components of the Yalobusha River and Grenada Reservoir. ND = not detected. X = not tested.

Herbicides in BOLD	**Water**		**Sediment**		
Insecticides in NORMAL	**Watershed**	**Reservoir**	**Watershed**	**Reservoir**	**Debris Jam**
ALACHLOR	**0.011**	**ND**	**0.863**	**157.381**	**ND**
ALDICARB	ND	X	ND	X	X
ATRAZINE	**1.513**	**0.317**	**3.698**	**322.257**	**3.697**
BIFENTHRIN	0.011	0.016	0.364	0.120	0.173
CHLORFENAPYR	0.014	0.059	0.413	ND	ND
CHLORPYRIFOS	0.002	< 0.001	0.023	12.528	ND
CYANAZINE	**0.007**	**0.131**	**0.994**	**1.523**	**ND**
CYFLUTHRIN	0.028	X	0.002	X	X
CYHALOTHRIN-LAMBDA	0.007	0.008	0.656	0.324	ND
DELTAMETHRIN	ND	X	X	X	X
FIPRONIL	0.001	0.003	ND	X	0.913
FIPRONIL SULFONE	0.004	ND	0.195	X	1.814
FLUOMETURON	**ND**	**ND**	**1.379**	**ND**	**X**
METHOXYCHLOR	ND	X	0.089	X	X
METHYL PARATHION	0.015	0.014	0.626	55.977	ND
METOLACHLOR	**0.090**	**0.118**	**0.005**	**127.388**	**ND**
METRIBUZIN	**ND**	**X**	**X**	**X**	**X**
MIREX	0.021	X	0.658	X	X
NORFLURAZON	**0.021**	**X**	**0.658**	**X**	**X**
PENDIMETHALIN	**0.001**	**0.001**	**0.223**	**0.148**	**ND**
TEFLUTHRIN	ND	X	X	X	X
TRALOMETHRIN	0.055	X	ND	X	X
TRIFLURALIN	**0.001**	**0.031**	**0.018**	**27.210**	**ND**

mean concentration (3.697 $\mu g\ L^{-1}$), a concentration similar to other locations in the watershed outside of the reservoir. The other three detected compounds were bifenthrin, fipronil and fipronil sulfone.

Conclusions

The Yalobusha River drainage upstream of Grenada Lake is representative of mixed cover, hill-land watersheds that empty into the Mississippi River floodplain. Watersheds like this one experience runoff and associated contaminants from both agriculture and municipalities. In addition to field erosion, deeply incised streams in the watershed provide an additional source of suspended sediment.

As expected, runoff in winter and spring provided the highest concentrations of suspended sediments. All concentrations above 800 mg/L were detected from late November through February. In-stream sediment sources provided additional material so that wet seasons were almost identical to each other in elevated suspended sediment concentrations, while dry seasons had considerably lower mean concentrations of moving sediments.

In addition to the river, its tributaries, and Grenada Lake, a large obstruction, a debris dam, in the channelized portion of the river bordering Grenada Lake provided insight into riverine (lotic) and lake (lentic) effects on sediment, metals, and pesticides. The debris dam slowed water so that sediments settled in both the main river stem and wetlands/floodplain to either side on the main channel. The same depositional phenomenon occurred in the reservoir. Overall, sediment deposition in the reservoir was not cause for concern.

Metals and pesticides were detected in 12-15% of water and sediment samples in all sampling habitats. Atrazine (corn and cotton herbicide) was ubiquitous; BHC (beta), an organochlorine insecticide, had concentrations which were high in all sediments. As expected, DDT, its derivatives and Aldrin were also evident in sediments.

Overall, pesticide and metal concentrations were well below levels of concern. A few insecticides and herbicides had sporadically high concentrations, especially in sediments, which merit future scrutiny. Atrazine was ubiquitous.

Corresponding Author:
Charles M. Cooper
Ecologist, USDA–ARS–National Sedimentation Laboratory
Water Quality and Ecology Research Unit
PO Box 1157, 598 McElroy Drive
Oxford, MS 38655
Phone: 662.232.2935
Fax: 662.232.2988
Email: ccooper@msa–oxford.ars.usda.gov

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Measuring Streambank Erosion Due to Groundwater Seepage

John W. Fuchs[1], Garey Fox[1], Glenn Wilson[2], Andrew Simon[2], Eddy Langendoen[2]

The National Water Quality Inventory continues to report sediment as the most or at least one of the most severe pollutants of surface waters. Excessive sediment causes numerous water quality problems. It increases the potential for downstream flooding, diminishes water quality and destroys aquatic habitat. The two primary sources of sediment entering streams are erosion from adjacent landscapes and erosion of streambank sediment. Research on upland erosion has led to methodologies for controlling erosion from adjacent landscapes such as the use of best management practices. However, excessive sediment continues to be a problem throughout the United States. USDA–ARS scientists report that streambank material in many watersheds may contribute as much as 80% of the total sediment, especially in some watersheds in Mississippi. There exists a lack of information on one of the basic mechanisms governing sediment input to streams from the streambank: erosion by groundwater flow or seepage erosion. The importance of seepage erosion is not understood at this time, even though such an erosion mechanism has been observed throughout the United States. The objective of this research was to characterize streambank properties where seepage erosion contributes sediment to streams and causes bank failure and quantify subsurface flows and seepage erosion of bank sediment that results in bank failure at Goodwin Creek. Subsurface flows and sediment loads were quantified using lateral flow collection pans placed against exposed faces of the stream bank at pre–identified seepage locations when seepage was occurring. Subsurface flow and sediment concentrations were measured following rainfall events at both sites. These measurements were correlated to precipitation data, soil pore–water pressure measurements, and stream stages to ascertain the timing and importance of seepage erosion.

[1] Department of Civil Engineering, University of Mississippi, University, MS;
[2] USDA–ARS National Sedimentation Laboratory, Oxford, MS

Influence of Slope on Streambank Seepage Erosion

Leili Gordji[1], Garey A. Fox

Erosion by lateral, subsurface flow is known to erode streambank sediment in numerous geographical locations. However, the role of seepage erosion on mass failure of streambanks is not well known. Previous research has investigated the mechanisms of erosions by concentrated, lateral subsurface flow and developed a sediment transport model for seepage erosion. As a continuation of this earlier research, slope destabilization driven by lateral, subsurface flow was studied to investigate the impact of slope on the suggested sediment transport model and to verify the model. Laboratory experiments were performed using a two-dimensional soil lysimeter. The experiments were conducted on two soils: a field soil (sandy loam) from Little Topashaw Creek (LTC) in Northern Mississippi and a sieved sand with greater sand content and less cohesion than the field soil. Simulated bank profiles were packed in the lysimeter with slopes of 60°, 75°, and 90°. Three zones of erosion were observed during the experiments: fluvial, sapping, and undermining zones. As opposed to the cohesion less sand soil, for LTC sand, a tension crack was not observed and bank failure did not occur for the 60° bank angle suggesting the importance of considering cohesion in bank failure analyses. Power law sediment transport models, derived based on a dimensionless sediment discharge and dimensionless flow shear stress in previous research, was investigated for dependency on slope. Observed sediment concentration data were compared to predicted sediment concentration data from the power law sediment transport model based on measured discharge. This research suggests that seepage erosion may be of critical importance for estimating sediment loads into streams by streambank failure even at relatively small streambank slopes.

Kaywords Sediments, Groundwater, Hydrology, Water quality

[1]Graduate Research Assistant, Department of Civil Engineering, University of Mississippi, 203 Carrier Hall, P.O. Box 1848, University, MS 38677-1848, E-mail: lgordji@olemiss.edu

Corresponding Author:
Dr. Garey Fox, Assistant Professor
Department of Civil Engineering
University of Mississippi
208 Carrier Hall, P.O. Box 1848
University, MS 38677-1848
Phone: 662-915-5618
Fax: 662-915-5523
E-mail: gafox@olemiss.com

Removal of Copper, Chromium, and Arsenic by Water Hyacinths

Cyle Keith[1], Hamid Borazjani[1], Susan V. Diehl[1], Y. Su[2], and B.S. Baldwin[3]

The removal of different concentrations of toxic metals by water hyacinths (Eichhornia crassipes) from a simulated aqueous environment was studied in an outdoor experiment. The hyacinth's tissues were analyzed to evaluate the removal of copper (Cu), hexavalent chromium (Cr VI), and arsenic (As) from CCA (Chromated Copper Arsenate) contaminated water (5, 10, and 20 mg/L arsenic content, 7, 14, and 28 mg/L chromium concentration, and 2.5, 5, and 10 mg/L copper concentration) over a 17 day period. The vigor of the plants was also recorded during this period. The results showed that the hyacinth was not a suitable plant to remediate arsenic and copper. Arsenic removal for 5, 10, and 20 mg/L concentrations were 4.8, 4.7, and 0% respectively and copper removal for 2.5, 5, and 10 mg/L showed 53, 0, and 0% respectively. However, water hyacinth did show promising results as a hyper-accumulator of chromium. Percent chromium removal for 7, 14, and 28 mg/L contaminated water was 72.3, 21, and 19% respectively. The amount of copper in the containers with water hyacinths present was higher than the containers without plants (controls) indicating that the water hyacinth exudates might cause copper to stay in aqueous solution longer. As for the vigor of the plants, 5 and 10 mg/L arsenic concentrations damaged the plants somewhat over the 17 day period but overall these plants remained alive for the duration. Plants that were treated with 20 mg/L arsenic began to wilt and change color after day 1 and by the end were lifeless.

Keywords: Treatment, Toxic metals, Water quality

Introduction

One of the major problems the public faces is contamination of drinking water. There are many industries that may or have contributed to the contamination of waterways by discharging toxic metals into rivers and streams.

One wood preservative that is used in the Forest Products industry is CCA (Chromated Copper Arsenate). During the spring of 2001, Florida newspapers began to report that arsenic was leaching from playground equipment made of CCA treated wood (Hauserman, 2001). In Dec. 2003, CCA treated wood was voluntarily taken off the residential market by the wood-treating industry.

Of the three metals in CCA, only two are extremely toxic: hexavalent chromium and arsenic. Chromium is a naturally occurring element found in rocks, animals, and plants. It is usually present in several different forms, the most common of which are hexavalent chromium (extremely toxic) and chromium III (nontoxic). Chromium III is essential for breaking down sugar, fat, and protein inside an animal's body, thus making it vital for good health. In contrast, hexavalent chromium can be detrimental to the health of anyone exposed over long periods of time. Inhaling or ingesting hexavalent chromium over time can cause nosebleeds, ulcers, convulsions, kidney and liver damage, various cancers, and/or death (ATSDR online).

Chromium can also contaminate soil, sediment, and groundwater (Mei et al., 2002). It is because of this pollution that scientists seek new methods to clean up waterways that contain chromium. In one study, water-hyacinths (Eichhornia crassipes) were used to accumulate hexavalent chromium from contaminated water. As the plants were put under an x-ray spectroscope, the plants had converted hexavalent chromium to chromium III in the lateral roots of the plant, thus detoxifying the water significantly (Lytle et al., 1998).

Arsenic is the 20th most abundant element found naturally in the Earth's crust. In very small quantities, it plays an essential role in animal metabolism but in large amounts it is damaging due to its carcinogenicity (Pickering et al., 2000). Arsenic is a heavy metal that can cause significant health problems by primarily attacking the immune system (Huang et al., 2004). When arsenic is ingested, it also increases the risk of bladder and prostate cancers. It can also cause other health disorders such as a decrease in hearing ability, skin thickening, and disturbances to the nervous system.

Because of the health risks involved with arsenic contamination in the drinking water in the US, the Environmental Protection Agency (EPA) had set a maximum arsenic contaminant level standard for drinking water at 50 ppb. Due to growing concerns from the general public and government officials the EPA considered dropping the standard to 10 ppb (EPA online, 7/13/04).

Copper, a metal that occurs naturally in rocks, soil, water, and air throughout the environment, is an essential element in plants and animals (including humans). Therefore, plants and animals must absorb some copper from eating, drinking, and breathing (ATSDR online). Although copper is not as toxic as hexavelent chromium or arsenic, it can be potentially serious if high levels are present in drinking water. The most common symptoms of copper toxicity are injury to red blood cells and lungs, as well as damage to liver and pancreatic functions (OR Online, 3/19/06). Long-term exposure to copper can also cause irritation of the nose, mouth and eyes, as well as headaches, stomachaches, dizziness, vomiting and diarrhea (Lenntech Online, 3/22/06).

At the present time, the most common way to clean water contaminated with heavy metals is a coagulation/filtration method that involves removing pollutants by chemically conditioning particles to agglomerate into larger particles that can be separated and settled, followed by running the contaminated water through various filters that trap the pollutants and hold them for disposal. One of the major problems with this method is the sludge-like by-product that is produced as a result of the settled and trapped contaminants (Huang et al., 2004). Use of these methods for cleanup/disposal of contaminated water is very expensive and disruptive to the habitats that surround the water (Tu and Ma, 2002). Scientists are searching for new and economic ways to alleviate this contamination problem.

One new and promising method that has been drawing interest for many years is called phytoremediation or phytoextraction. Simply put, phytoextraction is the use of plants to remove contaminants from water by pulling the contaminants out of the water through the root system and into the plant body (Huang et al., 2004). This makes disposal easier and much less expensive because properly destroying the plants and the contaminants held within is a relatively simple process. The major contaminants that are removed from water by plants are various carcinogenic metals such as copper, chromium, arsenic, mercury, etc.

One aquatic plant that has been studied is the water-hyacinth (Eichhornia crassipes). It is a floater plant with large, glossy leaves and a small blue or yellow flower in the center of its vegetative mat (USGS online, 8/22/03). It is native to the southeastern US and California and one of the most invasive aquatic plant species ever introduced into the US (USGS online, 8/22/03).

The objectives of this study were: 1) to evaluate the removal of arsenic, chromium, and copper by water-hyacinths in water contaminated with different CCA concentrations and 2) to observe the effects of these metals on the health and vigor of the plants.

Methods and Materials

Plant Preparation

A mat of water-hyacinths were purchased and placed in a plastic container filled with de-ionized water. The water hyacinth mat was then separated into individual plants.

Preparation of CCA Solution

A CCA solution consisting of 17% arsenic, 24% chromium, and 9% copper was obtained from the Mississippi State University Forest Products Laboratory. A stock solution was prepared by adding de-ionized water to the CCA until desired metal concentrations were obtained.

Project Setup

Twenty-four quart size mason jars were filled with 450 ml of water and 50 ml of Miracle-Grow solution (prepared according to label). After the solution was added, a black line was drawn on the jars so a 500ml water level could be kept constant. Plants were placed into jars three days prior to the application of the different concentrations of stock solution so the plants could establish themselves inside the jars. The jars were then sorted into three treatments with six jars in each treatment and a control group with the last six jars. Each group had three jars with plants and three jars without plants (figure 1).

Figure 1. Photo taken of water-hyacinths in different treatments and their controls.

For group 1 treatments (5 ppm arsenic, 7 ppm chromium, and 2.5 ppm copper), the water-hyacinths were removed temporarily from the jars and 1 ml of the above concentration the stock solution was added to the water/Miracle grow solution in each of the six jars. The solutions were then mixed and the plants were placed back into the jars. For Group 2 (10 ppm arsenic, 14 ppm chromium, and 5 ppm copper) and group 3 (20 ppm arsenic, 28 ppm chromium, and 10 ppm copper) treatments, 2 or 4 ml of stock solution was added to each jar in the same manner. The solutions were mixed and the plants were placed back into the jars. In the fourth group, the water-hyacinths were placed into the three plant jars and no stock solution was added to these six jars (controls). All 24 jars were then placed in an area outdoors and observed for 17 days.

Project Upkeep

Every 1-2 days, the plants were checked and de-ionized water was added to each jar so the 500 ml water level line remained constant. Pictures were taken also to document the health and vigor of the plants.

Preparation of Samples for Analysis

The mason jars were assorted into eight groups with three jars representing the respective arsenic concentration. For each of the eight groups, a 30 ml composite sample was generated by placing a 10 ml aliquote from each of the three jars and combining them in glass bottles. The eight composites were logged into the lab notebook and transported to the Mississippi State University Diagnostic Instrumentation & Analysis Laboratory for metal analysis. Also, plant and root tissues were dried, crushed, and digested with HNO3 and H2O2. The digested solution was filtered and then analyzed for As, Cr and Cu by atomic absorption analysis (Sridhar et al., 2002).

Results

Table 1 shows the levels of arsenic in the water after 17 days of observation. The plants appeared to have little remediatory affect on arsenic contaminated water. The levels of arsenic in the water after analysis remained around the same concentration (5 ppm) as the initial levels at the beginning of the experiment.

The greatest change in concentration occurred with chromium levels in the water samples. The data in Table 2 show that a significant amount of chromium was removed from the water.

Results from copper treatements show a different outcome in the data values in Table 3. In the water samples that contained plants, the copper values were higher than samples that without plant. One explanation could be that exudates on the roots of the plants may have kept copper in suspension.

The health and vigor of the plants were observed over the 17 days of this experiment. At the end, the plants that were treated with 5 mg/L arsenic stock solution were affected by a slight wilting of the stems and a rusty coloration on parts of the leaves (figure 2). The same results occurred at 10 mg/L arsenic treatment, except that there were more rust coloration present on the leaves (figure 3).

Table 1. Arsenic uptake by water-hyacinths during a period of 17 days.

Sample ID	Arsenic	% Removal vs. control
Ctr. w/ plant	0.056	0
Ctr. w/o plant	0.03	
5 ppm w/plant	5.4	4.8
5 ppm w/o plant	5.67	
10 ppm w/plant	10.2	4.7
10 ppm w/o plant	10.7	
20 ppm w/plant	20.4	0
20 ppm w/o plant	19.6	

The plants in the 20 mg/L arsenic treatment were near death after the first week as the leaves totally wilted and became completely covered with rust. In group three (20 ppm treatment), two of the three plants were found dead in the first week due to root damage from the higher metal concentrations. After 17 days, these plants were the only plants in the experiment to die (figure 4). Plant injuries could have attributed to much lower chromium uptake for 10 and 20 mg/L treatments.

Figure 2. Photo taken shows the health and vigor of the water-hyacinth at a treatment of 5ppm.

Figure 3. Photo taken shows the health and vigor of the water-hyacinth at a treatment of 10ppm.

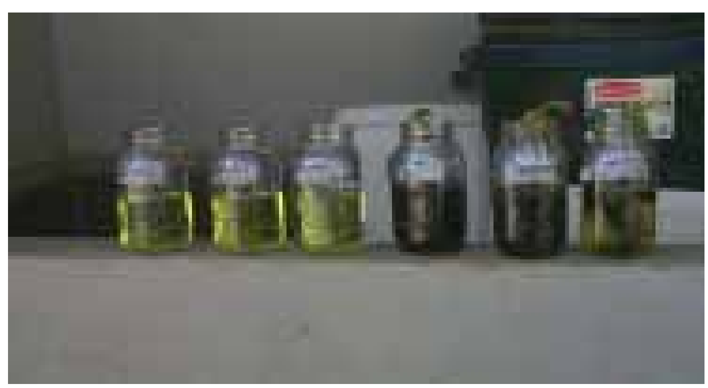

Figure 4. This photo shows the health and vigor of the water-hyacinth at a treatment of 20ppm.

Table 2. Chromium uptake by water-hyacinths during a period of 17 days.

Sample ID	Chromium	% Removal vs. control
Ctr. w/ plant	0.067	0
Ctr. w/o plant	0.014	
7 ppm w/plant	1.98	72.3
7 ppm w/o plant	7.16	
14 ppm w/plant	10.4	21
14 ppm w/o plant	13.1	
28 ppm w/plant	20.7	19.1
28 ppm w/o plant	25.6	

Table 3. Copper uptake by water-hyacinths during a period of 17 days.

Sample ID	Copper	% Removal vs. control
Ctr. w/ plant	0.567	0
Ctr. w/o plant	0.266	
2.5 ppm w/plant	0.949	53
2.5 ppm w/o plant	2.02	
5 ppm w/plant	3.45	0
5 ppm w/o plant	2.9	
10 ppm w/plant	5.38	0
10 ppm w/o plant	2.39	

Discussion

The results show how arsenic, chromium, and copper are affected by a floater plant like water-hyacinth. Water-hyacinths do not seem to remove large amounts of arsenic or copper from contaminated water. Post-analysis levels of arsenic were close to the beginning levels of arsenic, and it was concluded that water-hyacinth will not remediate arsenic contaminated water.

The water-hyacinth appeared to be a good choice for removing chromium from polluted water. At low concentrations, the plant removed about 70% of the chromium in the water (Table 2). As the concentrations increased, the plant appeared not to be able to take up as much percent chromium, but the amount the plant was able to take up was still significant (around 25%). This reduction could be attributed to plant tissue injuries caused by high arsenic levels.

The copper data showed directly opposite results when compared to chromium and arsenic. It seems that when a plant such as a water-hyacinth is in the presence of large amounts of copper, levels of copper in water are greater than when copper is present without any plants. With no plants present, copper is much more likely to precipitate out of an aqueous solution (Tucker and Hargreaves, 2003). Water samples for this study were collected from the center of the jars, if this copper had precipitated out, the analysis would show that levels were low in the jars that had no plants. The samples that had plants present had the higher values because of how plants and copper react with each other. Root exudates from the water hyacinth could have kept copper soluble or suspended in water, thus showing higher amount of copper in the samples with plants.

In conclusion, if a body of water is polluted with CCA and phytoremediation is the choice to clean it, water hyacinths are not the plants to be used for copper and arsenic removal. Although they might be effective against chromium, they do little to control an arsenic or copper contamination. To remedy this situation, use of more than one plant species could be the answer. More experiments on other aquatic plants are needed to determine which plants are the best to uptake arsenic or copper, then the two could be combined to help remediate the problem.

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[1]Forest Products Department, Forest and Wildlife Research Center, Mississippi State University, MS
[2]Diagnostic Instrumentation & Analysis Laboratory (DIAL), Mississippi State University, MS
[3]Plant & Soil Science Department, Mississippi State University, MS

Precise Elevation Modeling for Hydrologic Decision Support System in the Cabuçu de Baixo Urban Basin in Sao Paulo City–Brazil

Rodrigo A. A. Nobrega[1, 2], Mario Thadeu Leme de Barros[1], Jose Alberto Quintanilha[1], Charles G. O'Hara[2]

The Sao Paulo Metropolitan Region is composed by 35 municipalities, with a population estimated in 17 million people. The post–70's intense urbanization of the area has caused many hydrologic problems. Flood frequency has increased, as well as peak flows during the rainy season. Some hydraulic improvements have aimed to reduce the damages, however with limited success. The lack of integration decision–making can be considered as one of the most important causes of the increases in urban flooding. However, from a technical point of view, the lack of precise information maybe can be considered the most important factor vitally needed for delivering desired engineering solutions.

Regarding the hydrological investigation, urban basins are more affected by land cover changes than their rural counterparts. The Cabuçu de Baixo watershed is a typical Sao Paulo City area, characterized by recent and unplanned occupation. Dense urbanization can be found intermixed with some parts of preserved nature located in the upstream part of the basin. During the last three decades, its land cover was abruptly changed. Irregular and unplanned settlements were formed over hills and valleys, replacing the vegetation. Later, the number of buildings increased. Implementation of transportation, electrical and hydrological utilities have been required. Sometimes, many of these implementations were done without the use of actualized maps. Due the characteristics of these areas, the use of recent technologies like LIDAR or IFSAR could produce faster results for the geo–database actualization. However, regarding the difficulties for developing countries, the absence of maps, or the amount of obsolete data, has resulted in the adoption of alternative methodologies, aimed at faster and less onerous paths without the needed commitment to a quality solution.

Simulating surface water and flooding based on current geo–technologies is an efficient tool for planning and management for urban environments. In accordance with the principle that "the more actualized the reference database, the more accurate the investigation" this paper illustrates in a real–world example the necessity of precise geoinformation for urban basin investigation. A methodology of terrain modeling for hydrological studies based on breaklines extraction is reported. The coarse detailing was used for the mountainous regions and the finer one for other areas. The meadow areas were prioritized for precise simulation of flooding. Very high resolution orthophoto and contour maps were generated and applied to the Hydrologic Decision Support System. The final analysis shows the viability of the methodology for studies of urban hydrology.

Keywords: *Digital terrain model, flooding, surface water, urban basin, simulation*

[1] Escola Politécnica da Universidade de Sao Paulo
Av. Prof. Luciano Gualberto, trav 3, 380 – CEP 05508–900 São Paulo SP, Brazil +55 11 3091–5504
[2] GeoResources Institute, Mississippi State University
2nd Research Boulevard – ERC, Starkville MS 39759, USA +1 662 325–2067

Presenting Nutrient Data Online Using a Customized ArcIMS® Map Viewer

Harriet Perry[1], Christine Trigg[1], John Anderson[1], Faye Mallette[1], G. Alan Criss[2], Barbara Viskup[3]

The ability to easily share geographical and associated environmental data among researchers, managers, planners, and the concerned public is an important and powerful tool. Data on water quality was collected during 2003–2004 by the Center for Fisheries Research & Development, The University of Southern Mississippi, as part of a larger series of studies to establish numerical criteria for nutrients in the State's coastal waters. The water quality study was made accessible using an interactive online Geographic Information System. The first step in the creation of the web site was the development of a custom ArcIMS® DHTML viewer. The ArcIMS DHTML viewer is a collection of HTML and JavaScript files that dynamically generate map images of a specified Map Service based upon user interaction. The custom viewer maintains the uniform appearance of web pages in the web site and its more efficient page layout provides more area for map images. The custom viewer becomes a template to generate all subsequent web sites, allowing rapid deployment of new Map Services. (A Map Service is a process that runs on one or more ArcIMS® Spatial Servers.) Map Services are based on instructions written in ArcXML that specify the data sources to be used in the map and how to symbolize the different data layers.

[1]Center for Fisheries Research & Development, The University of Southern Mississippi
[2]Gulf Coast Geospatial Center, The University of Southern Mississippi
[3]Mississippi Department of Environmental Quality

Biological Remediation of Oriented Strand Board (OSB) Waste Water

Lawrence Brad Strombock[1], Susan V. Diehl[1], Hamid Borazjani[1], B. S. Baldwin[2], M. Lynn Prewitt[1]

Oriented strand board (OSB) is a wood composite product made from wood strands heat bonded with resin adhesives and waxes. In 2001, 22 billion square feet of OSB was produced in the US. The OSB manufacturing process generates large amounts of wastes including wood, water, resins, waxes and organic compounds such as terpenes, resin acids, phenol formaldehyde resins, and other wood leachates. High levels of organic materials as measured by its biological oxygen demand (BOD) determines whether this water is suitable for discharge. High BOD levels correlates to a high organic matter content and low oxygen availability. The wastewater used in these studies was collected from an OSB manufacturing plant and contained an initial BOD concentration of 1600mg/L. The first study evaluated free cell bioreactors to reduce the BOD concentration in this process waste water using four treatments with three replications within each treatment. Treatment one was the control treatment; treatment two added aeration; treatment three consisted of aeration plus an indigenous microbial consortium; treatment four consisted of aeration, an indigenous microbial consortium, plus a nitrogen–phosphorus based fertilizer. BOD levels were determined on day 0 and every 15 days until day 105. Toxicity was determined on day 0 and day 105. Results revealed after 30 days, a 57% BOD reduction in treatment 1, 72% and 73% reduction in treatment 2 and 3 respectively and 84% reduction in treatment 4. By day 105, all treatments showed a 93% BOD reduction and treatment 2 showed the greatest toxicity reduction.

The second study was to determine if plants could reduce BOD levels in OSB process waste water. Water hyacinth, Chinese water chestnut, Azolla, Small duckweed, Bullrush, Beak–rush, soft rush, Bald cypress and Black Willow were screened for survival in high BOD process waste water. Chinese water chestnut, Small duckweed, soft rush and water hyacinth survived the initial screening study and were further evaluated. Two clumps of each plant type were placed in fiberglass tubs containing OSB process waste water or natural lake water. Treatment one consisted of floating plants and treatment two consisted of emergent plants. After 75 days the BOD levels were reduced by 84% in the floating plants compare to 96% in the emergent plants.

Keywords: Treatment, Wastewater, Water Quality, Wetlands

[1] Forest Products Department, Forest and Wildlife Research Center, Mississippi State University
[2] Department of Plant and Soil Sciences, Mississippi State University

Corresponding Author:
M. L. Prewitt, Assistant Research Professor
Forest Products Department
Mississippi State University
100 Blackjack Road
Starkville, MS 39759
Phone 662–325–4083
E–mail: lprewitt@cfr.msstate.edu.

Mississippi Watershed Characterization and Ranking Tool (MWCRT)

John Storelli

The Mississippi Watershed Characterization and Ranking Tool (MWCRT) is an easy to use spatial tool of broad applicability. The MWCRT uses geographic information systems (GIS) to assess readily available statewide spatial datasets within the watersheds of major river basins. The watersheds are based on the 12 digit hydrologic unit code dataset created by the United States Geological Survey (USGS).

The tool helps users determine which watersheds are of higher value based on the presence of important natural resource features. These features are described by land uses and surface water uses within the watersheds of a major river basin. Each spatial layer is placed into a broad category to determine its resource value on the environment, its impact on human welfare and to assess the stressors placed on each watershed.

Each category can be used to describe or characterize the natural resource datasets within the watersheds of the major river basins. This information is used to generate a ranking system that determines scores for each watershed. The ranking system is generated based on the raw spatial data expressed as observations for point data, miles for line data and acres for polygonal datasets. These raw data values for each dataset are normalized using a linear scaling transform equation and assigned weights by relative importance. The final output provides a value for each watershed. Each watershed can be ranked by individual dataset or by combining datasets to produce a ranking system by category.

The MWCRT provides a way to identify watersheds of interest, make meaningful decisions and to prioritize watersheds for protection and restoration activities. The purpose of MWCRT is to provide the Mississippi Department of Environmental Quality (MDEQ) and its state and federal agency partners with a tool to help manage the state's water resources.

Keywords: Management & Planning, Conservation, Surface Water, Water Quality

Corresponding Author:
John Storelli, GIS Analyst
Mississippi Department of Environmental Quality
2380 Highway 80 West
Jackson Mississippi 39202
Phone 601-961-5375
E-mail: John_Storelli@deq.state.ms.us

Linking Watersheds: A Pilot Project in the Tombigbee–Mobile Bay Basin

Mary Love Tagert[1], R. Jackson[2], L. E. Reed[3], J. A. Ballweber[1], W. H. McAnally[3]

Partnering among natural economic development, resource management, and conservation groups in the Tombigbee–Mobile basin offers the possibility for sustained collaboration on data and decision making for the basin. NOAA's Coastal Services Center supported a pilot study that included workshops of key inland and coastal institutions to identify and integrate priority geospatial data and facilitate more effective and sustained stakeholder engagement. The primary partners on the project were Mississippi State University, Alabama Department of Conservation and Natural Resources, Tennessee–Tombigbee Waterway Development Authority, US Army Corps of Engineers Mobile District, Tennessee Valley Authority, Alabama SmartCoast, Tombigbee River Valley Water Management District, and the NOAA Weeks Bay Reserve. Data on the spread and flow of aquatic nuisance species, water quantity and quality, boundary and buffer information for wetlands and watersheds, fish and wildlife/water resources overlap information, water use, navigation, and bathymetric data were identified as most needed. A project web site containing survey results, a pilot database, and tools has been created to facilitate communication and data sharing.

Keywords: Institutions & Policy, Management & Planning, Water Quality, Water Quantity

[1] Mississippi Water Resources Research Institute, Forest and Wildlife Research Center, Mississippi State University
[2] GeoResources Institute, Mississippi State University
[3] Civil Engineering, Mississippi State University

Corresponding Author:
Mary Love Tagert, Assistant Research Professor
Mississippi Water Resources Research Institute
Mississippi State University
Box 9652; Mississippi State University
Phone 662–325–8081
E–mail: MLTagert@GRI.MsState.Edu

Synthesis of Manganese Oxide Coatings for Adsorption of Trace Metals from Ground Water

Amey. S. Tilak[1], Clint W. Williford[2], Garey A. Fox[1], and Terry Sobecki[3]

Manganese oxide (MnOx) occurs naturally in soil and has high affinity for trace metals such as lead, chromium, cadmium and zinc. Such heavy metals, when present in groundwater, will cause health risks if they enter aquifer drinking water sources. Our aim is to produce and characterize manganese oxide coatings on aquifer soil materials. The long term goal is to form a Permeable Reactive Barrier (PRB) to adsorb trace metals. Syntheses of manganese oxide coatings were carried out in the laboratory using continuous (column) reactor and batch reactor experiments on clean Ottawa sand. In the column experiment, manganese and bleach solutions were cycled alternately through a column of sand. The pH and ORP (Oxidation Reduction Potential) were monitored during coating process. The pH ranged from (4–9). The ORP was (600–800) mV when passing bleach and manganese solutions, and, (100–300) mV when passing DI–water. The coating process was carried out for increasing numbers of the cycles 24, 48 and 72. Flow rate was kept constant at 4ml\minute. After the coating process, a lead solution (50mg\L) was passed at 4ml\min through the coated sand to determine the lead adsorption capacity on manganese oxide coated sand. A lead selective electrode was used to plot breakthrough curves. The amount of lead adsorbed for 24, 48 and 72 cycles was 500mg\kg, 760mg\kg and 2106mg\kg respectively. An excavated aquifer soil will be investigated for coating synthesis and lead adsorption. The batch studies performed on Ottawa sand used three oxidants: ozone, hydrogen peroxide and bleach. For batch synthesis pH was investigated for a range of (6–9), and the initial amount of manganese was varied from 0.3 to 3.0g Mn per 100g sand. The manganese oxide formed on the sand ranged from 63.6mg\kg to 10372.8 mg\Kg. Results of batch syntheses indicate that manganese oxide on Ottawa sand increased with pH from 6 to 9 for hydrogen peroxide, ozone and bleach. Manganese synthesis was observed at the lower pH of 6 for bleach. Select coated sands from batch syntheses were also tested in column runs to determine lead adsorption. The batch synthesis was repeated three times on the same sand sample to examine the sequential buildup of manganese oxide coatings. The adsorption of cadmium, chromium, and zinc on the manganese coated sand was measured to determine relative/competitive adsorption capacity. Along with the experimental approach, modeling will allow us to estimate the service life for the Permeable Reactive Barrier to adsorb heavy metals.

[1] Amey S. Tilak, Graduate Student, Department of Civil and Environmental Engineering, Univ. Mississippi
[2] Dr. Clint W. Williford, Faculty in Department of Chemical Engineering, Univ. Mississippi
[3] Dr. Garey A. Fox, Faculty in Department of Civil and Environmental Engineering, Univ. Mississippi
[4] Dr. Terry Sobecki, Head of Environmental Processes Branch, ERDC, U.S. Army Corps of Engineers Vicksburg MS 39180–6199.

Corresponding Authors:
Dr. Clint W. Williford, Department of Chemical Engineering
232 Anderson Hall
University of Mississippi, MS 38655
Phone 662–915–5348
E–mail: drwill@olemiss.edu

Amey S. Tilak, Graduate student
Department of Civil and Environmental Engineering
University of Mississippi MS 38655
E–mail: astilak@olemiss.edu

Analysis of Mississippi and Alabama Water and Sediment Samples after Hurricane Katrina with H4IIE and YES Assays

Beth Emerson[1], Katherine Argote[2], Amit Chaudhary[2], James Weston[2], Shabana Khan[3], Kristie L. Willett[2]

Hurricane Katrina struck the Mississippi and Alabama Gulf Coasts on August 29, 2005. Following the extensive storm surges and flooding, this study was designed to test the toxicological effects caused by the storm. On September 13, 2005, the first water and sediment samples were collected with monthly collections following. Samples were collected from ten sites along the Alabama and Mississippi coasts: Mobile, AL, and Biloxi, Grand Bay, Gulfport, Ocean Springs, and Pascagoula, MS. Samples were extracted and filtered for analysis employing the H4IIE rat hepatoma and the Yeast Estrogen Screen (YES) Assay. The H4IIE assay is used to determine the induction of CYP1A by the presence of aromatic hydrocarbons. CYP1A activity is measured by the deethylation of ethoxyresorufin into a fluorescent pink product, resorufin (EROD). Maximal EROD activities for benzo(a)pyrene (BaP, 0.5 nM) and TCDD (0.5 nM) were 6.1 ± 0.05 and 3.0 ± 0.11, respectively. Ultimately activity from Katrina extracts will be expressed as percent of the highest concentration of BaP response measured in the EROD assays. The YES assay is used for predicting the presence of potential endocrine disruptors based on their binding and activation of the human estrogen receptor. The estrogen equivalents for the water samples ranged from non-detectable to 5.5 ng/L at a Grand Bay, MS site (site #8). The suspended sediment samples ranged from non-detectable to 1.4 ng/L at a Gulfport, MS site (site #2). By the February sampling, estrogen equivalencies for all but the Back Biloxi Bay site were below detection limits. Further analysis will be conducted with the H4IIE assay to measure CYP1A induction of the water and sediment samples. Results from the YES assay will be compared to NOAA's pre-Katrina data, as available, with monthly collections and analytical chemistry data to follow.

Corresponding Authors:
[1]Chemistry and Biochemistry
[2]Pharmacology and Environmental Toxicology
[3]National Center for Natural Products Research
University of Mississippi
University, MS

Session A: Katrina—Coastal Impacts

Joe Jewell, Moderator

Bill Cosgrove
Water Quality Study of Bays in Mississippi Following Hurricane Katrina and Rita

Charlie Demas
USGS Louisiana Water Science Center Assessment of the Water–Quality Impacts of Hurricane Katrina in Louisiana

Richard Rebich
Summary of Bacteriological Data Collected at Coastal Mississippi Sites Following Hurricane Katrina, September–October 2005

Jim Weston
Storm Effects on Estuarine Water Quality

Van Wilson
Summary of Hurricane Katrina Storm Surge on the Mississippi Gulf Coast

Water Quality Study of Bays in Mississippi Following Hurricanes Katrina and Rita

Bill Cosgrove

EPA's Region 4 Science & Ecosystem Support Division (SESD), in cooperation with the Mississippi Department of Environmental Quality (MDEQ), conducted a water quality study in the rivers and bays along the Mississippi coast following Hurricanes Katrina and Rita. The study was completed during the period September 26–30, 2005. The study area encompassed major bay systems on the Mississippi coast including Bangs Lake, Bayou Casotte, the Pascagoula and West Pascagoula River systems, the Back Bay of Biloxi, St. Louis Bay, and the Pearl River. The objective of this study was to provide sediment and water quality data in each major bay system along the Mississippi Sound. Flow was also measured at the seaward boundary of each system for estimating both conventional and toxic pollutant loadings entering the Mississippi Sound at the time of the study. This study was not designed to identify specific pollutant sources within each system or provide definitive information on the potential long term effects of the hurricanes on human or ecological health.

Findings from the EPA and MDEQ joint survey of coastal Mississippi following Hurricanes Katrina and Rita showed few detectable priority pollutant type compounds in the studied bays and rivers. In general, the pollutants detected were low in concentration when compared to EPA's National Ambient Water Quality Criteria for surface waters and the National Oceanic and Atmospheric Administration (NOAA) published effect levels for sediment. Dissolved oxygen concentrations were determined to be above the State's adopted minimum criteria at all but two of the thirty–nine surface water locations. Bacteriological densities at the study locations were less than EPA's promulgated enterococci criteria for coastal waters. Overall, the data collected by EPA shows that few water quality criteria were exceeded during the study. An exception was algal growth results in Back Bay of Biloxi and Bayou Casotte that exceeded 5 mg/l (dry weight). Dioxin results for the five sediment samples collected were all well below the EPA screening value for residential soils. The results of this study may be used as the basis for future targeted water quality studies by MDEQ and/or the EPA.

Keywords: Water Quality, Sediments, Surface Water

Corresponding Author:
Bill Cosgrove
US–EPA, Region 4, Science & Ecosystem Support Division
980 College Station Rd
Athens, GA 30605
(706) 355–8616
E–mail: cosgrove.bill@epa.gov

Summary of Bacteriological Data Collected at Coastal Mississippi Sites Following Hurricane Katrina, September–October 2005

Richard A. Rebich

On August 29, 2005, Hurricane Katrina devastated coastal Mississippi with 150 mile-per-hour winds and a storm surge in excess of 20 feet. Katrina moved inland and wreaked destruction on a broad swath of eastern Mississippi. Some eastern Mississippi counties were left without power and water, and some major roads were impassable for weeks. Disease transmission from contaminated water were major concerns.

As part of a multi-agency response to the disaster, the U.S. Geological Survey (USGS), in partnership with the Mississippi Department of Environmental Quality (MDEQ), acquired temporary space at a USGS facility at Stennis Space Center near Bay St. Louis, Miss., to operate a bacteriological laboratory. Weekly bacteriological samples were collected at 31 estuarine tributary sites and 13 beach monitoring sites in coastal Mississippi counties – Hancock, Harrison, and Jackson – for a period of 5 weeks beginning September 19, 2005. Samples were collected by MDEQ and USGS personnel and transported to the laboratory for analysis within 6 hours of collection. USGS personnel analyzed the samples primarily for enterococci, which is the standard fecal indicator bacteria for brackish waters (and can be used as a fecal indicator for freshwater). Enterococci concentrations were determined by the most probable value (MPN) method. About 14 percent of the enterococci concentrations were less than the detection limit, and 81 percent were lower than current standards. The highest enterococci concentration was 24,200 MPN per 100 milliliters, which occurred at Bayou Chico at Pascagoula, Miss. Concentrations at most of the bacteriological sites increased during the second week of sampling due to runoff associated with Hurricane Rita rainfall that occurred September 23-24, 2005.

The USGS also collected 19 water-quality samples at 12 inland freshwater sites for a period of 2 weeks starting on September 19, 2005. Sampling sites were located near established USGS stream gages. Physical properties of the streams were measured on site. Water-quality samples were collected, processed, and preserved on site according to standard procedures and then shipped to the USGS National Water Quality Laboratory in Denver, CO, for analysis, except for biochemical oxygen demand samples, which were analyzed by the MDEQ laboratory in Pearl, MS. Each sample was analyzed for multiple constituents including nutrients, major ions, trace metals, modern-use and polar pesticides, waste-water compounds, volatile organic compounds, and degradate organic compounds. Most detections were below current water-quality criteria for Mississippi streams.

Overall, the results from the bacteriological and water-quality samples indicated no systematic contamination in the sampled streams in the aftermath of Hurricane Katrina. This project demonstrated to the public that both Federal and State governments were concerned about public safety, were willing to respond and to respond quickly, and performed their missions under highly unfavorable conditions. The data from this project are now available online at http://pubs.usgs.gov/ds/ds174/.

