The Carolina Bay Restoration Project

Status Report II 2000-2004

Prepared by:
Christopher Barton

Center for Forested
Wetlands Research
A division of the Southern Research Station

The University of Georgia
Savannah River Ecology Laboratory

CLEMSON
UNIVERSITY

West Virginia University
Where Greatness is Learned

Executive Summary:

A Wetlands Mitigation Bank was established at SRS in 1997 as a compensatory alternative for unavoidable wetland losses. Prior to restoration activities, 16 sites included in the project were surveyed for the SRS Site Use system to serve as a protective covenant. Pre-restoration monitoring ended in Fall 2000, and post restoration monitoring began in the Winter/Spring of 2001. The total interior harvest in the 16 bays after harvesting the trees was 19.6 ha. The margins in the open-canopy, pine savanna margin treatments were thinned. Margins containing areas with immature forested stands (bay 5184 and portions of bay 5011) were thinned using a mechanical shredder in November 2001. Over 126 hectares were included in the study areas (interior + margin). Planting of two tree species and the transplanting of wetland grass species was successful. From field surveys, it was estimated that approximately 2700 *Nyssa sylvatica* and 1900 *Taxodium distichum* seedlings were planted in the eight forested bays resulting in an average planting density of ≈ 490 stems ha⁻¹. One hundred seedlings of each species per bay (where available) were marked to evaluate survivability and growth. Wetland grass species were transplanted from donor sites on SRS to plots that ranged in size from 100 – 300 m², depending on wetland size. On 0.75 and 0.6 meter centers, respectively, 2198 plugs of *Panicum hemitomon* and 3021 plugs *Leersia hexandra* were transplanted. New shoots originating from the stumps were treated with a foliar herbicide (Garlon® 4) during the summer of 2001 using backpack sprayers. Efforts to control some of the sprouts, particularly those of sweetgum (*Liquidambar styraciflua*), may be necessary in the near future.

The filling of drainage ditches was delayed approximately eight months after planting due to permitting constraints. However, most sites were effectively plugged by harvesting activities, when native soil was dozed into the ditch and compacted to facilitate movement of harvesting equipment. These actions, coupled with a regional drought, inhibited surface water loss through the drainage ditches. With the verification of the Corps of Engineers 404 permit (Nationwide Permit 27), received December 18, 2001, formal actions to plug the drainage ditches began. A clay plug was installed by excavating an area perpendicular to the drainage ditch at the location of the historical wetland boundary (rim). The excavated site was at least twice the width and depth of the original drainage ditch and extended 2 to 3 meters into the upland. The material removed was used as a surface cover on the impermeable clay plug. Subsoil clays obtained from SRS borrow areas were put in the excavation, and compacted. Water levels rose in the Winter 2003 and no leaks or undercutting of the plugs have been detected since.

A national ban (DOE) on prescribed burning following a Los Alamos National Lab wildfire; a Site burning ban following September 11, 2001; and extreme drought and unfavorable burning conditions combined to delay burning of the open-canopy, pine savanna margin bays. In 2003, site conditions improved and prescribed burning operations were re-initiated. Thus far, bays 5190, 131, 5092, 124, 126 and 5135 have been burned for the first time (February - May 2003). Bays 5011 and 5184 will be burned when field conditions are acceptable. The eight bays will be re-burned as soon as fuel levels permit (approx. 3 years).

Preliminary information from 2000-2004 regarding the hydrologic, vegetation and faunal response to restoration is presented in this status report. Post restoration monitoring will continue through 2005. A final report to the Mitigation Bank Review Team will be submitted in mid-2006.

Table of Contents

The Carolina Bay Restoration Project: Implementation and Management of a Wetland Mitigation Bank *by:* Christopher Barton, Diane DeSteven, Rebecca Sharitz, Donald Imm, Randal Kolka and John Blake. pp. 4-13.

Restoration of Carolina Bays on the Savannah River Site: Hydrological Response *by:* Christopher Barton. pp. 14-16.

Vegetation Response in Restorations of Small Carolina Bay Depressional Wetlands *by:* Diane De Steven, Rebecca R. Sharitz. pp. 17-18.

Aquatic Invertebrates in Carolina Bays and Other Wetland Ponds Before and After Restoration Treatments *by:* Barbara E. Taylor, Darold P. Batzer, and Adrienne E. DeBiase. pp. 19-21.

Effects of Hydrology Restoration and Landscape Pattern on Amphibians in Carolina Bays on the Savannah River Site *by:* Karen Kinkead and David Otis. pp. 22-23.

The Influence of Buffer Zone Management Techniques on Carolina Bay Herpetofauna *by:* Joseph A. Ledvina and Hugh G. Hanlin. pp. 24-27.

Bird Response to Carolina Bay Restoration *by:* John C. Kilgo, and Stephen J. Czapka. pp. 28-30.

Literature Cited. pp. 31-32.

Appendix I. pp. 33-39.

The Carolina Bay Restoration Project: Implementation and Management of a Wetland Mitigation Bank

Christopher Barton¹, Diane DeSteven², Rebecca Sharitz³, John Kilgo^{4,5}
Donald Imm⁵, Randal Kolka⁶ and John Blake⁵

¹*University of Kentucky, Department of Forestry, Lexington, KY;*

²*USDA Forest Service, Southern Research Station, Stoneville, MS;*

³*University of Georgia, Savannah River Ecology Lab, Aiken SC;*

⁴*USDA Forest Service, Southern Research Station, Center for Forested Wetlands Research, Charleston, SC;*

⁵*USDA Forest Service, Savannah River, New Ellenton, SC;*

⁶*USDA Forest Service, Ecology and Management of Riparian Systems, Grand Rapids, MN.*

Introduction. A Wetlands Mitigation Bank was established at the Savannah River Site (SRS) in 1997 as a compensatory alternative for unavoidable wetland losses associated with future authorized construction and environmental restoration projects in SRS wetlands. The Bank was intended not only to hasten mitigation efforts with respect to regulatory requirements and implementation, but also to provide onsite and fully functional compensation of impacted wetland acreage prior to any impact. Restoration and enhancement of small isolated wetlands, as well as major bottomland wetland systems scattered throughout the nonindustrialized area of SRS were designated for inclusion in the Bank. Based on information and techniques gained from previous research efforts involving Carolina bay wetlands (DOE, 1997), a project to restore degraded Carolina bays on SRS has been undertaken to serve as the initial “deposit” in The Bank. There are over 300 Carolina bays or bay-like depression wetlands on the SRS, of which an estimated two-thirds were ditched or disturbed prior to federal occupation of the Site (Kirkman et al., 1996). These isolated wetlands range from small ephemeral depressions to large permanent ponds of 10-50 hectares in size. They provide habitat to support a wide range of rare plant species, and many vertebrates (birds, amphibians, bats). Historical impacts to the Carolina bays at SRS were primarily associated with agricultural activities. Bays were often drained, tilled and planted to crops. The consequence was a loss in the wetland hydrologic cycle, the native wetland vegetation, and associated wildlife.

The purpose of this mitigation and research project is to restore the functions and vegetation typical of intact depression wetlands and, in doing so, to enhance habitat for wetland-dependent wildlife on SRS. Twenty Carolina bays in the nonindustrialized management area of SRS were identified as candidates for restoration (Figure 1.1). All twenty bays possessed an active drainage ditch and nearly all had a vegetation composition characteristic of a disturbed wetland system. Pre-restoration characterizations of soil, hydrology, vegetation and wildlife were performed within each site, to be used as a baseline for evaluating restoration success. Of the twenty bays, sixteen were initially being restored and the remainder were planned to serve as unrestored controls. Undisturbed bays of similar size have also been selected for use as reference sites. The use of reference and control systems will enhance our ability to assess response in the wetlands due to treatment implementation. The project is a collaborative effort that was designed jointly by researchers and the management staff of the US Forest Service, Westinghouse Savannah River Corporation – Environmental Protection Division, Westinghouse Savannah River Technology Center, and the Savannah River

Ecology Laboratory. Additionally, cooperators from the University of Kentucky, US Fish & Wildlife Service, Clemson University, the University of South Carolina at Aiken, the University of West Virginia, and the University of Georgia are participating in studies examining the responses of soils, hydrology, vegetation and animal communities.

Methods. To restore the wetlands, trees in the bay interior were harvested and drainage ditches were plugged with low permeable clay to re-establish prior hydrological conditions. Several strategies for restoring the vegetation in replicated sets of these bays and their associated uplands are being examined. Planned endpoints or treatments included pine and hardwood managed upland buffer zones, and forested and herbaceous bay interiors. In bays that are being restored as herbaceous communities, test plots were planted with obligate wetland grass species (*Panicum hemitomon* and *Leersia hexandra*). The majority of the area within these herbaceous bays was not planted, but efforts were taken to encourage natural succession through soil scarification. Bays intended for restoration to a forested community were planted on 4.5 m centers throughout the interior with swamp tupelo (*Nyssa sylvatica* var. *bifloura*) and baldcypress (*Taxodium distichum*). The role of the seed bank on revegetation and natural succession is also being examined in all bays to evaluate the need for out planting of wetland species and for further development of planting strategies in disturbed Carolina bays and similar depression wetlands.

The influence of upland margin management (buffer-zones) on wetland properties and wildlife usage is widely debated and relatively misunderstood (Semlitsch, 1998; Buhlmann and Gibbons, 2000). There are two principal upland landscapes on the SRS that are commonly associated with Carolina bays: 1) fire-managed, open-canopy pine forest savannas, and 2) closed canopy mixed pine-hardwood forests (principally mixed pine-hardwoods that occur in the absence of fire). To gain a better understanding of the relationship between buffer-zone management and wetland properties, these two upland management alternatives are being examined as bay margin treatments and as expected endpoints for the uplands surrounding the bays. The restoration sites are organized in an experimental design such that two planned wetland vegetation types (herbaceous and forested) are examined against the two buffer-zone types (open-canopy pine and closed-canopy pine-hardwood) for a total of four bay-margin communities (Figure 1.2). Bay margin treatments were applied to a 100-m buffer from the edge of each bay into the upland (i.e. essentially a 100 m radius from the bay rim). Selective harvesting of hardwoods and some pines was performed in the open-canopy pine forest savanna margins to reduce the basal area to approximately 5 m² ha⁻¹. Several young stands in this treatment group were thinned using a mechanical shredder to achieve the targeted basal area. Prescribed burning of the margins began in 2003 and will be repeated as dictated by fuel levels (approximately once every 3 to 4 years). Margins within the closed canopy mixed pine-hardwood forests were left unthinned.

Determination of whether restored systems and their accompanying buffers are moving toward planned endpoints will be accomplished by assessing trends and rates of change in biotic and abiotic metrics and comparing these to undisturbed reference bays and/or unrestored control bays. The monitoring program will record the progress of the restoration for five years after the treatment manipulations (2001 – 2005), and will be used as a guide for determining the final net improvement displayed for each individual wetland. A Carolina bay restoration will be deemed a success when the restored hydrologic regime has stabilized and the associated wetland community is dominated by hydrophytic vegetation more commonly found in wetlands than in the community occupying the

site immediately before restoration (DOE, 1997). Table 1.1 outlines specific criteria for determining restoration success.

Preliminary Results. By the end of 2002, all restoration treatments were successfully imposed (Table 1.2). Pre-restoration monitoring ended and post restoration monitoring began as harvesting activities in the bay interiors began. The total interior harvest for the 16 restoration bays was targeted at 15.5 hectares, and ultimately 19.6 ha. were cleared. Two bays (131 and 5016) were deemed too wet to harvest the entire wetland interior with the mechanical harvester; consequently, the trees in these two bays were felled with chainsaws and left in the bay center. Subsequent efforts were undertaken to cut these trees in small pieces, to facilitate decomposition, and to remove some of the slash from the interior for vegetation plot development. Bay 5 was too wet to perform any harvesting activities in 2001 and was not harvested until February 2002. Efforts to thin the margins in the open-canopy, pine savanna margin treatments were successful in sites that contained a mature forested stand (Table 1.3). Margins containing areas with immature forested stands (bay 5184 and portions of bays 5011 and 131) were not completely finished during the initial harvest. Two of these sites (5184 and 5011) were later thinned using a mechanical shredder in November, 2001. Additional thinning of immature trees in these areas is anticipated through mortality from future burning activities. Ultimately, over 126 hectares were included in the study areas (interior + margin). Planting of the two forested species and the transplanting of wetland grass species was successful. The exact number of either *Nyssa sylvatica* or *Taxodium distichum* that were initially planted in the bays was difficult to ascertain due to methods employed by the contracted planting crew. The total number of trees planted at each site was estimated by the number of seedling bags utilized at the site (50 and 100 seedlings per bag of cypress and tupelo, respectively) and the number of seedlings claimed planted by the contractor. Subsequently, field surveys were performed to evaluate the accuracy of these numbers. From these surveys it was estimated that approximately 2700 *Nyssa sylvatica* and 1900 *Taxodium distichum* seedlings were planted in the eight bays. One hundred seedlings of each species per bay (where available) were marked and measured and will be utilized throughout the study to evaluate survivability and growth. Wetland grass species were located at donor sites on SRS and transplanted to test plots that ranged in size from 100 – 300 m², scaled to correspond to the size of the wetland. On 0.75 and 0.6 meter centers, respectively, 2198 plugs of *Panicum hemitomom* and 3021 plugs *Leersia hexandra* were transplanted (Table 1.4). Annual surveys will also be performed in these plots to evaluate transplant survival and growth.

Efforts to curtail stump sprouting immediately following the interior harvest were not undertaken. However, new shoots originating from the stumps were treated with a foliar herbicide (Garlon® 4) during the summer of 2001 using backpack sprayers. Dieback of the sprouts was apparent within a week of the applications, but the treatment proved ineffective as evident by the numerous shoots that reappeared, and persisted in subsequent growing seasons. Efforts to control some of the sprouts, particularly those of sweetgum (*Liquidambar styraciflua*), may be necessary in the near future.

The filling of drainage ditches was intended to begin immediately after the planting had been completed; however, plug installation was delayed approximately eight months due to permitting constraints. Most of the sites were “effectively” plugged by harvesting activities, where native soil was dozed into the ditch and compacted so as to facilitate movement of mechanical harvesters and skidders in the wetland interiors. These actions, coupled with a regional drought, inhibited surface water loss through the drainage ditches at all sites during the year with the possible exception of bay 124, which may have drained actively for a portion of April/May 2001. With the verification of the

Corps of Engineers 404 permit (Nationwide Permit 27), received December 18, 2001, actions to plug the drainage ditches began. A clay plug was installed by excavating an area perpendicular to the drainage ditch at the location of the historical wetland boundary (rim). The excavated site was at least twice the width and depth of the original drainage ditch and extended two to three meters into the upland. The material removed from the ditch was set aside and reused as a surface cover on the impermeable clay plug. Once the excavated area was established, subsoil clays obtained from SRS borrow areas were dumped into the pits and compacted using a backhoe. The total amount of material used for each plug is presented in Table 1.5. Erosion control practices (seeded annual ryegrass and installed coconut/straw stitched erosion control blankets) were implemented at each site with the completion of the plug installation. Continuation of the regional drought through the 2002 calendar year inhibited our ability to evaluate the integrity of the plugs as surface water levels remained well below the discharge level of the historical drainage ditches. Water levels ultimately rose during the winter/spring of 2003 and no leaks or undercutting of the plugs were detected. Annual checks of the plugs have continued and they have remained stable.