Keywords: Water Quality, Nonpoint Source Pollution, Surface Water

Corresponding Author:
Richard A. Rebich, Hydrologist
U.S Geological Survey
308 South Airport Road
Jackson, MS 39208–6649
Phone: 601–933–2928
Fax: 601–933–2901
E–mail: rarebich@usgs.gov

Storm Effects on Estuarine Water Quality

James Weston

In 2005 the northern Gulf of Mexico experienced 1 tropical storm and 4 hurricanes, hurricane Katrina being the largest, while the frequency and severity of tropical storms is predicted to increase due to global warming (Webster et al., 2005). To better understand the impact of these pulse disturbances on nearshore marine communities temporally and spatially explicit data are needed. Within Grand Bay, MS, the National Oceanic and Atmospheric Administration (NOAA) and the Mississippi Department of Marine Resources established a National Estuarine Research Reserve (NERR) to monitor water quality and other estuarine conditions. Four automated dataloggers have been collecting water quality data in Grand Bay NERR since 2004. Temperature, depth, specific conductivity, salinity, pH, dissolved oxygen, dissolved oxygen percent saturation and turbidity are measured every 30 minutes. Because of Grand Bay NERR's temporal and spatial monitoring program a unique profile of storm related effects on abiotic factors has been captured but not analyzed. Water quality conditions before, during and after tropical storms and their change in magnitude and recovery to baseline conditions will be qualitatively and quantitatively assessed and compared within and between Grand Bay NERR sites. Data from last year, 2005, is currently being evaluated. Understanding the magnitude and duration of abiotic stressors in estuarine systems is a step toward developing better monitoring programs to assess ecosystem health and storm related disturbances.

Corresponding Author:
James Weston
Department of Biology and Environmental Toxicology
University of Mississippi
University, MS

Summary of Hurricane Katrina Storm Surge on the Mississippi Gulf Coast

K. Van Wilson, Jr.

Hurricane Katrina made landfall on the coast of Mississippi and Louisiana as a Category 4 hurricane during the early hours of August 29, 2005. Following the path of Hurricane Camille in 1969, Katrina moved into Mississippi with 140 MPH winds and a storm surge reported to be 30 feet above sea level. The USGS coastal monitoring network was destroyed along with most of the buildings and structures within the inundated area. The USGS Hydrologic Instrumentation Facility (HIF) at Stennis Space Center served as a refuge for hundreds of people during the storm. The facility operated without power, water, and communications for several days while continuing to provide shelter for the families of HIF staff and local residents who lost their homes to Katrina. The Facility also served as a base of operations for USGS staff from other locations as response efforts got underway.

Staff from the USGS Mississippi Water Science Center in Jackson deployed to coastal counties on August 31 to document storm surge elevations along the I–10 corridor, which is 6–10 miles inland. Initial data indicated that the storm surge varied from about 15 feet near the Alabama/Mississippi state line to about 28 ft in Hancock and Harrison Counties near St. Louis Bay. In the following weeks, USGS staff in Mississippi, Louisiana, and Alabama coordinated efforts with FEMA and The U.S. Army Corps of Engineers–Mobile District to complete the documentation of the storm surge elevation and extent.

A comparison of storm surge data from Hurricanes Katrina and Camille indicates that the storm surge generated by Katrina exceeded Camille by 5 or more feet at all locations in Mississippi and exceeded the Camille peak by more than 10 feet in some locations in Hancock and Harrison Counties.

Keywords: Hurricane Katrina, storm surge

Corresponding Author:
K. Van Wilson, Jr.
U.S Geological Survey
308 South Airport Rd.
Jackson, MS 39208
Phone: 601.933.2922
Fax: 601.933.2901
E–mail: kvwilson@usgs.gov

Session B: Selected Invasives

Fred Howell, Moderator

Gary Ervin
The Landscape Context of Plant Invasions in Mississippi Wetlands

John Madsen
Techniques for Managing Invasive Aquatic Plants in Mississippi Water Resources

Wilfredo Robles
Aquatic Vegetation Diversity in Lake Columbus, Lowndes County, MS

Ryan Wersel
Survey of Invasive and Native Aquatic Plants in the Ross Barnett Reservoir

The Landscape Context of Plant Invasions in Mississippi Wetlands

Gary N. Ervin and M. Jeffrey Linville

Invasive species are a known and growing threat to native ecosystems and the services they provide, and it is widely accepted that human activities contribute substantially to their spread. In a study of fifty-two north Mississippi wetlands, approximately 10% of the vascular plant species encountered were non-native, and 60% of the wetlands surveyed contained at least one plant species considered to be highly invasive. Furthermore, when highly invasive species were encountered, they were distributed across as much as 80% of the wetland area. Other work has shown that the degree of invasibility of a diversity of Mississippi wetlands was found to be much more strongly correlated with surrounding land use patterns than with the natural degree of connectivity among wetlands. For these reasons, we investigated the relationship of landscape features with exotic species richness in fifty-two freshwater wetlands across north Mississippi. Within those wetlands, invasibility was correlated only with certain forms of surrounding land cover, and inconsistently so. Agricultural land use appeared to enhance invasion of non-native plants, whereas density of surrounding wetlands and pine forest were correlated negatively with invasion. When wetland watersheds were classified based on the dominant land use as indicated by geospatial land cover data, no relationship was detected between dominant land use and degree of invasion. Indices of human activity surrounding the wetlands at the time of our vegetation surveys, however, did correlate closely with richness of exotic species, supporting the widely held notion that human alteration of the landscape can aid in the dispersal and establishment of non-native, weedy species.

Keywords: Conservation, Ecology, Invasive Species, Wetlands

Introduction

Exotic species include (but are not limited to) those species that are recognized noxious weedy invaders that may degrade wetland ecological integrity through multiple mechanisms. The presence of even one exotic species (such as *Hydrilla verticillata* (L. f.) Royle or *Lythrum salicaria* L.) may have disastrous consequences for wetland health, as many invasive species have been demonstrated to alter ecosystem properties such as hydrology, nutrient cycling, and native species regeneration (Gordon 1998).

In a 2004 survey of Mississippi wetlands, we found that just over 10% (on average) of the plants present in freshwater wetlands were not native to Mississippi, and 52 of 53 wetland surveyed contained at least one non-native plant species. Furthermore, 60% of the wetlands surveyed harbored at least one species that could be considered highly invasive. Examples of highly invasive plant species encountered were water hyacinth (*Eichhornia crassipes* (Mart.) Solms), alligatorweed (*Alternanthera philoxeroides* (Mart.) Griseb.), privet (*Ligustrum* spp.), and Japanese honeysuckle (*Lonicera japonica* Thunb.). These species, when present, occurred throughout an average of 16% of the vegetated portions of the wetlands, with this number as high as 80% in one wetland (40 of 50 plots contained a potentially highly invasive species).

In a separate study, we found that wetland type had no significant effect on the degree to which exotic species had invaded, despite high incidence of non-native species in lake fringe and river floodplain wetlands (Ervin et al. 2006). It was hypothesized that wetlands associated with lakes and wetlands would have a higher frequency of nonnative species because of the higher degree of connectivity to other aquatic systems – or because lakes and rivers tend to be more heavily used for recreation and navigation than other types of wetland. We did find, however, that an index of human activity surrounding the fifteen wetlands included in that work was significantly correlated with degree of invasion. That correlation of human use of the landscape with frequency of exotic plant species led to the present evaluation of the relationship of more quantitative measures of landscape use with exotic species prevalence in wetlands of Mississippi.

Methods

Site Descriptions

These analyses utilized vegetation data from 52 wetlands across northern Mississippi during summer of 2004 (plus one immediately east of the MS-AL border, near Pickensville, AL; Fig. 1). These sites were selected arbitrarily, based on our familiarity with potential study areas (public lands in Mississippi) and the ability to obtain permission to access wetlands across the northern half of the state. Some sites were selected specifically because of their proximity to or remoteness from human activities, in order to obtain data from sites along a more complete gradient of anthropogenic disturbance. For example, one wetland was located within the city limits of West Point, MS and immediately adjacent to US Highway 45 and another around the perimeter of a shopping mall parking

Figure 1. Map of wetlands included in this study. We surveyed 52 sites across north MS and one in extreme westcentral Alabama in 2004.

area in Tupelo, MS as representative of urban wetlands within the state. Wetlands were not selected, however, based on a priori knowledge that they contained a disproportionate complement of exotic species. These wetlands varied in degree of human impacts from beaver ponds to farm ponds located in recently fallowed cattle pastures, to moist soil management wetlands on MS National Wildlife Refuges, to active cattle pasture and other agricultural practices and the above mentioned urban wetlands. The result was an assortment of wetlands that had experienced a broad gradient of human activities on the surrounding landscape.

Plant Surveys

From March through September of 2004, sites were surveyed using a stratified random sampling design. A total of 50 plots per wetland (0.5 m2 circular plots) were spaced approximately evenly throughout the vegetated zones of each wetland, beginning at a random point, with quadrats placed by blind toss of the 0.5m2 sampling ring at the area of the appropriate spacing interval. The size of each wetland was estimated visually and plot spacing determined from that estimate such that plots covered the entirety of the vegetated wetland area, from the obviously saturated soils on the upland edge to open water, or water inaccessible by wading, on the wetland edge. Ten of these wetlands were surveyed twice during the growing season; the latter of the two surveys was used in these analyses.

Plants occupying the plots were recorded and collected if a definitive identification could not be made on site (and often collected when a reasonably certain identification was made, as a quality control measure). Collections were pressed, dried, and identified in the laboratory. Numerous species were identified with the assistance of herbarium personnel and vouchers were deposited in the Mississippi State University Herbarium (MISSA).

Species identified in these surveys were recorded as native or exotic (non-native) based on nativity data provided in regional keys (Godfrey and Wooten 1979, 1981 and Radford et al. 1968), the Flora of North America (FNAEC 1993+), and other sources (Miller et al. 2004; USDA NRCS 2004). Based on information provided by these sources, plant species were recorded as either exotic (non-native) or native.

Site Disturbance Index Evaluation

Using methodology presented by Lopez and Fennessy (2002; US EPA 2002), we ranked the wetlands included in this evaluation based upon intensity of human impact within the immediately surrounding landscape. Each site was evaluated, in a hierarchical manner, on whether it (1) was surrounded within the landscape by a) forest or grassland, b) fallow agricultural, c) active agricultural, or d) urban land use, (2) was surrounded by an immediately adjacent a) forest, b) grassland, or c) no buffer zone, and (3) possessed obvious signs of hydrologic alteration. Each of the landscape-scale factors has been linked to wetland plant response at the level of species and functional guilds (Table 1; Lopez et al. 2002). The possible Disturbance Rankings thus spanned from 1 to 24 with this ranking method. A second method of quantifying human disturbance to the wetland sites was developed to rank each form of disturbance separately, rather than in the interdependent manner provided by the flowchart ranking system above. This method was termed the Anthropogenic Activity Index (AAI; Appendix 1). The AAI also included other types of disturbances and measures of habitat heterogeneity not represented by the Lopez and Fennessy ranking method. This AAI is a modification of an index developed by the Minnesota Department of Environmental Quality (Gernes and Helgen 2002) and includes sections from the Ohio disturbance ranking system (Mack 2001). Sites were evaluated by five different metrics, each of which was divided into four categories that were scored 0 to 3. The metrics then were summed for a total index score. Sites that scored a 0 were considered to have very low levels of impairment, as would be expected for reference sites. At the opposite end of the gradient, sites that scored a 15 represented the highest levels of impairment. The results of each disturbance indexing method was used as a semi-quantitative evaluation of the relationship between then-current land use and degree of invasion of each wetland.

Land Use Data

In order to derive the area of each land use category within the specified radii of our sample points, a vector point file of GPS locations for the fifty-two Mississippi wetlands was created and combined with Mississippi GAP data (Vilella et al. 2003) in ArcMap. The MS GAP dataset used for this analysis was a thematically classified statewide geospatial data layer based on 40 distinct LULC types. Both the raster image and the vector point file then were

Table 1. Effects of within-watershed land cover on relative contribution of native and wetland-adapted species to wetland plant assemblages (Lopez et al. 2002).

Plant group	Land Cover	Direction of effect	Effect on
All vascular plants	Grassland	Positive	%FACW spp.
	Open water	Positive	%OBL spp.
	Forest	Positive	%Native spp.
	Agriculture	Negative	%Native spp.
All woody spp.	Urban	Negative	%OBL
	Agriculture	Negative	%Native
All herbaceous spp.	Grassland	Positive	%FACW & %Native
	Open water	Positive	%OBL
	Forest	Positive	%Native
	Agriculture	Negative	%OBL & %Native
Emergent herbaceous spp.	Grassland	Positive	%FACW
	Open water	Positive	%OBL
	Forest	Positive	%Native
	Forest	Negative	%Invasive
	Agriculture	Negative	%OBL & %Native

projected into a Transverse Mercator projection with the North American Datum 1983 (NAD83; the legal horizontal geodetic reference datum used by the US federal government). Using the editor toolbar, each wetland buffer was given a unique identifier equal to that of its corresponding wetland. Next each of the forty individual land class units was reclassified with the analysis mask and the extent of the spatial analyst set to the land cover image. In this reclassification, the land class unit of interest were given a value of one, while all other land class units were given a value of no data. This isolated the land class units into forty different raster images. The zonal statistics function then was executed on each raster image, with the analysis mask and the extent of the spatial analyst set to the buffer vector layer. The resulting tables were compiled into a single data set.

We used the area (ha) of seven collapsed LULC categories within a 100m or 1000m buffer as independent variables in analyses of the effects of land use on vegetation parameters. The collapsed LULC categories were: agricultural (including aquaculture and pasture), urban, pine forest, transportation, other/natural forest (hardwood and mixed forest), wetland, and other natural land cover (consisted of pine savannah, evergreen shrub, and sand bars/beaches). We used linear regression analyses of the land area (ha) of each land use category against the number of exotic species and percent of species identified as exotic for each wetland. We then assigned each wetland to one of the above categories, based on the predominant land use type within the surrounding 100m buffer to conduct ANOVA as another means of determining the relationship between land use and degree of invasion by non-native plant species (this was not performed for the 1km buffers).

Two additional analyses were carried out, following on results of the above tests. In the first, we categorized each of the wetlands as existing in an agricultural landscape, an urban setting, as a moist-soil managed wetland, a beaver pond, or "other" at the time of sampling. This served as a contemporary evaluation of "land use," as the data of Vilella et al. (2003) were based on Landsat imagery from 1991 through 1993. We also categorized each wetland as experiencing high, moderate, or low levels of human influence, based on the AAI scores assigned to each. The AAI scores spanned a possible gradient of 0 to 15, and intervals of 0 to 5 (low), 6 to 10 (moderate), and 11 to 15 (high) were used to develop these three AAI categories. For both groupings (current land use and AAI level), we carried out ANOVA using SYSTAT version 11 (SYSTAT Software Inc.). SYSTAT also was used for the above described linear regression analyses.

Results and Discussion

Both the Disturbance Rank and Anthropogenic Activity Index were correlated significantly with the number of non-native species present in the study wetlands. For simplicity, only the AAI data are shown (Fig. 2). Similarly, significant correlations were found between the AAI for these wetlands and land cover (Fig. 3). The land area within 100m of the wetlands used for agriculture was positively correlated with our AAI scores, while the area of pine or other forest was negatively correlated with our somewhat less quantitative index of human activity.

Regression analyses of the seven collapsed LULC categories against the number of non-native species present indicated correlations of agriculture, pine forest, and wetland LULC classes with exotic species in only one of the two buffer sizes (Table 2). Agricultural use of land within 1km of the wetlands appeared to enhance slightly the number of exotic species within the wetlands, whereas wetland area within 1km and pine forest within 100m were correlated negatively with the number of nonnative species observed. Although this does correlate to a large degree with the data of Lopez et al. (2002), the correlations were not terribly

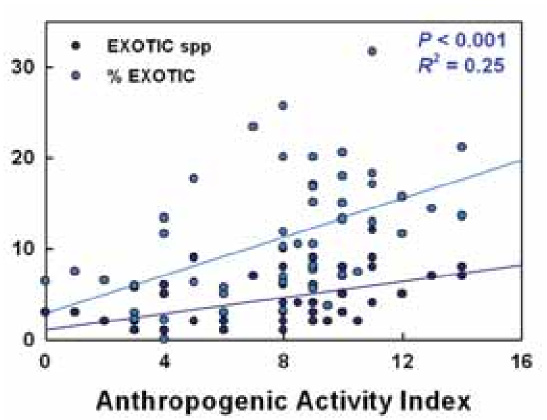

Figure 2. Correlations between non-native species and one of two indices of human activity surrounding our study wetlands.

strong (especially the enhancing effect of agricultural land use), and other land uses that would be expected to influence degree of invasion did not do so (e.g., urban land cover and "other" natural land cover). We found similar results following ANOVA of number of exotic species and the percent of species that were exotic against land use type (based on the dominant land use within the 100m buffer). That is, land cover type was not correlated with the richness of established nonnative species (Fig. 4)

The low degree of correlation between exotic species and land cover was surprising given the interactions observed in other studies between land use or human activity and exotic species establishment or other indicators of impairment in wetlands and other natural areas. In a study of Florida wetlands, Brown and Vivas (2005) found significant correlations between landscape development intensity (LDI, an index of the degree of human activity on the landscape based on land use) and wetland rapid assessment scores in studies of Florida (USA) wetlands. Those analyses included wetlands situated in agricultural, urban, and natural landscapes. Similarly, Cohen et al. (2004) found significant correlations between LDI and floristic "quality" in a study of 75 depressional wetlands in Florida. Both of the those studies (Cohen et al. 2004 and Brown and Vivas 2005) used buffer sizes of similar size as ours for calculation of land use intensity. One potential problem in our analyses was the age of the LULC data being used, relative to the date of our plant surveys. We censused these wetlands in 2004, but the land cover data used to build the MS GAP coverage was collected during 1991 to 1993. Indeed, land use surrounding some of the wetlands had changed markedly during the eleven to thirteen years that passed between the land cover survey and our vegetation surveys. For example, one wetland was immediately adjacent the Mall at Barnes Crossing (Tupelo, MS), which opened in 1990, but the surrounding area has been developed extensively during the five to ten years immediately prior to our vegetation surveys. Another wetland was situated immediately adjacent to a four-lane section of U.S. Highway 82, west of Starkville, that opened to

Table 2. Comparison of land cover against exotic species richness in the 52 Mississippi wetlands. Slope is the regression coefficient, equivalent to unit change in species richness per hectare of a given land use class within the given buffer diameter.

Land Cover	Slope	R^2	F	P
vs. Agriculture				
100 m	+0.42	0.02	1.28	0.27
1000 m	+ 0.01	0.07	3.6	0.06*
vs. Urban				
100 m	+3.84	0.01	0.5	0.5
1000 m	-0.03	0.14	0.5	0.5
vs. Pine Forest				
100 m	-1.43	0.09	4.9	0.03**
1000 m	-0.01	0.05	2.2	0.14
vs. Transportation				
100 m	+0.40	0.01	0.3	0.6
1000 m	+0.01	0.00	0.0	0.9
vs. Other Forest				
100 m	-0.40	0.00	0.1	0.7
1000 m	-0.00	0.00	0.1	0.7
vs. Other Wetland				
100 m	-0.45	0.01	0.3	0.6
1000 m	-0.01	0.11	6.3	0.02**
vs. Other land cover				
100 m	+0.01	0.00	0.0	1.0
1000 m	+0.01	0.04	2.1	0.2

* $P \leq 0.10$,
** $P \leq 0.05$
Mean percent cover of a 1-km buffer around the wetlands across all 53 sites was: Agriculture 31%, Urban 1%, Agroforestry 15%, Transportation 2%, Natural Forest 16%, Wetland 16%, Other 19% (2% non-wetland freshwater, 17% low herbaceous vegetation). These data essentially are identical to the statewide averages for Mississippi (paired t-test P = 1.00).

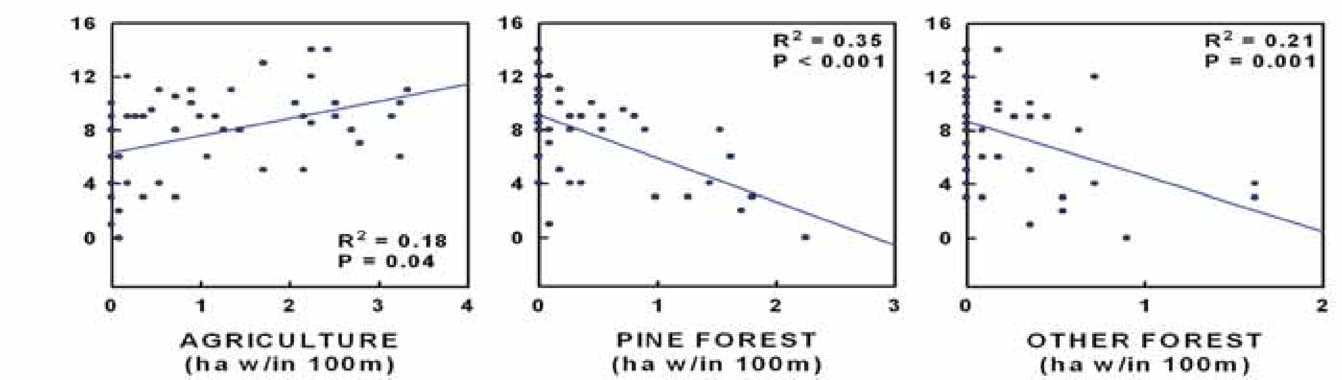

Figure 3. Correlations between Anthropogenic Activity Index scores for these wetlands and the more objective land cover data from the MS Gap Analysis Project (GAP). None of the other four land use categories exhibited significant correlations with AAI.

traffic in late 2004. When we examined the link between contemporary land use and richness of exotic plant species, we did find an effect of land use on non-native species (Fig. 5) . Wetlands in urban or agricultural settings were invaded to a much greater extent than unmanipulated wetlands in Wildlife Management Areas or beaver ponds. Wildlife Management Areas tend to consist of large tracts of forest with intermittent wetland areas, some of which may be quite remote from large-scale human activity. Similarly, beaver ponds usually are found on lands managed as or permitted to exist as forest and usually with low levels of human activity. A corresponding pattern was observed when sites were grouped into AAI "classes" and subjected to ANOVA (Fig. 6). Areas with low levels of human activity had roughly half as many non-native plant species as areas with high levels of anthropogenic disturbance. Furthermore, other analyses of these data have indicated that the AAI metric most closely correlated with number of exotic species in these wetlands was that which represented the degree of direct manipulation of areas within the wetlands themselves (Metric 4; Appendix 1), although all metrics were correlated statistically with degree of invasion (Herman 2005). Thus, it appears that a primary difficulty in our initial evaluations of links between exotic species and landscape factors was the lack of geospatial data contemporary to our vegetation sampling. One possible solution would have been to use the more recent National Land Cover Dataset, but even that dataset is based on data collected between 1994 and 1998 (Homer et al. 2004). Perhaps the most effective means of assessing the correlation of human land use and exotic species is to conduct visual assessments of land use during surveys, much as we did in assessing the Disturbance Rank and Anthropogenic Activity Index in this work. In particular, the AAI seemed to perform better in these analyses (see, e.g., Herman 2005), and this probably was because the AAI scoring system includes information on activities within the wetland as well as activity on the surrounding landscape. Finally, it is certainly worth mentioning again that 52 of the 53 wetlands surveyed were occupied by at least one non-native plant species at the time this work was performed. This is a clear indication of the potential that exists for native species to be replaced and for native ecosystems to be impacted as those species considered to be highly invasive expand their ranges throughout Mississippi.

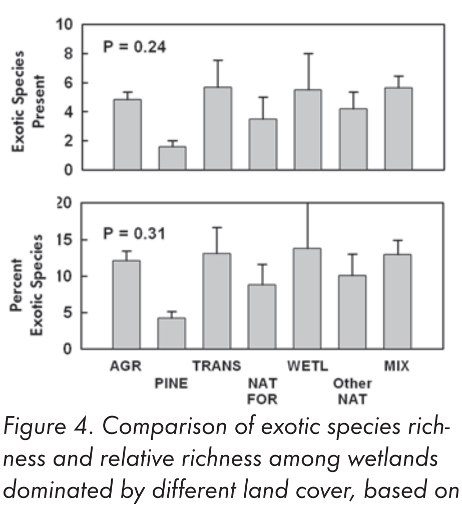

Figure 4. Comparison of exotic species richness and relative richness among wetlands dominated by different land cover, based on MS GAP data.

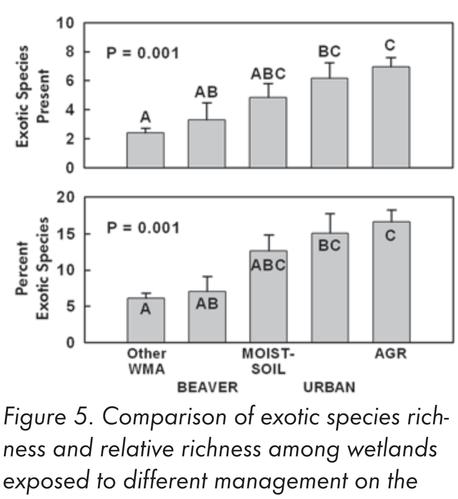

Figure 5. Comparison of exotic species richness and relative richness among wetlands exposed to different management on the surrounding landscape and/or within the wetland boundaries.

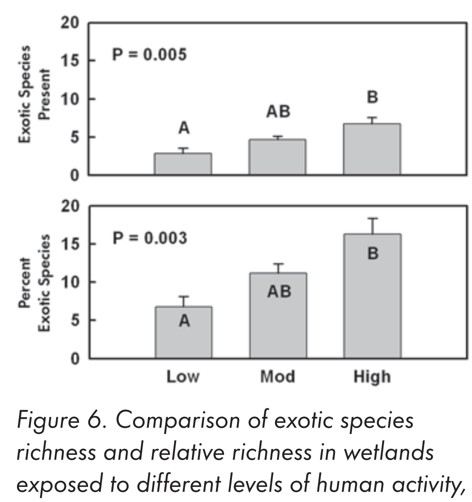

Figure 6. Comparison of exotic species richness and relative richness in wetlands exposed to different levels of human activity, based on ranked AAI scores.

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Corresponding Author:
Gary N. Ervin, Assistant Professor of Plant Ecology
Department of Biological Sciences
Mississippi State University
PO Box GY
Mississippi State, MS 39762
Phone 662–325–1203
Fax 662–325–7939
E–mail: gervin@biology.msstate.edu

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Appendix 1. Anthropogenic Activity Index score sheet (developed by Herman 2005).

ANTHROPOGENIC ACTIVITY INDEX
(AAI version specified for north and central Mississippi)

Site: Study: Crew: Date:

Metric 1. Surrounding Land Use Intensity (500 m surrounding buffer)

_______ Points

	Very Low- as expected at reference site	No evidence of disturbance, mature forest, grassland	0
	Low- mostly undisturbed, some human influence	Old fields, secondary forest, shrubby woodlots	1
	Moderate- a significant amount of human influence	Active pasture, high road density, newly fallowed fields, wildlife habitat management, other intermittent agricultural practices	2
	High- Intensive use of land up to buffer or wetland margin	Urban, residential, industrial operations, row cropping, other intensive agricultural operations	3

Metric 2. Intactness and Effectiveness of Buffer (up to approximately 50 m surrounding site)

_______ Points

	Best- ~50 m wide, as expected for reference site	Mature forest, grassland	0
	Moderate- 50-25 m wide, some human influence	Mixture of grassland and secondary forest, old fields, shrubby woodlots	1
	Fair- 25-10m wide with significant human influence	Active pasture, newly fallowed field, adjacent roads, wildlife habitat management, other intermittent agricultural practices	2
	Poor- no effective buffer	Row cropping, turf vegetation, adjacent urban development, impervious surfaces, other intensive agricultural practices	3

Metric 3. Hydrologic Alteration

_______ Points

	Very Low- as expected at reference site	No evidence of disturbance	0
	Low- low intensity alteration	or past alteration not currently affecting wetland	1
	Moderate- significant, visible influence	Current and active	2
	High- intensive activity	Major disturbance currently and actively effecting hydrology	3

Examples of Alterations

Ditch inlet		Point source inlet		Other (describe):
Tile inlet		Installed weir, outlet		
Berm/dam		Levee		
Road bed		Used for drainage		

Subtotal from this page: ________

Appendix 1 (continued). Anthropogenic Activity Index score sheet (developed by Herman 2005).

ANTHROPOGENIC ACTIVITY INDEX
(AAI version specified for north and central Mississippi)

Site: Study: Crew: Date: (page 2)

Metric 4. Habitat alteration (within wetland)
_______ Points

	Very Low- as expected at reference site	No evidence of human activity	0
	Low- low intensity, or not currently affecting wetland	Some removal of vegetation, but vegetation is recovering	1
	Moderate- significant alteration of either vegetation or substrate	Vehicle use, grazed, livestock hooves, coarse woody debris removal, mowed	2
	High- intensive disturbance of vegetation and substrate	Dredging, filling, tiling, disking, vehicle use, tree/shrub removal, removal of emergent vegetation	3

Specify Other Activities:

Metric 5. Habitat Quality and Microhabitat Heterogeneity
_______ Points

	Best- large amount of habitat heterogeneity, high diversity of microhabitats	Small proportion of open water, 0-25%, large amount of emergent and submersed vegetation and coarse woody debris, some standing dead trees	0
	Moderate- significant amount of habitat heterogeneity	25-50% open water, some woody debris	1
	Fair- small amount of habitat heterogeneity	50-75% open water, no woody debris	2
	Poor- small amount of habitat heterogeneity, low quality habitat	75-100% open water	3

Subtotal from this page: ________

Additional factors or concerns:

Metric 1 _____

Metric 2 _____ **Total Anthropogenic Activity Index Score**

Metric 3 _____

Metric 4 _____

Metric 5 _____

Techniques for Managing Invasive Aquatic Plants in Mississippi Water Resources

John D. Madsen

Invasive aquatic plants are an ever-growing nuisance to water resources in Mississippi and the rest of the United States. These plants are generally introduced from other parts of the world, some for beneficial or horticultural uses. Once introduced, they can interfere with navigation, impede water flow, increase flood risk, reduce hydropower generation, and increase evapotranspirational losses from surface waters. Invasive species also pose direct threats to ecosystems processes and biodiversity. A variety of techniques have been used to manage these invasive plants in waterways around the United States. These techniques can be classified as Biological, Chemical, Mechanical and Physical techniques. Biological techniques utilize an herbivore or pathogen to control the plant, or reduce the equilibrium level of the population to an acceptable level. Chemical techniques utilize US EPA-approved herbicides to control plants, from small plots to large areas. Mechanical techniques utilize machines or tools to harvest, cut, pulverize or otherwise damage the plant. Physical techniques involve altering the environment to prevent or reduce the growth of invasive plant species. I will describe specific techniques and their potential niches for managing invasive aquatic plant species in Mississippi. I will also present some resources available for assisting in selecting the best technique, including the APIS system from USAERDC, available Best Management Practices plans, and information resources available from Mississippi State University.

Keywords: Invasive Species, Management & Planning, Recreation, Wetlands

Introduction

Invasive aquatic plants, mostly nonnative species introduced for ornamental and aquarium applications, have become a widespread nuisance problem in the United States (Madsen 1997). Many of the species common throughout the southeastern United States also create nuisance problems in Mississippi water resources; including waterhyacinth (*Eichhornia crassipes* (Mart.) Solms), hydrilla (*Hydrilla verticillata* (L.f.) Royle), and Eurasian watermilfoil (*Myriophyllum spicatum* L.) as typical examples (Madsen 2004). More recently, giant salvinia (*Salvinia molesta* Mitchell) has been found in southern Mississippi, despite repeated management efforts. Invasive aquatic plants interfere with human uses of water resources, including increasing flood magnitude and frequency, interfering with commercial and recreational navigation, impeding fishing, boating, and swimming, and increasing the survival of some disease vector insects (Pimentel et al. 2000). Invasive species can also have deleterious ecosystem impacts, including reducing species diversity, suppressing the growth of desirable native species, reducing habitat value for fish and wildlife, increasing internal loading of nutrients, reducing water quality, and increasing the extinction rate of rare, threatened, and endangered species (Madsen 1997, Mullin et al. 2000). The total cost of managing invasive aquatic plants in the United States has been estimated at $100M (Pimentel et al. 2000, Rockwell 2003).

For the Mississippi Water Resources Conference in 2004, I reviewed the species that either currently impact Mississippi water resources or may pose a future threat (Madsen 2004). During the 2005 Mississippi Water Resources Conference, I explained the process by which aquatic plant management plans could be developed (Madsen 2005). In this paper, I will detail the currently used techniques for managing invasive aquatic plants.

Management Plans

Before invasive plant management begins, some effort should be made to make an effective management plan (Madsen 2000). An aquatic plant management plan should have eight components: prevention, problem assessment, project management, monitoring, education, management goals, site-specific management, and evaluation (Madsen 2005). If management goals are not made before implementation, the resource manager increases the likelihood of either selecting techniques that are contrary to long-term but unstated goals. In addition, a lack of education and outreach may result in public reaction to management, often based on incorrect information or misperception.

Management Techniques

Management techniques described below can be classified as biological, chemical, mechanical, or physical control techniques. I will review the major techniques available, and indicate their applicability to the five most likely invasive aquatic plants for larger water resource systems (Table 1).

Each technique should not be viewed as an exclusive choice; but rather the techniques should be selected based on the nuisance problem at a given site and the economic and environmental constraints of the resource.

Table 1. The five most likely invasive aquatic plant species in Mississippi.

Common name	Scientific name	Growth form
Alligatorweed	*Alternanthera philoxeroides*	Emergent
Eurasian watermilfoil	*Myriophyllum spicatum*	Submersed
Giant salvinia	*Salvinia molesta*	Floating
Hydrilla	*Hydrilla verticillata*	Submersed
Waterhyacinth	*Eichhornia crassipes*	Floating/ Emergent

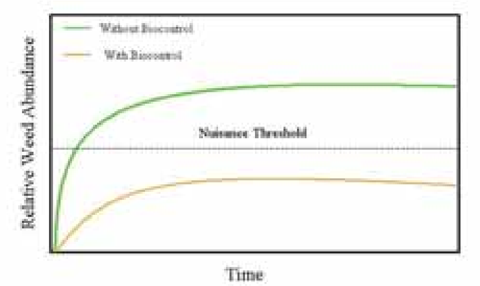

Figure 1. Relative weed abundance exceeds the nuisance-causing level without the biocontrol agent, and is reduced to below the nuisance threshold with an adequate population of the biocontrol agent.

Biological Control

Biological control is often misunderstood in terms of reasonable expectations and outcomes (Grodowitz 1998). This is in part because the first examples of successful biological control for terrestrial and aquatic weeds (pricklypear cactus and alligatorweed, respectively) were resounding successes in eliminating the nuisance problem (Grodowitz 1998). Rather, the typical expectation is that the established populations of the biological control agent will reduce the abundance of the target plant below nuisance-causing levels (Figure 1). Biological control insects may increase the competitive ability of native plants over the invasive species (Van et al. 1998).

The available biological control techniques include vertebrate generalist herbivores (specifically grass carp), insects, and pathogens (Table 2).

Grass carp can be effective at controlling hydrilla, but do little to control emergent or floating species (Van Dyke et al. 1994, Pine and Anderson 1991). Grass carp are not effective for control of Eurasian watermilfoil (Fowler and Robinson 1978). Grass carp can be economical and effective, particularly in small ponds with no outflow, but they tend to remove all submersed vegetation, travel at will, and migrate from large open aquatic systems (Haller 1994, Bonar et al. 1993). For these reasons, use of grass carp is not recommended for large waterbodies.

Introducing overseas insects that feed on invasive aquatic plants have been widely studied and utilized, with varying success (Grodowitz 1998). The first such project was to introduce the alligatorweed flea beetle for control of alligatorweed, which was a resounding success (Grodowitz 1998, Cofrancesco 1988). Several insects have been introduced to feed on both waterhyacinth and hydrilla, but neither has been nearly as successful under field conditions as releases for alligatorweed. Recently, releases of *Cyrtobagous salviniae* have been made in the U.S. to control giant salvinia, but it is too early to judge the results of those efforts. *Cyrtobagous salviniae* has been fairly successful in controlling giant salvinia in other countries (Oliver 1993, Thomas and Room 1986, Julien and Griffiths 1998)

In some instances, native or naturalized insects have been utilized in an attempt to control invasive weeds (Cofrancesco 2000). Several attempts have been made with various native insects to feed on Eurasian watermilfoil (Johnson et al. 2000), with the most common insect used being *Euhrychiopsis lecontei* (Creed 1998). To date, these attempts have had individual successes, but no long-term control or strategy for their implementation.

Pathogens have also been investigated for use in controlling invasive aquatic plants (Cofrancesco 2000). Thus far, the only current research and development is with *Mycoleptodiscus terrestris*, which acts like a contact bioherbicide (Shearer 1998, 2002). Much of the research has been focused on integrating the use of the pathogen with herbicides (Nelson and Shearer 2005, Nelson et al. 1998, Netherland and Shearer 1996). This pathogen is currently being formulated for demonstration use.

Revegetating with native plants after control is not a control technique in and of itself, but it may reduce the reinvasion rate and will definitely provide habitat and other valuable ecosystem services provided by plants in the littoral zone (Smart et al. 1996). The main problem is that this is very labor intensive and expensive, with some question as to whether this does more than reduce the time for recolonization (Madsen 2000).

Chemical Control

Chemical control of invasive plants has remained the mainstay of management techniques, with some good reason: chemicals are more effective, more predictable, and costs are competitive with most techniques. With the cost of most aquatic herbicides and application techniques, the costs typically range from \$150 to \$500 per acre, which is significantly more than terrestrial weed management. Herbicides formulated for aquatic use do not have

Table 2. Biological control techniques for managing invasive aquatic plants.

Type	Specifics	Activity	Applicability to MS Water Resources
Generalist vertebrate herbivore	Grass carp (*Ctenopharyngodon idella*)	Generalist feeder, preference for hydrilla	Small ponds with hydrilla
Insects	Alligatorweed flea beetle *Agasicles hygrophila* And others	Alligatorweed	Excellent, some herbicides may be needed
	Euhrychiopsis lecontei	Eurasian watermilfoil	Poor
	Cyrtobagous salviniae	Giant salvinia	Successful overseas
	Hydrellia spp. and *Bagous* spp.	Hydrilla	Some success
	Neocehtina spp. and others	Waterhyacinth	Some reduction in flowering and biomass
Pathogens	*Mycoleptodiscus terrestris*	Shows activity on submersed plants	Under development; Eurasian watermilfoil and hydrilla
Native Plant Restoration	Planting of desirable native plants	Possible restoration after control	Labor intensive and expensive, but possible

surfactants; so the applicator will have to add a surfactant appropriate for aquatic use when applying to the aerial portions of floating-leaved and emergent plants. For submersed plant applications, no surfactants are required. Lastly, it is imperative that applicators read the label before use, and only use herbicides that specify on the label that it is approved for aquatic use.

Nine active ingredients are currently approved for use in the aquatic environment for control of vascular aquatic plants (e.g., not algae), with several more being reviewed by the U.S. EPA. I have listed these nine active ingredients with the most common formulated products (Table 3). Four of these products (carfentrazone-ethyl, copper, diquat, and endothall) are contact herbicides, and work at the site of absorption. The remaining five products are slower-acting system herbicides that are translocated more readily throughout the plant. Some of these products are only for use on emergent plants, others only on submersed plants, and some are selective for certain groups of plants. Understanding the nature of each chemical and their use is critical for proper product selection and expectation of results.

To select an appropriate herbicide, the first step is to select an herbicide that is effective on the target species (Table 4). Proper identification of the target plant is critical to selecting an effective herbicide. In addition, different products or formulations of the same herbicide may vary in their efficacy on the target plant. Once the appropriate possibilities are identified, the use restrictions of the herbicides must be considered (Table 5). These use restrictions are generally limited based on the uses of the water, and are set based on toxicological data when the label is approved. Use restrictions are made to protect the health and safety of humans, animals, and crops using the treated water, so they should not be violated. For emergent and floating-leaved plants, this is typically the final consideration in selecting the right herbicide. For submersed plants, the herbicide is added to the water, and the plants take up the herbicide from the water.

For the herbicide to be effective, the plants must be in contact with an adequate amount of herbicide for a long enough period of time to be effective. For contact herbicides, contact times of 6 to 12 hours is often sufficient; whereas some of the systemic herbicides will require contact times ranging from 12 hours to 60 days (Table 6). Knowledge of the water exchange characteristics of the treatment site is critical for a proper herbicide treatment (Madsen 2000).

Herbicides may be used selectively to control emergent, floating-leaved, and submersed target plants while minimizing impacts on desirable native plants (Getsinger et al. 1997, Madsen et al. 2002). Selective use may be based on the timing of application, inherent selectivity of the molecule, or subtle differences in the metabolism of an herbicide in an otherwise "broad-spectrum" herbicide (Getsinger et al. 1997, Madsen et al. 2002, Netherland et al. 1997, 2000, Poovey et al. 2002, Skogerboe and Getsinger 2001).

Table 3. U.S. EPA-Approved aquatic herbicides for control of invasive aquatic plants.

Chemical	Product	Formulation	Company	Emergent, Floating or Submersed
2,4-D	Aqua-Kleen DMA IV Navigate	granular liquid granular	Cerexagri Dow AgroSciences Applied Biochemists	Submersed All Submersed
Carfentrazone-ethyl	Stingray	liquid	FMC	All
Copper	Captain Cutrine Plus Komeen	liquid liquid or granular liquid	SePRO Applied Biochemists SePRO	Submersed Submersed Submersed
Diquat	Reward Weedtrine	liquid liquid	Syngenta Applied Biochemists	All All
Endothall	Aquathol K Aquathol Super K Hydrothol 191	liquid granular liquid	Cerexagri Cerexagri Cerexagri	Submersed Submersed Submersed
Glyphosate	AquaPro Rodeo	liquid liquid	SePRO Dow AgroSciences	Emergent and floating Emergent and floating
Imazapyr	Habitat	liquid	BASF	Emergent and floating
Fluridone	Sonar	liquid and granular	SePRO	Submersed
Triclopyr	Renovate 3	liquid	SePRO	All

Table 4. Efficacy of U.S. EPA-Approved herbicides on Mississippi invasive aquatic weeds. E, excellent; G, good; F, fair; P, poor; NA, not applicable.

Chemical	Alligatorweed	Eurasian watermilfoil	Giant salvinia	Hydrilla	Waterhyacinth
2,4-D	E	E	P	P	E
Carfentrazone-ethyl	E	G	P	P	E
Copper	P	P	P	E	P
Diquat	G	G	G	G	G
Endothall	NA	G	NA	G	NA
Glyphosate	E	NA	G	NA	E
Imazapyr	E	NA	P	NA	E
Fluridone	NA	E	NA	E	NA
Triclopyr	E	E	P	P	E

Table 5. Water use restrictions, in days, for waters treated with U.S. EPA-approved herbicides. See the Mississippi Weed Management Guide or herbicide label for specific provisions or exemptions. An asterisk indicates examine the approved label.

	Treated Water Use Restriction (days)						
	Human			Animal	Irrigation		
Chemical	**Drinking**	**Swimming**	**Fish Consumption**	**Drinking**	**Turf**	**Forage**	**Food Crops**
2,4-D	21	0	0	0	21	21	21
Carfentrazone-ethyl	1	0	0	1	14	14	14
Copper	0	0	0	0	0	0	0
Diquat	1-3	0	0	1	1-3	5	5
Endothall	7-25	1	3	7-25	0	7-25	7-25
Glyphosate	0	0	0	0	0	0	0
Imazapyr	2	0	0	0	120	120	120
Fluridone	0	0	0	0	30	30	30
Triclopyr	*	0	0	0	0	120	120

Table 6. Herbicide exposure time for submersed applications, plant response, and application rate.