Planned burning of the open-canopy, pine savanna margin bays was postponed in both 2001 and 2002 due to a variety of reasons that included a national ban on prescribed burning following a wildfire incident at the Los Alamos National Lab in NM, a Site burning ban in response to terrorist activities of September, 2001, extreme drought and unfavorable burning conditions. The consequence of these delays on the vegetation and overall restoration response is difficult to ascertain at this time, though possibly it contributed to the lack of early control of woody resprouts. After soil moisture levels increased in response to rainfall in 2003, site conditions improved and prescribed burning operations were re-initiated. Bays 5190, 131, 5092, 124, 126 and 5135 were burned for the first time between February and May of 2003. With the exception of bay 5190, fires did not carry from the margins into the flooded wetland interiors. Efforts will be taken to burn bays 5011 and 5184 when field conditions are acceptable. The eight bays will be re-burned as soon as fuel levels are re-established to a point that will support the activity (approx. 3 years).

Details on the response of hydrology, vegetation and faunal communities to the restoration activities are presented in the following summaries. Post restoration monitoring of biotic and abiotic metrics shall continue through 2005. Photographs outlining the progress of the project are presented in Appendix I.

Figure 1-1. The Savannah River Site and location of Carolina bay restoration study sites.

Figure 1-2. Treatment pairs for the Carolina bay restoration project at SRS.

Table 1.1. Success Criteria Summary

Action #	Mitigation Action	Success Criteria
1	Habitat Protection	Areas are incorporated into the DOE-WSRC Site Use Program
2	Hydrological Restoration	Ditch plugs are installed and permanently stabilized. Water levels show an increase over baseline conditions and the depth and duration of flooding and/or groundwater saturation is comparable to the target reference ecosystems.
3	Vegetative Restoration Forested	Bay interiors are planted with appropriate wetland tree species at the density of 500 trees/hectare (200 trees/acre) with a 50% survival rate after 5 years. Plant species density and diversity are comparable to the target reference ecosystem.
4	Vegetative Restoration Herbaceous	Wetland emergent vegetation from plantings and/or natural seeding is established. Plant species and density and diversity are comparable to the target reference ecosystem.

Table 1.2. Status of milestones for the Carolina Bay Restoration Project.

Event	Begin Date	Finish Date
Bay Selection	1998	December, 1999
Restoration Plan	January, 1999	December, 1999; Rev. 1, November, 2001
COE 404 Permit (NW 27)	April, 2001 ³	January, 2002
Pre-restoration Monitoring	January, 2000	November, 2000
Harvest Trees	November, 2000	February, 2001 ⁴
Interior Tree Planting	February, 2001	March, 2001
Interior Herbaceous Planting	April, 2001	May, 2001
Sprout Control	September, 2001	September, 2001 ⁵
Ditch Plugging	January, 2002	February, 2002 ⁶
Post Restoration Monitoring	February, 2001	December, 2005 (tentative)
Burn Pine Savanna Margins¹	February, 2003	2003 ⁷ (repeat as soon as fuel levels permit)
Thin and Plant MPH Margins²	2006	
Final Report	2006	

¹Open-canopy pine savannah margin.

²Closed-canopy mixed pine/hardwood margin.

³Submitted "Pre-construction notification for the Carolina Bay restoration project- Nationwide Permit 27" to US Army Corps of Engineers. Permit approved January, 2002.

⁴Bay 5 could not be harvested at that time due to wetness. Trees removed from Bay 5 in February, 2002.

⁵Sprout control was performed using a foliar herbicide (Garlon 4), which proved inefficient for effective treatment of the resprouts. Additional sprout control efforts may be implemented in 2003 or 2004.

⁶Bay 5016 was not plugged at this time due to wetness and an inability to get equipment to the location where the plug was to be installed. The plug was subsequently installed at Bay 5016 in December 2002.

⁷Bays 5190, 131, 5092, 124, 126, 5135 burned between the dates of 2.12.03 and 4.22.03, respectively. Bays 5011 and 5184 have not been burned (to date) due to inappropriate site conditions and weather patterns.

Table 1.3. Total area of restoration sites and adjacent margins after treatment application.

Bay	Harvested Interior Area (ha)	Interior + Margin Area (ha)
<u>Thinned Margin</u>		
<i>124</i>	1.36	7.15
<i>126</i>	1.72	9.71
<i>131</i>	1.06	5.50
<i>5011</i>	1.08	7.69
<i>5092</i>	1.16	7.85
<i>5135</i>	0.79	6.43
<i>5184</i>	1.22	7.73
<i>5190</i>	<u>1.29</u>	<u>8.18</u>
<i>Total (thin)</i>	9.68	60.24
<u>Intact Margin</u>		
<i>5</i>	1.68	14.21
<i>171</i>	1.67	9.44
<i>5001</i>	0.57	7.69*
<i>5016</i>	0.70	7.68
<i>5071</i>	1.16	7.71
<i>5128</i>	1.65	10.74
<i>5204</i>	1.13	8.29
<i>5239</i>	<u>1.42</u>	<u>7.82</u>
<i>Total (intact)</i>	9.98	73.58
Grand Total	19.66	126.13

*Margins of 5 and 5001 overlap, so 7.69 ha is inclusive of both sites and factored only once in total area calculation.

Table 1.4. Final number of wetland grass transplants per site and plot size.

Bay	Block Size (m ²) [†]	# Panicum Transplants [‡]	#Lersia Transplants [*]
<i>5204</i>	180	156	195
<i>5071</i>	200	169	225
<i>171</i>	300	247	345
<i>5239</i>	300	247	345
<i>124</i>	200	169	225
<i>5092</i>	300	247	345
<i>5190</i>	162	144	196
<i>126</i>	300	247	345
<i>5128</i>	200	169	225
<i>5001</i>	162	144	196
<i>5135</i>	112	90	154
<i>5011</i>	<u>200</u>	<u>169</u>	<u>225</u>
Total	2616	2198	3021

[†]Each block was divided into two equal-area split-plots (one plot per species).

[‡]Planted on 0.75 m centers.

*Planted on 0.6 m centers.

Table 1.5. Volume of fill material used to plug drainage ditches.

Bay #	Ditch Area at Bay Rim (m ²)	Fill Volume (m ³)†	Clay Fill (cy)†
5	1.13 and 0.71	5.7 and 3.59	12.1
124	0.88 and 0.66	4.45 and 3.31	10.2
131	0.43	2.20	2.8
5001	1.82 and 1.71	9.11 and 8.6	23.2
5011	1.40	7.05	9.2
5016	0.67	3.35	4.4
5071	1.14 and 1.09	5.75 and 5.50	14.3
5092	2.99	14.98	19.6
5135	1.51	7.58	9.9
5184	0.39	2.02	2.6
5190	1.14	5.75	7.5
5204	0.24 and 0.53	1.29 and 2.69	5.2
Total	18.44	92.92	121.0

†Volume as cubic yards required for 404 permit.

Restoration of Carolina Bays on the Savannah River Site: Hydrological Response

Christopher Barton¹

¹*Department of Forestry, University of Kentucky, Lexington, KY*

Introduction. Hydrology is generally considered to be the primary controlling factor for the development and persistence of wetlands. However, characterizing wetland hydrology is difficult to perform and often compromised due to constantly changing hydrological/environmental conditions and to potential error associated with water budget accounting. Carolina Bays are shallow elliptical depressions found in the Atlantic Coastal Plain that are not only poorly understood with respect to hydrology, but have also been severely altered by human activity. Historical impacts to the Carolina bays at the Savannah River Site, SC were primarily associated with agricultural activities. Bays were often drained, tilled and planted to crops. The consequence was a loss in the wetland hydrologic cycle, the native wetland vegetation, and associated wildlife. Although speculations have been fabricated as to the effect that filling drainage ditches may have on bay vegetation and fauna at SRS, an increased hydroperiod will likely be the most definable constituent associated with any restoration activities. Considering that abiotic and biotic metrics of wetland function are greatly influenced by hydrology, an attempt must be established to thoroughly quantify and characterize bay hydrology so that an assessment for current and future restoration projects involving Carolina bays may be evaluated. As such, a study has been initiated to investigate hydrogeochemical processes in altered, restored and unimpacted bay systems and to use the hydrological data to assess the effectiveness of restoration practices within the current Carolina Bay restoration project. In addition, an overall evaluation of hydrology response to restoration activities is being monitored to fulfill reporting requirements for the Bank.

Methods. Bay hydrology was monitored bimonthly using a combination of piezometers, wells, and water level gages. Hydrological transects were established that traverse bays from the center of the bay to the upland in a perpendicular fashion following the long and short axis. Each transect was equipped with well nests within the bay interior, upland zone and a transitional point (hydric soil boundary or abrupt vegetation change) that is likely to exhibit hydromorphic change in response to filling the drainage ditch. The well nests were comprised of a shallow monitoring well at a depth of 200 cm and four piezometers at 50, 100, 200 and 300 cm depths. Information pertaining to the types of monitoring equipment installed and location within the bays was presented in the first SRS Mitigation Bank Status Report (Barton, 2003). Piezometers were constructed of 2.5 cm diameter schedule 80 PVC pipe with perforations drilled at the submerged end and covered with geofabric. Borings for the piezometers were drilled by hand using a 8.5 cm bucket auger. Washed sand was packed from the base of the borings to approximately 25 cm above the screened area and the remaining annulus will be filled to just below the surface with a slurry created from the bore cuttings. A plug of bentonite was placed at the surface to prevent short circuiting. Shallow wells were constructed in a similar fashion using 5.0 cm diameter schedule 40 PVC pipe that has been slotted along its entire length. Development of the shallow wells was achieved by surging with a bailer to remove water and any dislodged sediment. Water depths within the wells and piezometers were measured using a portable water level indicator. Surface water level (pond stage) was monitored using staff gages and semi-continuously recording monitoring wells (WL-40 measurements taken at 6-hr. intervals). The saturated water depth, piezometric surface, and bay hydroperiod were

determined from measurements accumulated with the above monitoring devices. Open precipitation and throughfall were measured in bay margins and interiors of each site. Other meteorological variables (net solar radiation, air temperature, humidity, wind speed) were provided by the SRS Weather Center. Surface evaporation within the bay interiors was determined using a modified Penman-Monteith equation.

Preliminary Findings. The hydrological response to the restoration treatments was initially complicated by a prolonged regional drought. For the first three year period examined, average monthly rainfall fell below the 50 year precipitation average at SRS for all but 6 months (Figure 2.1). As such, all treatment and reference bays were dry for most of 2002. As indicated in the first SRS Mitigation Bank Status Report (Barton, 2004), a change in hydroperiod (% time ponded per year) was detected in many of the treatment bays, and most of the restored bays exhibited an initial increase in hydroperiod after the treatments had been imposed. Interestingly, all control and reference bays responded to the drought conditions with initial post restoration hydroperiods that were lower than those exhibited prior to treatment implementation in the restored bays. The increased hydroperiod in the treatment bays was most likely the result of changes to the water budget via tree removal and subsequent lowering of water demand in these systems by transpiration. Similar findings pertaining to the role of forest harvesting on wetland hydrology have been noted elsewhere (Sun et al., 2000). One study indicated that the water table rise associated with harvesting is most expressed during periods when the water tables were low (Riekerk, 1989). By the end of 2002, normal rain conditions had returned to the region and all bays exhibited ponding through most of 2003 (Table 2.1). In 2004, ample rainfall was recorded during the summer months and all bays exhibited ponding or saturation for significant periods during the growing season (Table 2.2). An examination of the hydroperiod data was performed to determine whether or not the sites met the minimum hydrology criteria for wetland delineation as described in the 1987 COE Wetlands Delineation Manual (with reference to the 1991 interagency revisions). For this interpretation we determined that a site met the minimum hydrology criteria if inundation continues for 15 consecutive days during the growing season for most years (3 out of 5) after restoration (2001-2005). With this criteria, all bays except 5071, 5092 and 5135 exhibit wetland hydrology (Table 2.3). The 2005 hydrology data will ultimately determine whether or not the remaining three bays meet the hydrology requirements.

Efforts to quantify site specific hydrologic budgets for the bays are underway. In addition, the role of hydroperiod change on the biogeochemistry of the restored bays is currently being assessed. Comparisons between restored, control and reference bays will be available in the final report.

Table 2.1. Annual hydroperiod (portion of year bay is ponded) data for treatment bays* .

	<i>Treatment Bays</i>															
Bay	5	124	126	131	171	5001	5011	5016	5071	5092	5128	5135	5184	5190	5204	5239
2000	0.74	0.56	0	0.44	0.10	0	0.01	0.35	0.02	0	0.01	0.01	0	0	0.12	0.01
2001	0.79	0.67	0.33	0.81	0.23	0.38	0.40	0.41	0.01	0.15	0.67	0.05	0.24	0.28	0.65	0.47
2002	0.81	0.44	0.04	0.51	0.15	0.09	0.02	0.29	0.02	0	0.55	0.04	0.13	0.01	0.52	0.01
2003	1.0	0.88	0.85*	1.0	0.99	0.74	0.87	1.0	0.60	0.83	1.0	0.72	0.98	0.83*	1.0	0.75
2004	1.0	0.51	0.38*	0.71*	0.62	0.31	0.59	1.0	0.23	0.20	0.89	0.32	0.39	0.16*	0.92	0.36
2005																

*WL-40 damaged, hydroperiod estimates from manual sampling of A-well and observations of whether or not water was present during this period.

Figure 2.1. Monthly rainfall deviation from 50 year mean precipitation levels acquired by the C-Area weather station.

Table 2.2. Bay hydroperiod (portion of year bay is ponded) during growing season (April 1 – November 1: 214 days).

		<i>Treatment Bays</i>														
Bay	5	124	126	131	171	5001	5011	5016	5071	5092	5128	5135	5184	5190	5204	5239
2000	0.55	0.28	0	0.23	0.04	0	0.01	0.04	0	0	0.01	0.01	0	0	0.09	0.01
2001	0.78	0.81	0.29	0.97	0.24	0.51	0.50	0.44	0.01	0.26	0.95	0	0.29	0.47	0.97	0.51
2002	0.86	0.20	0	0.27	0.01	0.03	0	0.10	0.01	0	0.32	0	0.01	0.01	0.27	0.01
2003	1.0	0.94	1.0	1.0	1.0	0.86	1.0	1.0	0.76	1.0	1.0	0.96	1.0	1.0	1.0	0.92
2004	1.0	0.39	0.29	0.50	0.85	0.17	0.53	1.0	0.13	0.07	0.86	0.19	0.27	0.13	0.91	0.23
2005																

Table 2.3. Status of whether or not restored bays meet COE wetland hydrology criteria.