Chemical	Exposure Time (Submersed)	Plant Response	Maximum Application Rate
2,4-D	Intermediate (18-72 hours)	7-10 days	0.5 gal/acre (emergent) 2.84 gal/acre-ft (submersed)
Carfentrazone-ethyl	Unknown	7-14 days	0.2 lb ai/acre (emergent) 0.296 gal/acre-ft (submersed)
Copper	Intermediate (18–72 hours)	7-10 days	1.5 gal/acre-ft (submersed)
Diquat	Short (12-26 hours)	7 days	2 gal/acre (both)
Endothall	Short (12-36 hours)	7-14 days	3.2 gal/acre-ft (submersed)
Glyphosate	NA	Up to 4 weeks	2 gal/acre (emergent only)
Imazapyr	NA	Up to 8 weeks	.75 gal/acre (emergent only)
Fluridone	Very long (60 to 90 days)	Up to 90 days	5 oz/acre-ft (submersed application only, generally use much less)
Triclopyr	Intermediate (12-60 hours)	Up to 2 weeks	6 lb ae/acre (emergent) 2.3 gal/acre-ft (submersed)

Mechanical Control
Mechanical control techniques are often a useful tool for either small infestations, or in locations that cannot be treated with chemicals (Table 7). By far the most common mechanical technique used worldwide is manual removal, either with a bare hand or with a hand tool. In North America, this technique is most useful when only individual plants are found, particularly during noxious weed surveys. Cutting has been used in the past, where a sickle blade cutter or other device cuts the stem. While faster than other techniques, the fragments are often viable, so this technique mostly succeeds in spreading nuisance plant infestations.

Harvesting with aquatic harvesters has been widely used for all the species listed in Table 1 except alligatorweed. While immediate relief from the nuisance growth is achieved, the plants regrow rapidly and disposal of plant material may be problematic. Destructive machines like the cookie cutter or flail chopper have been used for herbaceous and woody mat-forming plants. While these machines may provide nuisance relief, the dead plant material may pose an environmental hazard and spread viable fragments. Diver operated suction harvesting has been widely used to remove small colonies of submersed plants like Eurasian watermilfoil and hydrilla, but is not practical for infestations larger than an acre (Eichler et al. 1993). Lastly, rotovating has been used in the Pacific Northwest and western Canada for control of Eurasian watermilfoil. Similar machines based on rototilling could be used for other invasive plants. These machines, however, will result in the spread of viable fragments.

Physical Control
Physical control techniques reduce or eliminate plant growth through altering the environment, rather than directly controlling plants. Types of physical control techniques include benthic barriers, drawdown, dredging, light attenuation or shading, and nutrient inactivation (Table 8).

Benthic barrier involves using a bottom covering of synthetic material to cover over a small colony of aquatic plants. This technique would be ineffective for free-floating plants, though it may be useful for rooted emergent plants. It has been most widely used for submersed plant control, particularly Eurasian watermilfoil, with excellent results (Engel 1984, Eichler et al. 1995).

In lakes or reservoirs that have a water level control structure, drawdown can be used to control plants by draining or dewatering the waterbody to below the level in which plants are rooted. Drawdown is most effective over the winter, especially if freezing temperatures occur. While inexpensive and effective for many species,

Table 7. Mechanical control technique advantages, disadvantages, and effectiveness. E, excellent; G, good; F, fair; P, poor; NA, not applicable.

			Efficacy on Target Species				
Technique	Advantages	Disadvantages	Alligatorweed	Eurasian watermilfoil	Giant salvinia	Hydrilla	Waterhyacinth
Hand cutting or pulling	Low technology and affordable	Labor-intensive, for individual plants	E	G	E	F	E
Cutting	More rapid than harvesting	Mats of cut plants my be environmental hazard and spread infestation	F	F	P	F	P
Harvesting	Removes plant biomass and nuisance	Slow and expensive, plants regrow	G	G	G	G	G
Cookie cutter	Rapid destruction of mat materials	Large amount of debris, may spread plants	G	NA	F	NA	G
Flail chopper	Rapid destruction of floating and emergent material	Fragments may spread plants	G	NA	P	NA	G
Diver-operated suction harvester	Direct removal of plants, no floating fragments	Slow and labor-intensive	NA	E	NA	G	NA
Rotovating	Disrupts root crown of submersed plants	Spreads fragments	G	G	NA	F	NA

Table 8. Physical control techniques for invasive aquatic plants: advantages, disadvantages, and effectiveness. E, excellent; G, good; F, fair; P, poor; NA, not applicable.

			Efficacy on Target Species				
Technique	**Advantages**	**Disadvantages**	Alligatorweed	Eurasian watermilfoil	Giant salvinia	Hydrilla	Waterhyacinth
Benthic Barrier	Direct and effective, may last several seasons	Expensive, small-scale, not selective	?	E	NA	E	NA
Drawdown	Inexpensive and effective	Requires water control structure, can have severe environmental effects and impacts on riparian users	P	E	G	P	E
Dredging	Creates deeper water, long term and effective	Too expensive if only goal is plant control, must deal with sediment disposal	G	E	P	E	P
Light Attenuation	Inexpensive and Effective	Nonselective, may not be aesthetically pleasing	G	G	G	G	G
Nutrient inactivation	Possible for floating plants, but not operations	Under research for rooted plants	NA	NA	F	NA	F, P

it may have significant environmental impacts and cause significant impairment to other water resource uses. Some plant species can be completely controlled (e.g., Eurasian watermilfoil and waterhyacinth), while others are resistant to water level drawdown through propagules tolerant to drying (e.g., hydrilla).

In some lake restoration projects, dredges are used to deepen the water by removing sediment. This will create water too deep for rooted plants to grow, resulting in a reduction of nuisance growth (Nichols 1984, Tobiessen et al. 1992). While effective, this method is too expensive for most situations.

Shading or light attenuation can control plant growth effectively, but the method may interfere with other water uses or otherwise be impractical. Light reduction can be created using shade trees plantings, covers, or fabric above the water surface (Dawson 1986, Madsen and Adams 1989). This may work either for emergent, floating-leaved, or submersed plants. The use of water-soluble dyes has also been used for submersed plant control, but this is best used for only small ornamental ponds (Madsen et al. 1999). Pond management in the southeast has long recommended the addition of fertilizer to create an algal bloom, which in turn reduces light availability to rooted plants. While this may be effective, it has other consequences, and should not be attempted in larger multi-purpose water resources.

Nutrient inactivation has been widely used for control of phytoplankton blooms through the addition of alum to bind phosphorus in the water column (Welch and Cooke 1995). Unfortunately, most invasive aquatic plants are limited by nitrogen availability in the sediment rather than phosphorus availability in the water column. To date, attempts to manipulate the nutrient concentrations of sediment have been unsuccessful, though water column manipulation of nutrients could control free-floating plants.

Information Resources

A number of Internet websites provide good authoritative information on aquatic plant management techniques (Table 9). The Aquatic Ecosystem Restoration Foundation site has up-to-date links to most aquatic herbicide manufacturers, and a regularly updated Best Management Practices manual. The Aquatic Plant Control Research Program of the US Army Corps of Engineers has a complete bibliography of their research articles and reports, as well as an online information system for aquatic plant management techniques that is periodically updated. This information system is also available as a CD-ROM.

The Center for Invasive and Aquatic Plants at the University of Florida has the premier collection of color photos and line drawings of invasive plants, as well as an online bibliographic service that includes both peer-reviewed articles and government reports

Table 9. Internet web page sources for information on invasive aquatic plants and their management.

Title	Purpose	Location
Aquatic Ecosystem Restoration Foundation	*Source for herbicide manufacturer information*	www.aquatics.org
AERF Best Management Practices	Report on best management approaches for invasive aquatic plants	www.aquatics.org/aquatic_bmp.pdf.
Aquatic Plant Control Research Program	Federal research program for invasive aquatic plants	el.erdc.usace.army.mil/aqua/
Aquatic Plant Information System	Information on aquatic plant management techniques	el.erdc.usace.army.mil/aqua/apis/
Center for Invasive and Aquatic Plants	Information, photos, and bibliography	aquat1.ifas.ufl.edu/
Florida Aquatic Plant Management Program	Full description of techniques and application procedures	www.dep.state.fl.us/lands/invaspec/2ndlevpgs/Aquaticplnts.htm
GeoResources Institute Invasive Species Program	Information and research on invasive species at Mississippi State University	www.gri.msstate.edu/lwa/invspec.php
Mississippi Weed Control Guidelines	Aquatic Weed Control Recommendations	msucares.com/pubs/publications/p1532aquatic.pdf
Mississippi State University Extension Service	Information from all of Extension Service	msucares.com

on invasive plant research. The Florida Department of Environmental Protection has developed an excellent web page of operational aquatic plant management techniques. The GeoResources Institute has information on invasive species research in Mississippi. The Mississippi Weed Control Guidelines are produced by the Mississippi Weed Science Consortium, and are updated annually. The link provided is specifically for aquatic weeds, but additional weed management information is available in this report. Lastly, the Mississippi State University Extension Service web page, msucares.com, has an extensive listing of fact sheets and reports on water resource management.

Conclusion

Aquatic plant management techniques are constantly updated and revised. While deciding on aquatic plant management techniques, look for the most current information available. If you are using herbicides, always read the label before using the product, as these regulations are constantly changing. Be prepared to use different techniques as each situation and infestation dictate.

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Corresponding Author:
John D. Madsen, Assistant Professor
Mississippi State University
GeoResources Institute
Box 9652
Mississippi State, MS 39762–9652
Phone 662–325–2428
Fax 662–325–7692
E–mail: jmadsen@gri.msstate.edu

Aquatic Vegetation Diversity in Lake Columbus, Lowndes County, MS

Wilfredo Robles and John D. Madsen

Successful management of water bodies requires monitoring aquatic vegetation diversity and abundance. Lake Columbus was surveyed in July 2005 using a point intercept sampling method to determine the presence and distribution of aquatic plants. Using ArcGIS software, Lake Columbus was divided into a 400m x 400m grid in which each survey point was located with GPS using UTM coordinates. Aquatic plant species presence was recorded by deploying a plant rake. A GPS unit was used to navigate sequentially through each point location. A total of 72 points were surveyed resulting in 27 vegetated points and 45 unvegetated points. Eichhornia crassipes was the most common aquatic species observed (77.7%). Other nonnative aquatic plants observed were Hydrilla verticillata, Myriophyllum spicatum, Ludwigia uruguayensis, and Nelumbo nucifera. Native emergent aquatic plants observed were Justicia americana and Nelumbo lutea. The native submersed plant Ceratophyllum demersum was observed at 3.7% of vegetated points. Nonnative aquatic plants dominated the vegetation of Lake Columbus, indicating the need for aquatic plant management. Survey information incorporated into GIS layers is useful for management of invasive aquatic plants, biodiversity, and fish and wildlife resources.

Keywords: Invasive species, Ecology, Wetlands

Introduction

For years, invasive species have been a worldwide concern. Invasive species exhibit negative effects on the environment. Aquatic invaders disrupt many bodies of water affecting ecological interactions, disrupting water supply, and impeding boat traffic. Successful colonization within the water body may be attributed to their rapid growth rates by means of vegetative reproduction.

In the US, invasive aquatic plants cost an estimated $110M annually through losses, damage, and management expenses for weeds such as *Hydrilla verticillata* (hydrilla), *Salvinia molesta* (giant salvinia), and *Eichhornia crassipes* (waterhyacinth) (Pimentel et al. 2000). Large populations of waterhyacinth reduce phytoplankton production by competing for light (McVea and Boyd 1975). Invasive weeds such as hydrilla out compete and displace native submersed flora such as *Vallisneria americana* (Van et al. 1998). Human diseases such as malaria, encephalitis and schistosomiasis and oxygen depletion have been related with waterhyacinth and giant salvinia (Barrett 1989, Oliver 1993). Therefore, the importance of monitoring and management cannot be overlooked.

The point intercept method is a ground-truth sampling method that involves detection of the presence or absence of aquatic plants on an evenly spaced grid. As a result, surveys can be performed to provide estimates of the percent frequency occurrence of aquatic plants as well as establishing relationships between water depth and plant occurrence (Wersal et al. 2006, Case and Madsen 2004). Point intercept sampling efficiently and effectively provides information about the occurrence of plant species and their spatial distribution throughout a given area. The objective of this study is to determine and document native and non-native aquatic plant species occurrence in Lake Columbus.

Methods

Point intercept sampling method

Studies were conducted in Lake Columbus, Columbus, MS in 2005. Lake Columbus is 1,208 ha in size and belongs to the Tennessee-Tombigbee Waterway System. A grid of points placed in a 400m x 400m array was constructed over the lake using ArcGIS software. At the center of each grid, one point location was established, resulting in 72 total points in the survey. Point locations were numbered in order to navigate sequentially on the lake (Figure 1).

A Trimble® XRS GPS system, which has estimated 1m accuracy, was used to navigate to each point location using Universal Transverse Mercator (UTM) coordinates, datum WGS 84. The GPS was set up before hand with a template of aquatic species names. The presence of each species was recorded for each point. The survey was conducted on July 8, 2005, when water temperature was 30 °C.

Documenting diversity of aquatic vegetation

Survey point locations and lake boundary file were added to ArcGIS-ArcMap. Points with aquatic plant species presence labeled as 1 were selected. This selection allows determining the total vegetated points in the survey. Once vegetated points were determined, an additional selection by each plant species was performed. The following formula was used to determine % of plant species occurrence in the survey:

% Occurrence = (number of points present/ total vegetated points in the survey) * 100.

Table 1. Aquatic plant species occurrence on Lake Columbus, Lowndes County, MS.

Plant Species	Common Name	Native or Nonnative	% Frequency of Occurence
Ceratophyllum demersum L.	coontail	Native	3.7
Eichhornia crassipes (Mart.) Solms	waterhyacinth	Nonnative	77.7
Hydrilla verticillatia (L.f.) Michx.	hydrilla	Nonnative	3.7
Justicia americana (L.) Vahl	waterwillow	Native	11.1
Ludwigia uruguayensis (Camb.) Hara	waterprimrose	Nonnative	3.7
Myriophyllum spicatum L.	Eurasian watermilfoil	Nonnative	3.7
Nulumbo nucifera Gaertn.	sacred lotus	Nonnative	3.7
Nelumbo lutea Willd.	American lotus	Native	7.4

Results and Discussion

The 72 points surveyed resulted in 27 (37.5 %) vegetated points and 45 (62.5%) unvegetated points or open water. Nonnative plant species observed were: *Eichhornia crassipes, Hydrilla verticillata, Myriophyllum spicatum, Ludwigia uruguayensis* and *Nelumbo nucifera* (Table 1). However, a few native plants species such as *Justicia americana, Nelumbo lutea* and *Ceratophyllum demersum* also were found (Table 1).

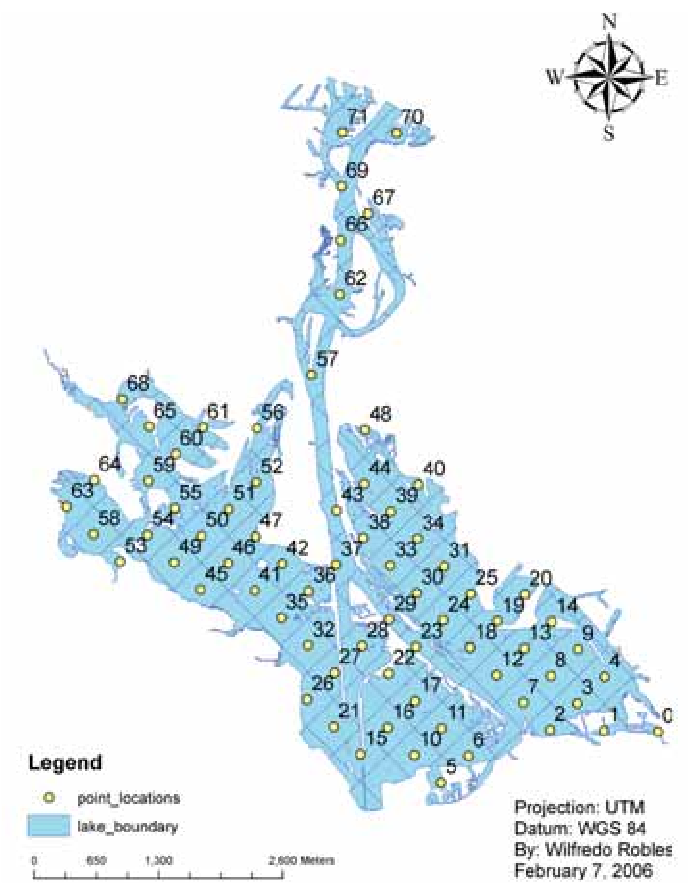

Figure 1. Point intercept survey locations on Lake Columbus, MS.

Two of the vegetated points were found with more than one aquatic plant species. *Eichhornia crassipes* was the most common aquatic plant species observed (77.7%). Similar results were reported by Ferrer-Montaño and Dibble (2002) in Lake Aliceville where *E. crassipes* was the most abundant aquatic plant species. Lake Aliceville is part of the Tennessee-Tombigbee waterway and is located south of Lake Columbus. Submersed nonnative plants such as: *H. verticillata, M. spicatum* and emersed *L. uruguayensis* and *N. nucifera* were also found. Although *H. verticillata* and *M. spicatum* are highly aggressive submersed plants, they were not very common in Lake Columbus. The limited distribution of these two submersed plants may be influenced by waterhyacinth shading at the vegetated points. *L. uruguayensis* and *N. nucifera* were the most dominant aquatic plant species at the points where they were found.

Conclusion

Aquatic plant species in Lake Columbus were dominated by nonnative plants, of which three of them (*E. crassipes, H. verticillata*, and *M. spicatum*) are considered among the most invasive aquatic plant species in United States. The abundance of *E. crassipes* suggests that management is necessary. Although *E. crassipes* was the most abundant species, the growth of *H. verticillata* and *M. spicatum* should be managed to prevent further infestation in the future. The current study may be used as a baseline for further surveys in Lake Columbus for monitoring and detecting changes in aquatic plant diversity.

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Corresponding Author:
Wilfredo Robles, Graduate Student, PhD
GeoResources Institute
Mississippi State University
117 Dorman Hall
Box 9555
Mississippi State, MS 39762

Survey of Invasive and Native Aquatic Plants in the Ross Barnett Reservoir

Ryan M. Wersal, John D. Madsen, Mary Love Tagert

Ross Barnett Reservoir is a 33,000-acre impoundment on the Pearl River and serves as the primary source of drinking water for the City of Jackson. It also provides recreational opportunities in the form of fishing, boating, water sports, and onshore camping and hiking; activities that bring revenue to the state. Invasive aquatic plants have become increasingly troublesome in recent years, specifically impacting navigation, fishing, and reducing the aesthetics of waterfront properties. To assess the distribution and abundance of invasive species in the Reservoir a point intercept survey was conducted on a 300 meter by 300meter grid in June of 2005. A plant rake was deployed at each of the 1,423 points visited. The primary areas of infestations were in the shallow upper reservoir, Pelahatchie Bay, and along the eastern shoreline. Alligatorweed and American lotus were the most common plant species observed (10.0 % and 8.2% respectively), followed by pennywort (3.0%), water hyacinth (2.4%), water primrose (2.3%), and parrotfeather (0.4%). Plant species presence may be influenced by light availability in different locations within the reservoir as noted by light profiles. Light transmittance at all sites was less than 20% in the upper 1 meter of the water column. The exotic invasive species, especially alligatorweed and water hyacinth, due to their growth habits can infest a large area of the reservoir if left unmanaged.

Keywords: Invasive species, Ecology, Wetlands

Introduction

Invasive aquatic plants are an increasing problem to water resources in Mississippi and most other states around the country (Madsen 2004). These plants generally are introduced from other parts of the world, some for seemingly beneficial or horticultural uses (Madsen 2004). Invasive plants affect aesthetics, drainage, fishing, water quality, fish and wildlife habitat, flood control, human and animal health, hydropower generation, irrigation, navigation, recreation, and ultimately land values (Rockwell 2003). It is estimated that over $100 million is spent annually in the United States for the control of aquatic weeds (Rockwell 2003). The Ross Barnett Reservoir is the largest surface water impoundment in Mississippi (33,000 acres) and serves as the primary drinking water supply for the city of Jackson, Mississippi's capital city. It is surrounded by extensive residential growth, approximately 50 residential subdivisions and over 4,600 homes. The reservoir provides recreational opportunities in the form of 5 campgrounds, 16 parks, 22 boat launches, 3 handicapped-accessible trails, and 2 multi-purpose trails.

Invasive aquatic plants have become increasingly problematic in the Ross Barnett Reservoir by impeding navigation channels, reducing recreational fishing opportunities, and reducing access for users of the reservoir. A long-term management plan is required to address the current and potential problems posed by the presence of invasive aquatic plants. The objective of this study was to assess the reservoir's plant community by mapping the current distribution of aquatic plants throughout the reservoir. The survey will serve as the starting point in the development of a long-term aquatic plant management plan.

Methods

A point intercept survey (Madsen 1999) was conducted on a 300 m grid in June of 2005 to assess the distribution and abundance of aquatic plants in the Ross Barnett Reservoir. Points were sampled throughout the Reservoir and along the main channel of the Pearl River as far north as Low Head Dam (Figure 1). The original boundary of the reservoir and a portion of the Pearl River were sampled, as well as areas of the reservoir that were not accessible by boat (Figure 1). Due to their inaccessibility these areas were not sampled. A hand-held computer (Hewlett Packard 2110 Ipaq) outfitted with a GPS receiver (Holux GM-270) was used to navigate to each point. Spatial and presence/absence data were directly recorded in the hand-held computer using Farm Works® Farm Site Mate software. Data were recorded in database

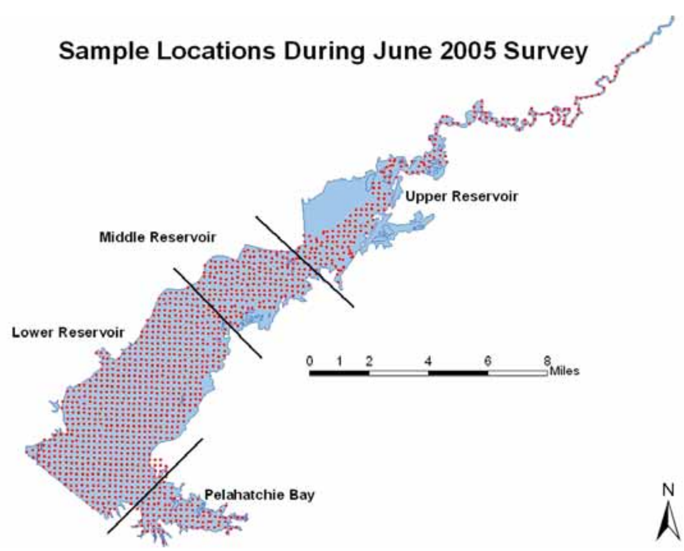

Figure 1. Points sampled on the Ross Barnett Reservoir during the survey conducted in June of 2005.

templates using specific pick lists constructed exclusively for this project. A total of 1,423 points were sampled during the survey by deploying a rake to determine the presence or absence of aquatic macrophyte species at these points. Percent frequency of occurrence was calculated using the total number of points sampled to give an estimate of plant occurrence throughout the entire reservoir not just littoral zone.

Water depth was taken at each of the sample points. Light intensity was recorded at Pelahatchie Bay, Lower Reservoir (2 sites), Middle Reservoir (2 sites), and Upper Reservoir sites using a LiCor light meter enabled with a submersible photosynthetically active radiation sensor as well as an incident PAR sensor. All measurements were taken in 0.5-meter intervals from the water surface to the Reservoir bottom. Light extinction coefficients (K_d) were calculated for each site as an index of how rapidly light is attenuated in the water column.

$$K_d = [\ln(I_{z1}) - \ln(I_{z2})] / (z_2 - z_1) \quad (1)$$

Where z = the water depth at a given point and I = the light intensity at that point. The greater the coefficient indicates the more rapidly light is attenuated.

Also, the maximum depth of plant colonization (Z_c) (Vant et al. 1986) was calculated using the light extinction coefficients (K_d) for each site.

$$Z_c = 4.34/K_d \quad (2)$$

Results

A total of 19 species of aquatic or riparian plants were observed during the survey. Of the 19 species, 14 are most often found in aquatic systems and 3 are exotic invasive species (Table 1). Alligatorweed, an exotic invasive, was observed most often (10%), followed by American lotus (8.2)%, a native species. The two other exotic invasive species observed during the survey were water-hyacinth and parrotfeather. The distributions of invasive species were located primarily in the Upper Reservoir, along the eastern shoreline of the Middle and Lower Reservoir, and in Pelahatchie Bay. Other species found during the survey include coontail, fragrant waterlily, American pondweed, duckweed, frog's-bit, cattail, soft-stem bulrush, and arrowhead (Table 1). Waterprimrose was the most common native species observed (2.3%).

In general, the occurrence of aquatic plants increased in the Upper Reservoir and Pelahatchie Bay in shallow water. Species occurrence was low in parts of the Middle and Lower Reservoir where water was deeper. Maximum depth of plant colonization was greatest in a portion of the Middle Reservoir and lowest in Pelahatchie Bay, indicating that water depths of less than 1.5 meters are favorable for rooted aquatic plants (Table 2). Light intensities

Table 1. Percent frequency of occurrence aquatic plant species mapped within the Ross Barnett Reservoir, June 2005.

Species Name	Common Name	Native (N) or Exotic (E), or Invasive (I)	% Frequency
Alternanthera philoxeroides	alligatorweed	E I	10.0
Nelumbo lutea	American lotus	N	8.2
Hydrocotyle ranunculoides	pennywort	N	3.0
Eichhornia crassipes	water hyacinth	E I	2.4
Ludwigia peploides	water primrose	N	2.3
Myriophyllum aquaticum	parrotfeather	E I	0.4
Ceratophyllum demersum	coontail	N	2.2
Nymphaea odorata	fragrant waterlily	N	2.1
Potamogeton nodosus	American pondweed	N	1.5
Lemna minor	common duckweed	N	1.3
Limnobium spongia	Frog's-bit	N	0.7
Typha sp.	cattail	N	0.6
Scirpus validus	softstem bulrush	N	0.6
Sagittaria latifolia	arrowhead	N	0.5

at these locations were generally reduced to less than 20 percent of surface light intensity within the upper 100 cm of the water column (Figure 2).

Discussion

Rooted submersed plants growing in the Ross Barnett Reservoir are limited by water depth and subsequent light extinction in the water column. Light extinction coefficients ranging from 0.5 to 4.0 are considered optimal in an aquatic ecosystem (Madsen et al. 1994). Extinction coefficients in the Reservoir tended to approach the middle to upper threshold of this range, indicating that light availability is likely limiting the growth of rooted plants. Water depths were shallower in the Upper Reservoir and in Pelahatchie Bay, areas where plant presence was greatest as they were better able to overcome light deficiencies in the shallower water. Data from this study indicate that light transmittance is less than 20% in

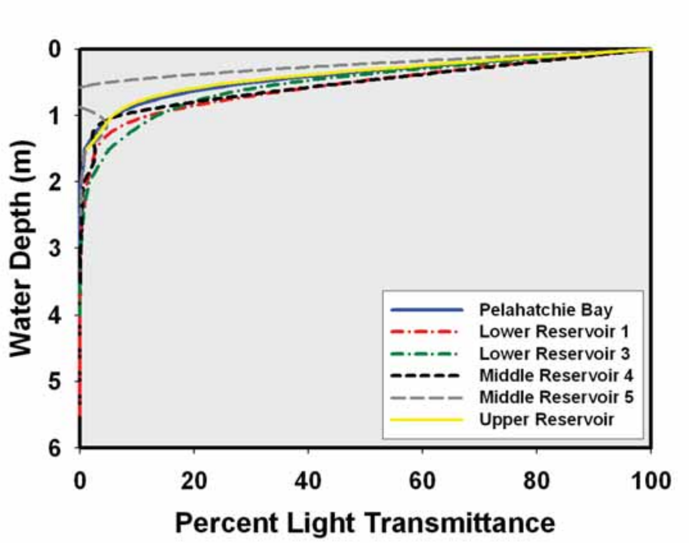

Figure 2. Light profiles for six sites in the Ross Barnett Reservoir.

the upper 100 cm of the water column. In general, submersed plants are located at depths where at least 21% of light reaches the bottom (Chambers and Kalff 1985). The corresponding mean maximum depth of plant colonization (Zc) is approximately 1.6 ± 0.1 meters based on light intensity measurements, meaning that rooted plants, preferably native species, would be able to colonize 22% or 7,200 acres of the Ross Barnett Reservoir. However, this relationship is exacerbated by the presence of floating invasive species, such as waterhyacinth, and canopy-forming species such as alligatorweed, pennywort, and waterprimrose, as they shade native species and can out-compete native species for available nutrients.

The Ross Barnett Reservoir has areas, mainly in the Upper Reservoir and Pelahatchie Bay, which can promote substantial macrophyte growth due to lower water depths and increased light availability. Currently, it appears that infestations of invasive species are confined to these two areas; however, the Middle Reservoir has large areas that could be colonized by nuisance species. Hydrilla (Hydrilla verticillata) was detected in July 2005 at three locations in the Reservoir. Two of the three infestations including the largest infestation (~ 67 acres) are in the Middle Reservoir site. The third and smallest (~ 3 acres) infestation of hydrilla is on the boarder of the Middle Reservoir and Upper Reservoir sites. Hydrilla is a submersed aquatic macrophyte introduced into the US in the 1960's. Hydrilla is considered a serious threat to water bodies in the United States as it can rapidly out-compete native plants establishing dense monotypic stands. If left alone these dense beds of hydrilla can be difficult and expensive to control and or manage. Florida spends approximately $14.5M each year on hydrilla control (Pimentel et al. 2000). If control efforts are not implemented in the Ross Barnett Reservoir the hydrilla infestation could encompass over 7, 200 acres. Future research needs to continue monitoring aquatic plant distribution to assess changes and spread in nuisance species populations; implement and assess techniques to control nuisance species and promote the growth of more desirable native plants; and implement and assess herbicide applications to control the hydrilla infestations.

Table 2. Light extinction coefficients (Kd), estimated maximum depth of macrophyte colonization (Zc), and maximum observed depth in the Ross Barnett Reservoir, June 2005.

Site	K_d	Z_c (m)	Maximum Depth (m)
Pelahatchie Bay	3.8	1.1	3.0
Lower Reservoir (1)	2.3	1.9	5.5
Lower Reservoir (3)	2.3	1.9	4.0
Middle Reservoir (4)	2.0	2.2	3.5
Middle Reservoir (5)	3.0	1.4	2.5
Upper Reservoir	3.0	1.5	1.5

Acknowledgments
We would like to thank the Pearl River Valley Water Supply District for funding this project. Thank you to Billy Lester, Bill Gallagher, Josh Cheshier, Bill Warren, and Matt Persaud for help during the survey and Wade Givens for setting up our hand held computers with the appropriate software. We also thank Gary Ervin, Victor Maddox, and Wade Givens for reviews of this manuscript. Publication number PS-10927 of the Mississippi Agriculture and Forestry Experiment Station.

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Corresponding Author:
Ryan M. Wersal
GeoResources Institute
Mississippi State University
Box 9652
Mississippi State, MS 39762
Phone 662–325–4595
Fax 662–325–7692
E-mail: rwersal@gri.msstate.edu

Session C: Katrina—Inland Impacts

Dean Pennington, Moderator

Jeannie Bryson
Use of Heat Tracing to Quantify Stream/Ground-water Exchanges During and After Hurricanes Katrina and Rita in the Bogue Phalia, Northwestern, Mississippi

Fred Howell
Short Term Impacts of Hurricane Katrina on the Dissolved Oxygen Content of the Lower Leaf River Water in South Mississippi

Jake Schaefer
Effects of Hurricane Katrina on the Fish Fauna of the Pascagoula River Drainage

Use of Heat Tracing to Quantify Stream/Groundwater Exchanges During and After Hurricanes Katrina and Rita in the Bogue Phalia, Northwestern, Mississippi

Jeannie R. Bryson and Richard H. Coupe

Heat provides a natural tracer of groundwater movement, which is readily tracked by measuring changes in streambed sediment temperature. Heat tracing has been used successfully in a number of studies to determine whether a stream segment is gaining or losing water. When water is moving into or out of the bed sediments it carries with it a measurable amount of heat, creating different temperature profiles in the bed sediments depending upon whether ground water is discharging into the surface water or surface water is moving into the sediments. This difference can be exploited to determine which direction the water is moving and, also, to estimate some hydrologic properties of the bed sediment. Quantitative estimates of streambed water fluxes into or out of the stream can be determined by using inverse modeling to fit simulated sediment temperatures to measure temperatures.

In June 2005, a set of three piezometers were installed in the Bogue Phalia upstream from the stream gage near Leland, Miss., to monitor groundwater temperature. The piezometers were installed laterally across the stream and equidistant from each other. In each piezometer was an array of four temperature loggers deployed at 20cm, 50cm, 1m, and 2m below the sediment water interface. These loggers, plus one installed in the Bogue Phalia to measure surface water temperature, recorded temperature at 15 minute intervals. Precipitation in the Bogue Phalia Basin for the months of June to October 2005 was below normal, and consequently, the streamflow generally was below the long-term average. The temperature profile from the piezometers indicates that the Bogue Phalia was a gaining stream during most of this time. However, two anomalous precipitation events, Hurricanes Katrina and Rita, caused a sharp rise in streamflow over a short period of time. The temperature profiles indicated that warmer surface water was pushed into the stream bed. The temperature increased from 3 to 4 °C at the 2 meter depth, beginning soon after the stream started to rise. The increase in temperature at the other depths was similar, but commenced sooner, depending upon proximity to the surface. There were spatial differences in the temperature profiles. The west side of the stream was warmer than the east side, and the temperature changes occurred more rapidly on the west side than the east and more closely followed the hydrograph. The peak temperature corresponded with the peak of the hydrograph on the west side, but was offset on the falling limb of the hydrograph on the east side. This would indicate that the hydraulic conductivity of the bed sediments on the west side was greater than on the east side.

As a quantitative tool, the U.S. Geological Survey (USGS) heat and groundwater transport model, VS2DI, was used to develop two-dimensional simulations of water fluxes into and out of the streambed sediments. Inverse modeling fits of simulated to measured sediment temperatures yielded estimates of fluxes across the streambed surface, which substantiated variations in hydraulic conductivity on the west versus east side of the Bogue Phalia site.

Keywords: Groundwater, Surface Water, Hydrology

Corresponding Author:
Jeannie R. Bryson, Hydrologist
U.S. Geological Survey

Short Term Impacts of Hurricane Katrina on the Dissolved Oxygen Content of the Lower Leaf River Water in South Mississippi

Fred G. Howell, Acker Smith, Brent Wally

One of the impacts of Hurricane Katrina on the quality of surface water in rivers and streams of the affected area was a serious depletion of dissolved oxygen content. Personnel at the Koch Cellulose pulp mill in New Augusta became aware of the problem shortly before they were to resume effluent discharge to the Leaf River; consequential contact with the Mississippi Department of Environmental Quality in Jackson dictated that dissolved oxygen concentrations in the receiving water (Leaf River) must be at least 5 ppm before such release could occur. Therefore, a monitoring effort was done at strategic points upstream and downstream of the mill's effluent outfall to monitor the dissolved content of the river water. Initial sampling was began on 7 September and terminated on 11 September. Dissolved oxygen readings ranged from 2.35 to 3.2 ppm for all sampling locations sampled on the September 7. Monitoring data were collected at three to four hours intervals until it was obvious that the sustained dissolved oxygen readings were well into the fives. The monitoring effort was terminated late afternoon September 11.

Keywords: Climatological Processes, Surface Water, Water Quality

Corresponding Author:
Fred G. Howell, Ph.D.
Professor of Biological Sciences
118 College Drive #5018
University of Southern Mississippi
Hattiesburg, MS 39406
Phone 601-266-4750
E-mail: Freddie.Howell@usm.edu

Effects of Hurricane Katrina on the Fish Fauna of the Pascagoula River Drainage

J. Schaefer, P. Mickle, J. Spaeth, B.R. Kreiser, S.B. Adams, W. Matamoros, B. Zuber, P. Vigueira

Large tropical storms can have dramatic effects on coastal, estuarine and terrestrial ecosystems. However, it is not as well understood how these types of disturbances might impact freshwater communities further inland. Storm surges can change critical water quality parameters for kilometers upstream, potentially causing subtle shifts in community structure or more drastic fish kills. Hurricane Katrina was one of the largest such storms to strike the coast of Mississippi and provides an opportunity to examine these effects in areas where we had pre-storm fish community data. In the weeks following the storm, fish kills were reported from some of the lower portions of the drainage. As part of a separate ongoing research project, monthly electrofishing surveys were conducted in 2004 and 2005 at ten sites throughout the Pascagoula River drainage. Survey effort was evenly divided among bank, sandbar and open channel habitats at each site from June through November. Sites ranged from 90 to 250 river km from the mouth of the Pascagoula River, creating a gradient of storm impact with which to judge storm effects on big river fish fauna. In addition, we compare pre- and post-storm communities along Black Creek, a medium-sized tributary, as well as in a group of small, headwater tributaries.

Keywords: Ecology, Surface Water

Introduction

Ecological disturbances are characterized by their unpredictability and destructive power (Resh et al. 1988; Poff and Ward 1990, Fausch and Bramblett 1991). While ecologists have long recognized the importance of understanding disturbance impacts, the most severe disturbances are difficult to study due to their rarity and force. Investigators often lack complete datasets of appropriate spatial and temporal scales to assess pre- and post-disturbance community structure. As a result, studies linking changes in community structure to major disturbances are generally lacking (Tillman 1989; Collins 2000).

Hurricanes are one of the most severe forms of disturbances to coastal ecosystems. While hurricanes are natural phenomena, they are disturbances and their impacts may be enhanced in areas where anthropogenic changes to the landscape have reduced ecosystem resilience to such stressors (Naiman and Turner 2000; Mallin et al. 1999; Mallin et al. 2002). While the mechanisms for hurricane impacts on estuarine (e.g., erosion, storm surge) or inland terrestrial systems (e.g. wind damage) are fairly obvious, similar mechanisms are not as well understood for river ecosystems. Floods are a necessary component of riverine ecosystems, and river discharges after hurricanes are often not significantly greater than typical high water levels. One might also expect riverine systems to be well sheltered from direct wind effects. However, the combination of high winds and intense precipitation can defoliate trees and deposit large amounts of litterfall material in streams. Litterfall mass from hurricanes Hugo and Gilbert were up to twice as high as mean annual inputs (Lodge and McDowell 1991). Fallen trees combined with intense precipitation might also increase erosional deposition of sediments. In smaller headwater streams, downed trees and debris can alter flows and habitat structure by creating debris dams (Golladay et al. 1986). Finally, downstream portions of coastal drainages might also be exposed to storm surges that could raise salinities or disturb sediments.

One of the ways hurricanes are thought to impact fish communities is through immediate drops in dissolved oxygen (DO). While the precise cause of the lower DO is not known, there are a number of possible contributing factors in our study system. First, a spike in the biological oxygen demand (BOD) caused by high temperatures and leaf and wood debris input. Second, the storm surge may have disturbed anoxic sediments in the Gulf of Mexico or lower Pascagoula River (Buck 2005). Finally, water treatment plants or other sources of polluted water might have found their way into the river as a result of flooding (MS Department of Wildlife, Fisheries and Parks 2005).

Study System and Hurricane Katrina

The Pascagoula River system is the largest unimpounded drainage remaining in the continental United States (Dynesius and Nilsson 1994). As a relatively pristine system, it is ideal for studying fish community dynamics and disturbance effects. Additionally, the system has been well studied in the past and the authors had two independent, ongoing projects documenting fish community structure before hurricane Katrina struck.

Hurricane Katrina struck the Gulf Coast on August 29, 2005. The eye of the storm struck the coast roughly 113 km west of the mouth of the Pascagoula River and proceeded to move in a north by northeast direction over much of the drainage. Winds in excess of 209 km/h were recorded throughout most of the drainage. The storm surge in Jackson County (where the Pascagoula River enters the Gulf of Mexico) was estimated at 5 m, meaning salt water was likely driven inward over many river kilometers. In the days and weeks following the storm, extremely low DO levels were recorded in large rivers and streams in the lower portions of the drainage (Howell 2006). In the weeks following the storm there were numerous reports of fish kills, primarily in the lower reaches of the Pascagoula River (Buck 2005; MS Department of Wildlife, Fisheries and Parks 2005; Todd Slack, pers. comm.).

The purpose of this study is to assess impacts of hurricane Katrina on fish communities in the Pascagoula drainage by comparing pre- and post-hurricane fish community structure. Data were included from three portions of the drainage, 1st-4th order smaller streams, Black Creek (a medium-sized tributary) and the three large rivers (Leaf, Chickasawhay and Pascagoula) in the drainage (Fig. 1).

Materials and Methods

Pre- and post-hurricane data were compiled for three portions of the drainage: 1st -4th order streams (hereafter referred to as small streams) in the headwaters, the length of Black Creek and the three large rivers (Leaf, Chickasawhay and Pascagoula). Sampling was done by seining in small streams and Black Creek and by boat electrofishing in large rivers. Seines varied in size based on the demands at particular sites but were always 4.8 mm mesh. Seining effort was typically 30-45 minutes per site during which time all available habitat was sampled. For electrofishing surveys, each

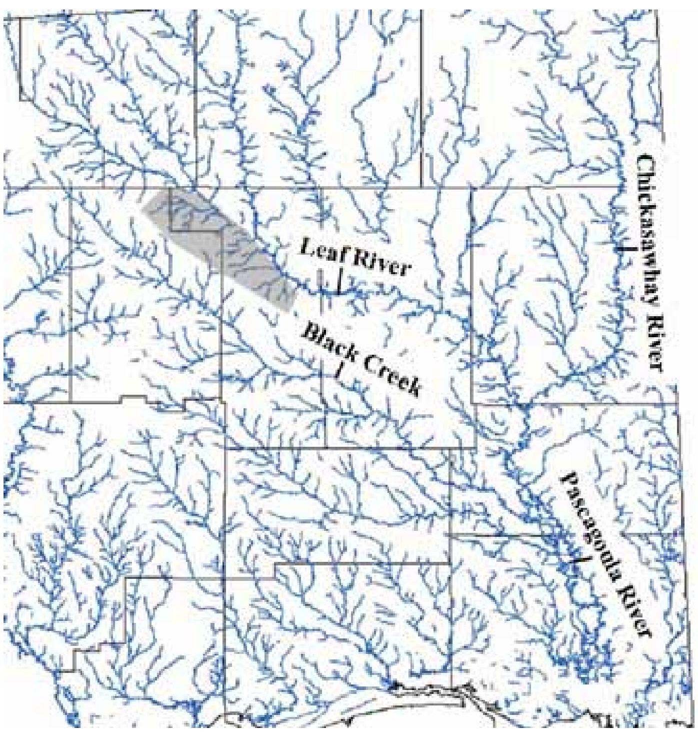

Figure 1. Map of the Pascagoula drainage with the rivers and creeks sampled labeled. The small streams sampled are not labeled but are all within the shaded polygon.

site contained sand bar, open channel and bank habitat that were electrofished for 400 seconds each. Fish from electrofishing surveys were identified in the field and then released. Fish from seining were preserved in 10% formalin, identified in the laboratory, transferred to 70% EtOH and then deposited in the USM Museum of Ichthyology. For most analyses, we will consider each dataset independently. Although we employed varied sampling methods, the approaches taken were well suited for each type of stream. Finally, pre- and post-hurricane sampling methods were consistent within stream types and datasets.

Small Stream Dataset

Five to seven small stream sites were sampled monthly from April 2005 through April 2006. Some site locations changed in certain months depending on water levels or other conditions. Sites were all within six streams: the Bouie River, Hayden Creek, Big Creek, Shelton Creek, Beaver Creek and Okatoma Creek. There was a total of 36 pre- and 44 post-hurricane samples.

Black Creek Dataset

Steve Ross, along with colleagues, conducted extensive surveys of the Black Creek fish fauna in the 1970's and 1980's and from 2000-2003. These surveys involved a variety of gears and efforts and were sometimes targeted on a single taxa. To control for this, we examined all 176 collections made in the creek during this period. We eliminated records that did not use seines (e.g. some sites were sampled with eel traps) or that did not sample on the appropriate scale (e.g. some sites were divided into very small subsections). Original field notes were available to clarify questions we had about gear or sampling protocol. Because Black Creek represents a gradient from a smaller stream to a larger river, we divided it into five longitudinal sections defined by road or boat access points. We compiled 63 pre- and 11 post-hurricane samples among the five sections. All post-hurricane data were collected in April 2006 by re-sampling localities that were part of the pre-hurricane database.

Large Rivers Dataset

Ten large river sites were sampled by monthly boat electrofishing from June through November in 2004 and 2005. Four sites were in the Leaf River, three in the Chickasawhay River and three in the Pascagoula River (Mickle 2006). There was a total of 67 pre- and 14 post-hurricane samples. Post-hurricane sampling was not conducted in September 2005 but began the next month.

Statistical Analyses

The complete dataset consisted of 235 collections (166 pre- 69 post-hurricane) and 92 species. Descriptive statistics calculated included the proportion of samples that contained individuals of a species and the mean rank abundance of a species. To calculate mean rank abundance, rank abundances for a species (within individual collections) were averaged across all collections for that

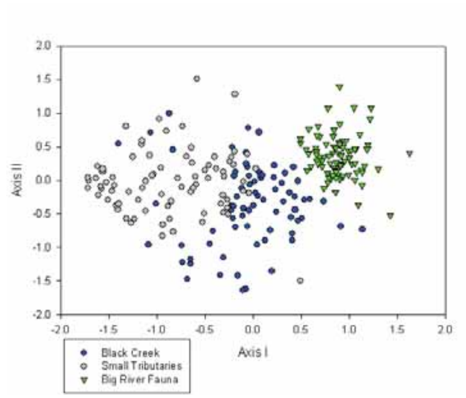

Figure 2. NMS plot of all three datasets from Mississippi rivers.

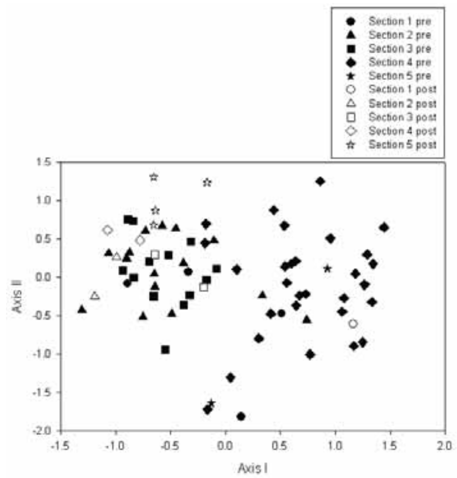

Figure 3. NMS plot of samples from Black Creek. Longitudinal creek sections are represented by different symbols. Open symbols represent post-hurricane samples, closed symbols are pre-hurricane.

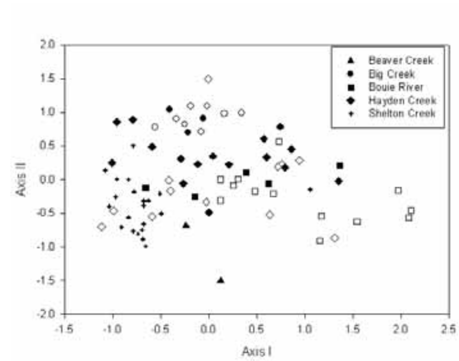

Figure 4. NMS plot of samples from the small streams. Individual streams are represented by different symbols. Open symbols represent post-hurricane samples, closed symbols are pre-hurricane.

locality or stream. For example, if a species was present in 4 out of 5 samples at a site and it was most abundant in two samples and fourth in rank abundance in the other two, we would report its mean rank abundance as 2.5 and present in 80% of samples. Community similarity was assessed visually by clustering patterns in two-dimensional ordinations (nonmetric multidimensional scaling, NMS) based on Bray-Curtis similarity among sites. Ordinations were run for the entire matrix to assess community differences among the three datasets and then on each of the datasets individually to assess hurricane effects. All analyses were performed on raw abundance data using the software package Primer (version 5.0).

Results

Stress values on all NMS analyses were less than 0.24 indicating that the analysis was able to accurately place samples in two dimensions so that Euclidian distance among samples in NMS space and Bray-Curtis similarity were highly correlated. Stress values in this range are considered acceptable for community analyses.

Overall Community Patterns

The large river fish communities were distinct from those in Black Creek and the small streams (Fig. 2). Large river collections contained a number of species not found in the other two datasets (Tables 1-3). Some of the more abundant distinct large river species were: *Alosa alabamae, Alosa chrysochloris, Aplodinotus gruniens, Anchoa mitchilli, Carpiodes velifer, Hiodon tergisus, Lepisosteus oculatus* and *Mugil cephalus*. Black Creek and the small streams partially overlapped (Fig. 2) and contained a number of species not found in the large river samples: *Etheostoma lynceum, E. stigmaeum, E. swaini, Luxilus chrysocephalus, Lythrurus roseipinnis, Notropis baileyi*. (Tables 1-3). Within Black Creek, the pre-hurricane upstream and downstream sections (1-3 and 4-5, respectively) generally clustered separately (Fig. 3). In the small streams dataset, Shelton Creek and Big Creek clustered separately from the other three small streams, which broadly overlapped (Fig. 4).

Hurricane Impacts

There were no clear hurricane impacts in the small streams (Fig. 4), upstream sections of Black Creek (Fig. 3) or the Leaf and Chickasawhay rivers (Fig. 5). In each of these cases, the post hurricane community samples were clearly clustered with pre-hurricane samples. In contrast, the hurricane clearly affected fish communities in the Pascagoula River (Fig. 5) and downstream portions of Black Creek (Fig. 3). In downstream portions of Black Creek, post hurricane samples (open diamonds and stars of Fig. 3) did not cluster with pre-hurricane samples. In the Pascagoula River, post-hurricane samples (open circles in Fig. 5) did not cluster with pre-hurricane samples or with any of the other large river samples. In the Pascagoula River, the post-hurricane communities were numerically dominated by two species (*Anchoa mitchilli* and *Mugil cephalus*) not particularly abundant in previous samples in the large rivers (Table 3).

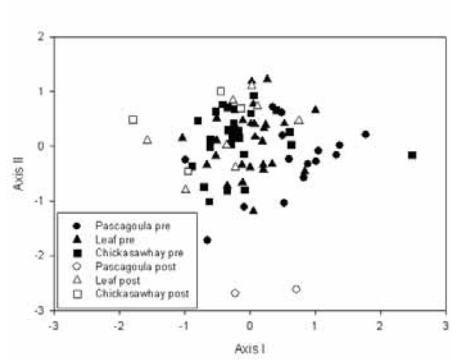

Figure 5. NMS plot of samples from the Leaf, Chickasawhay and Pascagoula rivers. Each river is represented by a different symbol. Open symbols represent post-hurricane samples, closed symbols are pre-hurricane.

Table 1. Pre- and post-Katrina mean rank abundance and proportion of samples containing species in the smaller tributaries sampled.

	Small Tributaries			
	Pre		Post	
	Abund.	Prop.	Abund.	Prop.
Ambloplites ariommus	8.5	5.6		
Ameiurus natalis	8.0	2.8		
Ammocrypta beani	5.3	27.8	5.4	31.8
Ammocrypta vivax	10.2	13.9	10.4	18.2
Cyprinella venusta	5.0	55.6	4.9	36.4
Ericymba buccata	6.0	38.9	5.8	27.3
Etheostoma chloroso-mum	16.0	2.8	6.0	2.3
Etheostoma lynceum	5.5	47.2	2.5	47.7
Etheostoma parvipinne	6.7	16.7	6.0	2.3
Etheostoma stigmaeum	6.4	55.6	3.5	54.5
Etheostoma swaini	6.2	50.0	6.6	34.1
Fundulus olivaceus	8.6	38.9	6.8	11.4
Gambusia affinis	6.9	30.6	8.4	25.0
Hypentelium nigricans	6.8	58.3	7.8	25.0
Ichthyomyzon gagei	6.7	8.3	4.0	2.3
Labidesthes sicculus			10.8	9.1
Lepomis cyanellus	7.0	8.3		
Lepomis gulosus			4.0	2.3
Lepomis macrochirus	6.6	38.9	7.0	13.6
Lepomis megalotis	7.0	16.7	9.2	11.4
Luxilus chrysocephalus	2.9	91.7	3.2	79.5
Lythrurus roseipinnis	4.0	58.3	5.3	27.3
Micropterus punctulatus	8.1	41.7	8.0	18.2
Micropterus salmoides	8.5	11.1	11.0	2.3
Micropterus sp	10.0	2.8		
Moxostoma poecilurum	10.5	22.2	12.3	9.1
Nocomis leptocephalus	6.4	52.8	7.5	31.8
Notemigonus crysoleu-cas	10.0	2.8		
Notropis baileyi	5.1	66.7	3.3	54.5
Notropis longirostris	4.7	58.3	4.2	52.3
Notropis texanus	8.7	27.8	6.9	31.8
Notropis volucellus	8.0	5.6	5.5	18.2
Notropis winchelli	9.3	36.1	6.7	27.3
Noturus leptacanthus	9.1	33.3	6.9	15.9
Opsopoeodus emiliae			8.8	13.6
Percina nigrofasciata	4.0	91.7	3.8	86.4
Percina sciera	8.1	27.8	6.0	38.6
Pimephales vigilax	8.0	2.8		
Pomoxis annularis			12.0	2.3

Discussion

The observed differences in community structure among the large rivers and the other two river types are not unexpected. While one would expect to see differences among such disparate habitats, it should be noted that different sampling techniques likely exacerbated these effects. Many of the species only sampled in the large river habitat are large bodied species that are extremely difficult to sample by seine. Seining and electrofishing are both known to selectively sample some taxa more effectively than others. With this sampling bias in mind, all of our tests for hurricane impacts were conducted within the three separate datasets where sampling gear and effort was standardized.

Community structure differed significantly between upstream and downstream sections of Black Creek. The upstream portions of Black Creek and many of the small stream samples had higher abundances of species that are considered riffle and gravel substrate specialists (Tables 1 and 2).

Hurricane Impacts

Hurricane impacts were most pronounced in the areas closest to the Gulf of Mexico. There were large changes in community structure in the Pascagoula River and lower portions of Black Creek. The remaining sites are all further (in river km) from the Gulf of Mexico and did not show any change in fish community structure. Pre- and post-hurricane samples in the smaller streams and upstream portions of Black Creek showed no shifts in community structure.

After the hurricane, fish kills were reported in the Pascagoula River and generally attributed to low DO. It is difficult to directly link species absence in post hurricane samples to tolerance of these

Table 2. Pre- and post-Katrina mean rank abundance and proportrion of samples containing species in the five sections of Black Creek. Section one is in the headwaters, section five near the confluence with the Pascagoula River.

	Section 1				Section 2				Section 3				Section 4				Section 5			
	Pre		Post		Pre		Post		Pre		Post		Pre		Post		Pre		Post	
	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.
Ambloplites ariommus													11.2	20.7						
Ammocrypta beani	3.8	83.3			3.7	76.9			7.0	92.3	9.0	100.0	7.1	31.0	4.0	100.0				
Aphredoderus sayanus													9.6	24.1			1.0	50.0	8.0	25.0
Carpiodes cyprinus									11.0	7.7					13.0	50.0				
Cyprinella venusta	2.4	83.3			1.8	92.3	4.0	100.0	1.7	100.0	2.5	100.0	4.4	65.5	1.5	100.0	3.0	50.0	1.3	100.0
Elassoma zonatum													10.9	24.1						
Ericymba buccata	4.0	16.7			3.9	61.5			4.5	84.6	1.0	50.0	5.8	20.7	7.0	100.0				
Erimyzon tenuis									14.0	7.7	9.0	50.0	13.5	6.9						
Esox niger			9.0	100.0	10.0	7.7	8.0	50.0					12.0	13.8			6.0	50.0		
Etheostoma lynceum	6.5	33.3			9.3	30.8			6.5	84.6	8.0	100.0	7.8	27.6						
Etheostoma stigmaeum	7.0	16.7			7.0	38.5	7.0	50.0	10.3	84.6	7.0	100.0	7.1	65.5	9.0	50.0				
Etheostoma swaini	1.0	16.7					7.0	50.0	10.5	15.4			6.0	55.2			6.0	50.0		
Fundulus notti													7.5	6.9	8.5	100.0				
Fundulus olivaceus	7.3	66.7	6.0	100.0	5.9	61.5	4.0	100.0	8.6	84.6	10.0	50.0	4.7	79.3	6.0	100.0	4.0	50.0	4.0	25.0
Gambusia affinis	5.0	16.7	3.0	100.0	8.5	15.4	6.0	50.0					8.3	31.0			6.0	50.0	1.0	25.0
Hypentelium nigricans	7.0	33.3			8.2	38.5			8.8	69.2	13.0	50.0	12.8	13.8						
Hybognathus nuchalis																			5.0	50.0
Ichthyomyzon gagei					7.0	7.7							20.0	3.4			6.0	50.0		
Ictalurus punctatus					7.0	15.4			12.0	15.4									4.0	25.0
Labidesthes sicculus	4.5	33.3	2.0	100.0	5.3	30.8	3.0	100.0	11.3	30.8	8.0	50.0	5.4	44.8	5.5	100.0	4.0	50.0	3.0	25.0
Lepomis cyanellus					15.0	7.7			11.0	7.7			20.0	3.4						
Lepomis gulosus													14.0	10.3						
Lepomis macrochirus	4.0	16.7	9.0	100.0	7.3	30.8	2.5	100.0	8.3	30.8			5.9	44.8	9.5	100.0	6.0	50.0	10.0	25.0
Lepomis marginatus	7.0	16.7											7.0	6.9						
Lepomis megalotis	7.5	33.3	5.0	100.0	6.3	53.8	5.0	50.0	6.5	46.2	9.5	100.0	5.2	51.7	3.0	50.0	3.0	50.0	5.5	50.0
Lepomis microlophus	2.0	16.7			9.0	7.7	5.0	50.0	5.0	7.7			25.0	3.4	13.0	50.0				
Lepomis miniatus													9.5	6.9						
Luxilus chrysocephalus					8.8	30.8			10.2	46.2	9.0	50.0	7.0	3.4	10.0	50.0				
Lythrurus roseipinnis	1.0	16.7	1.0	100.0	5.0	46.2	1.5	100.0	7.3	92.3	5.5	100.0	1.9	82.8	10.0	50.0	1.0	50.0		
Macrhybopsis aestivalis					14.0	7.7			10.3	23.1										
Macrhybopsis storeriana					15.0	7.7													4.5	50.0
Micropterus punctulatus	6.0	50.0			5.8	92.3	7.0	50.0	7.8	100.0	10.0	50.0	11.7	20.7	8.0	100.0				
Micropterus salmoides	8.0	16.7			10.5	15.4	7.0	50.0	19.0	7.7			8.6	17.2						
Minytrema melanops					10.0	7.7							10.7	10.3			6.0	50.0		
Moxostoma poecilurum					6.0	7.7			12.0	23.1			5.0	3.4						
Notemigonus crysoleucas									21.0	7.7			12.0	3.4						
Notropis atherinoides					11.0	15.4														
Notropis baileyi			7.0	100.0	10.0	7.7			16.0	7.7										
Notropis longirostris	2.0	100.0			2.9	100.0			3.3	100.0	1.5	100.0	7.4	17.2	1.5	100.0			2.0	50.0
Notropis texanus	4.0	50.0	9.0	100.0	6.0	7.7	7.0	50.0	11.0	15.4			2.8	72.4	7.0	50.0	2.0	50.0	3.7	75.0
Notropis maculatis			7.0	100.0			8.0	50.0												
Notropis volucellus					7.4	38.5			9.5	46.2	3.0	50.0	6.0	17.2	5.0	50.0	4.0	50.0	5.0	25.0
Notropis winchelli					7.4	38.5			11.8	30.8			9.7	10.3						
Noturus funebris	7.0	16.7											4.0	3.4						
Noturus leptacanthus	7.5	66.7							7.3	30.8			4.4	34.5	13.0	50.0	2.0	50.0		
Noturus nocturnus													12.0	6.9						
Opsopoeodus emiliae			4.0	100.0									12.0	17.2			4.0	50.0		
Percina nigrofasciata	5.8	66.7			7.1	53.8			6.3	100.0	3.5	100.0	4.6	86.2	13.0	50.0	6.0	50.0		
Percina sciera	8.3	50.0			7.7	53.8			9.0	69.2	8.0	100.0	9.0	20.7						
Percina sp					10.0	7.7			6.0	7.7										
Percina vigil	8.0	16.7			8.0	30.8			11.0	7.7					9.0	50.0			6.0	25.0
Pteronotropis welaka													7.0	6.9						
Trinectes maculatus	5.5	33.3							12.0	7.7									3.0	50.0

Table 3. Pre- and post-Katrina mean rank abundance and proportrion of samples containing species in the three large rivers of the Pascagoula drainage.

	Chickasawhay River				Leaf River				Pascagoula River			
	Pre		Post		Pre		Post		Pre		Post	
Species	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.	Abund.	Prop.
Alosa alabamae	11.5	8.3			6.0	39.3	3.7	37.5	6.0	37.5		
Alosa chrysochloris	8.0	20.8	9.0	25.0	8.9	25.0			11.5	12.5		
Ambloplites ariommus	7.6	29.2	7.0	50.0	11.2	17.9			11.0	6.3		
Ammocrypta beani					9.0	3.6						
Anguilla rostrata	10.5	12.5	9.0	25.0								
Aplodinotus grunniens					7.5	21.4	7.7	37.5				
Anchoa mitchilli					4.0	3.6					3.0	50.0
Carpiodes velifer	3.4	87.5	3.3	75.0	2.9	92.9	2.8	75.0	3.3	100.0		
Cycleptus meridionalis	7.0	16.7			10.7	10.7	6.0	12.5	7.5	25.0		
Cyprinella venusta	1.3	95.8	2.0	75.0	1.9	92.9	1.3	75.0	2.9	100.0	4.0	50.0
Dorosoma cepedianum	5.4	62.5	3.5	50.0	7.4	57.1	5.4	87.5	3.8	50.0	2.0	50.0
Dorosoma petenense	2.0	4.2	9.0	50.0			5.0	12.5	5.0	6.3	2.0	50.0
Hypentelium nigricans	7.4	25.0	15.0	25.0	8.7	35.7	7.8	62.5	11.0	6.3		
Hybognathus nuchalis	3.5	37.5	6.0	25.0	5.9	42.9	5.0	37.5	2.4	87.5	4.0	50.0
Hiodon tergisus	8.4	33.3	4.0	50.0	8.9	39.3			9.0	6.3		
Ictiobus bubalus	7.3	12.5			7.0	17.9	12.0	12.5	8.0	25.0		
Ictalurus furcatus	8.5	8.3	11.0	25.0	11.7	10.7			7.0	6.3		
Ictaluriu punctatus	7.8	33.3	8.0	75.0	8.6	46.4	5.5	50.0	7.8	31.3		
Labidesthes sicculus					8.0	7.1	6.0	25.0	7.0	12.5		
Lepomis macrochirus	5.8	50.0	6.5	50.0	8.3	25.0	4.0	12.5	6.0	25.0		
Lepomis megalotis	4.5	62.5	3.7	75.0	6.1	78.6	6.4	62.5	6.3	68.8		
Lepomis microlophus	6.3	12.5	9.0	25.0	8.0	7.1	8.3	37.5	9.0	12.5		
Lepisosteus oculatus	6.6	58.3	8.0	75.0	7.4	32.1	8.0	12.5	6.5	87.5	2.0	50.0
Lepisosteus osseus	14.0	8.3	6.0	25.0	8.8	14.3	7.0	25.0	8.0	18.8		
Macrhybopsis storeriana	4.7	37.5	15.0	25.0	7.4	39.3	8.3	37.5	8.3	18.8		
Micropterus punctulatus	5.6	83.3	6.0	50.0	5.8	82.1	5.8	75.0	6.4	75.0		
Micropterus salmoides	8.5	33.3	5.7	75.0	7.4	25.0	5.0	25.0	8.9	43.8		
Moxostoma poecilurum	5.7	75.0	2.3	75.0	5.5	82.1	4.7	87.5	8.0	31.3		
Mugil cephalus	4.5	33.3	7.5	50.0	3.9	85.7	5.7	75.0	4.5	81.3	1.0	100.0
Notropis atherinoides	4.3	79.2	2.0	50.0	5.6	71.4	2.6	62.5	6.2	93.8	4.0	50.0
Notropis longirostris	6.3	41.7			8.0	46.4	8.7	37.5	6.9	50.0		
Notropis texanus	7.7	12.5			8.3	10.7	9.0	12.5	2.0	6.3		
Notropis volucellus	9.0	4.2	9.0	25.0	10.0	3.6	9.0	12.5	9.0	6.3		
Notropis winchelli					9.0	3.6	9.0	12.5				
Noturus leptacanthus	14.0	4.2										
Percina nigrofasciata	9.0	4.2			9.0	3.6						
Percina lenticula	10.5	8.3			11.0	3.6			7.0	6.3		
Pimephales vigilax	8.3	25.0	11.0	25.0	8.0	10.7	7.0	37.5	14.0	6.3		
Pomoxis nigromaculatus			9.0	25.0	8.3	10.7	9.0	12.5				
Strongylura marina									7.0	6.3		

conditions. However, post-hurricane samples from the Pascagoula River contained no individuals from two typically abundant families, Centrarchidae and Catostomidae, that made up roughly 18% of all individuals captured in pre-hurricane samples. Some of the most abundant species (e.g. *Mugil cephalus* and *Anchoa mitchilli*, Table 3) in post-hurricane Pascagoula River samples were diadromous or peripheral marine-estuarine species that would likely be tolerant of salinity spikes associated with storm surge.

The community changes in lower sections of Black Creek included decreases in the abundance of some previously abundant darter species (*Percina* sp. and *Etheostoma* sp.) and *Lythrurus roseipinnis* which had been the most abundant species. Conversely, some species (e.g., *Hybognathus nuchalis* and *Trinectes maculatus*) that were relatively abundant post-hurricane had not been seen in pre-hurricane samples of Black Creek. In the Pascagoula drainage, these two species are typically found in sandy or silt bottom portions of larger rivers. One might hypothesize that these species used portions of Black Creek as refugia (Sedell et al. 1990; Chapman et al. 1995) from low DO conditions in the Pascagoula River. Finally, it should be pointed out that our post-hurricane samples from the small streams and large rivers were taken in fall 2005, whereas the post-hurricane samples from Black Creek were not taken until spring 2006.

While there were no immediate impacts on community structure in upstream portions of Black Creek or the smaller streams, these are areas that might have sustained the greatest change in habitat structure. In fall 2005, debris dams altered streamflows at many small-stream sites, and shallow, riffle-type habitat was scarce. Progressing downstream, once channel widths increased beyond what a fallen tree could block, riffle type habitats returned. Community change induced by habitat changes may not be readily visible on the temporal scale at which we collected our data. Longer term sampling (multiple years) of these same sites is necessary to assess recovery of communities closer to the Gulf of Mexico as well as potential shifts in upstream communities due to habitat alteration.

Acknowledgements

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Corresponding Author:
Jake Schaefer
Department of Biological Sciences
University of Southern Mississippi
118 College Drive #5018
Hattiesburg, MS 39406
Phone 601–266–4928
Jake.schaefer@usm.edu

Session D: Stream Processes

Jake Schaefer, Moderator

Joann Mossa
Channel Planform Stability and Instability in the Pascagoula River and Tributaries, Mississippi

Gaurav Savant
Predicting Long–term Estuarine Sedimentation

David Welch
LMRFC Dambreak Operations

Channel Planform Stability and Instability in the Pascagoula River and Tributaries, Mississippi

Joann Mossa and David Coley

Because the Pascagoula River and its tributaries traverse a number of geologic units and the basin shows a diverse set of land cover and land uses, it is not surprising that different reaches in the basin show varying degrees of stability. To better understand this, planform changes were mapped for the Leaf, Chickasawhay, Pascagoula, and some of their major tributaries in a GIS using varied sources of data (aerial photographs, topographic maps and digital ortho quarter quadrangles). Unstable reaches have varied forms, including reaches with cutoffs, reaches with avulsions, reaches with direct anthropogenic channel alterations, reaches with appreciable lateral migration, and reaches that have altered in width. Using channel boundary data that was created in the GIS, this paper discusses the types and spatial patterns of varying degrees of stability/instability in the system. Polygon areas derived from overlays characterizing changes (areas of erosion, deposition, no change, and areas between channels) in channel position and planform were extracted from the GIS and normalized for scale, so that larger rivers could be compared with the headwaters, as well as smaller rivers and creeks. This study provides insights into the natural factors and anthropogenic activities that influence channel changes in this basin, and thus where future change might be expected to occur. Those insights, in turn, can be applied to management and engineering in the basin.

Keywords: Geomorphological and Geochemical Processes, Management and Planning, Methods

INTRODUCTION

Rivers are dynamic by nature, changing in response to variations in discharge and sediment supply. Such changes are quite variable in space, depending in part upon position within the basin, and influenced by local variations in geology, soils, bank characteristics, vegetation, hydraulics, and other factors that influence vulnerability such as various types of land use. Additionally, channel changes are variable in time, depending somewhat on the timing of floods and droughts, and the timing of fires and land use changes, and other factors that would influence vulnerability. Human activities, including land use changes, affect discharge and sediment supply indirectly, typically by increasing peak flows and increasing the quantities of sediment considerably (Wolman and Schick 1967, Gregory and Walling 1973, Dury 1977, Allan 1995). In addition, dams and floodplain mining are some activities that also have indirect effects on channel form and hydrology. Other human activities have a direct impact on river systems, particularly channelization and instream mining. Resulting physical changes in rivers may include deposition of channel bars, transportation of coarser sediments, erosion of channel banks, shifting channel bottoms, and changes in channel position and pattern.

Particular human activities, such as removal of floodplain vegetation, have a second influence, decreasing the resistance of banks to erosion thus making rivers more susceptible to channel changes. Various land use activities have been documented as having specific effects on river geomorphology, in addition to the effect associated just with vegetation removal. For instance, the presence of cows in riparian areas has been well documented as a notable geomorphic agent (Trimble and Mendel, 1995). Also, specific practices associated with floodplain and in-channel mining can promote various forms of instability. For instance, pits and ponds created by mining provide various routes for diversions (changes in channel position), especially during floods (Mossa and McLean, 1997). Removal of large woody debris may also induce channel changes (Keller and MacDonald, 1995). Construction activity has been well documented as increasing sediment supply up to several orders of magnitude (e.g., Wolman and Schick, 1967). As a consequence, rivers with particular types of land uses are particularly dynamic and unstable in comparison with unmodified rivers.

While becoming increasingly common, the ramifications associated with river instability are numerous. Although there has been some qualitative study of the association of floodplain alterations and instability (Bull, 1973, Graf, 1979, Kondolf, 1994, Mossa, 1995, etc.), only few studies have examined this problem from a detailed quantitative and spatial perspective (Mossa and McLean, 1997). Problems include: bank erosion and riparian property disputes associated with channel shifting, which sometimes leads to litigation; structural problems associated with undermining or filling at bridges and reservoirs; changes in channel capacity which affect flood patterns and increases the need for flood control; changes in floodplain habitat and effects to aquatic biota; and reductions in the quantity and diversity of fishes and mussels (e.g. Allan and Flecker, 1993; Brim Box and Mossa, 1999). Thus, it is important to riparian property owners, state and federal regulators, local communities and governments, industries, as well as other scientists and other individuals, to understand spatial and temporal variations of river channels, and how various factors contribute to instability and channel change.

RESEARCH OBJECTIVES

The stability of rivers and river reaches vary widely based on geologic and other natural factors. Based largely on curvature and the nature of bank sediments, some reaches are more prone to migration and cutoffs. The types and magnitude of anthropogenic impacts along rivers varies considerably. Scientists have had difficulty comparing the quantities of change, independent of scale. It is also difficult to classify the types of change in quantitative ways, and separate the background or natural rate(s) of change in a river system from change accelerated by direct and indirect human impacts. This paper discusses a technique that compares channel planform changes along and between rivers, independent of scale. The ratios characterize the types of change in quantitative ways, and can be used to separate the background or natural rate(s) of change in a river system from change accelerated by direct and indirect human impacts. The major rivers in the Pascagoula basin (Leaf, Chickasawhay, and Pascagoula) and two disturbed tributaries to the Leaf (Bowie River, Thompson Creek) are used as examples.

STUDY AREA DESCRIPTION

Natural Setting

The Pascagoula Basin is located in southeastern Mississippi and drains about 25,000 km2 (9700 mi2), of which the Leaf and Chickasawhay are the principal tributaries (Fig. 1). The Leaf occupies the northwestern portion of the basin and drains about 9280 km2 (3580 mi2), and the Chickasawhay occupies the northeastern portion of the basin with an area of about 7700 km2 (2970 mi2). The Pascagoula River drains southward into the Mississippi Sound, which is connected to the Gulf of Mexico. It drains all or parts of 21 counties in Mississippi. The state of Mississippi experiences abundant rainfall, with different locations in the basin averaging from 1300 to over 1700 mm (52 to 68 in) annually, yet on the Pascagoula, some years average four times the flow as other years (Lamonds and Boswell, 1985).

The topography of the Pascagoula basin is generally rolling to hilly with low to moderate relief, with the highest elevations in the northern part of the Chickasawhay basin exceeding 180m (600 ft). The basin has sediments of diverse lithologies that are Eocene and younger (Bicker, 1969) (Fig. 2). The mineralogy and lithology of these deposits in the central part of the basin is further characterized in Li and Meylan (1994) and Meylan and Li (1995).

Land Use/Land Cover, Changes and Disturbances

The total estimated surface area in the Pascagoula River watershed is 342,700 acres (U.S. Department of Agriculture-NRCS, 2005). Based on data extracted from the 1992 USGS Land Use/Land Cover classification, most of the land is in forest (67.8%) (Fig. 3). There are 542 farms within this watershed, and the average farm size is 94 acres. There are about 22,100 acres of cropland and 20,800 acres of pasture, each over 6% of the total area, and less than 3% of the basin is considered developed.

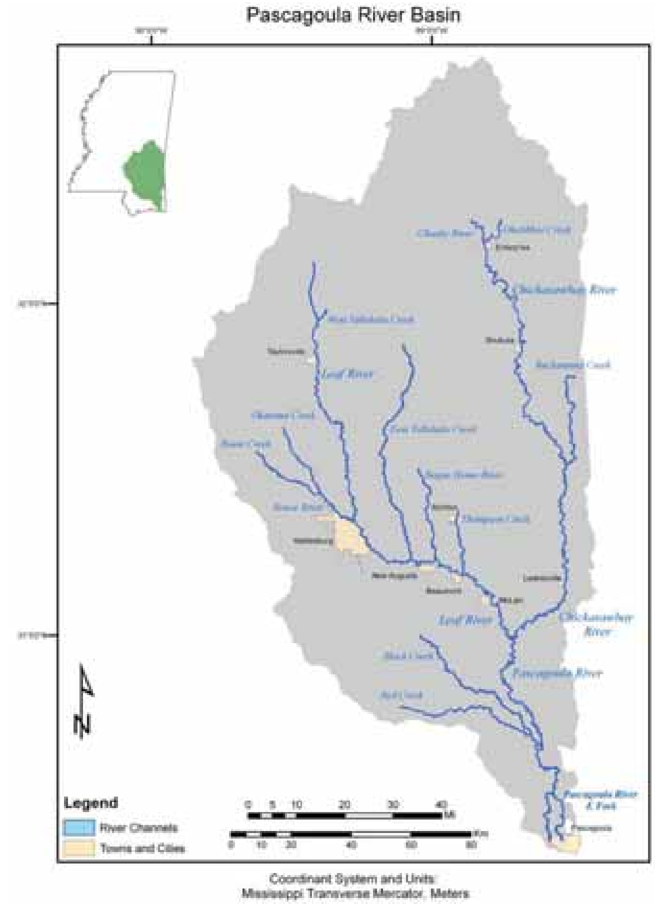


Figure 1. Map showing the Pascagoula River drainage basin.

Different disturbances in the basin have affected the patterns and timing of sediment supply. The European history in the basin begins in the 1830s. Although the area has remained predominately forested there is a strong history of agriculture and timber harvesting. The first complete census of agriculture is from 1920. The census of agricultural data for the counties in the study area show a peak in agriculture related landuse in the 1950s. In the decades since the amount of cropland has declined. Timber harvesting began in the area in the 1840's but was limited to the banks of streams until the Mississippi timber boom that began in the 1890's. From 1904 to 1915, Mississippi was ranked third nationally in timber production (Howe, 2001). Although the boom ended in the 1930s, and the production of lumber declined since then, it is still a major source of revenue for the region. While agricultural and timber activity has declined in the region in recent years, other forms of land use have increased their impact. In-channel mining occurred on the Bowie River in 1946 and was prohibited in 1995. Floodplain sand and gravel mining presently occurs along the Bowie River, Thompson Creek and Leaf River. Concurrently, development has been occurring in many parts of the basin, including the cities of Pascagoula, Moss Point, Meridian, Hattiesburg and Laurel.

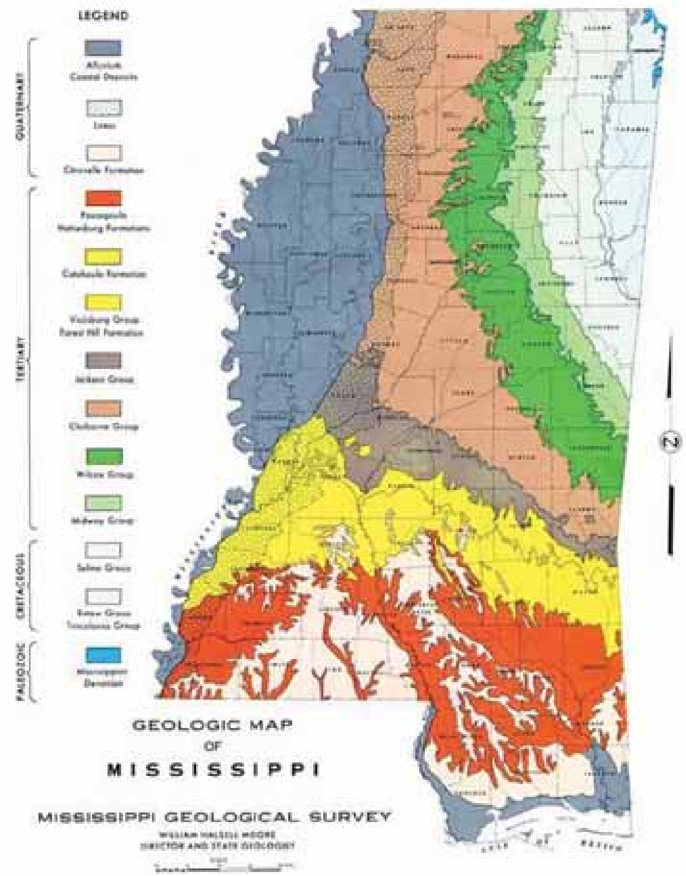


Figure 2. Map showing geologic formations in study area with county names and rivers, courtesy Mississippi Geological Survey. Source: www.ces.clemson.edu/semaps/jpeg/8dmsr/8d2.html

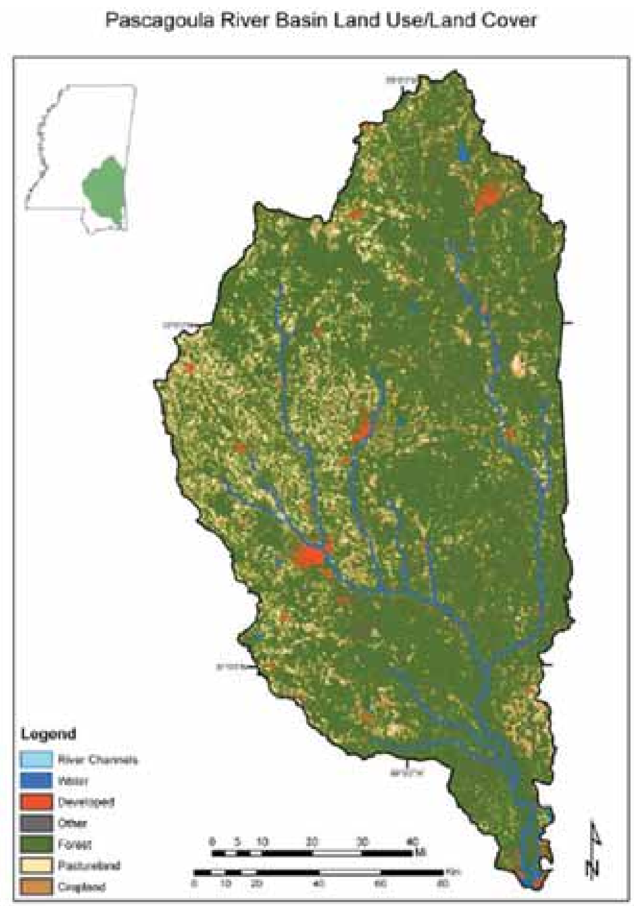


Figure 3. Land use/Land cover of the Pascagoula River Drainage basin.

METHODS

Mapping of Channel Boundaries for this Change Analysis

Over 2600 km (1640 mi) length of channel banks were digitized from the 1995-96 DOQQs. Given that only channels with both banks visible were digitized, this corresponds to over 1300 km (800 mi) river length and about 1000 km (600 mi) of valley length. Comparable lengths were digitized for the 1982-86 channels. Somewhat less channel length was digitized for the 1955-1960 period because of image quality and difficulties discerning the channel under vegetative coverage in smaller tributaries and channels. Map coverage for the 1947-1951 period was spotty and concentrated largely in the southern portion of the basin, thus the least channel length was digitized. Best available geospatial methods were used and stream channels and other all other data digitized were checked for quality assurance and spatial accuracy. Channel change maps were presented in Mississippi State Transverse Mercator (MSTM, a custom projection). Specific information on coverage accuracy, registration procedures, channel boundary mapping, minimum mapping units and attribute data are in Mossa and Coley (2004).

The magnitude of work involved is better understood when compared to other studies. For example, the only other study in the basin involving planform change (Turnipseed, 1993) assesses about 4 km (2.5 mi) valley length total. Published studies involving GIS and channel change are of comparatively short reaches, including Cluett and Radford (2003) who examined 5.5 km (3.4 mi) of the Lower Ord River in Australia, Gurnell (1997) who examined 18 km (11.2 mi) of the River Dee in England and Mossa and McLean (1997) who examined 50 km (31.1) of the Amite River in Louisiana.

Channel Change Analysis

By producing overlays of channel boundaries in the same projection for different time periods, the degree and type of instability in various reaches could be assessed. The polygons produced by overlaying channel boundaries from two time periods were labeled according to whether and how they have changed from the prior time period into the following categories: E (erosion), D (deposition), B (between), or U (unchanged) (Mossa and McLean, 1997) (Fig. 4; Table 1).

Normalizing by initial channel area or I (also D+U) allows comparison of different size channels, either along-stream or between streams. Thus, the proportional area change ratios (U-I, D-I, E-I, B-I) compare the type and amount of areal planform change across space for different periods (Mossa, 1999). U-I shows what proportion is unchanged or in its initial position. D-I shows what proportion of it has been deposited or abandoned, E-I shows what proportion of the initial channel area has been eroded, and B-I is a measure of displacement through cutoffs, rapid migration, or local avulsions into pits and secondary channels. Because rivers migrate naturally, some erosion and deposition is expected in an interval of a decade or longer, but high values of B (area between channels), other than occasional cutoffs in sinuous reaches, are usually indicative of instability. As a number of other planform geomorphic variables, the above measures are somewhat stage- or discharge dependent and should be interpreted accordingly. However, B is less stage-dependent than U, D or E.

Planform change variables and ratios (Table 1) were derived from reach blocks of 1km for small tributaries and 2 km for the major rivers in the system. This decision about reach block size was a compromise between reducing landscape noise and capturing local spatial variability. Different types of natural and disturbed reaches will change in different ways and their planform changes can be quantified using areal change ratios and visualized in Figures 5a and 5b.

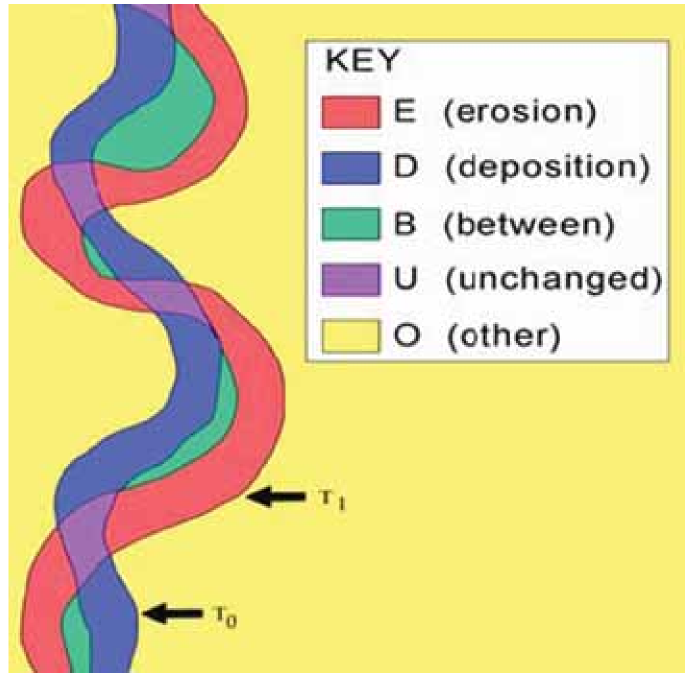


Figure 4. Channel boundaries for different time periods are used to make overlays in a GIS. Polygons are then labeled as E, D, B, or U according to whether and how they have changed from the prior time period.