		<i>Treatment Bays</i>														
Bay	5	124	126	131	171	5001	5011	5016	5071	5092	5128	5135	5184	5190	5204	5239
2000	Yes	Yes	No	Yes	No	No	No	No	No	No	No	No	No	No	Yes	No
2001	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	Yes	Yes	No	Yes	Yes	Yes	Yes
2002	Yes	Yes	No	Yes	No	No	No	Yes	No	No	Yes	No	No	No	Yes	No
2003	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
2004	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No†	Yes	Yes	Yes	Yes	Yes	Yes
2005																
Wet?*	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes			Yes		Yes	Yes	Yes	Yes

*Meets wetland hydrology criteria of inundation for 15 consecutive days during the growing season for most years (3 out of 5) after restoration (2001-2005).

†Does not meet ponded criteria, but soils were likely saturated for greater than 30 days.

Vegetation Response in Restorations of Small Carolina Bay Depressional Wetlands

Diane De Steven¹ and Rebecca R. Sharitz²

¹*USDA Forest Service, Southern Research Station, Stoneville, MS;*

²*Savannah River Ecology Laboratory, Aiken SC*

Introduction. On the Savannah River Site, SC, a landscape-scale experiment was initiated to test strategies for implementing and managing restorations of depressional wetlands. In 16 small “Carolina bay” depressions that historically had been ditched and drained, successional forest canopies of facultative pines and hardwoods were harvested to open the sites and promote revegetation from seed banks, and surface ditches were plugged to re-establish natural hydrologic patterns. Vegetation is a sensitive indicator of site conditions and processes, thus it constitutes an important success criterion for wetland restorations. Our studies are addressing two questions: 1) whether existing seed banks are sufficient to establish desired wetland vegetation, and 2) if selected tree and herbaceous species can be planted successfully to accelerate vegetation restoration. Recruitment from seed banks plus natural seed dispersal was the basic revegetation method used in all 16 experimental wetlands. In addition, eight of the wetlands were planted through at low density with seedlings of wetland tree species (cypress, *Taxodium distichum*, and swamp tupelo, *Nyssa biflora*) to promote development of wetland forest. Finally, in 12 of the wetlands, test blocks of characteristic wetland grasses (*Leersia hexandra* and *Panicum hemitomon*) were set up to determine if these species could be established successfully from transplants. Previously (2002 status report), we reported that the seed banks of the experimental sites had substantial numbers of wetland plant species that contributed to the restored vegetation in the first two years. The focus of the study has shifted to monitoring the progress of vegetation development and the success of planted species.

Methods. Percent covers of all plant species are recorded annually in permanent 2 x 2 m ground-layer plots within each experimental wetland. As needed, larger plots are also sampled to assess the extent of woody resprouting. In the 8 wetlands where wetland trees were planted, a sample of marked seedlings (generally 100 per species per wetland) is monitored annually for survival and height growth. In the 12 wetlands with test blocks of planted grasses (one block per species per wetland), Percent covers of all plant species present are also assessed annually in a permanent 2 x 2 m plot within each planted block.

Results and Discussion. Since the start of the experiment, the vegetation has varied annually in response to climatic and hydrologic fluctuations (Table 3.1). With the return of more “normal” rainfall after the 1999-2002 drought, it is likely that the “typical” hydrological behaviors of each wetland have been established. Thus, wetland vegetation composition will increasingly be shaped by these individual hydrologic regimes. By the fourth year (2004), wetland plant species averaged 57% (range: 34–76%) of all species and 57% (range: 11–96%) of total vegetative cover. To date, roughly half of the experimental sites have achieved >50% cover of “wetland” species. Several wetlands have substantial resprouting from harvested stumps, so this remains a persistent effect of conditions at initial site preparation.

By spring 2004, three years after planting, average survival of tree seedlings was 81% for cypress (range: 54–96) and 24% for swamp tupelo (range: 2–75%) (Table 3.2). The tupelo were younger and smaller at the time of planting; their higher mortality likely resulted from multiple factors including

drought stress and competition from other vegetation. Survival rates for both species appear to have stabilized between 2003 and 2004. By spring 2004, attained heights typically ranged between 1.2–1.8 m for cypress and between 0.7–1.2 m for tupelo.

Most plantings of the clonal grasses have established successfully, probably because plants were transplanted as tillers rather than started from seed. By the fourth year (2004), planted grasses attained anywhere from 1% to 98% cover within the test blocks. To date, 9 of 12 *Panicum* plantings and 5 of 12 *Leersia* plantings have achieved >60% cover.

TABLE 3.1. Change over time in species richness, vegetative cover, and the percent contributions of wetland species (OBL, FACW) in the experimental wetlands. Year 0 is pre-restoration. Data are means (s.e.) averaged over *n* wetlands.

	<i>n</i>	Total number of species	Percent wetland species	Total percent cover	Percent cover of wetland species
year 0	16	23 (2)	27 (3)	148 (12)†	25 (5)
year 1	15*	43 (4)	50 (3)	78 (12)	48 (8)
year 2	16	36 (2)	41 (2)	65 (5)	41 (6)
year 3	16	17 (2)	48 (4)	37 (6)	40 (7)
year 4	16	30 (3)	57 (4)	94 (5)	57 (8)

* excludes one wetland that was not manipulated until the end of year 1

† includes pre-harvest forest canopy cover

TABLE 3.2. Survival and average attained heights of planted baldcypress and swamp tupelo seedlings planted in early 2001. At planting, the cypress were 3 yr old (3-0 seedlings) and tupelo were 2 yr old (2-0 seedlings). Data are means (s.e.) averaged over *n* wetlands.

year	<i>n</i>	% seedlings surviving since planting		average seedling height (m)	
		cypress	swamp tupelo	cypress	swamp tupelo
2001*	8	100*	100*	1.03 (0.02)	0.56 (0.01)
2002	8	88 (4)	63 (5)	1.04 (0.06)	0.54 (0.03)
2003	8	82 (5)	25 (8)	1.26 (0.08)	0.62 (0.06)
2004	8	81 (5)	24 (8)	1.49 (0.10)	0.91 (0.09)

*in 2001, 100% of marked seedlings alive at planting; heights are initial sizes at planting

Aquatic Invertebrates in Carolina Bays and Other Wetland Ponds Before and After Restoration Treatments

Barbara E. Taylor¹, Darold P. Batzer², Adrienne E. DeBiase¹

¹*University of Georgia, Savannah River Ecology Laboratory, Aiken, SC;*

²*Department of Entomology, University of Georgia, Athens, GA*

Introduction. Aquatic invertebrates are abundant and diverse in Carolina bays and other wetland ponds of the Savannah River. This study was conducted in cooperation with the U.S.D.A. Forest Service with cooperation and collaboration of Savannah River Ecology Laboratory, the U.S. Fish and Wildlife Service, and several universities (Barton and Singer, 2001). All of these ponds had been ditched before 1951, when the land was acquired by the federal government. Most of the ditches were probably dug between the mid-19th and early 20th centuries to facilitate use of the land for crops or livestock. Since the creation of the SRS, all of sites had become forested, mainly with bottomland hardwoods or pine.

Methods. The pre-treatment information on the wetland ponds provides a baseline for detecting faunal changes after the ditches were plugged to restore natural hydrology in 2001-2002. Bimonthly sampling was conducted for three years during the pre-treatment phase (1998-2000, except August-October 1998). Sampling continued through 2001-2004 to characterize responses to treatment. Microcrustaceans were sampled qualitatively with a small, hand-held 100- μ m mesh net; macroinvertebrates were sampled semi-quantitatively with a standard 1-mm mesh D-frame sweep net. Because the set of ponds for the restoration study was changed after we began sampling, we have 3 years of pre-treatment data for 17 of the 20 ponds, 1 year for 2 ponds, and none for 1 pond.