RESULTS AND DISCUSSION

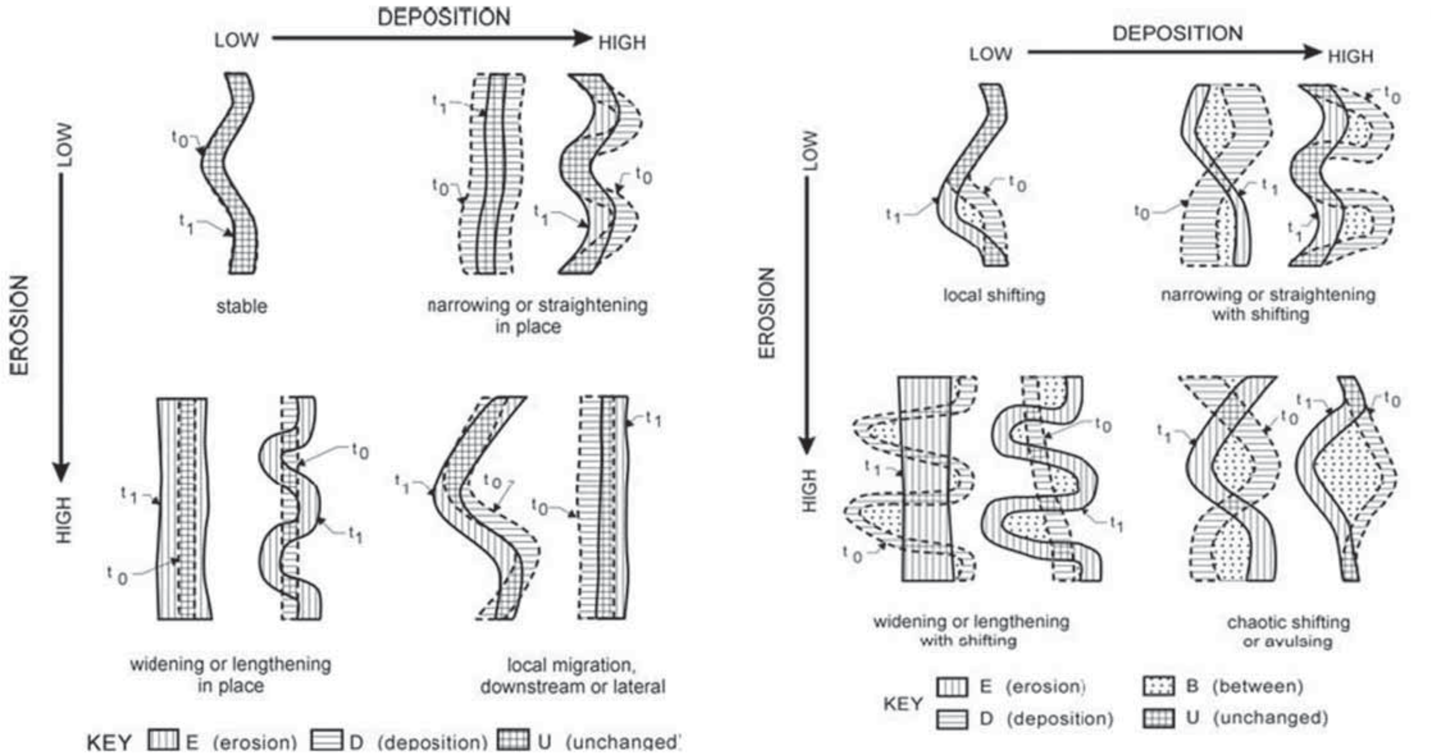
Major Rivers: Chickasawhay, Leaf and Pascagoula Rivers

For this paper, changes over an approximately 40-year period are compared for 5 rivers in the basin. Planform change indices using historical data on three major rivers, the Chickasawhay, the Leaf, and the Pascagoula, are evaluated. Two modified tributaries.

The Chickasawhay is for the most part a stable river. It crosses several different geologic units, some of which are far more cohesive than others. There are localized cutoffs, shown by B-I values up to 0.4, suggesting that the area between channels was as much as 40% of the initial channel area in places. In general, most of these cutoffs occur in areas where the bank sediments are noncohesive,

Table 1. Planform change variables used in GIS overlays and computations

AREAS
U: Portion or area of channel that is unchanged or remains in same position D: Portion or area of channel abandoned/deposited when comparing two periods E: Portion or area of channel created/eroded when comparing two periods B: Portion or area between channels when comparing two time periods, Reflects amount of displacement from cutoffs, rapid migration or avulsions I: Area of channel during the initial period in reach block, also equals D + U
TIME INTERVALS
$\Delta T_{ly-ey} = T_{ly}$ (Latter year of comparison) - T_{ey} (Earlier year of comparison): Number of years between selected planform change comparisons, can be expressed in decimals if dates of photographs are known
PROPORTIONAL AREA CHANGE RATIOS
U-I: U divided by I, shows proportion of initial channel area in same position D-I: D divided by I, shows proportion of initial channel area abandoned E-I: E divided by I, shows proportion of initial channel area eroded or created B-I: B divided by I, shows proportion of initial channel area between channels



Figures 5a and 5b. Types of planform changes can be interpreted from the absolute/relative amounts of erosional and depositional areas (a, left) or the area between channels and the absolute/relative amounts of erosional and depositional areas (b, right).

and the channel is sinuous. The straight reaches tend to be semi-consolidated mudstones and show less change or migration than sandier materials. Over the 40 year time period, about 60-90% of the channel remained in the same position (U-I values >0.6), except for a few local reaches with more rapid migration (Fig. 6).

The Leaf is less stable than the Chickasawhay (Fig. 7). The reaches with cutoffs are more numerous shown by the peaks in B-I along the channel. Over the 40 year time period, about 60-80% of the channel remained in the same position, except for a few local reaches with more rapid migration. One of these areas occurs upstream of the juncture with the Bowie River where a headcut is likely moving upstream. Prior study showed that the Leaf River shows reduced stage elevations, decreased mean and thalweg elevations, and increased planform instability at and upstream of its juncture with the Bowie River, a tributary with considerable in-channel mining (Mossa, 2003). Degradation and planform change on the Leaf was likely due to in-channel mining in this tributary, where a pit boundary on the channel bed expanded migrated into adjoining mainstem waters and then upstream.

The Pascagoula is the most stable of the three major rivers, especially in the lower portion (Fig. 8) The stability, indicated by U-I, mostly hovers around or above 0.8 and in the lowermost reaches exceeds 0.95, suggesting that over 80%, and in places 95% of the channel was in the same position in 1992 as in 1955. The increasing stability in the lower reaches is likely in part related to the bank sediments becoming finer, more cohesive and less erodible downstream. However, over the 40 year time period, cutoffs have occurred in a few very sinuous meanders. B-I values spike at these locations because the areas between the former (deposited) and new (eroded) channels are quite substantive at these sinuous cutoffs. Concomitantly, E-I and D-I increase as new channels are created, and U-I plummets to 0.2 to 0.4.

Disturbed Rivers: Bowie River and Thompson Creek

The Bowie River in southern Mississippi is a tributary to the Leaf River (Fig. 1) with a drainage area of 1732 km2 (Mississippi DEQ, 2005a). The lower 5km before it joins the Leaf River has been altered by floodplain sand and gravel mining, but perhaps more so by historical in-channel mining that began in the 1940s and ended in 1995 by environmental regulation. On topographic maps, portions of the channel are several times wider than other portions such that the channel resembles a chain of lakes. This form is due to mining along the channel perimeter in selected reaches plus some avulsions into floodplain pits. Sections between mined reaches,

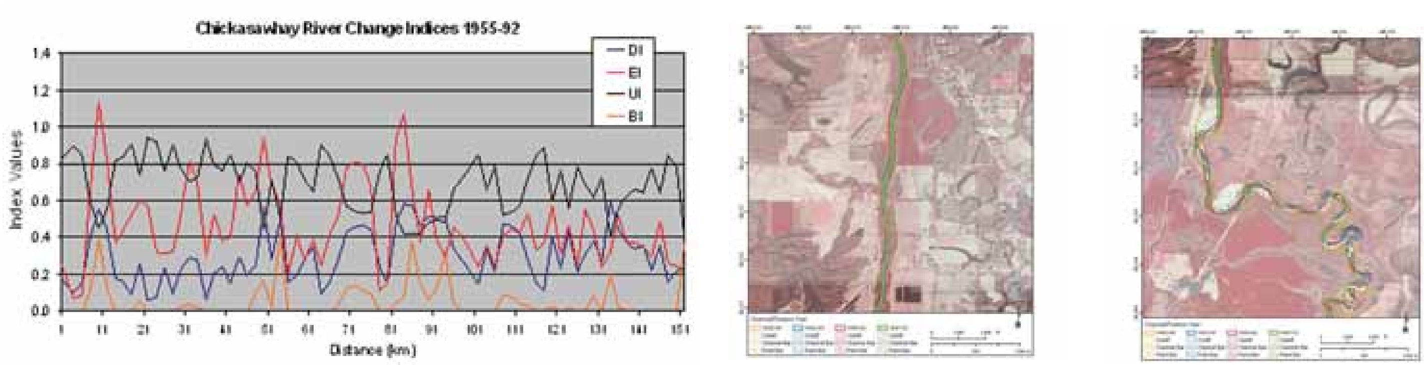


Figure 6. Planform change indices along the Chickasawhay River. Photos to the right show stability of straight reaches vs. greater migration in sinuous reaches.

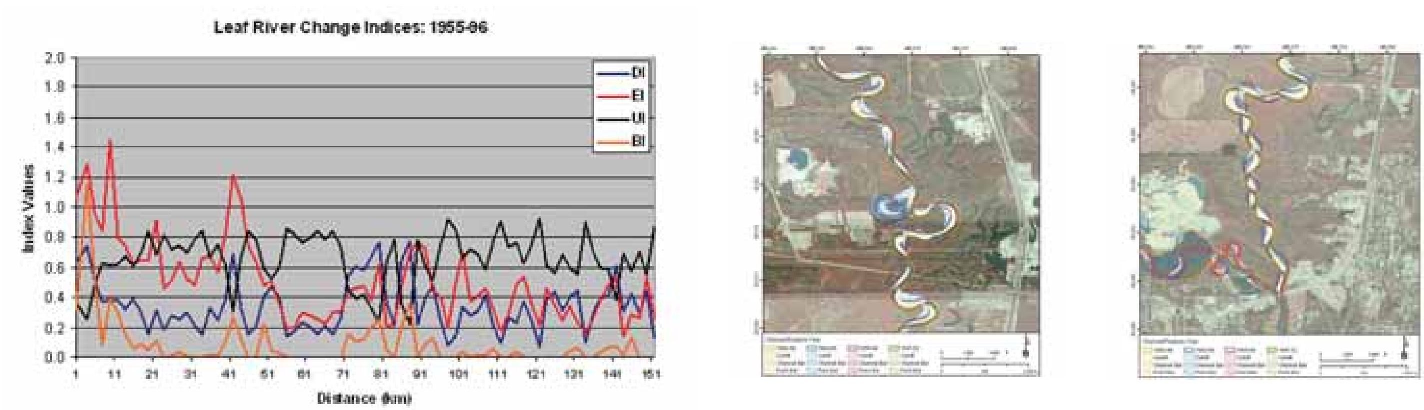


Figure 7. Planform change indices along the Leaf River. Photos to the right show recent instability just upstream of the juncture with the Bowie River.

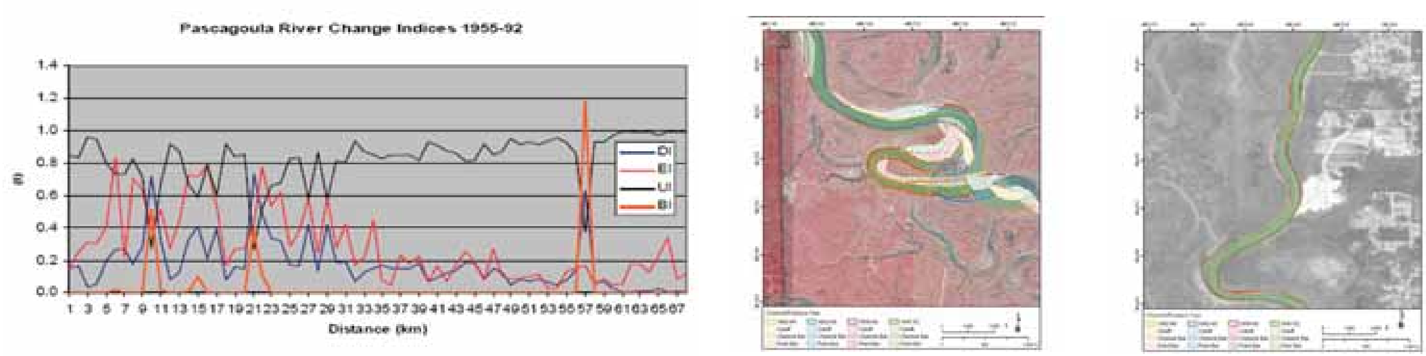


Figure 8. Planform change indices along the Pascagoula River. Photos below show a cutoff, a location where the B-I index would spike on the graph, and a very stable reach characteristic of much of the lowermost portion of the river where U-I values exceed 0.95 (95% of the channel is in the same position over the 40-year period).

and upstream of the mined stretch, tend to be fairly stable and in cohesive sediments. The planform change indices show enormous channel enlargement (high E-I values) due to instream mining of 4 to 5, about 400% to 500% increases from the initial channel area in locations (Fig. 9). There are also some avulsions into floodplain pits where the B-I values spike. Upstream reaches, and segments between the channel pits which resemble lakes, show much smaller variations. The rectangular box on the graph shows the range of variations in less modified reaches, which can be interpreted as the background values. Anything above this can be considered human impact, and clearly the mined areas show appreciable channel widening.

Thompson Creek is a relatively small tributary to the Leaf River (Fig. 1). Its drainage area is 493 km2 (Mississippi DEQ, 2005b). The upper portions of the creek have had localized cutoffs and some channel shifting through migration. The middle and lower portion of the creek has had historical and active floodplain mining, with large bare areas, sandy mounds and pits on the floodplain. In some cases, the channel has avulsed into these pits, shown by the higher B-I values in the planform change indices (Fig. 10). More change has occurred due to mining than these values suggest, because one photo (Fig. 10) shows that there was mining connected to the channel, giving it a large initial area, prior to 1955. For the most part, U-I values are 0.6 or lower, suggesting that only 60% or less of the channel was in the same location in 1996 as in 1955. Clearly, this creek is more disturbed than the larger rivers in this system, and the mined lower portion shows more change than the upper portion.

CONCLUSIONS

The major and minor tributaries in the Pascagoula system show a wide range in planform change indices or lateral stability-instability conditions, related to geology, anthropogenic alterations and other factors. Unstable reaches have varied forms, including reaches with cutoffs, reaches with avulsions, reaches with direct anthropogenic channel alterations such as channel widening due to instream mining, reaches with appreciable lateral migration, and reaches that have altered in width for other reasons. In some places, the cause of the instability is evident, and in other cases it is not.

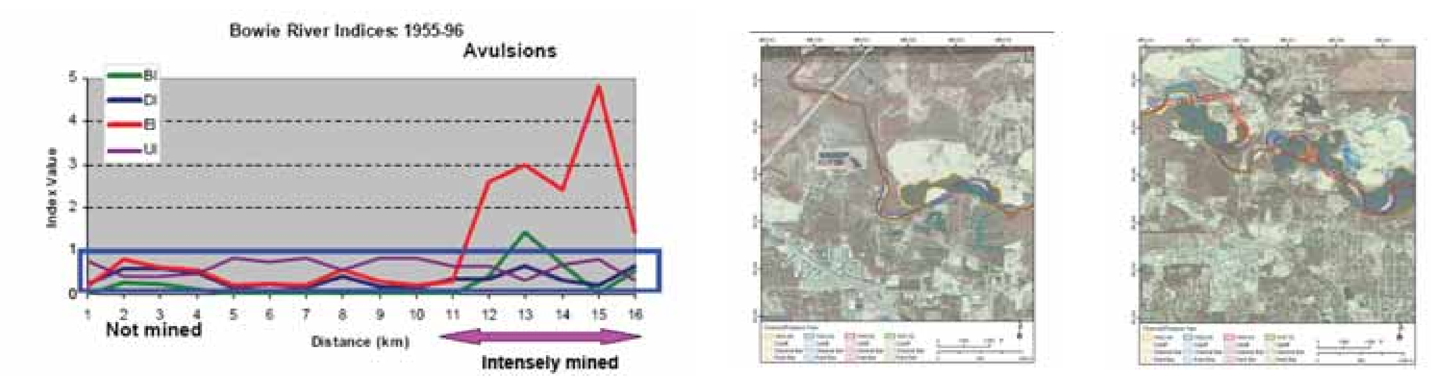


Figure 9. Planform change indices along the Bowie River. Photos to the right show enormous channel enlargement (high E-I values) due to instream mining, about 400% to 500% increases from the initial channel area in locations and some avulsions where the B-I values spike. The box shows the range of variations in less modified reaches, compared, and the intensively mined reach is well above these background values.

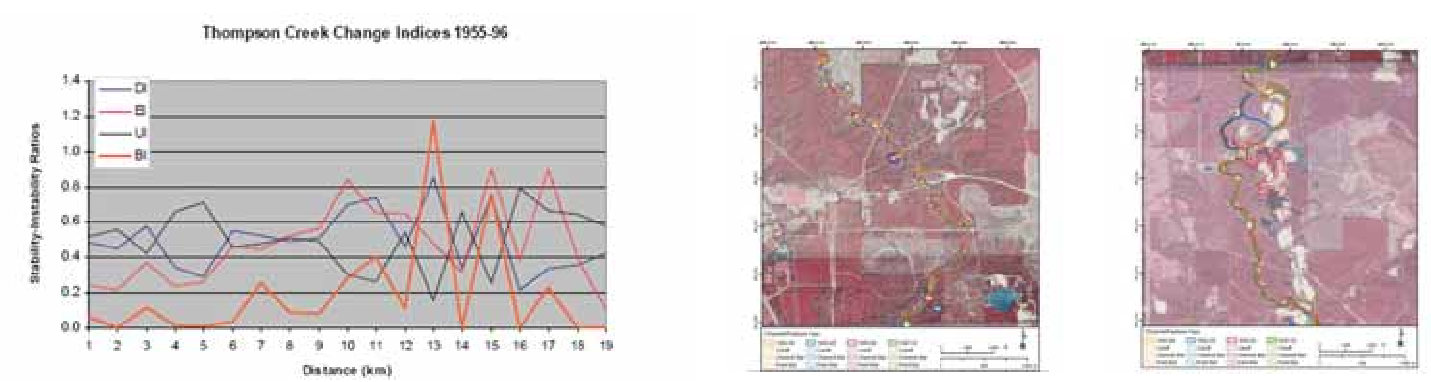


Figure 10. Planform change indices along Thompson Creek. Photos below show an area with that has changed, primarily due to natural factors, and an area with avulsions into floodplain pits associated with sand and gravel mining.

Of the major rivers in the Pascagoula basin, the Leaf is the least stable. Some of its tributaries (especially Bouie River and Thompson Creek) are even less stable as channel planform changes greatly exceed background rates. For instance, appreciable channel enlargement (E-I values of 4 to 5) due to instream mining occurs on the Bowie, indicating about 400% to 500% increases from the initial channel area in locations. Both the Bowie River and Thompson Creek showed higher than background B-I values in locations associated with avulsions into floodplain pits. Besides the rivers in this paper, several other major tributaries in the basin have been digitized and mapped. Individuals interested in historical changes at specific locations in the basin should consult Mossa and Coley (2004), a report with over 200 color maps on the Pat Harrison Waterway District website (http://www.phwd.net/district_water_quality.asp, Year 2 final report).

Many places that have been unstable historically are likely to go through additional adjustments in the future. Further changes are likely in the Bowie River and Thompson Creek, and in the Leaf River upstream of the Bowie juncture. Application of best management practices for different land use/land cover categories, including the timber industry, agricultural practices for cropland and pastureland, and practices to limit the impacts of urbanization and development on runoff and sediments, such as retention-detention ponds and construction fencing, should be adopted to minimize future impacts. Mining has impacted a few streams in the basin and more work should be done to document which types of mining activities create the most serious impacts and to develop best management practices for this industry.

The methods described and applied herein, involving planform change indices using GIS, are useful for comparing the quantities of change, independent of scale, for classifying the types of change in quantitative ways, and for separating the background or natural rate(s) of change in a river system from change accelerated by direct and indirect human impacts. The maps and the change indices, when applied wisely, can be used in planning, management, engineering, and stream restoration in the basin.

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Corresponding Author:
Joann Mossa, Associate Professor
Department of Geography, University of Florida
P.O. Box 117315, 3141 Turlington Hall
Gainesville, FL 32611–7315
Phone 352–392–4652, Fax 352–392–8855
E–mail: mossa@geog.ufl.edu

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Predicting Long Term Estuarine Sedimentation

Gaurav Savant and William H. McAnally

The purpose of this work is to develop an engineering method for predicting the morphologic behavior of estuaries. Where the river meets the sea, tides and density currents tend to trap sediments delivered by inflowing rivers, creating a natural sediment trap. For that reason estuaries can eventually fill with sediment and become tidal rivers. Human activities such as farming, mining, and dam building can accelerate or retard the infilling, as in San Francisco Bay and Atchafalaya Bay. Predicting estuarine infilling over long periods (decades to centuries) is made difficult by complex physical processes, highly variable sediment supply, and a lack of reliable long term data. Empirical orthogonal function (EOF) analysis, a technique for decomposing spatial and temporal data, was used to examine water depth trends in Suisun Bay, California. The analysis showed that EOF eigenfunctions demonstrate both depositional and erosive modes over a 40 year period and those modes can be correlated with freshwater and suspended sediment inflows with appropriate time lags. The findings suggest that EOF analysis can provide a method for predicting long term sedimentation, and experiments on larger and longer datasets are planned to develop the technique.

Keywords: Models, Sediments, Wetlands

Corresponding Author:
Gaurav Savant, Graduate Student
Department of Civil Engineering
Mississippi State University
Box 9546
Mississippi State University
Mississippi State, MS 39762
Phone 662–325–8081
Fax 662–325–7692

LMRFC Dambreak Operations

David Welch, John Lhotak, and Dave Reed

The National Weather Service (NWS) has the responsibility to issue flood warnings regardless of a flood's origin; including, those resulting from dam failures. The Lower Mississippi River Forecast Center (LMRFC) provides hydrologic forecasts and support to assist NWS Weather Forecast Offices in the issuance of flood warnings. When a dam failure occurs or has the potential to occur, LMRFC hydrologists use various forecast tools to develop dam break forecasts in providing this support.

A basic component of the LMRFC dambreak operations is the NWS Dam Catalog (Damcat) database derived from the US Army Corps of Engineers, National Inventory of Dams (NID) database. In addition to the NID information, the Damcat database contains estimates of breach width, breach height, time to breach, forecast flood elevation and discharge at the dam, and the nearest downstream point of interest. Estimates of these parameters were developed from numerous historical dam breaks and provide general guidance that a NWS forecaster can quickly find and include in flood warnings.

As time permits the LMRFC runs the NWS Simplified Dambreak model (SMPDK) to determine water surface elevations downstream of a dam failure. Inputs include estimated Damcat breach parameters, NID storage, and cross-sections derived from a commercial off-the-shelf mapping database. A Visual Basic GUI is available to refine the estimated breach parameters as needed.

To ensure the quality control of all dam break forecasts, the LMRFC has developed "rules of thumb" based on numerous historical dam failures and hydraulic principles. These rules of thumb can also be used to provide general guidance so flood warnings can be released to the public in a timely manner to reduce the loss of property or lives.

Keywords: Floods, Hydrology, Methods, Models, Surface Water

Corresponding Author:
David Welch, Senior Hydrologist
National Weather Service – Lower Mississippi River Forecast Center
62300 Airport Road
Slidell, LA 70460
Phone 985-641-4343
Fax 985-643-1226
E-mail: david.welch@noaa.gov

Session E: Groundwater Assessment & Management

Jamie Crawford, Moderator

Charlotte Bryant–Bryd
Irrigation in the Mississippi Delta: A Historic Perspective

Antonia Cerdeira
Effect of sugarcane coverage on the behavior of Tebuthiuron in soil in Brazil

Steve Jennings
Groundwater Resources of the Jackson Metropolitan Area

Charles Wax
A Climate-Based Management Plan to Conserve Groundwater and Reduce Overflow in Aquacultural Ponds in the Southeastern U.S.

Heather Welch
Water Quality in the Mississippi Embayment–Texas Coastal Uplands Principal Aquifer

Irrigation in the Mississippi Delta: A Historic Perspective

Charlotte Bryant–Byrd

In order to gain some insight into the present socio–economics of the Mississippi Delta, it is important to understand the past history of the region including its historical geology. The hydrogeologic setting of the Delta was influenced by its juxtaposition along the axis of the Mississippi Embayment, indirectly by the glacial episodes of the Pleistocene, but most significantly by the depositional and erosional sequences associated with meandering streams across the region during Recent times.

Vast stretches of the Delta were covered in thick vegetation during much of its formative years. While large tracts were being cleared, the economy of the region was based predominantly on the cutting of virgin timber and the sawing and transportation of large volumes of lumber. As more cleared acreage became available during the 1800's, the economy began a transition to that of more traditional agriculture with "King Cotton" reigning as the dominant commodity.

The discovery of a prolific shallow aquifer underlying the Mississippi Delta was realized in the early 1900s. The water–bearing beds in the Mississippi River alluvium form the Mississippi River valley alluvial aquifer, the most extensive aquifer in the Lower Mississippi region. Pumping from the alluvium for irrigation began to gain momentum following a severe drought in the early 1950s. However, the number of alluvial wells in the Delta increased dramatically during the 1970s and 1980s with the introduction of pond–raised channel catfish to the region. The economy of the Delta today is dependent upon the alluvial aquifer as the source of water for over 14,000 irrigation and catfish wells.

Corresponding Author:
Charlotte Bryant–Byrd
Mississippi Department of Environmental Quality
Office of Land and Water Resources

Effect of Sugarcane Coverage on the Behavior of Tebuthiuron in Soil in Brazil

Antonio L. Cerdeira[1], Manoel D. DeSouza[1], Vera L. Ferracini[1], Sonia C. N. Queiroz[1], Denizart Bolonhezi[2], Marco A. F. Gomez[1], Regina Helena Costa Queiroz[4], Otavio Balderrama[3], Paulo Rampazzo[3]

The region of Ribeirao Preto City located in Sao Paulo State, southeastern Brazil, is an important sugarcane, soybean and corn producing area. This region is also an important recharge area for groundwater of the Guarany aquifer, a water supply source of the city and region. The herbicide tebuthiuron (N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2 yl]-N,N'-dimethylurea) is regularly applied in the area. In order to understand the movement of tebuthiuron, the herbicide was applied at the recommended label rate with and without sugarcane coverage, on sandy soil area in Santa Rita do Passa Quatro County in Brazil, located in the region. Soil samples were collected at each 20 cm down to 120 cm and taken to the laboratory for determination of tebuthiuron. Tebuthiuron was measured at those depths mentioned before in ten intervals of time up to 300 days. Tebuthiuron half-lives varied from 69 days in sugarcane cropped area to 49 days in non-cropped area. After 180 days there were no measurable residues in the soil and tebuthiuron was not found below 40 cm depth in any time.

Key Words: Agriculture, Ground Water, Nonpoint Source Pollution, Solute Transport, Water Quality

INTRODUCTION

The region of Ribeirao Preto City (Figure 1), located in Sao Paulo State, southeastern Brazil, is an important sugarcane, soybean and corn producing area. This region is also an important recharge area for groundwater of the Guarany aquifer, a water supply source of the city and region. It has an intercontinental extension that comprises areas of eight Brazilian states, as well as significant portions of other South American countries like Argentina, Uruguay, and Paraguay, with a total area of approximately 1,200,000 Km^2.

Intensive cultivation in this area has required the constant use of pre-emergent herbicides and fertilizers. The risk of groundwater contamination by those chemicals, which are normally reapplied annually, has been a major concern. Due to the high permeability of some soils present in this region, the mobility of the herbicides and fertilizers applied, and being a recharge area, it is important to investigate the potential transport of applied herbicides to underlying aquifer. The herbicide tebuthiuron is regularly applied in the area. Tebuthiuron (N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2 yl]-N,N'-dimethylurea) is a phenylurea herbicide used in sugarcane culture for pre emergence control of weeds.

Although it was generally accepted that pesticides would not leach to groundwater, recent studies indicate agrochemical leaching as an important source of agricultural non-point-source pollution (Smith et al. 2001), particularly in the last decade (Bouwer 1990). Other studies have indicated that some American aquifers were contaminated with both inorganic and organic compounds, some of which were pesticides (Williams et al. 1988).

Figure 1. Map of South America showing the city of Ribeirao Preto, Brazil, where the recharge area is located.

MATERIALS AND METHODS

In order to understand the movement of tebuthiuron in the region it was chosen a sandy soil area located in the city of Santa Rita do Passa Quatro, Sao Paulo state, Brazil, with and without sugarcane cover. Soil samples were collected and taken to the laboratory. Soil density (Mg/m3), the relationship of soil mass to volume was measured using the Kopeck ring method described by Black (1965). Total porosity was measured based on the percentage of saturation in volume (Vomocil 1965). Microporosity was determined by the tension table method at a potential of 0.006 Mega Pascal (Mpa). After saturation and drying under tension, the samples were oven dried at 105 °C to obtain the volume of micropores ≤ 0.05 mm. Macroporosity was obtained by the difference between micro and total porosity. Also evaluated were the % organic matter and physical properties of the soils for each depth (Klute 1986). Soil samples were collected in trenches at 10 cm depths and their properties are shown in Table 1.

RESULTS AND DISCUSSION

The experiments conducted in Santa Rita area with sand soil have shown that tebuthiuron half-lives varied from 69 days in sugarcane cover area to 49 days where there was no cover crop. After 240 days there was no measurable residue in the soil and it was not found below 40 cm. depths, any time. (Tables 2 and 3, and Figures 2 and 3).

CONCLUSION

The half-life of tebuthiuron was much lower than expected. Literature indicates a period from 12 to 15 months (Weed Science Society of America) and we found a maximum of 69 days. There was an apparent effect of sugarcane coverage on the degradation, being quicker where there was no cover. The herbicide did not move deeper than 40 cm. depth anytime.

Table 1. Soil micro, macro, total porosity, density and moisture at various depths.

Depths (cm)	Macro (%)	Micro (%)	Total Porosity (%)	Density (kg/ dm3)	Moisture 0.1 bar (%)
0-10	11,88	30,72	42,59	1,51	8,6
10-20	11,30	30,23	41,53	1,55	8,4
20-30	8,68	30,64	39,32	1,64	7,6
30-40	11,32	30,31	41,63	1,56	8,8
40-50	10,40	31,07	41,46	1,57	9,1
50-60	10,60	31,50	42,10	1,54	10,8
60-70	12,65	29,19	41,84	1,54	10,4
70-80	13,13	29,14	42,27	1,52	9,7
80-90	14,90	27,57	42,47	1,49	9,4
90-100	15,38	27,55	42,93	1,44	9,2
100-110	16,20	25,93	42,13	1,46	9,2
110-120	14,11	28,76	42,87	1,42	9,8

Table 2. Amount (mg/kg) of tebuthiuron found at various depths and time with sugarcane cover.

Depth (cm)	Control	0[1]	3	30	60	90	120	150	180	240
0-20	ND[2]	0.33	0.115	0.12	0.080	0.075	0.08	0.025	0.035	ND
20-40	ND	ND	ND	0.02	0.02	ND	0.03	ND	0.02	ND
40-60	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
60-80	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
80-100	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
100-120	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

[1]DAA=Days after application. [2]ND= No detection.

Table 3. Amount (mg/kg) of tebuthiuron found at various depths and time with no sugarcane cover.

Depth (cm)	Control	0[1]	3	30	60	90	120	150	180	240
0-20	ND[2]	0.41	0.39	0.08	0.12	0.07	0.06	0.025	0.025	ND
20-40	ND	0.15	0.02	ND	ND	ND	ND	ND	ND	ND
40-60	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
60-80	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
80-100	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
100-120	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

[1]DAA=Days after application. [2]ND= No detection.

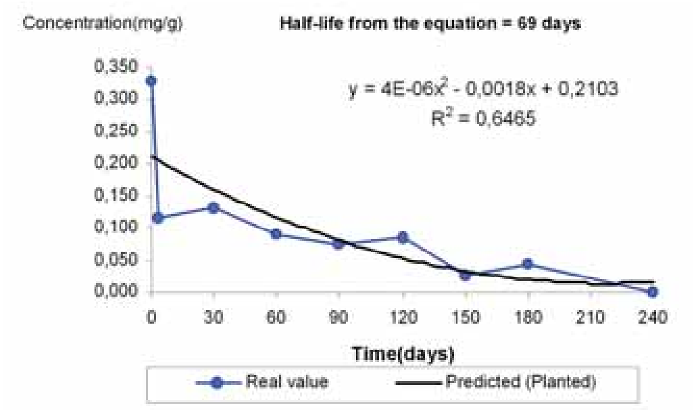

Figure 2. Tebuthiuron dissipation in sandy soil with sugarcane cover.

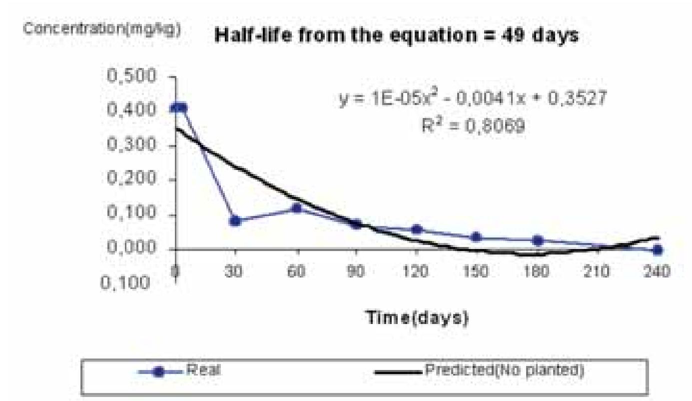

Figure 3. Tebuthiuron dissipation in sandy soil without sugarcane cover.

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Corresponding Authors:
[1]Embrapa, Meio–Ambiente, Research Division of the Brazilian Ministry of Agriculture, C.P. 69, Jaguariuna, SP, 13820–000, Brazil.
[2]Agronomic Institute of Campinas, APTA–Centro Leste, C.P. 271, Ribeirao Preto, SP, 14001–970, Brazil.
[3]Dow AgroSciences, Mogi Mirim Regulatory Laboratory, Mogi Mirim, SP, Brazil, 13800–970
[4]Pharmacy School of Sao Paulo
University, USP, Ribeirao Preto, SP, 14040–903, Brazil.

Groundwater Resources of the Jackson Metropolitan Area

Steve Jennings

About 98 percent of the groundwater used in the tri–county (Hinds, Madison, and Rankin) Jackson metropolitan area is pumped from three major aquifers – the Cockfield, Sparta, and Meridian–upper Wilcox. Sand intervals within each of these major hydrostratigraphic units, though not strictly contiguous across the area, are present in significant thicknesses and with sufficient storage and transmissive properties to provide fresh water to 101 cities, water associations, industries, and other users within the Metro area. As an aid to all stakeholders, an on–going effort to better delineate the extent of the aquifer sand intervals through subsurface geologic mapping is being conducted by the Office of Land and Water Resources (OLWR) of the Mississippi Department of Environmental Quality. Structural mapping of the tops of pertinent geologic units and construction of several regional geologic cross sections have been completed as a first major step in this effort. Mapping of the total sand thickness for the Meridian–upper Wilcox aquifer has been completed, and mapping of the Cockfield and Sparta sands is proceeding.

Concerns about water level declines in these aquifers prompted OLWR in 1997 to more closely monitor the water levels and publish summaries of the data. To this end a monitor well network of 83 wells has been used to periodically measure water levels and plot trends in the declines. The addition of the more recent water–level data with historical data indicate water level declines in the three main aquifers at average rates of approximately 1.5 feet per year in the Cockfield aquifer, 2.5 feet per year in the Sparta, and 2.6 feet per year in the Meridian–upper Wilcox. While of concern to all, these declines in water levels represent aquifer pressure declines and not actual dewatering. At the current rates of decline, the aquifers are capable of providing ample water supplies for many years.

Linking the hydrologic data such as water levels and water quality to the geologic parameters such as depths and sand thicknesses is an anticipated step in the overall study of the groundwater resources of the area. Successful completion of this phase of the project along with the data and analyses outlined above should provide water resource planners and users valuable tools to utilize and manage this vital natural resource.

Corresponding Author:
Steve Jennings
Mississippi Department of Environmental Quality
Office of Land and Water Resources

A Climate-Based Management Plan to Conserve Groundwater and Reduce Overflow in Aquacultural Ponds in the Southeastern U.S.

Charles L. Wax, Jonathan W. Pote, Thomas P. Cathcart

The potential for conservation of groundwater used in warm water aquaculture in the southern region is evaluated. Daily water balances for ponds are constructed by computing precipitation minus 0.8 * pan evaporation over a 40–year period (1961–2000) at five sites in the region: Fairhope, AL; Clemson, SC; Thompsons, TX; Stuttgart, AR; and Stoneville, MS.

Through computer simulation, daily water level fluctuations are controlled under two management schemes over the 40–year period: 1) pond surfaces are kept constantly at gage level by daily addition of sufficient groundwater to "make–up" evaporative losses not countered by precipitation, with excess precipitation lost to overflow; and 2) pond surfaces are allowed to drop six inches below gage level by cumulative evaporative loss not countered by precipitation, with addition of just three inches of groundwater at that point, leaving three inches of storage capacity for any subsequent excess precipitation. Amounts of groundwater used under the two management schemes are calculated and compared.

Under the "make–up" water use method, amounts used range from about 34–44 inches across the region, and average about 38 inches annually. Under the alternative "6/3" method of management, amounts used range from about 4–15 inches regionally, and average about 11 inches annually. Implied conservation potential from adoption of the "6/3" scheme ranges from 64%–88%, and averages 72% for the region.

Keywords: Climatological processes, groundwater, models

Introduction

Fish is considered a healthy food choice in developed regions and may provide the bulk of total protein intake in developing regions. World production and consumption of fish is geographically variable, but in 2004 over 30% of fish food consumed worldwide was provided by aquaculture (NMFS, 2004). In the U.S., 9% of seafood consumed is supplied by domestic aquacultural production of species such as tilapia, trout, striped bass, salmon, crawfish, oysters, and catfish. The U.S. aquaculture industry has expanded rapidly and has experienced substantial growth in the last ten years. Nationwide, aquacultural production increased from about 18 million tons in 1990 to over 32 million tons in 2004 (NASS, 2005).

Catfish is the predominant species produced, accounting for 46% of total U.S. aquacultural production, with U.S. catfish consumption (1.2 lbs/capita) ranking fifth behind shrimp, tuna, salmon, and pollock (NMFS, 2004). Four states in the southern region (AL, AR, LA, and MS) accounted for the overwhelming majority of this total national production (NASS, 2005). These four states accounted for 95% of the U.S. total of both surface water in aquaculture production in 2005 (170 million acres) and in value of sales in 2004 (over $480 million) (NASS, 2005).

The southern region has risen to prominence in the aquaculture industry largely because of its climatic characteristics. The humid subtropical climate of the region has temperate winters, evenly distributed rainfall through the year, and long, hot summers. With most places in the region receiving between 45 inches to 65 inches of precipitation annually, and with groundwater being relatively abundant and accessible across the region, water availability for aquaculture has been taken for granted. Recently, increased demand for groundwater, aggravated by a series of drought years, has prompted growing concern over future availability and quality of this vital natural resource, and the need for conservation of groundwater especially has become clear.

Agriculture is the major water consumer in the region, and aquaculture has the potential to become disproportionately consumptive. For example, most row crops in the region require 12-16 in/yr, whereas catfish farming requires up to 40 in/yr under current practices. In the "Delta" region of Mississippi where nearly 60% of U.S. farm raised catfish are produced, catfish production accounts for about 28% of all water used (Pennington, 2005). The production value of water used in aquaculture exceeds that of other food production methods. Boyd and Gross (2000) calculate the production value of water used for soybean production as $13/acre-in, compared to a value of $80/acre-in for catfish. As a valuable and fast-growing industry that requires large amounts of clean water, the future growth of warm water aquaculture requires that conservation methods for production be developed.

Another consideration is that processors of farm raised catfish prefer larger fish. Harvested fish increased in size from an average of 1.3-1.5 lb in the early 1990s to an average of 1.5-3 lb by the end of that decade (McGee and Lazur, 1998). Production of the larger fish requires a longer production cycle (almost five months more), increases production risks (8% increase in death loss), and increases production costs (McGee and Lazur, 1998). Additionally, prices for the fish are not increasing while prices of feed are rising. Only the most efficient producers will continue to make profits and stay in business. Any strategy that enhances conservation of groundwater and reduces production costs in this industry should be adopted.

A previous study (Pote, et al, 1988) showed that rainwater harvesting and storage was a promising groundwater conservation strategy in the aquaculture-intensive Delta region in Mississippi. The Extension Services in Alabama and Louisiana include variations of the "drop/add" strategy proposed by Pote, et al (1988) as industry best management practices for reducing effluent release in those states (Auburn University, 2002; LCES, 2003), but such recommendations remain largely unsubstantiated by research on a regional level. The best management scheme for this region would be one that would minimize pond overflow, capture rainfall and store it to offset evaporation losses.

This study evaluates the results of the techniques established by Pote, et al, (1988) for other locations in the southern region. The other locations were chosen to assess the effectiveness of the scheme throughout the region by taking into account the east-west precipitation gradient and the south-north change from maritime to continental characteristics. These spatial disparities produce minor climatic variations within the region that are potentially significant in their impacts on management strategies implemented to conserve water. This investigation develops climatological water budgets for ponds to evaluate these slight climatic differences.

Background Information

Commercial warm water aquaculture in the southern region is practiced in three types of ponds–watershed ponds, excavated ponds, and levee ponds. Levee ponds are most commonly used because their rectangular shape is convenient for management operations, and water levels can be controlled better than in other types of ponds (Boyd and Gross, 2000). These ponds are usually constructed on heavy clay soils with excellent water-holding capacity. A typical pond has about 17 acres of water on 20 acres of land, averaging about four feet in depth. Ponds are built with drains to control and adjust water level.

Although precipitation falling into ponds has the potential to supply up to 65" per year in some parts of the region, in practice nearly all this water is lost to overflow due to the management practice of holding pond levels at or near maximum. Levee ponds have virtually no watersheds, so sources of water other than surface runoff must be available. Because of water quality advantages, groundwater is presently by far the source of choice (EPA, 2004). Usually wells yielding 2000-2800 gal/min are adequate for four ponds of about seven water ha each (Wellborn, 1987). Theoretically the ponds can be operated indefinitely without draining. In practice, however, ponds are drained every two to 10 years for repair of levees.

Evaporation is a major and unavoidable water loss in the production process, because an open pond surface is necessary for gas exchanges as well as cultural activities such as feeding, aerating, and harvesting. Infiltration losses are minimal because leaking ponds are repaired or taken out of production in favor of better sites. Overflow losses are determined by both precipitation events and management practices.

In light of the above considerations, a realistic water budget for ponds can ignore surface water inflow and infiltration losses, but must account for precipitation and evaporation. Hydrologic modeling using historical meteorological records represents a method to assess rainwater storage in aquaculture ponds. In addition to the work by Pote, et al (1988), this approach has been used to determine effluent release from ponds in Mississippi (SRAC, 1998; Tucker, et al, 1996)) and as a tool for estimating pond requirements at individual facilities located in different geographical regions (Bolte and Nath, 1999).

Methods and Procedures

Site selection

Five sites were chosen for the analyses. Selection was based on one or more of the following three criteria:

1) sites were located in states that have significant aquacultural production;
2) sites had serially complete and homogeneous daily precipitation and evaporation records; and
3) sites were spatially dispersed to provide representation of the moister eastern and drier western portions of the region as well as the coastal (maritime) and interior (continental) characteristics of the regional climate.

Based on the best possible combination of these criteria, the following sites were selected: Thompsons, TX; Stuttgart, AR; Stoneville, MS; Fairhope, AL; and Clemson, SC. Figure 1 shows the location of each of these sites and demonstrates the P-E gradients that exist within this humid subtropical climate region.

Climatological Data

Daily observations of precipitation and evaporation for the period 1961 to 2000 were obtained from the National Weather Service Cooperative Observation System. The data were loaded into Excel spreadsheets for quality control and simulation analyses.

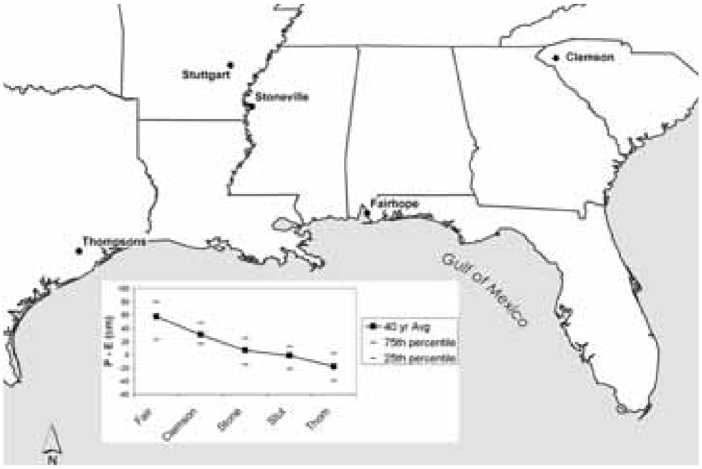

Figure 1. Location map illustrating regional P-E gradient.

While the precipitation data were essentially useful straight from the archived records, the evaporation data were much less complete and reliable (Irmak and Haman, 2003). In order to produce a complete daily evaporation record for each location, the existing daily observations were used to compute an average for each day. That average was then substituted where daily values were missing. Other errors appeared to be either typographical mistakes or excessively large values following long strings of missing observations, which were adjusted accordingly. If the observation in question appeared obviously wrong but no cause was readily evident, data from the next nearest location or the average value for that day was substituted. The result of this procedure, followed for all 40 years at all five locations, was a reasonably accurate and complete record of daily evaporation that could be used to quantify evaporative losses of water in the southern region.

Pan evaporation is not directly comparable to true evaporative loss from the surface of a lake or pond because of different heating characteristics and differing degrees of exposure to wind and sun of evaporation pans as compared to large bodies of water. Therefore, a "pan coefficient" is generally used to correct measured pan evaporation to a more realistic estimate of actual evaporative loss from ponds or lakes.

In early studies (Schwab, et al, 1955) a constant coefficient of about 0.7 was used to convert class A pan evaporation to an estimate of evaporation from large reservoirs. However, in more recent years it has been determined that the relationships vary from month-to-month and possibly from location-to-location (Kohler, et al, 1959; Ficke, 1972; Hounam, 1973; Yonts, et al, 1973; Linacre, 1994).

In a study conducted on a small, shallow pond near Auburn, AL, pan-to-pond coefficients were found to range from 0.72 in March

Table 1. Summary of average daily pond evaporation rates during summer, southern region (range, in inches).

Site Location	Evaporation Rate Range (in inches)
Fairhope, AL	0.18 - 0.20
Clemson, SC	0.17 – 0.19
Stoneville, MS	0.20 – 0.24
Stuttgart, AR	0.20 – 0.24
Thompsons, TX	0.21 – 0.27

to 0.90 in September, with an average of 0.81 for all months (Boyd, 1985; Boyd and Gross, 2000). This value is also consistent with work by Linacre (1994). Because the environment in which these coefficients were determined is closely related to the environments of aquacultural production ponds in the southeastern U.S., a correction coefficient of 0.8 was used in this study to correct the pan evaporation data to estimates of evaporative losses from ponds. Table 1 shows, for spatial comparison, the range of pond evaporation values thus determined from pan evaporation records for the five sites in this study.

Pond Simulations

The water balance described above can be written:

$$\Delta V = P*A + Q_{in} - Ev*A - Q_{out}$$

where ΔV is pond volume change, P is precipitation, A is pond surface area, Q_{in} is volume of groundwater pumped, Ev is evaporation (0.8*class A pan evaporation), and Q_{out} is volume of pond effluent released. This balance can be rewritten:

$$\Delta E = P + Q_{in}/A - Ev - Q_{out}/A$$

with ΔE (cm), the change in pond elevation, replacing volume as the dependent variable and all terms converted for consistency of units.

Summation of the daily amounts of water added to ponds by precipitation and lost from ponds through evaporation provides an accounting of water level fluctuations resulting from climatic processes. Pumping and effluent release are intermittent events based largely on pond management decisions. Such a water balance approach is used here to assess the impact of climatic variability (seasonally, annually, and spatially) with the objective of determining the effectiveness of a management plan that takes advantage of regional climatic characteristics.

Simulations for an Infinitely Deep Pond

These simulations assumed $\Delta E = P - Ev$ (i.e., no pumping or effluent release) and were conducted to illustrate temporal availability of the precipitation resource for use in a "drop/add" rainwater storage scheme. A daily comparison of precipitation (P) and pond evaporation (0.8E) was conducted at each of the five locations for the 40-year period Jan. 1, 1961 - Dec. 31, 2000. Cumulative summation of these daily values provided seasonal and annual variations in precipitation availability.

For perspective, the daily elevations for the average year, the wettest year, and the driest year were singled out for each location. These years were not necessarily the same at each site. The average year was computed as the mean of the 40 P-0.8E values for each day through the year. The extreme years were analyzed using actual, not averaged, daily P-0.8E values for the years with the largest and smallest total precipitation, respectively. The daily water level patterns during these selected years at each site were then graphed by cumulative addition of daily P-0.8E values through each of the years.

This analysis shows the pond level regimes that might be expected from a purely climatological standpoint; that is, if the daily interaction between precipitation and evaporation were the sole factors in determining pond water levels, and if no water was added from any source other than precipitation, no overflow occurred, and no loss other than evaporation was encountered. Comparison of results at the five sites documents the regional variation in potential for precipitation to exceed pond evaporation on a cumulative, daily basis, and therefore indicates the comparative advantage of different areas within the region to use precipitation in a management scheme to keep ponds filled and thereby conserve groundwater.

Pond Water Management Simulations

Two approaches to pond water management were simulated. The first, in use in much of the pond aquaculture industry (Boyd and Gross, 2000), maintains ponds at the full mark (an arbitrary level "0" in the simulations) through addition of groundwater to replace water lost via evaporation. All precipitation is therefore lost as effluent release. This simulation represents a worst-case scenario and provides a basis for comparison with the second management approach, described below.

The second method is the "drop/add" scheme proposed by Pote, et al (1988), a management option in which the water level of the pond is allowed to drop 6" from the starting point before addition of any groundwater occurs. When groundwater is added, the amount is only enough to raise the level of the pond 3", leaving a 3" capacity to store any precipitation which might subsequently occur. Groundwater pumped is represented by the elevation increase (Q_{in}/A) required to fill the pond to –3" (recall that "0" is the arbitrary outflow elevation). Effluent release (Q_{out}/A) is calculated as any positive water elevation (i.e., E – 0 for all positive E's). When water is neither added via pumping nor lost via overflow, the water level of the pond fluctuates with positive or negative daily P-Ev values. The rainwater storage capacity of the pond using the "drop/add" approach is always present except when precipitation fills the pond to capacity.

The application of the two management systems was simulated using daily time steps for each of the five locations for the 40-year period 1961-2000. Water elevations for ponds at all five locations were calculated using the water balance described above, daily precipitation, and 80% of class A pan evaporation. Groundwater pumped and effluent lost was calculated and conservation of volume was observed.

Results and Discussion

Simulations for an Infinitely Deep Pond

Results from the cumulative analyses of the average year, the wettest year, and the driest year are shown in Figure 2. If all precipitation was captured and retained in the pond to offset subsequent evaporation, pond levels at all five sites during an average year would rise and remain above the starting level through about May 1st. In an average year, pond levels at all five sites end the year at or above the starting level for all but the western-most (Thompsons) and most interior (Stuttgart) locations. The worst case is an 8" drop at Thompsons.

Conceptually, these results document the regional climatic advantage available for aquaculture at these locations. The shapes of the average curves also reveal two other regional climatic traits relevant to management schemes for pond water levels. First, on the average, precipitation at all the sites is fairly evenly distributed through the year, so the shapes of the curves (sigmoid or double-sigmoid) are more dependent on the strong seasonality of evaporation. Secondly, two of the sites, Fairhope and Thompsons, show the marked effect of the tropical storm season on the average daily P-Ev.

Comparison of the wettest years at each of the five sites shows that the daily pond levels experienced a nearly continuous increase through those entire years at each location as precipitation exceeded evaporation most days of the year. The extreme wet years show the considerable differences in pond elevations as influenced by climate across the region. The Fairhope site ended its wettest year (1978) with pond levels 53" above the starting level while the Stuttgart site ended the wettest year there (1990) with water only 20" above the starting level– a regional range of 33".

Evaluation of the driest year at each of the five locations showed that pond water levels stayed near or above the starting level at all sites for the first two to five months of those years before starting to drop. Thereafter all sites showed a nearly continuous decrease in daily pond levels throughout most of the remainder of the year, as cumulative P-Ev remained consistently negative. Also, the extreme dry years exhibited almost as large a regional range of year-end water level as did the extreme wet years. For example, the Thompsons site ended its driest year (1988) nearly 33" below the starting level, but the Fairhope site ended the driest year there (1968) only 3" below the starting point – a range of 30".

A notable aspect of pond elevations that clearly emerged from this part of the analyses was the effect of wet and dry periods. Whereas major precipitation events dramatically and suddenly increased water levels, during dry periods the daily pond evaporation rates remained remarkably stable, damping the dramatic changes in water levels caused by precipitation. It was also noted that variability between the extreme wet and dry years was greatest at Thompsons and Fairhope, the two most coastal locations.

Water level fluctuations to the extent outlined above under a "climate control only" scheme cannot be tolerated in conventional warm water aquacultural ponds. Levees cannot economically be constructed high enough to allow for 56" to 33" changes in storage, the magnitude shown in Figure 2. Some degree of management must be introduced to control fluctuations in water level.

Pond Water Management Simulations

The results of the "make-up" management scheme are not illustrated in figures, since pond water elevations were constant throughout the simulations. Amounts of water added on a daily basis were small. They were only enough to replace daily evaporation.

The daily water level patterns created by the "drop/add" management scheme for the wettest years and the driest years at each of the five locations are shown in Figures 3 and 4. The days in the year when pond levels dropped to the 6"threshold (and 3" of groundwater was consequently added) are marked on the date of pumping in these Figures. The differences in the daily fluctuation patterns from place-to-place, the amounts of water added, and the distribution of pumping events through each of the years are evident in the Figures. These analyses illustrate the effects of rainfall, pumping, and evaporation on pond water levels, and document the capability of this management scheme when annual climatic variability is most extreme.

Figure 3 shows predicted daily water levels for the wettest years at all locations. Pond levels fell no more than 1-2" through April, and overflowed on many occasions with excess rainfall. Notably, the western-most and most interior locations began their wettest years with large deficits that were promptly made up by rainfall. No water had to be added at Fairhope, and only one pumping event was required at Clemson. The other three sites required only two or three additions of pumped water during the wettest year. Further-

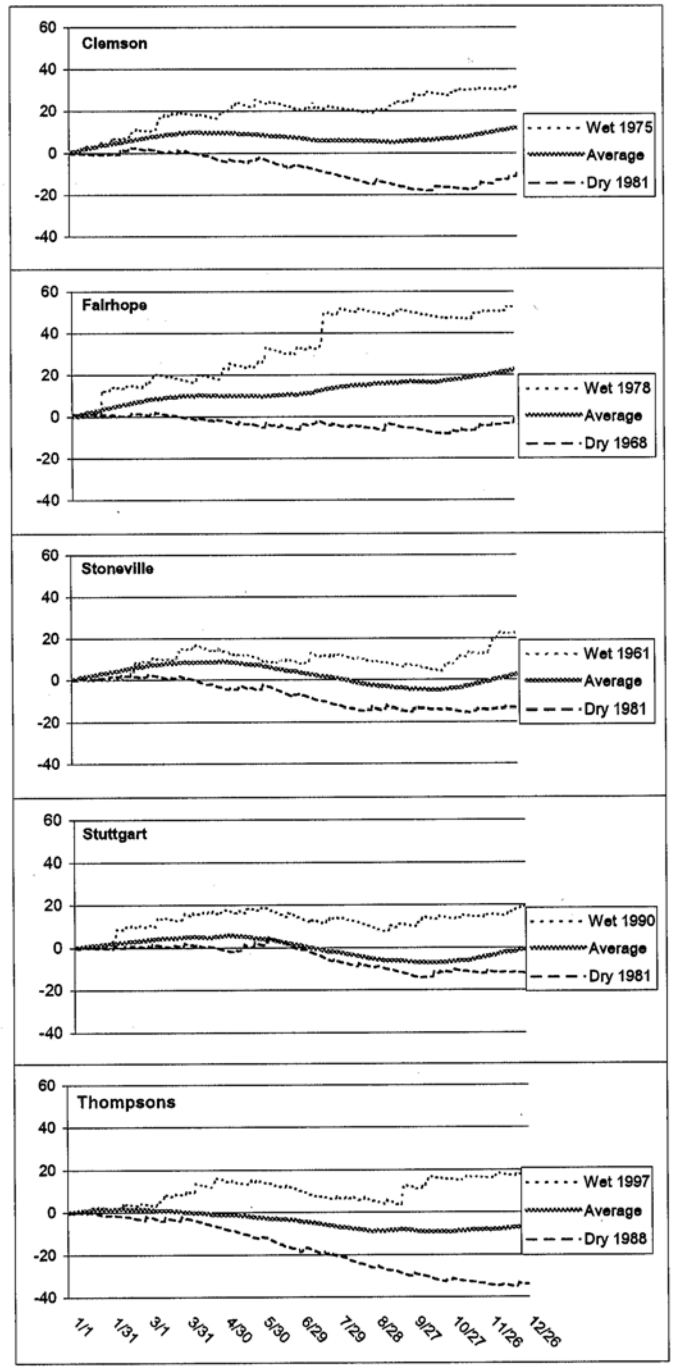

Figure 2. Daily water level determined by cumulative annual P-E, five locations–wettest and driest rears and 40-year average.

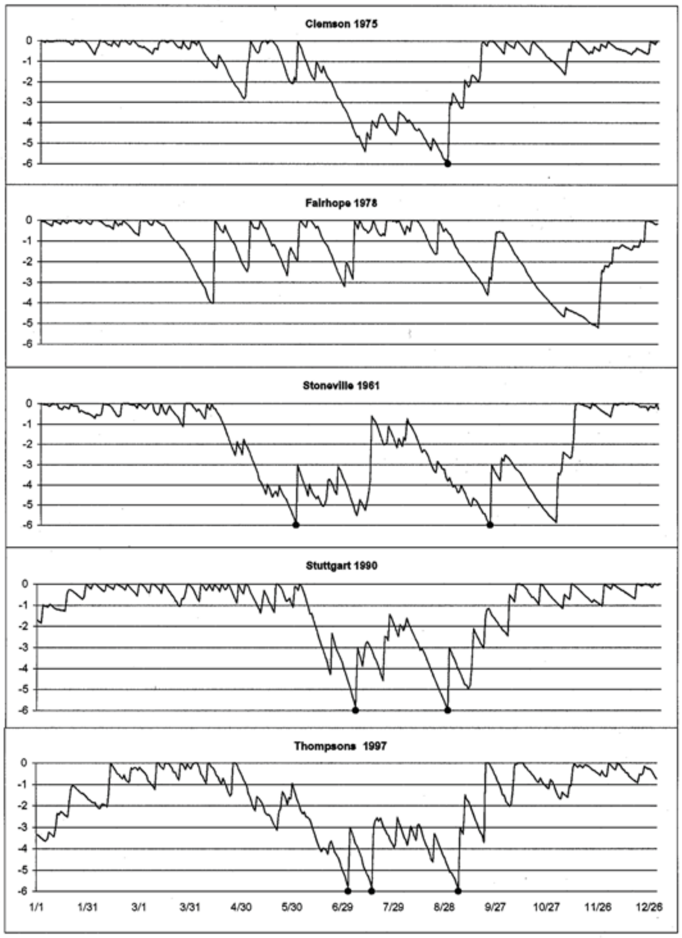

Figure 3. Daily water level, drop/add system, wettest years at each location.

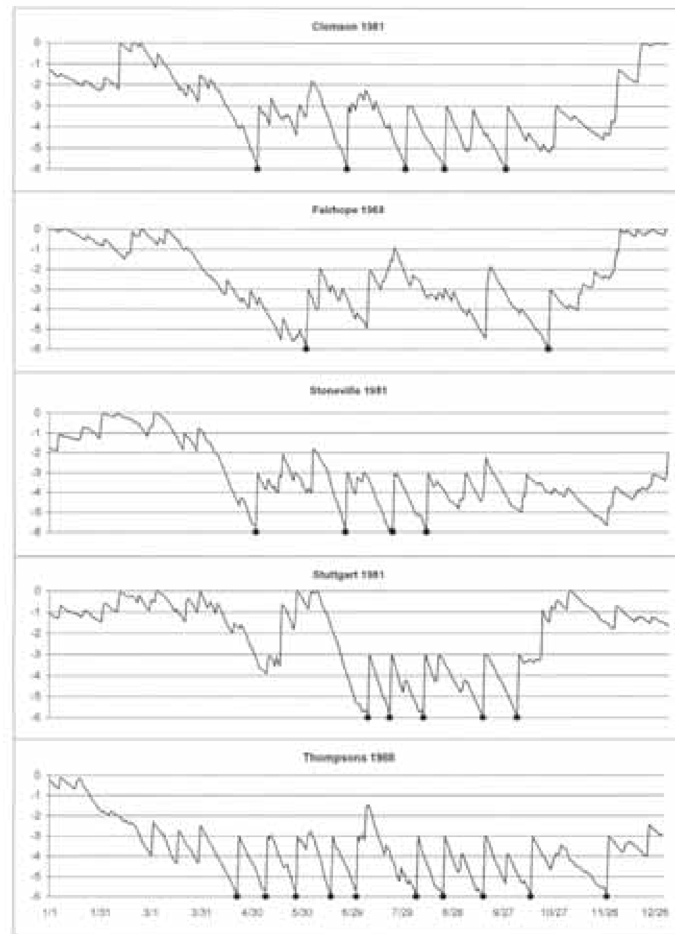

Figure 4. Daily water level, drop/add system, driest years at each location.

more, since only 3″ of water was added in those pumping events, subsequent large rainfalls at each of the sites were successfully captured. These simulations suggest that the climate in the Southern Region will provide nearly all the water necessary to maintain ponds within the 6″ design limit in wet years.

Figure 4 shows results of the management scheme in the opposite type of year–a year when rainfall was scarce and evaporation dominated. Water overflowed only one or two times through the year at all locations, and all locations required multiple additions of groundwater to maintain the pond levels within the 6″ design limit. Fairhope, the most coastal site, required only two pumping events, compared to 10 pumping events at Thompsons, the western-most site. The graph further reveals that, even in the extreme dry years, required pumping is limited to the time period between early April and late October. The comparative advantage of the maritime environment (Fairhope) is especially evident in the driest years. Figure 4 also clearly indicates how often 3″ additions of groundwater must be applied when no daily precipitation occurs, and thereby emphasizes the potential for conservation of this management method.

The years that required the most groundwater to maintain ponds within the 6″ design limit were typically those with the longest strings of dry days, rather than the driest years based on total precipitation. Figure 5 shows how the "drop/add" management strategy worked at Stoneville in 1966, the year requiring the most pumping events at that location (although 1981 was the driest year there). The addition of 24″ of groundwater (eight pumping events) was necessary. Pond level began dropping early in the year and no overflow occurred between mid-February and the last few days of the year. Repetitive additions of groundwater occurred at nearly uniform time intervals since no significant rainfall events took place during the part of the year when evaporation rates were consistently high. In comparison, the driest year (1981) was characterized by few major rainfall events but by multiple small events that occurred when water was most needed. The occurrence of periodic small rain events is historically more characteristic of the rainfall

Table 2. Forty-year averages of annual groundwater use (inches), two management methods, with conservation potential indicated

Test Site	"make-up"	"drop/add"	Amount Conserved
Fairhope	34.0	4.0	30.0 (88%)
Clemson	33.9	6.7	27.2 (80%)
Stoneville	40.1	14.5	25.6 (64%)
Stuttgart	39.7	13.5	26.2 (66%)
Thompsons	43.9	15.4	28.5 (65%)
Region	38.3	10.8	27.5 (72%)

regime in the Southern Region than are extended periods with no rain during the periods of greatest evaporative loss. This characteristic reveals that, even in dry years, there is a natural climatological advantage for the aquaculture industry in the South.

Table 2 compares water use under the "drop/add" scheme to the "make-up" water approach. Use of "make-up" water required a regional average of 38" of groundwater each year to keep ponds full. By comparison, the amount of groundwater required to keep ponds within the design limits of the "drop/add" scheme averaged about 11" a year regionally. These results indicate an average regional reduction in groundwater use of about 72% through employment of the "drop/add" management strategy. Specific figures for each site are given in the table.

Regional variation and spatial progression in both water use requirements and potential for conservation of groundwater are also evident in Table 2. For example, a pronounced regional east-west gradient occurred in the water required under the "drop/add" scheme at eastern-most Clemson (about 7") as compared to western-most Thompsons (about 15"). The maritime-to-continental gradient is also evident in the differences detected between the coastal environment represented by Fairhope (about 4") and the inland locations of Stoneville and Stuttgart (about 14-15" each).

Table 3, which shows the amounts of overflow calculated for both management schemes at each site, accentuates the efficiency of

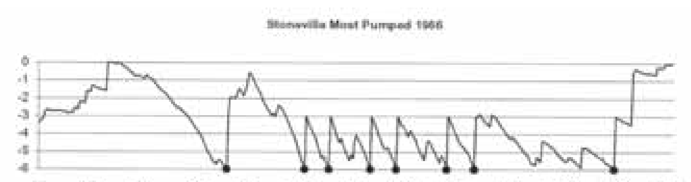

Figure 5. Daily water level 1966, drop/add system, year requiring most pumping, Stoneville, MS.

Table 3. Forty-year averages of annual overflow (inches), two management methods, with conservation potential indicated

Test Site	"make-up"	"drop/add"	Amount Conserved
Fairhope	56.5	26.5	30.0 (53%)
Clemson	45.8	18.7	27.1 (59%)
Stoneville	42.7	17.1	25.6 (60%)
Stuttgart	38.8	12.6	26.2 (68%)
Thompsons	36.8	8.3	28.5 (77%)
Region	44.1	16.6	27.5 (62%)

capturing rainfall rather than letting it escape to the environment. Use of the "make-up" water management scheme resulted in a regional average loss of 44" of useable precipitation each year. In contrast, the amount of useable precipitation lost to overflow when using the "drop/add" scheme averaged only about 17" per year regionally. These results thus indicate an average regional reduction in potentially-useable rainwater lost to overflow of about 62% through employment of the "drop/add" management strategy. Specific figures for each site are given in the table.

It is interesting to note that worst-case water use under the "drop/add" scheme was clearly superior to the best-case water use under the "make-up" scheme. Therefore, notwithstanding the spatial advantages of maritime environments or the temporal advantages of wetter years, the demonstrated potential for groundwater conservation is not limited by location in the region or by the annual climatic variability that is characteristic of the region.

Although the "drop/add" scheme is easily implemented and conceptually simple, there are some management considerations that must accompany its use. Although the 6" variation of water level was selected in part to minimize erosion of the levees, some protection and maintenance of exposed areas may be required. Also since in rainy years the primary source of water under this scheme will be rainfall, salinity levels should be monitored, especially after heavy rains. Finally, while it is unlikely that the water level fluctuations involved in the "drop/add" scheme will encourage increased benthic growth, if it becomes necessary to consider plans involving more extreme water level variations, deeper ponds may be necessary to maintain shading of pond bottoms.

Conclusions

It is becoming increasingly vital for warm water aquaculture producers to minimize production costs. Additionally, sustainable water use and avoidance of groundwater mining are conservation imperatives. Boyd and Gross (2000) state that reduction in effluent volume is the most effective way to save water in aquaculture production, and not only reduces water consumption but also reduces the pollution potential of pond aquaculture. Use of the suggested "drop/add" scheme has the potential to drastically reduce both groundwater use and loss to overflow in the southern region.

Pote, et al (1988) predicted a 50% reduction in groundwater use in catfish ponds in the Mississippi Delta. This investigation, using an expanded time frame and several locations throughout the southern region, concludes that actual reductions will exceed that 50% point at all sites in the region. The potential average conservation of groundwater indicated by this study ranges from 64% to 88%, and averages 72% regionally. Additionally, losses due to overflow were reduced from 53% to 77% regionally, with an average of 62%.

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Corresponding Author:
Charles L. Wax, Professor and State Climatologist
Department of Geosciences
Mississippi State University
Phone 662–325–3915
Fax 662–325–9423
E–mail: wax@geosci.msstate.edu

Water Quality in the Mississippi Embayment–Texas Coastal Uplands Principal Aquifer

Heather L. Welch[1], Roland W. Tollett[2], and Ronald C. Seanor[2]

The Mississippi Embayment–Texas Coastal Uplands principal aquifer comprises the Mississippi River Valley alluvial aquifer and Mississippi Embayment and Texas Coastal Uplands aquifer systems, and is one of sixteen principal aquifers identified by the U.S. Geological Survey's National Water–Quality Assessment (NAWQA) Program for further ground–water studies and regional synthesis of ground–water quality data. The principal aquifer covers approximately 197,187 square miles from southern Texas to southwestern Alabama and includes parts of Alabama, Arkansas, Illinois, Kentucky, Louisiana, Mississippi, Missouri, Tennessee, and Texas. According to 1990 water–use data, the principal aquifer ranks 9th in drinking water use, and 85 percent of the ground water withdrawn is used for irrigation, public–supply, industrial, and domestic needs. Three metropolitan areas use the principal aquifer as their primary source for drinking water – Memphis, Tennessee; Jackson, Mississippi; and Tyler, Texas.

From 1994 through 2004, water–quality samples were collected at 169 domestic, monitoring, irrigation, and public–supply wells ranging in depth from 21 to 1466 feet below land surface. Dissolved–solids concentrations generally were less than the U.S. Environmental Protection Agency National Secondary Drinking Water Regulation (SDWR) of 500 milligrams per liter (mg/L). Thirty samples exceeded the SDWR, and two samples were slightly saline (>1000 mg/L in concentration). Calcium and sodium were the dominant cations, and bicarbonate, chloride, and sulfate were the dominant anions detected. Over half of the samples had concentrations above the SDWR of 300 micrograms per liter (µg/L) for iron and 50 µg/L for manganese. Trace element (i.e. aluminum, chromium, manganese, etc.) concentrations were low. Two values for lead exceeded the USEPA's Maximum Contaminant Level (MCL) of 15 µg/L, and five values of arsenic were above the MCL of 10 µg/L.

Of the 77 pesticides and 8 degradation products analyzed, 27 pesticides and 2 degradation products were detected in water from 53 wells. The most frequently detected pesticides were bentazon, simazine, atrazine, and metolachlor which are all herbicides. The highest concentration of a pesticide detected was 14.8 ug/L for 2,4–D, an herbicide used on agricultural crops such as wheat, sorghum, corn, rice, and low–till Non agricultural uses are in rights–of–way, roadsides, non–crop areas, forestry, lawn and turf care, and on aquatic weeds. One pesticide, atrazine, had a concentration of 3.14 ug/L that exceeded the MCL of 3.0 ug/L. Atrazine is an herbicide that is used to control broadleaf weeds and some grassy weeds. Thirteen values of nitrate plus nitrite were above 2 mg/L. Of the 87 volatile organic compounds (VOCs) analyzed, 35 compounds were detected in water from 108 wells. The most frequently detected VOC was 1,2,4–trimethylbenzene, which is a gasoline additive. The highest concentration of a VOC detected was 22 ug/L for diisopropyl ether, a solvent. All concentrations of nutrients and VOC's were below MCL's.

Keywords: Ground water, water quality, drinking water

Corresponding Authors:
[1]United States Geological Survey, Jackson, MS
[2]United States Geological Survey, Ruston, LA

Session F: Water Quality Analyses

Phil Bass, Moderator

Onur Akay
Experimental Verification of the Interconnectivity of Macropores and Subsurface Drainage

Brian Alford
Environmental Relationships of Wadeable Stream Fisheries Resources in Mississippi

Peter Ampim
Runoff Losses of Pesticides and a Conservative Tracer from Warm–season Turf using Simulated Rainfall

Kimberly Caviness
Modeling the Big Black River: Evaluation of a Simplistic Water Quality Model

Alison Faulkner
Greenhouse Modeling of Nitrogen Use Efficiency in Two Wetland Cyperus Species at the University of Mississippi Field Station

Environmental Relationships to Wadeable Stream Fisheries Resources in Mississippi

John B. Alford and Donald C. Jackson

Wadeable streams in Mississippi do not garner the attention of most anglers or fishery managers; consequently, these streams hold an unmanaged recreational fishery. However, they can support quality sizes and abundances of largemouth bass (*Micropterus salmoides*), spotted bass (*Micropterus punctulatus*), and sunfishes. We used the U.S. Environmental Protection Agency's Wadeable Stream Assessment (WSA) to identify local and landscape-scale environmental features of Mississippi wadeable streams associated with relative abundances and size structure of catchable basses and sunfishes. In addition, we developed a testable regression model that can potentially be used as a rapid assessment tool to locate candidate streams in Mississippi that support a bass and sunfish fishery. Canonical correspondence analysis (CCA) suggested that increases in relative abundances of largemouth bass, longear sunfish (*Lepomis megalotis*), total bass combined and total sunfish combined were associated with small, meandering stream channels with residual pools, heavily forested watersheds and riparian canopy cover, and poor rapid habitat scores. Increases in relative abundances of spotted bass were associated with increasing stream size, flow and nitrogen concentration, decreases in channel incision height and sand substrates in favor of gravel and wood, as well as benthic macroinvertebrate assemblages with increasing proportions of scrapers and decreasing proportions of collector-gatherers and collector-filterers. These local-scale characteristics reflect forested riparian zones that minimize erosion and sedimentation from landscapes and supply woody debris for invertebrate colonization. In contrast, increases in bluegill (*L. macrochirus*) relative abundances were associated with more impacted systems, especially large, straight channels with open canopies, increased nutrient runoff, and landscapes with small to moderate increases in agricultural and urban cover (1-18% of watershed area). Our regression model suggests that, on average, as one proceeds towards the Mississippi Gulf Coast and 30 m wide riparian corridors are covered by increasing proportions of forest, then relative abundances and growth of age-1 of basses and sunfishes tend to increase. If this model can be validated, then fishery managers can use the model as a first-order assessment of wadeable streams in Mississippi with regard to their potential to support a recreational bass and sunfish fishery.

Keywords: Ecology, Recreation, Water Quality

Introduction

Wadeable streams support recreational fisheries throughout most of North America (Griffith 1999; Fisher and Burroughs 2003). The abundances and size structure of fisheries resources from these ecosystems tend to be related to local and landscape-scale environmental processes. For example, substrate size (Sowa and Rabeni 1995), water chemistry (Matthews 1987; Kwak and Waters 1997), woody debris inputs, (Angermeier and Karr 1984) and stable riparian corridors (Shields et al. 2000; Opperman and Merenlander 2004) are examples of local-scale processes that influence the abundances and size structure of fisheries resources from wadeable streams. However, landscape-scale attributes of wadeable streams typically influence local-scale processes, and they have been related to changes in abundance and biomass of trout (Jones et al. 1999; Kiffney et al. 2003; Wang et al. 2003; Kocovsky and Carline 2004), diatoms (Naymik et al. 2005) and aquatic insects (Nislow and Lowe 2006).

Clear-cutting of forest stands, row-crop agriculture, and pastoral activities on watersheds have lead to channel incision and sedimentation in wadeable streams (Roth et al 1996; Kauffman et al. 1997; Shields et al 2000; Dodds et al. 2004). Short-lived, small, fluvial generalists usually comprise fish assemblages from these streams (Schlosser 1982; Shields et al. 2003). Consequently, these wadeable streams typically do not support sport fisheries (Krueger and Waters 1983), especially for centrarchids (i.e., black basses, sunfishes and crappies) and salmonids (i.e., trout and salmon). In Mississippi, wadeable streams tend to be low-gradient, warmwater systems that support largemouth bass (*Micropterus salmoides*), spotted bass (*M. punctulatus*), and a variety of sunfishes (*Lepomis* spp.) (Robinson and Rich 1980; Robinson and Rich 1981; Robinson and Rich 1984). Compared to impoundments and large rivers (Jackson and Brown-Peterson 1995), local and landscape-scale processes that influence centrarchid fisheries resources in Mississippi wadeable streams are relatively unknown.

The goal of our research was to identify environmental influences on centrarchid fisheries resources in wadeable streams from Mississippi at two spatial scales. We define local-scale environmental features as those occurring at the riparian zone (0-300 m from channel edge) and the stream channel. Environmental features at the landscape-scale are those occurring throughout the entire upstream drainage area (i.e., watershed) of a reach. Our research had the following objectives: 1) identify local and landscape-scale environmental influences on abundance and size structure of centrarchid fisheries resources in wadeable streams from Mississippi and 2) develop a parsimonious and comprehensive model that can potentially predict centrarchid abundances and size structure from these systems.

Study Area

In 2004, the U.S. Environmental Protection Agency (USEPA) conducted the National Wadeable Streams Assessment (WSA) (USEPA 2004). From this program, we included the sampling units (i.e., stream reaches) from Mississippi. Sampling units consisted of one reach within each of 12 wadeable streams (Figure 1). Wadeable streams were identified as stream orders 1 through 5 (Strahler 1964). Reaches that were accessible only by boat were not considered wadeable. Sampling occurred during low flow conditions of summer months. The sample reach length was 40 times the mean wetted width at the X-site, where the X-site is the mid-point of the randomly-determined reach (Figure 2). This reach length was chosen by the USEPA to capture most of the variability in physical characteristics and benthic macroinvertebrate assemblages within the streams (USEPA 2004).

Materials and Methods

Environmental and benthic macroinvertebrate sampling protocol

Environmental data and benthic macroinvertebrates were collected once during the summer of 2004. The environmental data were collected at local and landscape scales, and variables from four categories were measured. The categories included physical characteristics of the channel (Table 1), taxonomy and size structure of riparian vegetation, land use and land cover (LULC) characteristics of the upland watersheds (Table 2) and water chemistry. Sampling methods for each of these data followed the environmental monitoring and assessment protocol in the WSA (USEPA 2004).

Benthic macroinvertebrates were collected in 2004 with a 500 μm mesh D-frame kicknet at left, center or right portions of transects for a period of 30 seconds. Samples from each transect were

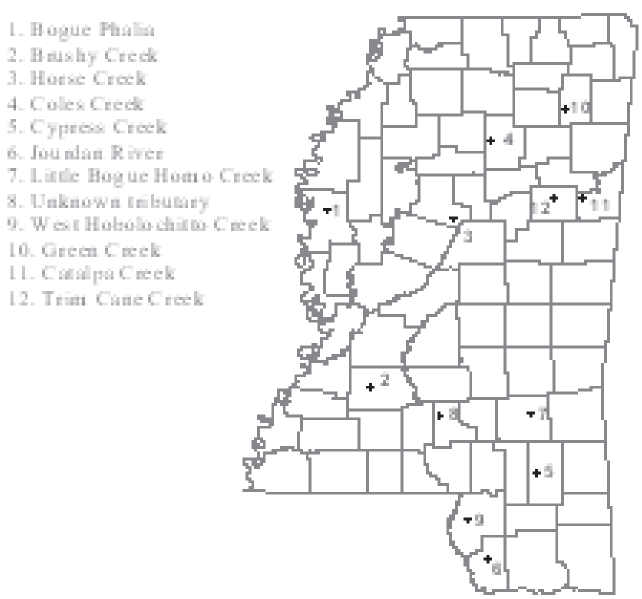

Figure 1. Locations of the 12 wadeable stream reaches in Mississippi that were sampled during the summers of 2004 and 2005.

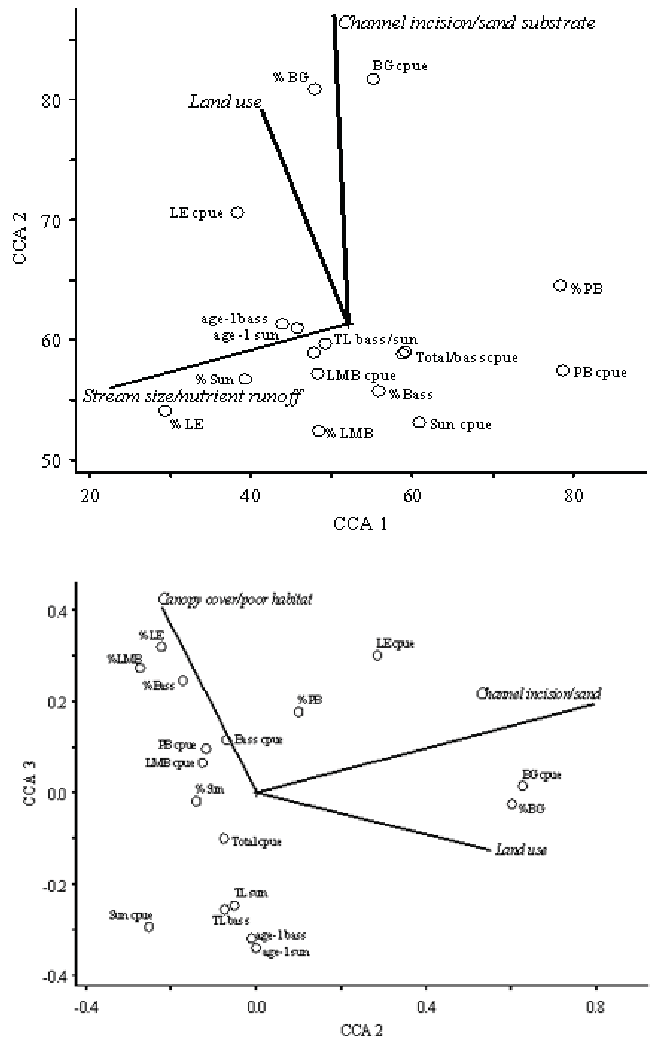

Figure 2.–Joint-plots describing a canonical correspondence analysis (CCA) ordination of centrarchid catch characteristics (circles). Environmental variables associated with catch characteristics are in italics. LMB=largemouth bass; PB=spotted bass; LE=longear sunfish; BG=bluegill sunfish; Total cpue=all fishes combined; Total bass=largemouth and spotted bass combined; Sun=all Lepomis spp. combined; TL=mean total length; age-1=mean length at age 1

Table 1. A summary of environmental data collected from reaches within 12 wadeable streams in Mississippi during summer 2004.

Environmental variables	Mean	Std. error
Channel		
CV of bank angle	73.0	8.16
Wetted width	9.2	1.79
Bankfull width	12.3	2.78
Bar width	2.2	1.36
Bankfull height	0.6	0.09
Incision height	2.8	0.27
CV of cross-sectional depth	104.4	5.19
Substrate size-class	2.6	0.36
% fines	29.2	9.26
% sand	42.1	8.58
% gravel	12.8	5.36
% wood	6.2	1.56
Total LWD units	36	10.94
Canopy cover (densiometer count: 0-17)	11.8	1.20
% water surfact slope	1.14	0.06
CV of bearing (i.e., sinuosity)	24.3	5.39
CV of thalweg depth	49.2	3.93
% of reach with bars	51.4	7.8
% of reach with soft sediment	64.2	10.11
% of reach with backwater	52.8	13.03
% of reach with residual pools	7.8	3.26
Discharge (m^3/s)	1.2	0.69
Rapid habitat assessment (0–180)	131.3	10.9
Intensity of beaver dams (0–3)	0.5	0.26
Riparian zone		
% deciduous vegetation	74.9	10.7
% mixed vegetation	25.1	10.7
Large tree cover class (0–4)	1.6	0.18
Small tree cover class (0–4)	2.5	0.17
Understory shrubs class (0–4)	2.8	0.15
Understory grasses class (0–4)	0.9	0.14
Ground shrubs class (0–4)	3.2	0.18
Ground grasses class (0–4)	2.3	0.27
Barren/dirt class (0–4)	1.3	0.12
Legacy tree dbh class (0–4)	2.3	0.30
Legacy tree distance from bank (m)	13.5	3.72
Total human disturbance score (0–6)	0.18	0.05
Aesthetic appeal (0–10)	6.8	1.08
Fish cover density (0–4)		
Small woody debris	1.5	0.33
Live trees/rootwads	0.5	0.08
Overhanging vegetation	2.3	0.34
Undercut banks	0.17	0.04
Watershed: physiographic features		
Stream order	3.3	0.30
Land area (km^2)	100.6	25.0
Latitude	32.5	0.38
Longitude	89.4	0.19
Land use/land cover		
Human impact score (0–26)	12.8	1.59
% forest	76.6	8.45
% non-forest	23.0	8.54
% deciduous vegetation	22.8	7.40
% conifer vegetation	37.3	10.18
% mixed vegetation	13.8	4.85
% agriculture	15.6	7.82
% pasture	4.5	1.54
% urban	3.8	1.42
Water chemistry		
pH	6.6	0.16
Temperature	25.9	0.76
Dissolved oxygen (mg/L)	4.4	0.50
Specific conductance (μS/cm)	179.8	80.98
Total alkalinity (mg/L $CaCO_3$)	43.1	10.67
Hardness (mg/L $CaCO_3$)	47.1	12.14
Turbidity (NTU's)	17.0	6.08
NH_3-N (mg/L)	0.58	0.13

Table 1. A summary of environmental data collected from reaches within 12 wadeable streams in Mississippi during summer 2004. (continued)

Environmental variables	Mean	Std. error
NO_3-N (mg/L)	0.10	0.02
Total nitrogen (mg/L)	0.33	0.05
Total phosphorus (μg/L)	40.2	9.13
Chlorine (mg/L)	11.9	2.12
Dissolved CO_2 (mg/L)	7.5	1.90
Dissolved organic carbon (mg/L)	4.3	0.68
Dissolved inorganic carbon (mg/L)	10.4	4.62
Total suspended solids (mg/L)	18.5	5.91

Table 2. Metrics describing benthic macroinvertebrates collected from 12 wadeable streams in Mississippi during summer 2004.[1]

Metrics	Mean	Std. error
Functional feeding groups		
% collectors (filterers + gatherers)	51.2	4.53
% shredders	15.5	3.18
% predators	19.2	3.46
% scrapers	9.5	2.83
Habit-types		
% burrowers	22.8	4.74
% climbers	18.6	3.28
% clingers	28.3	4.98
% miners	3.2	2.12
% sprawlers	20.6	4.80
% swimmers	1.8	0.91
Size structure and abundance		
Total length (mm)	3.4	0.54
% large individuals (>15 mm TL)	2.9	0.41
Abundance (individuals/m^2)	5,802	1,202

[1]Functional feeding and habit-type groups follow Merritt and Cummins (1996)

combined into one jar and preserved in 70-80% ethanol at the site. The reach-wide sample was then brought back to the fish research laboratory at Mississippi State University, where the invertebrates were sorted and identified. All benthic macroinvertebrates were measured (TL, mm), identified to genus using dichotomous keys and assigned to functional feeding and behavioral groups (e.g., collector-gatherer or burrower) (Merritt and Cummins 1996). The samples then were sent to Rithron Associates, Inc. (Billings, Montana) for taxonomic verification (a QA/QC procedure).