Predicted effects of treatments. For the microcrustaceans, we predicted that species richness would increase with hydroperiod and with conversion from forested to herbaceous vegetation. We predicted that new species would colonize slowly. We also predicted that ephemeral habitat specialists, such as the clam shrimp *Lynceus gracilicornis*, would disappear in from ponds where hydroperiod was substantially increased. These predictions were based on extensive previous studies of local wetland ponds (for example: assemblages Mahoney, Mort, and Taylor, 1990, DeBiase and Taylor, in prep.; population dynamics and production Taylor and Mahoney, 1990, Leeper and Taylor, 1998; genetics Boileau and Taylor, 1994; life histories Taylor, Wyngaard, and Mahoney, 1990, Medland and Taylor, 2001).

For the insects, by analogy to microcrustaceans, we expected that species richness would increase with hydroperiod and with conversion from forested to herbaceous vegetation. We predicted that richness of large predatory insects, such as odonates, would increase with hydroperiod and that new species would colonize rapidly. These predictions were based on a few previous studies (assemblages McClure, 1994; population dynamics and production Schalles and Shure, 1989, Leeper and Taylor, 1998; life histories Cross, 1955, Kondratieff and Pyott, 1987).

Preliminary Findings.

Responses to treatment. Restoration treatments were partially completed in 16 ponds in 2001, while 4 ponds were left untreated as controls. Because the designated set of study ponds changed after we

began to sample, we have three years of pre-treatment data for 17 of the 20 ponds, 1 yr for 1 pond, and none for 2 ponds. Most of the treatments were completed in January 2002. The drought that began in late 1998 continued through 2002: one of the control ponds did not hold water on any post-treatment sampling dates in 2001-2002. All microcrustacean samples from February 2001 to December 2004 have been processed and analyzed. Processing of macroinvertebrate samples is underway.

Although the ditches affected the timing and duration of inundated periods, all of the basins supported functioning wetland ponds during the pre-treatment phase of the project (Dietz, 2001; Dietz et al., 2001).

Restoration extended the hydroperiods in all but two of the treatment ponds (Fig. 4.1). Species richness (3 to 35 species per pond before treatment; 12 to 37 species per pond after treatment) increased substantially in most of them. Responses were quicker than we predicted. Because all of the basins did support functioning wetland ponds before restoration, some species may have been present but dormant during the pre-treatment phase. Some may have been inadvertently imported during treatment activities. We have not yet detected loss of ephemeral hydroperiod specialists or responses to vegetation treatments.

The most serious limitation to the use of microcrustaceans in restoration studies has probably been the lack of baseline data for interpreting the results. Our protocols yielded many species (86 total) with modest effort. The taxonomy is manageable, we believe, especially among the wetland pond specialists. Because microcrustacean assemblages are dynamic, we do caution that sampling across seasons is critical. In further analyses, we will evaluate other metrics of response, such as indicator taxa or trajectories of assemblage composition (see Philippi, Dixon, and Taylor, 1998).

Figure 4.1. Response of microcrustacean species richness to restoration. Change in hydroperiod index is the difference between numbers of sampling dates when the pond held water during pre-treatment (1998-2000) and post-treatment (2001-2003) phases. Change in number of species is the difference between totals for the pre- and post-treatment phases. Three ponds are omitted because we lack complete pre-treatment data.

Effects of Hydrology Restoration and Landscape Pattern on Amphibians in Carolina Bays on the Savannah River Site

Karen Kinkead^{1,2} and David Otis²

¹*South Carolina Cooperative Research Unit, Clemson University, Clemson, SC;*

²*Department of Natural Resource Ecology and Management, Iowa State University, Ames, IA*

Introduction. Commonly, guidelines for the assessment of success in recreated or mitigated wetlands require only the estimation of the survival of the planted trees and herbaceous wetland species. However, survivorship and growth of vegetation does not necessarily indicate functional success of the site. Therefore, more meaningful prescriptions are needed to define and evaluate success in restored or regenerated wetland areas. One recognized function of wetlands is in providing habitat for wildlife. Amphibians are sensitive within a narrow range of environmental conditions due to their permeable skin and biphasic life cycles, and many species are obligate wetland breeders. Therefore, amphibians are a logical choice to consider as a metric for restoration success due to their life history characteristics. Carolina bays serve as an important amphibian breeding sites. Many amphibians are philopatric and move less than two hundred meters from the breeding area. The temporal and spatial processes that affect colonization of restored sites have not been rigorously investigated.

This study examines the effects of hydrology and spatial context within a landscape of Carolina bays and neighboring wetlands on amphibian communities at the Savannah River Site. Variability in population and community parameters of amphibian species in healthy Carolina bay wetlands will be modeled for restoration and mitigation objectives.

Methods. In 1999, drift fences for capturing herptofauna were constructed around 16 bays which represent 5 bay classifications for this study: 6 bays to be restored (representing 2 treatment classes), 3 bays to serve as controls, 3 bays to serve as reference sites, and 4 bays that represent potential sources for recolonization of restored bays. Because the vegetation treatments may take 20+ years to reach a mature growth stage, the treatment bays this study is examining were chosen based on distance to other wetlands, which may serve as a source for re-colonization. Three of the treatment bays all have wetlands within 300 meters. The other three treatment bays include one with no wetlands within three hundred meters, one with only a large, fish infested lake within 300 meters, and one with a small Carolina bay within 300 meters, however, these two bays are separated by railroad tracks and paved roads.

Amphibians were monitored at the bays using drift fencing with pitfall traps. A total 420 pitfall traps were opened during January of 2000-2003 and closed during July of each year. In addition, 214 upright PVC pipes were used to capture treefrogs. Captured amphibians were given batch marks using Visible Implanted Elastomers (VIE) that designated the year and bay of capture.

Preliminary Findings. A total of 43,342 amphibian captures representing 24 species occurred during the study. While salamander population sizes appeared to decrease overall during the study (at all bay classifications), the restoration appears to have provided additional breeding habitat for several anuran species. Several species of anurans, including eastern spadefoot toads and southern

toads, appear to have responded well to the restoration efforts with respect to both increased adult breeding attempts and the number of metamorphic juveniles leaving the breeding bays.

The decrease in salamander populations at all treatment classifications during the last 2 years of the study was most likely attributable to the weather conditions, which included a severe drought during 2002 and a delay in the rainfall of 2003 resulting in drought-like conditions during the normal salamander breeding season. The only statistically significant result determined by a repeated measure ANOVA was that the bays that underwent additional logging in the buffer experienced a significantly larger decrease in salamander populations than did the restored sites that were not logged in the buffer area. It is expected that these areas will recover in terms of salamander populations but a longer monitoring period is recommended to support this assumption.

This study indicates that amphibians are a good metric with which to quantify wetland restoration success. More amphibians used the restored areas post-restoration. This study has shown that salamanders may be more susceptible to the restoration techniques (i.e. the use of heavy equipment) than anurans, suggesting that more than 3 years may be needed for salamander populations to recover from the process of habitat restoration. One potential drawback for using amphibians as a metric for success is the natural variation in amphibian breeding numbers. There is very high annual amphibian species composition and abundance among Carolina bays, making it difficult to distinguish a failed restoration from a drought year. Post-restoration monitoring needs to continue over a time frame long enough to account for normal variation in weather patterns, which heavily influence amphibian breeding, such that restoration assessment is not completely confounded with abnormal weather patterns.

On a shorter timeframe, anuran species, toads in particular, may provide a more immediate indicator of the potential success of a wetland restoration project. These species appear to require less time to recover from impacts associated with restoration and need shorter hydroperiods to complete breeding and juvenile metamorphosis. Combining information, from both anurans, as a short term indicator, and salamanders, as a long term indicator, should provide ample information as to the success of a wetland restoration project.

The Influence of Buffer Zone Management Techniques on Carolina Bay Herpetofauna

Joseph A. Ledvina¹ and Hugh G. Hanlin^{1,2}

¹*Department of Forestry and Natural Resources, Clemson University, Clemson, SC*

²*Department of Biology and Geology, University of South Carolina Aiken, Aiken, SC*

Introduction. Many of the thousands of acres of managed forests on the Savannah River Site (SRS) near Aiken, South Carolina, are contiguous with Carolina bays, isolated wetlands unique to the southeastern Atlantic Coastal Plain. At least 34 species of amphibians have been reported from Carolina bays on the Savannah River Site, and 31 of these are terrestrial for a portion of their life cycle (Schalles et al., 1989; Gibbons and Semlitsch, 1991). These temporary ponds and isolated wetlands provide critical habitat for many vertebrate and invertebrate species in upland landscapes. The Department of Energy implemented a wetland mitigation and banking program with 16 bays in 2000-2001. The program was initiated as an experimental study of two interior (harvested and converted to cypress-tupelo, or harvested and converted to herbaceous vegetation) and two buffer treatments (open canopy burned pine savannah, and closed canopy mixed pine hardwood) replicated as alternative end points for restoration. Monitoring was conducted to address questions about the dynamics of response by wetland dependent species to restoration and the impact of vegetation structure and composition in the surrounding uplands on the species composition and abundance of the wetland dependent community.

The initial questions associated with monitoring the herpetofauna community included the expectation that restored bays would support a greater richness and abundance of amphibians than control (i.e., non-restored) bays, but less than reference bays. It was expected that richness and abundance would increase over time following restoration as a result of the improved hydrology and immigration of species, but only two full years of monitoring data have been obtained. Lastly, the contrasting upland landscapes in terms of canopy cover and burning were expected to alter the composition of the herpetofauna community. Many amphibians are philopatric and move less than 200 m from their wetland breeding sites. Therefore, alteration of terrestrial habitats, particularly those contiguous with these natural wetlands, may affect the life histories of the amphibian species. Many reptiles which may utilize aquatic habitats for food and cover move to upland sites for nesting. Forested buffers adjacent to isolated wetlands may also be important to fully terrestrial species. The relationship of forest habitat to herpetofaunal abundance and distribution has been shown to vary with the amount of coarse woody debris, leaf litter depth and type humus/soil acidity, and hardwood shrub abundance. However, the relationship of overstory vegetation structure and composition to abundance and community structure of herpetofauna is unclear, since most studies have shown that near surface conditions of shrub cover, litter duff, and debris are more important habitat variables for many species.

Methods. In the ongoing Carolina bay restoration project at SRS, formerly ditched and drained Carolina bays are being restored. In the process, the upland bay margins within 100 m have been manipulated to represent alternative natural landscapes at SRS. Restoration treatments include establishment of either forested or herbaceous bay interiors (both harvested and disturbed) and either non-harvested mixed pine-hardwood or harvested, fire maintained pine savannah in surrounding upland margins. Additional ditched bays remain unrestored as controls (termed control

bays), and natural unditched bays serve as reference bays. The restoration project provides a unique opportunity to study the dynamics of herpetofauna recovery following restoration and the influence of upland forest structure on Carolina bay herpetofauna communities. Additional monitoring will serve to document the changes in the community structure by long-term monitoring of the adjacent upland forests following treatment. Direct comparison or contrasts between restored and reference bays, restored and control bays, and restored bays with contrasting margin treatments can be obtained and related to hydrology, vegetation structure and composition.

In 2001 six study bays were selected with regard to the type of buffer vegetation: three fire-maintained pine savannahs (around bays # 126, # 5092, and # 5135) and three mixed pine-hardwood (around bays # 5071, # 5128, and # 5204). Upland forests surrounding three control bays (# 108, #118, and # 5055) and three reference bays (#79, #153, and #5048) were also monitored. Two herpetofaunal sampling arrays were established within the 100 m upland buffer zone of each bay during 2001. Each sampling array contained the following standard components: (1) Three 15 m aluminum drift fences positioned relative to one another to form a "Y" configuration with pitfall traps (19-liter buckets) sunk to ground level at the ends on both sides of each fence; (2) double-ended 1/4" mesh hardware cloth funnel traps placed at the center and on each side of each drift fence; (3) three vertically positioned PVC pipes placed within 10 m of the ends of each drift fence to attract tree frogs; and (4) an array of two individually numbered plywood and galvanized tin coverboards (0.61 m X 1.22 m) placed within 10 m of the ends of each drift fence. Arrays were checked every other day from January through July, 2002 and 2003.

A ground survey of vegetation characteristics of each buffer site was conducted in August, 2001, prior to treatment and again in August, 2002, 2003, and 2004 following harvest treatment. That data is currently being analyzed. Vegetation sampling of the upland buffers centered at each herpetofaunal sampling array included over-story, mid-story, ground cover, leaf litter depth, and course woody debris. Vegetation data from upland forests will provide the opportunity to test the hypothesis that landscape features of forest margins surrounding Carolina bays influence amphibian community structure and colonization of restored bays. Specifically, I wished to determine (1) if there are differences in specific herpetofaunal population or community parameters among forests exhibiting different vegetation structure; (2) if bay attributes like hydro-period, rather than bay margin vegetation structure, dominate variance in herpetofaunal community structure; and (3) if there is a correlation between vegetation parameters and herpetofaunal parameters.

Preliminary Findings. Delays in harvesting precluded drift fence construction in time for Spring 2001 sampling. However, from 1 January 2002 to 31 July 2004 a total of 36,763 individual amphibians and reptiles representing 62 species were captured (see Tables 1 and 2). Amphibians comprised 95.6% of the total captures. There were 20 species of frogs and toads ($n=32,661$; 88.8% of total captures) and seven species of salamanders ($n=2474$; 6.7% of total captures). The eastern spadefoot (*Scaphiopus holbrookii*) was the most frequently captured frog species ($n=16,859$, 45.9% of total captures), followed in abundance by the southern toad (*Bufo terrestris*; $n=10,175$, 27.7% of total captures). The two most abundant salamanders were the mole salamander (*Ambystoma talpoideum*; $n=1820$, 5.0% of total captures) and the marbled salamander (*Ambystoma opacum*; $n=371$, 1.0% of total captures). Thirty-five species of reptiles ($n=1008$) comprised only 2.7% of the total captures. Seven species of lizards ($n=758$) represented 75.2% of the reptile captures (2.0% of total captures).

The extreme drought during 2000-2002 likely had a strong influence on overall captures and thus minimized difference among treatments for the first two years. A comparison of mean species richness and abundance of both amphibians and reptiles among treatments revealed no significant differences between reference, control and restored bays, or between upland buffers being managed for pine savannahs and any of the other three treatments in which the upland forest margins were not harvested. Although the differences were not significant, there was a trend toward more species being captured in forested buffers surrounding reference bays, with the least number of species being captured in buffers where no harvest took place, neither interior nor buffer. The greatest total number of individual captures occurred in buffers surrounding bays in which only the interior was harvested. This may have been due to the movement of animals into the surrounding uplands from the disturbed bay interiors. For buffers surrounding bays in which harvesting was conducted, the relative abundance of both amphibians and reptiles was higher in sampling arrays proximal to the nearest wetlands than in those distal. Although these are preliminary data, this suggests that movements of herpetofauna from disturbed wetlands may be directional.