Fish sampling protocol

Centrarchid fisheries resources were sampled by angling with ultra-light fishing gear (spinning reels with 1.8 kg test line and 1.7 m rod). We chose this sampling technique because it focused the sampling effort on the species of interest in Mississippi's wadeable streams (i.e., catchable centrarchids). Angling is an important sampling technique for fisheries resources in streams, because it applies directly to management of recreational fisheries in these systems (Hudgins 1984). In addition, angling has been a valuable sampling technique for small stream fish stocks in Alaska (Hetrick and Bromaghin 2006) and catfish stocks in South Dakota streams (Arterburn and Berry 2002), especially when other gear types were ineffective at sampling certain sizes or species of fish.

Beetlespin lures were used to fish all reaches and consisted of yellow grub bodies (5.1cm long), 3.5 g yellow jig heads and #0 nickel spinner blades. Two anglers entered a reach at its downstream end and fished in an upstream direction for three hours. To address temporal variation in angling, reaches were fished on three separate occasions from June through September 2004-2005. Upon capture, fish were measured (TL in mm), and scales were removed for age analysis. Mean length-at-age was determined by back-calculation regressions following the procedures of Gulland (1969). All fish were released back into the stream at their point of capture.

Statistical analyses

We used canonical correspondence analysis (CCA) to identify environmental influences on centrarchid fisheries resources in wadeable streams from Mississippi. A CCA is essentially a multivariate linear regression analysis. Unlike multiple linear regressions (MLR), which use only one dependent variable, CCA uses multiple dependent variables and accounts for multicollinearity between the dependent variables. In our analyses, CCA simultaneously ordinated reaches and centrarchid catch characteristics while running a multiple linear regression on the fish matrix and the environmental and benthic macroinvertebrate data.

As the number of independent variables approaches the number of samples units (i.e., reaches), CCA results become increasingly unreliable. Therefore, we ran a principal component analysis (PCA) on the original environmental matrix (67 variables) and the original benthic macroinvertebrate matrix (13 metrics). We reduced the dimensionality of these original datasets to ≤6 principal components for each matrix. These new independent variables (i.e., the components) were used in subsequent CCA with the original fish matrix (17 catch characteristics). To improve univariate normality, we arcsine-square root-transformed proportion data and log-transformed non-proportion data before any analyses occurred. All multivariate analyses were conducted with PC-ORD version 4.1 (McCune and Mefford 1999).

To develop a testable model, we used MLR to describe the relationships between centrarchid fisheries resources and environmental features of wadeable stream ecosystems in Mississippi. First, we performed a PCA on the centrarchid catch matrix to reduce the dimensionality of the original fish matrix and define gradients in abundance and size structure. Then, we selected candidate environmental variables that reflected the results of the CCA. The independent variables in the MLR included latitude, longitude and drainage area (km^2), an unnatural watershed vegetation index (e.g., agriculture or timber plantations), percent of the 0, 30 and 300 m wide riparian corridors forested and percent of the 0, 30 and 300 m wide riparian zones with human activities (e.g., parking lots, golf courses, residences). At α=0.10, we used backward elimination and best-subsets regression (Draper and Smith 1998) to select a parsimonious (i.e., simple) MLR model with the best fit. Finally, we checked the assumptions regarding constant variance, independence and normality of the errors by inspecting studentized residual plots and normal probability plots of the studentized residuals (Draper and Smith 1998). Regression analyses were conducted with SAS version 9.1 (SAS Institute Inc., Cary, NC).

Results

Description of environmental characteristics

The reaches in this study (Table 1) occurred primarily in forested landscapes (mean forest cover 76.6%). They were low-gradient (mean water surface slope 1.14%), warm-water (mean temperature 25.9°C) systems with unstable substrates (fines and sand 29.2% and 42.1%, respectively). On average, stable substrates (i.e., woody debris and gravel) comprised only 19% of the available substrates in the reaches. The cross-sectional depth varied twice as much as the longitudinal thalweg depth (Table 1), reflecting the formation of point bars (on average, median bar width 2.2 m; bars cover 51.4% of reach). Channel units were mostly glides, rather than riffles or pools (on average, <8% of reach length contained riffles or pools). On average, the channels were relatively narrow (median bankfull width 12.3 m) and straight (CV of mid-channel bearing 24.3).

Deciduous vegetation dominated the 10 m wide riparian zones (74.9% cover), followed by mixed coniferous/deciduous vegetation (25.1% cover). On average, the size structure of the riparian canopy (Table 1) was composed of small trees <0.30 m diameter-at-breast height (dbh), and the understory and ground cover consisted mainly of woody shrubs and saplings (i.e., height between 0.50 and 5.0 m). Total human disturbance scores averaged only 0.18 (where possible scores were 0-6). Therefore, the 10 m wide riparian corridors were very minimally impacted by human activities. Woody debris and overhanging vegetation from riparian zones provided most of the in-stream fish cover, followed by live trees/rootwads and undercut banks.

Conifers comprised most of the forests in the upland watersheds (37.3% cover), followed by deciduous (22.8% cover) and mixed (13.8% cover) vegetation. On average, row-crop agriculture (15.6% cover) was the dominant human land use activity on the watersheds, followed by pastures/hay fields (4.5% cover) and urban areas (3.8% cover of roads, residences, commercial and industrial facilities, etc.). Percent cover of agriculture varied from 0 to 97%, while percent of pastures and urban areas varied from 1 to 18% of the watershed areas.

Dissolved oxygen levels were low (on average, 4.4 mg/L), but the oxygen measurements were taken during summer low flow conditions. Total alkalinity varied widely among reaches. The reaches in the southeastern coastal plain region averaged <20 mg/L $CaCO_3$, while northern reaches in the Blackland prairie region averaged up to 155 mg/L $CaCO_3$. On average, specific conductance, turbidity, ammonia-nitrogen, nitrate-nitrogen, phosphorus, dissolved carbon and total suspended solids were below the recommended minimum criteria for Mississippi streams (see USEPA 2000 and MDEQ 2003 for minimum criteria).

Description of benthic macroinvertebrate assemblage

On average, collectors (i.e., filterers and gatherers) comprised the dominant functional feeding group (51.2% relative abundance) followed by predators (19.2%), shredders (15.5%) and scrapers (9.5%) (Table 2). Clingers (28.3%), burrowers (22.8%), sprawlers (20.6%) and climbers contributed relatively even abundances to the composition of habit-type groups, followed by miners (3.8%) and swimmers (1.8%). Overall, the invertebrates were abundant (on average 5,802 individuals/m^2), but very small (on average TL<3.5 mm and only 2.9% of individuals in a sample were >15 mm TL).

Description of fisheries resources

Over the course of the study, 298 fish were caught in 216 hours of angling effort. The mean total catch rate was 1.4 fish/angler-hour (Table 3). Total bass catch rates (0.90 fish/angler-hour; all Micropterus spp. combined) were nearly twice as large as total sunfish catch rates (0.53 fish/angler-hour; all Lepomis spp. combined). Likewise, largemouth bass and spotted bass catch rates (0.31 and 0.38 fish/angler-hour, respectively) were larger, on average, than that for longear sunfish and bluegill sunfish (0.26 and 0.13 fish/angler-hour, respectively). Basses comprised 47% of the total catch, while sunfishes comprised 40% of the total catch. However, longear sunfish (22.85 of total catch) were proportionately more abundant than spotted bass (11.7% of total catch). On average, basses and sunfishes were relatively small (212.4 mm TL and 135.4 mm TL, respectively). Mean age-1 length of sunfishes (92.1 mm) was nearly the same as that for basses (113.0 mm).

Reduction of environmental and benthic macroinvertebrate matrices

We used PCA to reduce the dimensionality of the original environmental and benthic macroinvertebrate matrices. As a result, we obtained new independent variables for subsequent CCA with the fish catch matrix. The environmental matrix (67 variables) was reduced to six new principal components that explained 78.2% of the variation in the original environmental data (Table 4). These new environmental variables (i.e., the components) suggested that wadeable stream ecosystems in Mississippi were influenced by combinations of landscape and local-scale features. In addition, these new environmental variables suggested that anthropogenic activities and geographic/physiographic characteristics influenced the structure of wadeable stream ecosystems in Mississippi.

We interpreted the new environmental variables as 1) human land use; 2) stream size and nutrient runoff (i.e., ammonia, nitrates, dissolved organic carbon, phosphorus); 3) canopy cover and habitat quality (i.e., amount of stream shade and habitat assessment scores); 4) alkalinity and small riparian trees; 5) sedimentation and straightened channels; 6) channel incision and sand substrate. The benthic macroinvertebrate matrix (13 metrics) was reduced to six components that explained 94.3% of the variation in the original data (Table 4). We interpreted the new benthic macroinvertebrate variables as 1) burrowers; 2) collectors and scrapers; 3) large individuals and predators; 4) small individuals, climbers and shredders; 5) miners and sprawlers and 6) swimmers. Because PCA is an orthogonal analysis, the new independent variables (i.e., components) were not linearly related. Therefore, multicollinearity between these new variables was minimized in subsequent CCA.

Environmental influences on fisheries resources in wadeable streams

Centrarchid fisheries resources in wadeable streams from Mississippi were associated with the following environmental gradients: 1) land use, 2) stream size and nutrient runoff, 3) canopy cover and habitat quality, and 4) channel incision and sand substrate (Figure 2). The CCA results suggested that largemouth bass abundance was correlated with minimally incised channels and proportionately less sand substrate in favor of other substrates. Additionally, largemouth bass, spotted bass, total bass and total sunfish abundances were correlated with increasing amounts of canopy cover (i.e., shaded channels with forested riparian zones) and relatively low habitat assessment scores (i.e., some sedimentation, few pools, relatively straight channels). Spotted bass were also correlated with wider streams, larger drainage areas and discharges, as well

Table 3. A summary of centrarchid catch characteristics from reaches within 12 wadeable streams from Mississippi. Centrarchids were sampled with ultra-light fishing gear on three occasions during summers 2004 and 2005.[1]

Catch characteristics	Mean	Std. error
Catch rate (CPUE: fish/angler-hour)		
Total	1.4	0.27
Total bass	0.90	0.17
Total sunfish	0.53	0.16
Largemouth bass *Micropterus salmoides*	0.31	0.06
Spotted bass *M. punctulatus*	0.38	0.15
Longear sunfish *Lepomis megalotis*	0.26	0.06
Bluegill sunfish *L. macrochirus*	0.13	0.04
Catch composition		
% Total bass	47.0	7.10
% Total sunfish	40.3	6.60
% Largemouth bass	35.0	6.60
% Spotted bass	11.7	5.61
% Longear sunfish	22.8	5.77
% Bluegill sunfish	9.7	3.19
Size structure (mm)		
Mean total length basses (spotted bass + largemouth bass)	212.4	10.85
Mean total length sunfishes (all *Lepomis* spp. combined)	135.4	6.32
Mean length age-1 basses	113.0	6.52
Mean length age-1 sunfishes	92.1	5.85

[1]Total=all fish species combined
Total bass=spotted bass and largemouth bass combined
Total sunfish=longear, bluegill, green, and redear sunfishes combined.

Table 4. PCA ordination results of an environmental matrix and a benthic macroinvertebrate matrix. Components with broken-stick eigenvalues ≥ 1.0 were retained as new independent variables in a subsequent CCA with a fish matrix. The purpose of these PCA ordinations was to reduce the number of environmental and benthic macroinvertebrate variables to a manageable set for further CCA.

New variable name	Proportion of variance explained in original data	Broken-stick eigenvalue
New environmental variables	78.2% (cumulative)	
Land use	24.8%	4.79
Stream size/nutrient runoff	18.1%	3.79
Canopy cover/ habitat quality	11.3%	3.29
Alkalinity/small riparian trees	9.2%	2.96
Sediment/ straightened channels	8.6%	2.71
Channel incision/ sand substrate	6.2%	2.51
New invertebrate metrics	94.3% (cumulative)	
Burrowers	29.0%	3.18
Collectors/scrapers	19.9%	2.18
Large/predators	16.8%	1.68
Small/climbers/ shredders	13.7%	1.35
Miners/sprawlers	9.6%	1.10
Swimmers	5.3%	0.90

as increasing nutrient levels. In contrast to spotted bass, longear sunfish were correlated with narrower streams, smaller drainages and discharges, as well as decreasing nutrient levels. Longear sunfish were similar to largemouth bass in that they were correlated with increased canopy cover and relatively low habitat assessment scores. Bluegill sunfish were correlated with more heavily impacted conditions than other fishes, especially increased land use and channel incision as well as proportionately more sand substrate instead of wood and gravel. Juvenile growth (i.e., from agew-0 to age-1) and adult size of basses and sunfishes were correlated with open canopies and large habitat assessment scores (i.e., stable substrate, pools, sinuous channels).

Centrarchid fisheries resources also were correlated with benthic macroinvertebrate groups that reflected environmental gradients of wadeable stream ecosystems in Mississippi (Figure 3). The CCA results suggested that spotted bass, total bass, total sunfish and total (i.e., all fishes combined) CPUE were correlated with increasing proportions of scrapers and decreasing proportions of collectors. In contrast, longear sunfish and growth of age-0 sunfish were correlated with proportionately more collectors and proportionately less scrapers. Benthic macroinvertebrate assemblages with abundant scrapers and fewer collectors reflect stream environments with stable substrates such as wood and gravel, allowing periphyton to attach and grow. Periphyton is the main food source for scraping invertebrates. In addition, periphyton and scrapers tend to be found in larger (e.g., 4-6th order), open canopy streams with abundant sunlight and nutrients. Conversely, benthic macroinvertebrate assemblages with abundant collectors and fewer scrapers reflect environments with less primary productivity in favor of detrital energy pathways.

Largemouth bass abundance was correlated with swimmers, while bluegill sunfish and percent total sunfish were correlated with proportionately fewer swimmers. These results suggested that largemouth bass were associated with lentic habitats (i.e., pools and backwater) provided by beaver dam pools, meandering channels with scour holes and/or pools formed by woody debris jams. In contrast, bluegill sunfish were associated with relatively straight channels, less stable substrate and fewer pools. These conditions are advantageous for bluegill sunfish because they can escape predation by pool-dwelling largemouth bass and spotted bass.

Longear sunfish were correlated with proportionately few burrowing invertebrates, while percent total bass were correlated with proportionately more burrowers. In addition, longear sunfish were correlated with large and predatory benthic macroinvertebrates (i.e., higher trophic levels), while total catch rates and largemouth bass catch rates were correlated with small, non-predatory invertebrates (i.e., lower trophic levels).

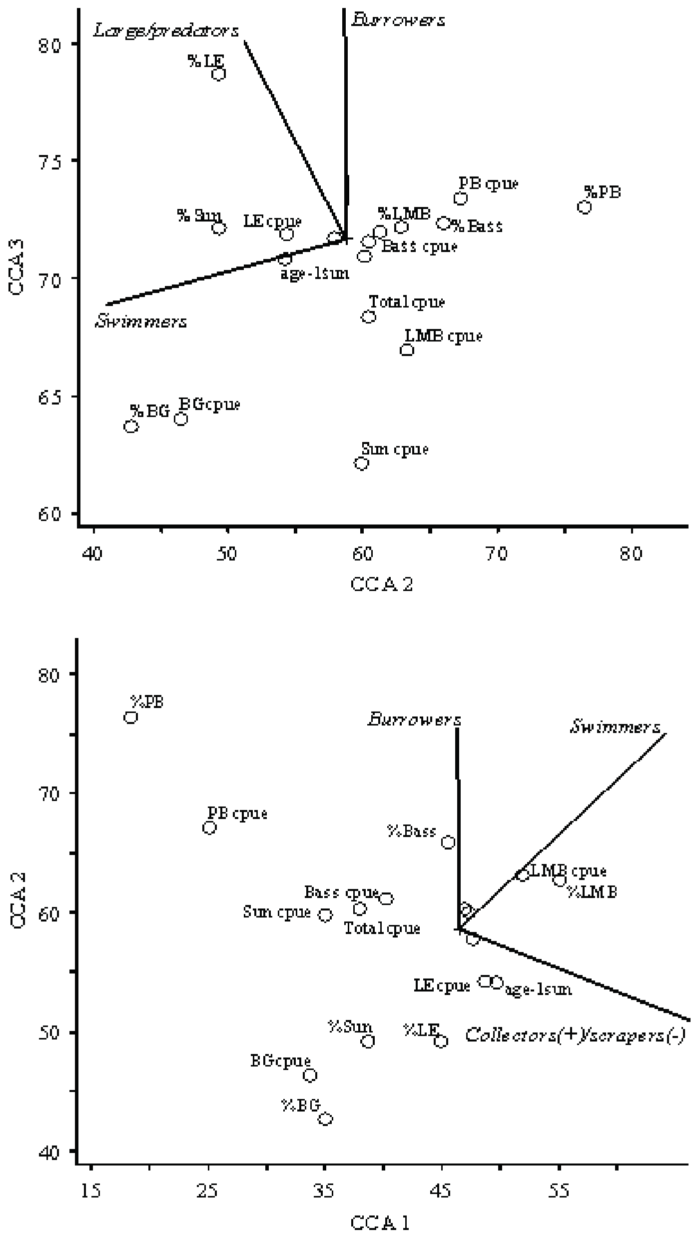

Figure 3.— Joint-plots describing a canonical correspondence analysis (CCA) ordination of centrarchid catch characteristics (circles). Benthic macroinvertebrate metrics associated with catch characteristics are in italics. LMB=largemouth bass; PB=spotted bass; LE=longear sunfish; BG=bluegill sunfish; Total cpue=all fishes combined; Total bass=largemouth and spotted bass combined; Sun=all Lepomis spp. combined; TL=mean total length; age-1=mean length at age 1.

A testable model for centrarchid fisheries in wadeable streams from Mississippi

To reduce the dimensionality of the original fish matrix, we used PCA to define gradients in centrarchid catch characteristics. The first three principal components explained 76.6% of the variation in the original fish matrix (Table 5). The first principal component (i.e., axis) represented a catch diversity and size gradient. The variables total catch rate, total sunfish catch rate, percent total basses, and mean total length of basses and sunfishes were negatively correlated with the first principal component (eigenvectors≥-0.32). The second principal component represented a spotted bass/longear sunfish gradient. Spotted bass were negatively correlated with principal component two (eigenvectors≥-0.34), while longear sunfish were positively related to the component (eigenvectors≥0.35). The third principal component represented a gradient in largemouth bass/bluegill sunfish abundance and growth of bass and sunfish. Largemouth bass were negatively correlated with the third component (eigenvectors≥-0.40), while bluegill sunfish and growth were positively correlated with the component (eigenvectors≥0.30). To develop a testable model for centrarchid catches in Mississippi wadeable streams, the site scores (i.e., locations of the reaches in species space) were used as new dependent variables in subsequent MLR analyses.

We developed a testable model that can potentially predict centrarchid catch characteristics in Mississippi's wadeable streams. The model describes the relationship between centrarchid abundance and size (i.e., site scores from principal component one) and environmental characteristics of Mississippi wadeable streams. We used the information from the CCA results to select independent variables reflecting landscape and local-scale processes as well as anthropogenic activities and natural geographic/physiographic characteristics. Using best subsets regression and backward elimination, we found a comprehensive and parsimonious MLR model (N=12; F=10.87; P=0.0007; R^2=0.76) for centrarchid fisheries resources in Mississippi's wadeable streams:

$$Y_{ij} = -37.562 + 1.298 * \text{latitude} - 0.072 * \text{rfor30};$$

where the dependent variable (Y_{ij}) is a gradient of large to small abundances and sizes of bass and sunfish. The independent variable "latitude" is the y-coordinate of the reach mid-point in decimal degrees, and "rfor30" is the percent of the 30 m wide riparian corridor that is forested. To address assumptions of linear regression analysis, inspections of the model's studentized residuals suggested the errors have constant variance, and they are independent and normally distributed.

Conceptually, the model says that as one proceeds towards the northern region of the state and riparian corridors (30 m wide) have proportionately less forest cover, wadeable stream reaches in Mississippi tend to support, on average, smaller total catch rates and sunfish catch rates, proportionately fewer basses, and smaller sizes of basses and sunfishes (Figure 4). Likewise, as one proceeds

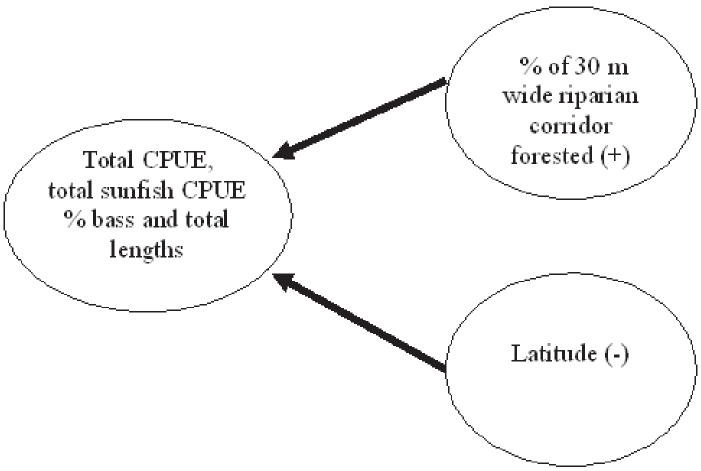

Figure 4. A conceptual illustration of a testable MLR model developed for centrarchid fisheries resources in Mississippi's wadeable streams.

towards the coastal region of the state and riparian zones are covered by proportionately more forest, wadeable stream reaches in Mississippi, on average, have larger total catch rates and sunfish catch rates, increased proportions of basses, and larger sizes of basses and sunfishes. The predictive ability of the model can be tested by incorporating a new, independent dataset from different wadeable stream reaches in Mississippi. The differences between observed and predicted values can then be evaluated. This validation procedure is required to evaluate the effectiveness of this model to predict the abundance and size of centrarchid fisheries resources in Mississippi's wadeable streams.

DISCUSSION

We found that centrarchid fisheries resources in Mississippi wadeable streams were correlated with a combination of local and landscape-scale human impacts to streams. Channel incision, substrate size, nutrient runoff, canopy cover and habitat quality were local-scale environmental characteristics associated with wadeable stream fisheries resources. These local attributes were probably influenced by landscape-scale characteristics, especially land use activities and stream size. Our results suggest that wadeable streams in Mississippi can support a centrarchid fishery in landscapes with small to moderate levels of land use (e.g., 1-18% of watershed area) in conjunction with forested riparian buffers.

Shields et al. (1998) studied the effects of channel rehabilitation on channel incision and sedimentation in reaches that drain large agricultural landscapes in Northern Mississippi. They found that fish abundance and fish size (especially largemouth bass and other fisheries resources) were greater in reaches where the banks were rehabilitated with stone spurs and willow posts. The bank modifications increased channel sinuosity and bank stability, providing sport fishes with larger residual pool area and slower mean current velocities. We found that increased channel incision and proportionately more sand substrate were negatively correlated with total catch rate, largemouth bass, total bass and sunfish abundances. Our results suggest that forested riparian zones minimize channel incision and excessive sedimentation, maintain sinuous channels that facilitate pool formation and provide backwater areas for centrarchid fisheries resources. In addition, these pools and backwater areas benefit anglers by providing access to catchable fish.

Landscapes covered by small to moderate amounts of agriculture, yet buffered by forested riparian corridors apparently increase primary and secondary productivity in wadeable streams, which can increase fish production (Nislow and Lowe 2006). For example, Krueger and Waters (1983) showed that fish stock biomass (all species combined) and invertebrate biomass differed significantly among three headwater trout streams in Minnesota with respect to human impacts to their watersheds. In their study, a stream within an agricultural landscape had large mean annual alkalinity concentrations of (245 mg/L), nitrates (5.0 mg/L), and proportionately more sand substrate compared to two streams in forested landscapes. Annual herbivore-detritivore biomass was larger in the agricultural stream (119.6 g/m2) compared to the forested streams (27.0 g/m2 and 36.9 g/m2, respectively). Similarly, the annual fish stock biomass was substantially larger in the agricultural stream (466.4 kg/ha) than the forested streams (46.6 kg/ha and 134.9 kg/ha, respectively).

We hypothesize that small to moderate levels of landscape development in combination with locally forested riparian zones increase nutrients inputs (Herlihy et al. 1998), such that fungal and bacterial productivity increases (Wainright et al. 1992; Koetsier et al. 1997; Benstead et al. 2004). This increase in primary productivity leads to an increase in secondary invertebrate production (Sponseller and Benfield 2001; Shieh et al. 2002), followed by increases in growth and abundance of basses and sunfishes. Microbial production increases dramatically when organic inputs from forested riparian zones (especially leaf litter) are available (Peckarsky 1980; Pomeroy and Wiebe 1988; Peterson et al. 2001). In Oregon, diatom production and diversity (Naymik et al. 2005) streams were maximized at moderate levels (~ 30% of watershed area) of watershed timber harvest compared to watersheds with no timber harvest. Kiffney et al. (2003) showed that periphyton and aquatic insect biomass were greatest when riparian buffer strips were 10-30 m wide (i.e., moderate timber harvest) compared to no riparian buffer (i.e., heavy timber harvest).

Landscape-scale impacts to streams must be mediated through forested riparian buffers. In the western United States, local impacts to wadeable stream riparian zones have reduced survival and recruitment of salmonids. Riparian zone deforestation has lead to siltation of spawning sites, declines in food resources by reducing coarse substrate availability for invertebrates, and sedimenta-

Table 5. PCA ordination results of a centrarchid catch matrix describing abundances and size structure of centrarchids caught by angling in reaches within 12 wadeable streams in Mississippi. Site scores represent graphical locations of the sample reaches along a particular axis. Percentages are the percent of the total variation in the original data explained by a particular axis. Eigenvectors (i.e., loadings) are analogous to correlations of centrarchid catch variables to a particular axis (i.e., component). [1]

	Site scores				Eigenvectors		
Reach name	**Axis 1 37.4%**	**Axis 2 22.1%**	**Axis 3 17.1%**	**Catch characteristic**	**Axis 1**	**Axis 2**	**Axis 3**
Coles Creek	-1.40	1.31	-0.99	Total cpue	-0.34	-0.19	-0.03
Green Creek	0.17	0.35	0.71	Total bass cpue	-0.23	-0.04	0.39
Jourdan River	-2.86	-2.89	3.23	Total sunfish cpue	-0.35	-0.15	-0.04
Bogue Phalia	7.01	-2.51	0.94	Largemouth bass cpue	-0.18	-0.13	-0.40
Brushy Creek	-0.25	-0.95	-0.79	Spotted bass cpue	-0.25	-0.35	0.08
Horse Creek	0.39	1.88	-1.62	Longear sunfish cpue	-0.17	0.31	0.07
Unknown Trib.	2.07	-2.13	-0.47	Bluegill sunfish cpue	-0.24	-0.19	0.40
Cypress Creek	0.46	0.05	-2.12	% Total basses	-0.32	-0.06	-0.24
L. Bogue Homo	-0.42	-0.03	-0.43	% Total sunfishes	-0.18	0.35	0.15
W. Hobolochitto Cr.	-2.17	0.95	0.64	% Largemout bass	-0.17	0.19	-0.45
Catalpa Creek	-2.70	-3.20	-2.10	% Spotted bass	-0.20	-0.34	0.12
Trim Cane Creek	-0.31	2.92	3.00	% Longear sunfish	-0.05	0.45	0.08
				% Bluegill sunfish	-0.22	0.05	0.37
				Mean length total bass	-0.34	0.17	-0.13
				Mean length total sunfish	-0.34	0.17	-0.10
				Mean length age-1 bass	-0.20	-0.05	0.32
				Mean length age-1 sunfish	-0.10	0.35	0.30

[1] Total bass=largemouth bass and spotted bass combined
Total sunfish=longear, bluegill, green, and redear sunfish combined

tion of holding areas such as pools and eddies (Chiasson 1996; Jones et al. 1999; Zimmerman et al. 2003). In contrast, riparian zone recovery, or return time, has improved habitat availability for catchable salmonids in California wadeable streams. For example, Opperman and Merenlender (2004) compared salmonid habitat parameters between clear-cut reaches (control) and reaches that were naturally reforested from clear-cutting. The reforested sites were historically clear-cut for timber harvest, but in 1982 and 1991 they were bought by private landowners. The landowners then built large fences around these stream reaches for hunting. These experimental exclosures made the channels narrower and deeper, lowered maximum water temperature by 1 °C on average, and provided large woody debris that helped to create pools.

Geographic location and riparian land cover influence centrarchid fisheries resources in wadeable streams from Mississippi. We found that a gradient representing centrarchid abundance and size increased in response to decreases in latitude and increases in percent cover of forest in 30 m wide riparian corridors. Our model should be tested with independent data before it can be used to predict abundances and sizes of centrarchids from wadeable streams in Mississippi. However, the model provides guidance towards the future management of a wadeable stream fishery in Mississippi. If the model is validated, it can be used to locate candidate streams in Mississippi that potentially support centrarchid fisheries resources.

We focused primarily on density-independent factors that influence sport fishes in Mississippi wadeable streams. We assumed density-dependent factors (predation, competition, angler harvest, etc.) did not influence our catch characteristics. Additionally, we assumed that temporal variation in angling did not affect our results. Only two of the reaches in our study were fished by anglers other than our research team (Brushy Creek and L. Bogue Homo Creek), but these anglers fished only the area immediately around access points. During each successive fish sample throughout our study, fish were always caught in the same portion of a reach. Therefore, temporal variation in angling was negligible.

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Corresponding Author:
John B. Alford
Department of Wildlife and Fisheries
Mississippi State University
Box 9690
Mississippi State, MS, 39762
Phone 662–497–1026
Fax 662–325–8726
E–mail: jba81@msstate.edu

Runoff Losses of Pesticides and a Conservative Tracer from Warm–season Turf using Simulated Rainfall

P. A. Ampim[1], J. H. Massey[1], B. R. Stewart[1], M. C. Smith[1], A. B. Johnson[2],
R. P. Maiers[1] and A. A. Andrews[1]

This study is part of a larger national research effort designed to improve the understanding and modeling of turf pesticide runoff. The specific objectives of the project are to investigate the effects of warm–season turfgrass species, mowing height and plot size on pesticide runoff. The turfgrass species include bermudagrass (*Cynodon dactylon* [L] Pers. x *Cynodon transvalensis* Burtt–Davy) and zoygrass (*Zoysia japonica* Steud.). The turfgrass species were maintained as either golf course fairways or residential lawns. The runoff studies were conducted on eighteen 3.7 x 9.1 m and four 6.1 x 24.4 m plots. These plots were arranged in a split plot design and were sloped at 3 % with minimal cross slope. Following a standardized field protocol, 2, 4–D herbicide, flutolanil fungicide, and chlorpyrifos insecticide were co–applied at 1.12 kg ai/ha, 2.24 kg ai/ha and 2.24 kg ai/ha respectively. A conservative tracer, KBr, was also applied at 10 kg/ha immediately before initiation of simulated rainfall. Simulated rainfall was applied to the plots at a rate of 38 mm/h for 90 min. Runoff water from the plots was collected at approximately five–minute intervals. The runoff and application monitor samples were analyzed by reverse phase High Performance Liquid Chromatography (HPLC) using UV–Vis detection. Maximum observed concentrations (ppb), total masses (g) and percentages of applied chemicals observed in runoff water were determined.

Keywords: Model, non–point source pollution, water quality

Corresponding Authors:
[1]Mississippi State University, Mississippi State, MS
[2]Alcorn State University, Lorman, MS

Modeling the Big Black River: Evaluation of a Simplistic Water Quality Model

Kimberly S. Caviness, Garey A. Fox, Patrick N. Deliman

The Mississippi Department of Environmental Quality (MDEQ) uses the Steady Riverine Environmental Assessment Model (STREAM) to establish permitted effluent limitations for industrial, commercial, and municipal facilities. While the U.S. Environmental Protection Agency (EPA) has approved of its use, questions arise regarding the model's simplicity. This research first evaluated STREAM using a statistical evaluation procedure based on sensitivity analyses, input probability distribution functions, and Monte Carlo simulation with site-specific data from a 46-mile reach of the Big Black River in central Mississippi. STREAM reasonably predicted dissolved oxygen (DO) based on a comparison of output probability distributions with observed DO. The observed DO was consistently within 80% confidence intervals of model predictions. This research also evaluated STREAM by comparing observed DO with predictions by both STREAM and the Enhanced Stream Water Quality Model (QUAL2E). One version of the QUAL2E and STREAM models utilized site-specific input data. A second version of each model involved additional calibration. A third version of STREAM was an uncalibrated model developed following MDEQ Regulations (1995) for cases where intensive input data are unavailable. All versions of the models were simulated at the 7Q10 flow for the Big Black River, the minimum flow expected for seven consecutive days during a period of ten years. STREAM over predicted while QUAL2E under predicted DO with the site-specific input data. Percent errors ranged between 4.8% and 11.2% for STREAM and 3.3% and 5.1% for QUAL2E. The uncalibrated STREAM model predicted the lowest DO for all scenarios and correspondingly provided the most conservative DO predictions.

Keywords. Dissolved Oxygen, Numerical Modeling, Rivers/Streams, Total Maximum Daily Load, Wastewater Discharge, Water Quality Modeling.

Introduction and Background

Water quality management attempts to protect of the uses of a water body while using the water as an economical means of waste disposal. The amount of waste a water body can assimilate depends on numerous factors (McBride, 2002; Mohamed et al., 2002; de Azevedo et al., 2000; Somlyódy et al., 1998). The water quality parameter of concern for waste load allocations (WLAs) and water quality based effluent limitations (WQBEL) is often the instream concentration of dissolved oxygen (DO). The Mississippi state standard for DO, as defined in the State of Mississippi Water Quality Criteria for Intrastate, Interstate, and Coastal Waters (2003), is a daily average concentration of at least 5.0 mg/L with an instantaneous minimum of no less than 4.0 mg/L. In order to sufficiently protect water quality, these standards must be attained at the low-flow, critical condition. Mississippi Department of Environmental Quality (MDEQ) regulations (1995) define the low-flow, critical condition as the 7Q10 flow: the minimum flow that occurs for seven consecutive days during a period of ten years.

The most commonly utilized model for predicting the impact of discharges on DO is the Enhanced Stream Water Quality Model (QUAL2E) (Bowen and Hieronymus, 2003; Chaudhury et al., 1998; Lung, 1998; Melching and Yoon, 1996; Walton and Webb, 1994). The QUAL2E model is a comprehensive and versatile water quality model widely used for WLAs, discharge permit determinations, and other conventional pollutant evaluations (Bowen and Hieronymus, 2003; Brown and Barnwell, 1987; Chaudhury et al., 1998; Lung, 1998; Melching and Yoon, 1996; Walton and Webb, 1994). It can simulate up to 15 water quality constituents in any combination desired by the user.

In lieu of the complexities of QUAL2E, MDEQ currently uses the Steady Riverine Environmental Assessment Model (STREAM) to establish permitted effluent limitations (MDEQ, 2004). STREAM is a simplistic, steady state, daily average, water quality model that utilizes a modified Streeter-Phelps (1923) DO sag equation. STREAM simulates DO, biochemical oxygen demand, and ammonia nitrogen concentrations. STREAM models DO as a function of the predicted dissolved oxygen saturation concentration (DOsat):

$$DO_{sat} = 14.652 + \left[-0.41022 + \left(0.007991 - 7.7774 \times 10^{-5} T\right) T\right] T \qquad (1)$$

where T is the temperature in degrees Celsius (Elmore and Hayes, 1987). The model solves for the steady state DO deficit (Di) concentration along a river reach using the following equation:

$$\begin{aligned} D_i = {} & D_{i-1} \exp\left(-k_a \frac{\Delta x}{U}\right) + \left[\frac{k_d}{k_a - k_r}\left\{\exp\left(-k_r \frac{\Delta x}{U}\right) - \exp\left(-k_a \frac{\Delta x}{U}\right)\right\}\right] L_{i-1} \\ & + \frac{4.57 k_n}{k_a - k_n}\left[\exp\left(-k_n \frac{\Delta x}{U}\right) - \exp\left(-k_a \frac{\Delta x}{U}\right)\right] N_{i-1} - \left[1 - \exp\left(-k_a \frac{\Delta x}{U}\right)\right] \frac{P}{k_a} \\ & + \left[1 - \exp\left(-k_a \frac{\Delta x}{U}\right)\right] \frac{R}{k_a} + \left[1 - \exp\left(-k_a \frac{\Delta x}{U}\right)\right] \frac{S}{k_a} \end{aligned} \qquad (2)$$

where D_{i-1} is the dissolved oxygen deficit concentration in the upstream stream reach (mg/L), k_a is the reaeration (McBride, 2002) rate (d^{-1}), Δx is the computational element distance (mile), U is the reach velocity (fps), k_d is the carbonaceous biochemical oxygen demand (CBOD) decay rate (d^{-1}), k_r is the overall CBOD removal rate (d^{-1}), L_{i-1} is the CBOD concentration in the upstream stream reach (mg/L), k_n is the nitrification rate (d^{-1}), N_{i-1} is the ammonia-nitrogen (NH_3-N) concentration in the upstream stream reach (mg/1), P is the photosynthesis rate (mg/L/d), R is the respiration rate (mg/L/d), and S is the sediment oxygen demand (Hatcher, 1986) rate (mg/L/d).

The major differences between QUAL2E model and STREAM are that QUAL2E simulates the complete nitrogen cycle, the complete phosphorus cycle, and growth cycle of algae. The growth cycle of algae is directly influenced by the concentrations of nitrogen and phosphorus. STREAM does not simulate phosphorus, and the only form of nitrogen simulated is ammonia nitrogen. Within STREAM, the user has the option of entering a value for the photosynthetic production of oxygen, oxygen utilized by aquatic plants through respiration, and community substrate oxygen demand. However, these values are constant within the model and do not respond to changes in the ammonia nitrogen concentrations.

Questions arise regarding the simplicity of STREAM in comparison to more commonly used models such as QUAL2E. Minimal analysis has been performed on STREAM to assess the level of confidence associated with model predictions, especially with respect to more commonly utilized water quality models. The objectives of this research were to evaluate STREAM using a statistical procedure that converts input parameter uncertainty into output prediction uncertainty and to compare STREAM predictions to predictions made with the more commonly utilized QUAL2E model. Evaluation of STREAM and other hydrologic and water quality models have been based on comparison of a model prediction with a single observation for a specific location (Sabbagh and Fox, 1999). However, modelers rarely know input parameters with exact certainty (Haan et al., 1995; Parker et al., 1995). An alternative evaluation strategy being increasingly used in hydrologic and water quality modeling involves uncertainty and probability analyses. The procedure is valid when input parameters are represented by singular values. Researchers have applied this statistical evaluation to a number of different models (Haan et al., 1995; Haan and Zhang, 1996; Prabhu, 1995; Zhang et al., 1995; Sabbagh and Fox, 1999; Haan and Skaggs, 2003) but not in-stream water quality models.

Big Black River Study Site

MDEQ received a request in 2001 to perform a WLA for a new National Pollutant Discharge Elimination System (NPDES) permitted facility in Madison County, Mississippi. This facility, known as the Canton Municipal Utilities Beattie's Bluff Wastewater Treatment Facility (CMU), planned to locate near the city of Canton. CMU was designed to treat wastewater from several local sources including a new Nissan facility that was under construction at that time. CMU proposed two discharge scenarios into the Big Black River: 4.0 and 8.0 million gallons per day (MGD). MDEQ's Office of Pollution Control, Water Quality Assessment Branch completed the WLA for the CMU project. The WLA established WQBELs for CMU to discharge treated wastewater into the Big Black River. MDEQ used an application of STREAM with very limited site-specific data to determine the discharge limitations for the proposed discharge scenarios. They assembled STREAM in accordance with MDEQ Regulations (1995). This application, like many WLA applications of STREAM, had no site-specific field data available for model input or calibration. In the absence of field data, MDEQ Regulations (1995) specified the methods and assumptions for the input data.

WQBELs were assigned to CMU for each discharge scenario. At the proposed flow of 4.0 MGD, MDEQ granted the facility permit limits of a monthly average of 22.0 mg/L 5-Day carbonaceous biochemical oxygen demand (CBOD5) and 2.0 mg/L ammonia nitrogen. At the proposed flow of 8.0 MGD, MDEQ granted the facility permit limits of a monthly average of 11.0 mg/L CBOD5 and 2.0 mg/L ammonia nitrogen. STREAM predicted no associated DO problems with the WQBELs set for each discharge scenario. However, MDEQ and U.S. Environmental Protection Agency (EPA) Region 4 proposed that an intensive WLA study be conducted within the Big Black River. The intensive study was conducted for one week in September 2002 during low flow conditions. A 46-mile segment of the Big Black River and its major tributaries were selected for intensive study. Water quality and hydraulic data were collected at multiple locations along the Big Black River and its major tributaries in order to develop a calibrated model of the system at low flow conditions.

Figure 1 shows the study area and all sampling locations including the main stem, tributaries, and wastewater treatment plants. The proposed CMU facility discharge location was at river mile 123. Seven monitoring stations were established within this segment of the Big Black River. Additional monitoring stations were also established within the major tributaries to the river including Pepper Creek, Bear Creek, Panther Creek, Cypress Creek, and Bogue Chitto Creek. The intensive study also sampled the effluent from five existing NPDES WWTPs in the area. These WWTPs discharge into the tributaries not the Big Black River itself. The samples taken from the wastewater treatment plants were analyzed for ultimate carbonaceous biochemical oxygen demand (CBODU), ammonia nitrogen, nitrate nitrogen, nitrite nitrogen, total kjeldahl nitrogen (TKN), total nitrogen, total dissolved phosphorus, total phosphorus, and total organic carbon (TOC). The data collected at the Big Black River and tributary water quality monitoring stations included DO, community oxygen metabolism, oxygen production and respiration, reaeration measurements, water quality, physiographic measurements, meteorologic measurements, time-of-travel and other hydraulic data.

Figure 1. Big Black River Study Area – September 2002

METHODOLOGY

Statistical Evaluation

This research used a five-step statistical evaluation procedure based on Monte Carlo simulation. First, a sensitivity analysis was conducted on STREAM to determine the input parameters with the greatest impact on model predictions. The sensitivity of each parameter was determined by holding all other parameters constant while small (i.e., ±10%) variations were made to the parameter of interest. The second step developed a probability distribution function (PDF) for each parameter based on available data for the southeastern United States. A combination of chi-square goodness of fit and Kolmogorov-Smirnov tests (Haan, 1991) determined best-fit distributions. The third step generated unique PDFs for the sensitive input parameters based on observed data from the Big Black River intensive study. The intensive study collected six to seven measurements of sensitive model parameters over the 46-mile reach. This measured data allowed the derivation of a unique PDF for each parameter. The type of distribution was constrained to be that determined for the larger datasets (i.e., throughout the southeastern United States) mentioned above, but the parameters of the distribution were allowed to vary.

All available data regarding the hydraulics of the river as well as all discharges into the river were entered into the model. Model predictions were verified to ensure accurate flow predictions in the Big Black River before continuing with the water quality modeling. Output probability distributions of predicted DO were generated using Monte Carlo simulation (Cheney and Kincaid, 1994). Values for each of the sensitive parameters were derived by applying a random number to each unique probability distribution generated based on observed data from the study. As a conservative approach, each sensitive parameter was assumed constant along the entire reach. The Monte Carlo simulation included 1000 model runs generating 1000 output values for DO at each river mile. The final step used the output probability distributions to assess the model using 80% confidence intervals. Confidence intervals were assumed symmetric with respect to probability (Haan and Skaggs, 2003). Therefore, the 10% and 90% quantiles from an empirical probability plot of the output estimated the 80% confidence interval.

Comparison of Water Quality Models

This research then developed a more detailed, calibrated, model of the river segment using QUAL2E and data from the intensive study. The QUAL2E model of the Big Black River was developed as a steady-state simulation. The model was calibrated to simulate the conditions measured at the time of the study. The model simulated temperature, DO concentrations, BOD concentrations, the phosphorus cycle, the nitrogen cycle, and algae as chlorophyll a. Two versions of QUAL2E (QUALMOD1 and QUALMOD2) were assembled. One version used all input parameters exactly as measured from the study. The second version used all input parameters as measured with the exception of the community substrate oxygen demand. In-situ measurements of this parameter can be quite variable exhibiting standard deviations anywhere from 0.2% to 150% with an average standard deviation of 44% (Hatcher, 1986). The measured values of community substrate oxygen demand were reduced up to 35% within QUAL2E to provide a better calibration to the observed DO data.

In addition to the complex QUAL2E models, this research also developed three versions of STREAM (STREAM1, STREAM2, and STREAM3). Data collected from the intensive study was used to complete a steady state, calibrated STREAM model. STREAM1 utilized all input parameters exactly as measured from the study. STREAM2 used all input parameters as measured with the exception of the community substrate oxygen demand. The community substrate oxygen demand was increased up to 44% to provide a better calibration to the observed DO. This research assembled a third version of STREAM as if no site-specific data were available. This uncalibrated version used assumptions given in MDEQ regulations (1995) to determine model inputs. According to regulations, stream flow should be assumed the 7Q10 flow based on nearby USGS flow gages and drainage areas, and water temperature should be assumed based on flow: for streams with minimum low flow greater than 8.5 $m^3\ s^{-1}$, the summer temperature should be assumed 30°C and the winter temperature should be assumed 20°C; for streams with minimum low flow greater than 1.4 $m^3\ s^{-1}$ and less than 8.5 $m^3\ s^{-1}$, the summer temperature should be assumed 28°C and winter temperature assumed 20°C; and for streams with minimum low flows less than 1.4 $m^3\ s^{-1}$, the summer temperature should be 26°C and winter temperature should be 20°C. The regulations also provide guidance on estimating velocities, background conditions, and other parameters when site specific data are not available. The models predicted DO under the

7Q10 scenario, which is the low-flow critical condition according to MDEQ Regulations (1995). In all cases, the models analyzed the two discharge scenarios of 4.0 MGD and 8.0 MGD.

RESULTS AND DISCUSSION

Statistical Evaluation

Four parameters had the most significant influence on predicted DO concentrations: (1) the reaeration coefficient, (2) photosynthetic oxygen production, (3) oxygen utilized by aquatic plants through respiration, and (4) oxygen demand of bottom deposits, otherwise known as community substrate oxygen demand or sediment oxygen demand (SOD). Measured values of the reaeration coefficient were available for numerous studies occurring in the southeastern United States from 1991 to 1999 (Koenig, 2004). This data include over sixty measurements of the reaeration coefficients (d^{-1}) at the reference 20°C temperature for stream reaches in Alabama, Georgia, Tennessee, North Carolina, and South Carolina. Reaeration coefficients ranged between 0.02 and 19.29 d^{-1} with an average of 3.05 d^{-1}. Chi-square and Kolmogorov-Smirnov statistics indicated that the measured reaeration coefficients conformed to a Weibull distribution with a shape parameter (α) = 0.88 and a scale factor (β) = 3.44 (Devore, 1995).

The use of a Weibull distribution is uncommon in many uncertainty analyses that assume the distributions to be either normal or lognormal (Gibbons, 2003). Scientists commonly assume environmental data to be lognormally distributed. No theoretical justification exists for the use of the Weibull distribution in this research other than the fact that the Weibull distribution simply provided an improved fit to the observed data. Figure 2 illustrates the measured reaeration coefficient data along with the best-fit Weibull cumulative distribution function (CDF). The lognormal CDF is also shown for comparison.

A similar dataset was available for in-situ chamber measurements of the SOD. Measured values of SOD were available for numerous sites in the southeastern United States from 1977 to 2001 (Koenig, 2004). Available data included over 100 measurements in streams in Alabama, Florida, Georgia, Kentucky, Mississippi, North Carolina, South Carolina, and Tennessee. SOD ranged between 0.11 and 7.90 g $O_2/m^2/d$ with an average of 1.81 g $O_2/m^2/d$. Measured SOD conformed to a lognormal distribution with a mean (μ) = 0.51 and a standard deviation (σ) = 0.94 (Devore, 1995).

Limited information was available for photosynthetic oxygen production and oxygen used through respiration. This research identified approximately 20 measurements of both variables in numerous case studies. Oxygen production rates at 20°C ranged between 0.64 and 18.65 g $O_2/m^2/d$ with an average of 7.89 g $O_2/m^2/d$. Respiration rates at 20°C ranged between 2.70 and

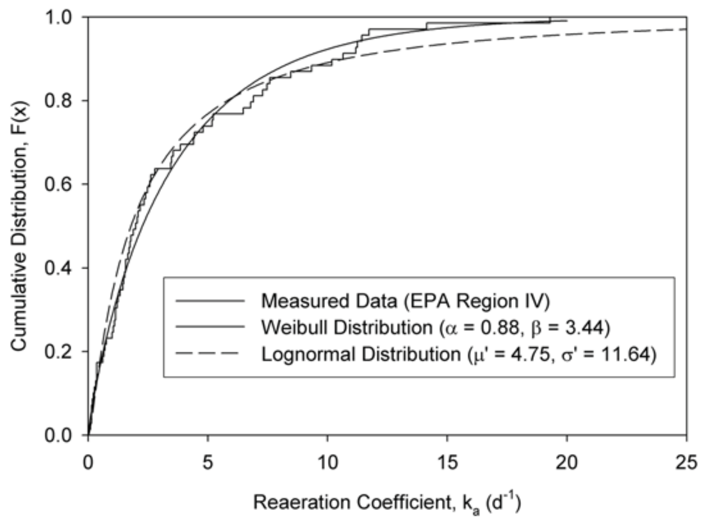

Figure 2. Measured reaeration coefficients from studies occurring in the southeastern United States from 1991 to 1999 and best-fit Weibull and Lognormal distributions.

30.70 g $O_2/m^2/d$ with an average of 10.36 g $O_2/m^2/d$. This limited information suggested a Weibull distribution with α = 1.47 and β = 8.72 for the photosynthetic oxygen production rate and a lognormal distribution with μ = 2.16 and σ = 0.78 for the oxygen utilized by aquatic plants through respiration.

The Big Black River intensive study included measurements of the four sensitive parameters. The intensive study included measured production and respiration rates at six locations along the 46-mile study reach with a range of 0.11 to 4.07 g $O_2/m^2/d$ and 0.04 to 2.72 g $O_2/m^2/d$, respectively. Reaeration coefficients measured at six locations ranged between 0.95 and 1.65 d^{-1}. SOD was measured at seven locations ranged between 0.80 and 1.2 g $O_2/m^2/d$. These measurements resulted in site-specific PDFs, assuming the representative distributions for each parameter as determined from the southeastern U.S. data. All distributions were representative of the range of values measured in the intensive study.

The DO concentrations measured at six locations (river miles 123, 115, 108, 105, 96, and 88) were used to evaluate STREAM. Figure 3 illustrates an example of the probability distribution results from the Monte Carlo simulations for river miles 108. Table 1 summarizes results from the statistical procedure for the rivers miles with measured DO concentrations. At each location the observed DO concentration fell within the 80% confidence intervals indicating that STREAM reasonably predicted DO at each river mile. Although the observed DO fell within the 80% confidence intervals at all river miles, STREAM over predicted DO. The observed DO at each river mile was generally less than the 25th percentile on the probability plot.

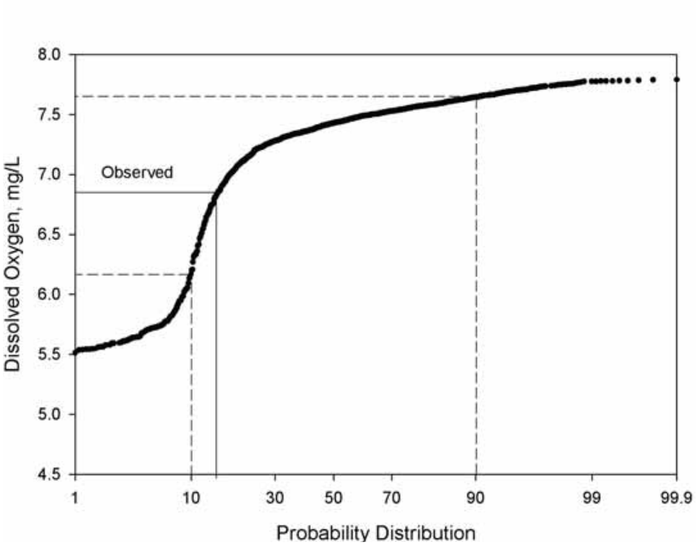

Figure 3. Comparison of observed DO concentration at river mile 108 versus the 80% confidence intervals on the distribution of predicted DO from the Monte Carlo simulations.

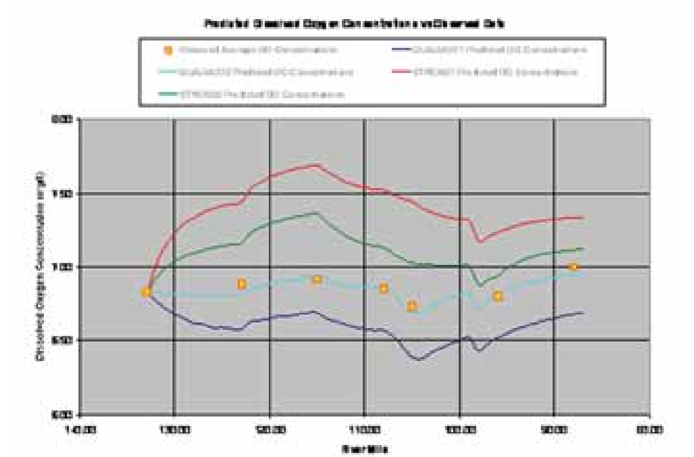

Figure 4. Observed versus predicted DO concentrations for river miles 132 to 88 along the Big Black River under study conditions.

Comparison of Water Quality Models

QUALMOD1, QUALMOD2, STREAM1, and STREAM2 simulated conditions along river miles 133 through 87. Verification of hydrologic predictions ensured that the models were properly simulating observed flows. Figure 4 compares observed DO to predicted DO using QUALMOD1, QUALMOD2, STREAM1, and STREAM2 at study conditions. Applying the models with detailed site-specific input data resulted in percent errors between predicted and observed DO concentrations ranging between 4.8% and 11.2% for STREAM and between 3.3% and 5.1% for QUAL2E. After calibration by adjusting the site-specific measured value of SOD within reported ranges, STREAM had percent errors in the range of 1.7% to 6.5% compared to percent errors of 0.1% to 1.0% for the calibrated QUAL2E model. QUALMOD1 and QUALMOD2 both predicted DO concentrations less than the concentrations measured in the field. STREAM1 and STREAM2 both predicted DO concentrations greater than the concentrations measured in the field. Modified QUALMOD1, QUALMOD2, STREAM1, and STREAM2 predicted DO under 7Q10 flow conditions with the addition of the CMU facility. The uncalibrated version of STREAM (STREAM3) also simulated this scenario. The 7Q10 flow was estimated to be 1.7 m^3 s^{-1} based on historical flow data from United States Geological Survey (USGS) Station 07289730 near Bentonia, MS and USGS Station 07290000 near Bovina, MS. Equivalent predicted flow verified the hydrologic component of the models. Figures 5 and 6 along with Table 2 summarize the lowest simulated DO and DO sag concentrations for each model simulation. The DO concentrations predicted by the QUALMOD1 and QUALMOD2 applications were lower than the DO concentrations predicted by the STREAM1 and STREAM2 applications. The DO concentrations estimated by STREAM3 were the lowest of all simulations indicating that use of MDEQ regulations (1995) produced conservative

Table 1. Summary of statistical evaluation of STREAM for predicting DO concentrations at locations along the 46-mile Big Black River intensive study.

	River Mile 123	River Mile 115	River Mile 108	River Mile 105	River Mile 96	River Mile 88
Observed DO Concentration (mg/L)	6.88	6.92	6.85	6.73	6.80	7.00
Predicted DO - Average of 1000 Simulations (mg/L)	7.22	7.30	7.22	7.24	7.10	7.10
Predicted DO - Minimum (mg/L)	4.96	4.86	4.79	4.80	4.72	4.71
Predicted DO - Maximum (mg/L)	7.70	7.84	7.79	7.81	7.61	7.62
80% Confidence Level (mg/L)	6.20–7.60	6.24–7.70	6.17–7.64	6.18–7.66	6.08–7.48	6.08–7.49

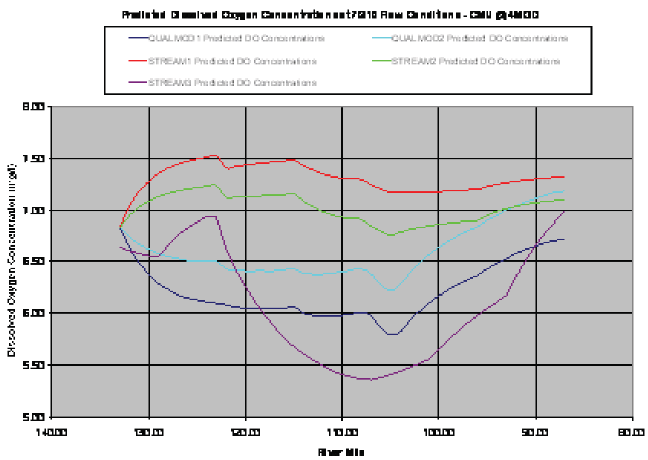

Figure 5. Predicted DO concentrations versus river mile-QUAL2E and STREAM simulations at 7Q10 flow and CMU facility at 4.0 MGD.

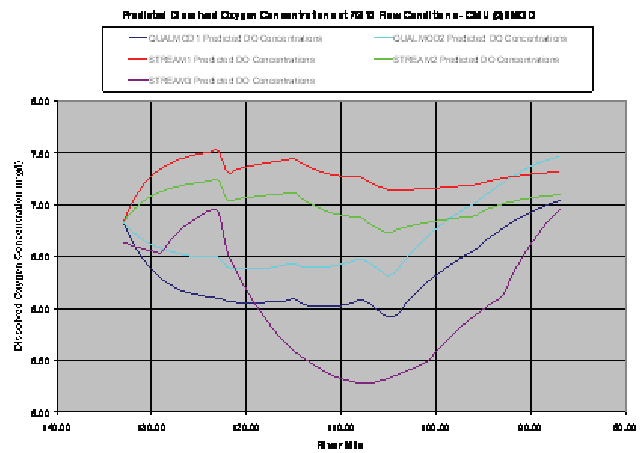

Figure 6. Predicted DO concentrations versus river mile-QUAL2E and STREAM simulations at 7Q10 flow and CMU facility at 8.0 MGD.

predictions for this segment of the Big Black River. All model applications of QUAL2E and STREAM confirmed the WQBELs assigned to the CMU facility. The models predicted no violations of the daily average DO standard of 5.0 mg/L.

SUMMARY AND CONCLUSIONS

MDEQ currently uses a simplistic water quality model for determining effluent limitations. This model, called STREAM, has been approved for MDEQ's use by EPA. However, concerns arise regarding the simplicity of the model and the lack of a detailed analysis of its predictive capability. STREAM was evaluated using a statistical evaluation technique based on sensitivity analysis, PDFs for input parameters, and Monte Carlo simulation using data from an intensive study along a 46-mile reach of the Big Black River in central Mississippi. The most sensitive input parameters in STREAM were the reaeration coefficient, photosynthesis and respiration rates, and SOD. Using datasets from the EPA's Region IV Science and Ecosystem Support Division and other studies, governing distributions (Weibull or lognormal) were developed for the sensitive STREAM parameters. Site-specific distributions were then generated for the sensitive parameters using data collected during an intensive study along the Big Black River. These distributions were then incorporated through Monte Carlo simulation. The Monte Carlo simulations allowed the development of output distributions for predicted DO at each river mile. Comparison of the observed DO concentrations with the confidence intervals on the model predicted distributions suggested that the model reasonably predicted DO. The model did tend to over estimate DO, which users of STREAM need to realize when using the model for defining effluent limitations. A second objective of this research was to evaluate the performance of STREAM in comparison to the more commonly utilized and complex water quality model, QUAL2E. Model evaluation

Table 2. Lowest predicted DO and DO sag concentrations for the 7Q10 flow.

Simulation	CMU Scenario	Lowest DO (River Mile)	DO Sag (River Mile)
QUALMOD1	4.0 MGD	5.79 mg/L (105)	5.79 mg/L (105)
QUALMOD1	8.0 MGD	5.92 mg/L (105)	5.92 mg/L (105)
QUALMOD2	4.0 MGD	6.22 mg/L (105)	6.22 mg/L (105)
QUALMOD2	8.0 MGD	6.32 mg/L (105)	6.32 mg/L (105)
STREAM1	4.0 MGD	6.83 mg/L (133)	7.17 mg/L (101-105)
STREAM1	8.0 MGD	6.83 mg/L (133)	7.15 mg/L (102-105)
STREAM2	4.0 MGD	6.76 mg/L (105)	6.76 mg/L (105)
STREAM2	8.0 MGD	6.73 mg/L (105)	6.73 mg/L (105)
STREAM3	4.0 MGD	5.76 mg/L (107)	5.76 mg/L (107)
STREAM3	8.0 MGD	5.28 mg/L (107-108)	5.28 mg/L (107-108)

involved comparing two versions of QUAL2E with three versions of STREAM for the Big Black River case study. The first versions of QUAL2E and STREAM utilized intensive field measurements collected on site. The second versions of QUAL2E and STREAM involved calibration beyond the intensive field collected data. A third version of STREAM involved parameters suggested by MDEQ regulations in cases where site-specific field data is unavailable. These models simulated the 7Q10 flow scenario with discharges of 4.0 MGD and 8.0 MGD from a wastewater treatment facility. Although the QUAL2E model is a much more sophisticated model that simulates more in stream processes than STREAM, comparison of the STREAM and QUAL2E models indicated that the models produced similar predictions for instream DO. The DO concentrations predicted by the calibrated STREAM simulations were consistently higher than those predicted by the calibrated QUAL2E simulations. The uncalibrated version of STREAM predicted the lowest DO concentrations of all the simulations; therefore, the uncalibrated model produced the most conservative estimates of waste load allocation.

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Corresponding Authors:
Kimberly S. Caviness, P.E.
Environmental Engineer
Mississippi Department of Environmental Quality
Office of Pollution Control
TMDL / WLA Branch

Garey A. Fox, Ph.D.
Assistant Professor
University of Mississippi
Department of Civil Engineering

Patrick N. Deliman
Environmental Engineering Branch Chief
U.S. Army Corps of Engineers
USAE Research and Development Center

Greenhouse Modeling of Nitrogen Use Efficiency in Two Wetland *Cyperus* Species at the University of Mississippi Field Station

Alison A. Faulkner[1], Marjorie M. Holland[2], M.T. Moore[3] and C.M. Cooper[3]

Emergent wetland perennials are an effective component of wastewater treatment wetlands. Members of family Cyperaceae such as *Scirpus*, *Carex*, and *Eleocharis* are commonly used to treat nonpoint source contaminants. However, scientific literature regarding the use of the genus *Cyperus* as potential wastewater treatment species focuses on the C_4 photosynthetic types, which comprise ~80% of this genus. The C_4 pathway in *Cyperus* is evidently an adaptation to temperate wetlands and sandy, infertile environments (Li, et al., 1999). *Cyperus* species using the C_4 pathway have a high photosynthetic nitrogen use efficiency (NUE), which appears to confer a high degree of success in wetland environments with low nitrogen concentrations. This translates to a possible competitive advantage over their C_3 *Cyperus* counterparts where the latter conditions exist.

Our six–month greenhouse experiment, which began in early December 2005, was designed to quantify and differentiate nitrogen use efficiency in two facultative wetland species: *Cyperus haspan*, a C_3 sedge, and *Cyperus strigosus*, a C_4 sedge. Both species co–occur in shallow wetlands and ditches at the University of Mississippi Field Station in Lafayette CO., MS; this situation presented the opportunity to determine each species' response to long–term nitrogen dosing. Each species was subjected to nitrogen dosing regimens of both 2.5 ppm and 4.0 ppm, representing typical lower and higher nitrogen concentrations in agricultural runoff in Lafayette Co. Our expectations are that (1) *Cyperus strigosus* may have higher above– and belowground biomass in the low (2.5 ppm) nitrogen treatments than *C. haspan* at the same dose; (2) however, *C. haspan* may display higher above– and belowground biomass in the high (4.0 ppm) nitrogen treatments than *C. strigosus* at that dose level, and (3) photosynthetic pathway (C_3 versus C_4) may differentially affect the abilities of these two sedge species to sequester nitrogen in their tissues. The combination of both C_3 and C_4 sedge species planted in wastewater treatment wetlands and agricultural drainage ditches may be a more effective method of partially treating fluctuating levels of nonpoint source pollution than using either of the two types singly, since C_3 sedges may function to remove nitrogen at higher concentrations while C_4 sedges may be able to remove nitrogen more efficiently at lower concentrations.

Keywords: Nitrate contamination, nonpoint source pollution, wastewater, wetlands

Introduction

Nitrogen is frequently a component in nonpoint source pollution from agricultural activity (Cooper and Moore, 2003). Ditches and downstream wetlands intercept nonpoint source pollution, where emergent perennials serve to slow water flow and promote transformation of nitrogen via nitrification, denitrification, and volatilization (Cronk and Fennessey, 2001). The denitrification process converts organic nitrogen to NO_2 or NO_{3-}; NO_{3-} and NH_4+ are inorganic forms of nitrogen readily available for plant uptake by both aboveground and belowground structures (Larcher, 1995). Common emergent perennials found in Southeastern wetlands include members of Families Cyperaceae, Poaceae, and Juncaceae. Scientific literature regarding the use of the genus *Cyperus* as potential wastewater treatment species focuses on the C_4 photosynthetic types, which comprise ~80% of this genus. The C_4 pathway in *Cyperus* is evidently an adaptation to temperate wetlands and sandy, infertile environments (Li, Wedin, and Tieszen, 1999). *Cyperus* species using the C_4 pathway have a high photosynthetic nitrogen use efficiency (NUE), which appears to confer a high degree of success in wetland environments with low nitrogen concentrations. This translates to a possible competitive advantage over their C_3 *Cyperus* counterparts where the latter conditions exist. Our six-month greenhouse experiment was designed to quantify and differentiate nitrogen use efficiency in two facultative wetland species: *Cyperus haspan*, a C_3 sedge, and *Cyperus erythrorhizos*, a C_4 sedge. Both species co-occur in shallow wetlands and ditches at the University of Mississippi Field Station (UMFS) in Lafayette CO., MS; this situation presented the opportunity to determine each species' response to long-term nitrogen dosing. Each species was subjected to nitrogen dosing regimens of both 2.5 ppm and 4.0 ppm, representing typical lower and higher nitrogen concentrations in agricultural runoff in Lafayette Co. Our expectations are that (1) *C. erythrorhizos* may have higher above- and belowground biomass in the low (2.5 ppm) nitrogen treatments than *C. haspan* at the same dose; (2) however, *C. haspan* may display higher above- and belowground biomass in the high (4.0 ppm) nitrogen treatments than *C. erythrorhizos* at that dose level, and (3) photosynthetic pathway (C_3 versus C_4) may differentially affect the abilities of these two sedge species to sequester nitrogen in their tissues. The combination of both C_3 and C_4 sedge species planted in wastewater treatment wetlands and agricultural drainage ditches may be a more effective method of partially treating fluctuating levels of nonpoint source pollution than using either of the two types singly, since C_3 sedges may function to remove nitrogen at higher concentrations while C_4 sedges may be able to remove nitrogen more efficiently at lower concentrations.

Materials and Methods

Forty-five 55-gallon split barrels were filled with approximately 12 gallons of clay/sand sediment obtained from UMFS property. Drums were filled to 4 inches above sediment surface with groundwater from UMFS and allowed to stabilize and become anoxic for approximately two weeks. An outlet for overflow was also installed on each barrel at 4 inches above the sediment surface to keep water level constant. Interstitial (pore) water sampling wells were installed to 8 cm below the sediment surface in order to collect pore water samples. *Cyperus haspan* and *Cyperus erythrorhizos* specimens were collected from wetlands at UMFS and ten individuals of each species were transplanted to designated containers and allowed to acclimate for a month. Pretreatment data (plant height, surface water, soil, interstitial pore water, pH, temperature, humidity, and daylight hours) were collected, measured and analyzed at the beginning of January 2006 to establish a baseline with which to compare the subsequent months when nitrogen dosing is taking place. Five-gallon Aquadosers™ were set up to deliver concentrations of either 0, 2.5, or 4.0 ppm ammonium nitrate (NH_4NO_3) at a rate of 3.024 L/day. Dosing began 10 January 2006. Data are being collected every 30 days for 180 days.

Results and Discussion

Figures 1A & 1B show mean $NH_{3-}N$ and NO_{3-} values for surface water, by treatment, from January to April 2006. As expected, $NH_{3-}N$ levels remain relatively low (>1 mg/L) in the surface water where nitrifying bacteria are absent. The increase in NO_{3-} concentration is reflective of nitrogen dosing over time as the systems receive more ammonium nitrate than they can process at this point in time. Figures 2A & 2B show the same analysis on soil extracts. Both inorganic NH_{4+} and NO_{3-} concentrations in soil are generally low compared with organic soil N. Since N cycles tend to stabilize and move towards a steady state, concentrations of NH_{4+} and NO_{3-} can be used to indicate nitrogen availability for plants (Bohn, McNeal, O'Connor, 1979). Both $NH_{3-}N$ and NO_{3-} shows no significant difference between treatments over time. NO_{3-} is expected to be low in soil where nitrifying bacteria quickly convert it into ammonium form. The $NH_{3-}N$ in soil is relatively low as well (>1.5 mg/L) and will likely increase slightly over time as systems become loaded with NH_4NO_3. Figure 3 shows $NH_{3-}N$ in interstitial water samples. NO_{3-} has remained at virtually undetectable levels throughout the experiment; the absence of NO_{3-} combined with $NH_{3-}N$ concentrations that are as much as two orders of magnitude higher than either $NH_{3-}N$ in soil or surface water indicates that the N cycle is proceeding effectively and that ammonium is the dominant form of inorganic nitrogen available for plant uptake. Fig. 4A and 4B illustrates mean plant height for aboveground parts in *C. haspan*. C_4 species begin emerging in late spring and early summer, so *C. erythrorhizos* has only just begun emerging in the past two months. Plant height has significantly increased over three months in the *C. haspan* group, although heights are similar between treatments at this point. Temperature and daylight hour increase likely play a role in this response rather than N concentra-

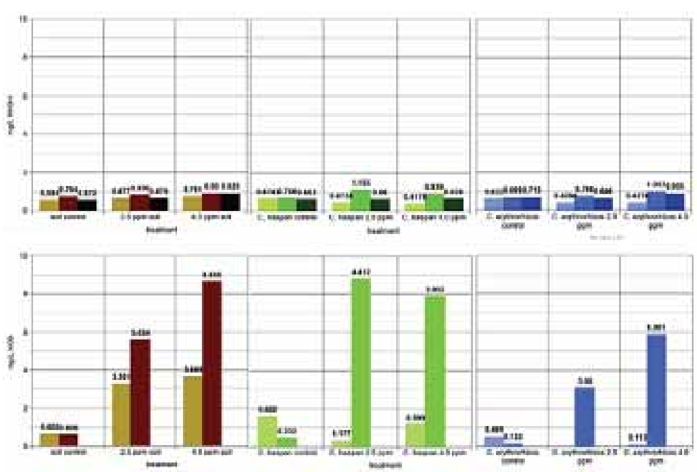


Figure 1A & 1B: Surface water nitrates and Kjeldahl nitrogen from January to March 2006.

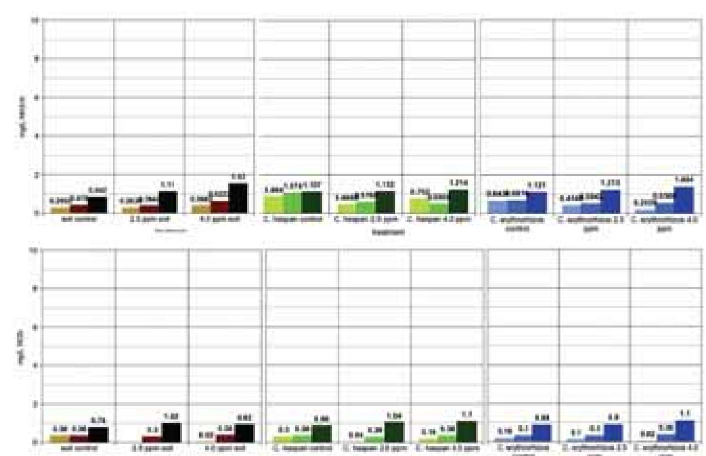


Figure 2A & 2B: Soil extracted nitrates and Kjeldahl nitrogen from January to March 2006.

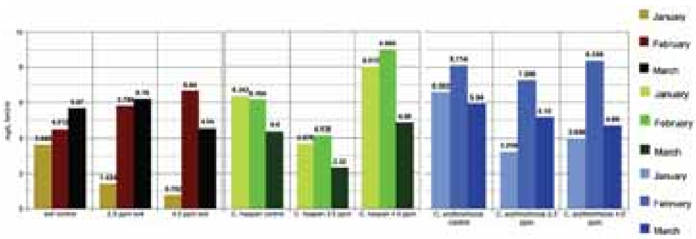
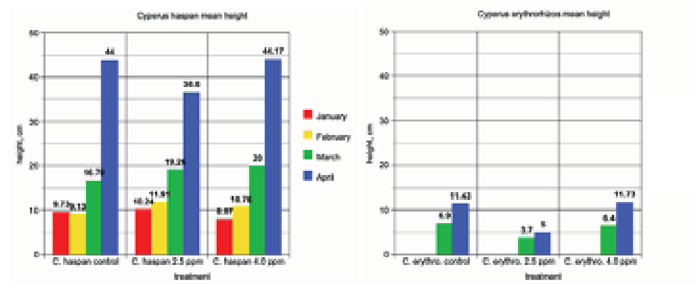


Figure 3: interstitial pore water Kjeldahl nitrogen from January to March 2006.



Figures 4A & 4B: Cyperus haspan and C. erythrorhizos mean heights, from January to March 2006.

tion; however, TN analysis on both species will be performed at a later date to determine N contribution to plant biomass.

CONCLUSIONS

Our preliminary data suggest that all groups (soil only, *C. haspan*, and *C. erythrorhizos*) are functioning as reducing wetland systems. We have recently qualitatively observed several differences in both *C. haspan* and *C. erythrorhizos'* general appearance among treatments. *C. haspan* at the 2.5-ppm dose appears to have weaker culms, smaller culm diameter and density, and appear less healthy than either the control or 4.0-ppm dosed *C. haspan* groups. The same differences have been noted among the *C. erythrorhizos* group as well, with plantlets being taller, healthier in appearance, and more abundant in control and 4.0-ppm treatments but less so in the 2.5 ppm treatment. Further analyses and observations may elucidate the reasons for these observed differences.

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Corresponding Authors:
[1]Alison A. Faulkner
University of Mississippi
Department of Biology
Oxford, MS
Phone 662–915–7203
E–mail: aafaulkn@olemiss.edu
[2] University of Mississippi Department of Biology, Oxford, MS.
[3] USDA–ARS National Sedimentation Laboratory, Oxford, MS.

Conference Attendees

Judy Adams
ACEC-MS
cecms@netdoor.com

Onur Akay
University of Mississippi
oakay@olemiss.edu

Vladimir Alarcon
GeoResources Institute
alarcon@gri.msstate.edu

Brian Alford
MSU, Dept. of Wildlife & Forestry
jba81@msstate.edu

Valerie Alley
MDEQ
valerie_alley@deq.state.ms.us

Nestor Anzola
University of Southern MS
pijume@hotmail.com

Trent Baldwin
USGS
tbaldwin@usgs.gov

Jeff Ballweber
MS Water Resources Research Institute
ballweber@cfr.msstate.edu

David Barnes
MDEQ
david_barnes@deq.state.ms.us

Mike C. Beiser
MDEQ
mike_beiser@deq.state.ms.us

Hamid Borazjani
MSU Forest and Wildlife Research Center
hborazjani@cft.msstate.edu

Shirley Bounds
MS Public Service Commission

Chris Bowen
Pat Harrison Waterway District
grolen@phwd.net

Karen Brasher
MSU Forest and Wildlife Research Center
kbrasher@cfr.msstate.edu

Wayne Brown
MDOT
seskew@mdot.state.ms.us

Jeannie Bryson
USGS
jbryson@usgs.gov

Thomas D. Burns
MDEQ Off. Land & Water Resources
thomas_burns@deq.state.ms.us

David Burt
USGS
deburt@usgs.gov

Charlotte Bryant Byrd
MDEQ Off. Land & Water Resources
charlotte_bryant_byrd@deq.state.ms.us

Lynn Carlisle
MS Public Service Commission

Laurance Carter
MS Levee Board
peter@msleveeboard.com

Kim Caviness
MDEQ
kim_caviness@deq.state.ms.us

Antonio Cerdeira
Embrapa-Brazil
alcerdei@olemiss.edu

Joshua Cheshier
GeoResources Institute
cheshier@gri.msstate.edu

Charles Chisolm
MDEQ

David Clarius
DGC Environmental
dgcinc@bellsouth.net

Kim Collins
MSU
kac65@engr.msstate.edu

Bill Cosgrove
US EPA
cosgrove.bill@epa.gov

Emily Cotton
MDEQ
emily_cotton@deq.state.ms.edu

Richard Coupe
USGS
rhcoupe@ugs.gov

Jamie Crawford
MDEQ Off. Land & Water Resources
jamie_crawford@deq.state.ms.us

George Alan Criss
USM Gulf Coast Geospatial Center
alan.criss@usm.edu

Sabrina Cummings
University of Southern MS
scummings68@yahoo.com

Mike Davis
Pearl RiverBasin Dev. District
mdavis@prbdd.state.ms.us

William K. Dean
US EPA, Region 4
dean.william-kenneth@epa.gov

Scott Dennis
USGS

Ayanangshu Dey
MSU Civil Engineering
ad196@msstate.edu

Jairo N. Diaz
MSU
jd216@msstate.edu

Alice T. Dossett
MDEQ
alice_dossett@deq.state.ms.us

Conference Attendees

Tom Doyle
USGS - Invited Speaker

Jeremiah Estes
University of Southern MS
freddie.howell@usm.edu

Alison Faulkner
University of Mississippi
aafaulkn@olemiss.edu

Henry G. Folmar
MDEQ
henry_folmar@deq.state.ms.us

Garey Fox
University of Mississippi
gafox@olemiss.edu

Mark Gilbert
MS Soil & Water Conservation Commission
mgilbert@mswcc.state.ms.us

Mandy Gong
MDOT - Bridge Division
mgong@mdot.state.ms.us

Leili Gordji
University of Mississippi Civil Engineering
lgordji@olemiss.edu

Will Green
MDEQ
will_green@deq.state.ms.us

Beau Griffin
University of Southern MS
bgriffin@hotmail.com

Ken Griffin
Pickering, Inc.
kgriffin@pickeringinc.com

Frank Gwinn
YMD Joint Water Management District

Bonnie Hamiter
MSU - College of Forest Resources
blh74@msstate.edu

Cynthia Hemphill
MSU Forest and Wildlife Research Center
chemphill@cfr.msstate.edu

Robert Hirsch
Assoc.Dir.-Water USGS
rhirsch@usgs.gov.

James Hoffman
MDEQ Off. Land & Water Resources
james_hoffman@deq.staet.ms.us

George M. Hopper
MSU Forest and Wildlife Research Center
ghopper@cfr.msstate.edu

Peter Howd
USGS - Invited Speaker

Fred Howell
University of Southern MS
freddie.howell@usm.edu

Russ Howell
USGS
jrhowell@usgs.gov

Joe Jewell
MS Dept. of Marine Fisheries
joe.jewell@dmr.state.ms.us

Cyle Keith
MSU College of Forest Resources
ckeith21@aol.com

Joyce Kellum
USGS
gabourie@usgs.gov

Sam Mabry
MDEQ Off. Land & Water Resources
sam_mabry@deq.state.ms.us

John Madsen
MSU GeoResources Institute
jmadsen@gri.msstate.edu

Sitaram Makenia
MDEQ
sid_Makena@deq.state.ms.us

Faye Mallette
Gulf Coast Research Lab
faye.mallette@usm.edu

David Massingill
USGS
massingd@usgs.gov

William H. McAnally
MSU
mcanally@engr.msstate.edu

Debbie McBride
MSU GeoResources Institute
dmcbride@gri.msstate.edu

Deirdre McGowan
MS Water Resources Association
insightltd@msn.com

Matthew P. Miller
The Nature Conservancy
matthew_miller@tnc.org

Lea Millet
Southern Company Services
llmillet@southernco.com

Bill Minor
MDOT
seskew@mdot.state.ms.us

Chukwukelue K. Monwuba
MSU Civil Engineering
ckm49@msstate.edu

Joann Moosa
University of Florida
mossa@geog.ufl.edu

Fred Morris III
USGS
fmorris@usgs.gov

Carol Moss
USGS
bpmoss@usgs.gov

Peter Nimrod
MS Levee Board
peter@msleveeboard.com

Conference Attendees

Rodrigo Nobrega
MSU GeoResources Institute
rodrigo@gri.msstate.edu

Glen Odom
Dept of Env Quality

Larry Oldham
MSU

Dean Pennington
YMD Joint Water Management District
dean@ymd.org

Harriet Perry
Gulf Coast Research Lab
harriet.perry@usm.edu

Patricia Phillips
MDEQ Off. Land & Water Resources
pat_phillips@deq.state.ms.us

Mickey Plunkett
USGS
plunkett@usgs.gov

Jonathan Pote
MSU-Ag & Bio Eng.
jpote@abe.msstate.edu

Lynn Prewitt
MSU Forest and Wildlife Research Center
lprewitt@cfr.msstate.edu

Sam T. Qarqish
MDEQ
sam_qarqish@deq.state.ms.us

Richard Rebich
USGS
rarebich@usgs.gov

Bo Robinson
MS Public Service Commission

Johnny Robinson
MS Levee Board
peter@msleveeboard.com

Wilfredo Robles
MSU GeoResources Institute
wr40@pss.msstate.edu

Kevin Roper
University of Southern MS
kevinroper@gmail.com

Mike Runner
USGS
msrunner@usgs.gov

Abby Salinger
USGS

Rickey L. Saucer
USACE
rickey.l.saucer@sam.usace.army.mil

Gaurav Savant
MSU
gaurav@eng.msstate.edu

Jake Schaefer
University of Southern MS
jake.schaefer@usm.edu

Robert H. Seyfarth
MDEQ
robert_seyfarth@deq.state.ms.us

Jeremy Sharp
MSU
zd9@msstate.edu

Beverly A. Smith
MSU Forest and Wildlife Research Center
bsmith@cfr.msstate.edu

Charlie Smith
MDEQ Off. Land & Water Resources
charles_smith@deq.state.ms.us

Trent Snellings
USGS
gtsnell@usgs.gov

John Storm
USGS
jbstorm@usgs.gov

Mary Love Tagert
MS Water Resources Research Institute
mltagert@cfr.msstate.edu

Samuel Testa
USDA-ARS-NSL
stesta@ars.usda.gov

Sid Thompson
Pickering, Inc.
rforsythe@pickeringinc.com

Robert Thompson
MS Levee Board
peter@msleveeboard.com

Amey S. Tilak
University of Mississippi
astilak@olemiss.edu

Christine Trigg
Gulf Coast Research Lab
christine.trigg@usm.edu

Don Underwood
MS Soil & Water Conservation Commission
dunderwood@mswcc.state.ms.us

Barbara Viskup
MDEQ
barbara_viskup@deq.state.ms.us

Patrick Vowell
MS Soil & Water Conservation Commission
pvowell@mswcc.state.ms.us

Rachel Waits
MDOT - Bridge Division
rwaits@mdot.state.ms.us

John Warner
MDEQ Off. Land & Water Resources
john_warner@deq.state.ms.us

Charles Wax
Mississippi GeoResources Institute
wax@geosci.msstate.edu

David Welch
National Weather Service LMRFC
david.welch@noaa.gov

Conference Attendees

Heather Welch
USGS
hllott@usgs.gov

Ryan Wersal
MSU GeoResources Institute
rwersal@gri.msstate.edu

Linda White
MSU GeoResources Institute
linda@erc.msstate.edu

Mark Whitney
DGC Environmental
dgcinc@bellsouth.net

Kay Whittington
MDEQ
kay_whittington@deq.state.ms.us

Van Wilson
USGS
kvwilson@usgs.gov