There are two factors unrelated to design that make it difficult to draw strong conclusions from the preliminary data. Amphibian populations often show annual, dramatic fluctuations simply due to climatic conditions (Pechman et al. 1991). In 2002 South Carolina was in the grips of an historic six-year drought. Not surprisingly, although captures at all sites was low, the most captures occurred at those sites which retained moisture in the Carolina bay for the longest period of time, regardless of treatment. The following year the area experienced a record rainfall resulting in significantly more captures, including one rare, sensitive species (*Rana capito*) which was captured only in one reference bay (#79) and all bays in which the buffer had been harvested (#126, #5092, and #5135). However, no conclusions can be drawn regarding possible correlations without additional sampling. Rainfall in early spring of 2004 was once again far below average, though most of the Carolina bays had retained moisture from the previous year. Although the amount of water in the bays facilitated the possibility that the larvae of most amphibian species could complete metamorphosis, the lack of rainfall during key breeding periods likely reduced reproductive migrations. At a minimum, an additional year of sampling will be required to ascertain any trends that may have resulted from restoration and the forest treatments.

A second factor is related to the timing of prescribed burns. There were delays in the burning schedule for those bays in which the buffers are being managed for fire-maintained pine savannahs. Burning did not occur until the summer of 2003 at the end of the sampling period. Our data, then, does not fully reflect the effects of prescribed burns on these herpetofauna. An additional year of sampling, however, would provide a valuable opportunity to assess those effects; we would have two years of data from harvested, pre-burned sites as a baseline to compare with numbers during the first two years following a prescribed burn.

Funding has been secured to continue this research one more year. Not only will this allow us to complete the analysis of the vegetation data for any possible correlations with herpetofaunal distribution or community structure, but it may allow us to assess the confounding factors of dramatic climatic fluctuations and prescribed burning.

From 1 January to 31 July 2002 a total of 1312 individual amphibians and reptiles representing 36 species were captured (Table 5.1). Amphibians comprised 86.4% of the total captures. There were 12 species of frogs and toads (n=998; 76.1% of total captures) and five species of salamanders

(n=136; 10.4% of total captures). The southern toad (*Bufo terrestris*) was the most frequently captured frog species (n=576, 43.9% of total captures), followed in abundance by the eastern narrowmouth toad (*Gastrophryne carolinensis*; n=306, 23.3% of total captures). The spotted salamander (*Ambystoma maculatum*) was the most frequently captured salamander (n=49, 3.7% of total captures). Nineteen species of reptiles (n=178) comprised only 13.6% of the total captures. Seven species of lizards (n=146) represented 82.0% of the reptile captures (11.1% of total captures).

Table 5.1. Herpetofaunal species richness in Carolina bay buffer zones by taxonomic group and treatment

Abundance	<i>Control</i>				<i>Mixed</i>				<i>Pine</i>				<i>Reference</i>				
Group	108	118	5055	Control	5071	5128	5204	Mixed	126	5092	5135	Pine	79	153	5048	Ref.	Total
Frog	294	1447	2019	3760	4303	5073	1543	10919	692	6204	785	7681	7189	1125	1678	9992	32352
Salamander	82	24	345	451	35	323	242	600	45	136	134	315	361	373	374	1108	2474
Treefrog	13	1	7	21	112	22	52	186	28	8	3	39	19	17	27	63	309
Amphibian	389	1472	2371	4232	4450	5418	1837	11705	765	6348	922	8035	7569	1515	2079	11163	35135
Mammal	31	70	37	138	45	48	77	170	57	111	40	208	40	41	23	104	620
Lizard	86	37	55	178	49	60	39	148	33	78	96	207	76	62	87	225	758
Snake	22	13	9	44	21	20	20	61	17	11	24	52	35	12	13	60	217
Turtle			1	1		1	6	7	3	4	1	8	5	7	5	17	33
Reptile	108	50	65	223	70	81	65	216	53	93	121	267	116	81	105	302	1008
Grand Total	528	1592	2473	4593	4565	5547	1979	12091	875	6552	1083	8510	7725	1637	2207	11569	36763

Richness	<i>Control</i>				<i>Mixed</i>				<i>Pine</i>				<i>Reference</i>				
Group	108	118	5055	Control	5071	5128	5204	Mixed	126	5092	5135	Pine	79	153	5048	Ref	Total
Frog	4	8	8	8	8	8	9	11	8	9	10	12	12	10	8	13	14
Salamander	5	4	5	6	4	6	4	6	3	5	4	5	3	4	4	5	7
Treefrog	5	1	3	5	3	4	5	5	3	4	1	4	3	6	5	6	6
Amphibian	14	13	16	19	15	18	18	22	14	18	15	21	18	20	17	24	27
Lizard	7	5	7	7	7	7	6	7	6	7	7	7	7	7	7	7	7
Snake	10	6	7	13	10	8	8	14	7	5	9	14	9	9	8	13	23
Turtle	0	0	1	1	0	1	1	1	3	2	1	3	2	1	3	4	5
Reptile	17	11	15	21	17	16	15	22	16	14	17	24	18	17	18	24	35
total	31	24	31	40	32	34	33	44	30	32	32	45	36	37	35	48	62

Bird Response to Carolina Bay Restoration

John C. Kilgo¹ and Stephen J. Czapka¹

¹USDA Forest Service, Center for Forested Wetlands Research, New Ellenton, SC;

Introduction/Methodology. We assessed bird response to bay restoration and to two alternative goals within the 100-m upland margin surrounding each bay (pine savanna versus mixed pine-hardwood forest). We compared bird use of restored bays to that of control (drained, un-restored) and reference (undisturbed) bays. We compared the following among restoration treatments: density of breeding forest birds and frequency of use by foraging wetland birds (waterfowl and wading birds).

Preliminary Findings. Drainage of bays does not seem to affect birds using the uplands surrounding bays but it eliminates wetland birds, as this group was less abundant in drained bays (pre-restoration) than in herbaceous and forested reference bays.

Table 6.1. Mean pre-restoration bird observations.

<i>Birds</i>	<i>Drained (mean ± SE)</i>	<i>Reference (mean ± SE)</i>	<i>P</i>
Total no. species	21.0 ± 1.0	23.5 ± 1.8	0.220
Total no. individuals	98.7 ± 8.0	116.3 ± 11.3	0.202
Wetland Species			
No. species	0	8	N/A
No. individuals	0	26	0.004
Upland Species			
Brown-headed Nuthatch (<i>Sitta pusilla</i>)	0.69 ± 0.51	0.54 ± 0.75	0.631
Indigo Bunting (<i>Passerina cyanea</i>)	2.23 ± 0.79	3.68 ± 1.23	0.309
Summer Tanager (<i>Piranga rubra</i>)	3.03 ± 0.61	3.22 ± 0.72	0.840

Bird response to restoration was monitored during the breeding seasons for one year pre-restoration (2000) and four years post-restoration (2001-2004) using strip transects through the bays and upland buffers.

Species richness and abundance were lowest in restored bays prior to their restoration, but by the first breeding season post restoration, both community indices were higher in restored bays than in either reference bays or control bays. This was likely due to a response by upland birds, which dominate the avian communities of the treatment areas, to the buffer treatments (see below).

Figure 6.1. Species richness and abundance of restored, reference and control bays.

Prior to restoration (2000), no wetland birds were observed in any bay under study, including reference bays, presumably due to drought conditions. During the first year post-restoration, restored bays supported approximately half as many wetland birds as reference bays, while unrestored bays (controls) had none. In 2002, water levels were again low and no wetland birds were observed in the bays. However, in both 2003 and 2004 when the drought had ended, restored bays supported as many or more wetland birds than reference bays, both of which supported more than control bays.

Figure 6.2. Number of wetland birds observed in restored, reference and control bays.

Species richness and the abundance of most bird species (primarily upland species) responded to the margin treatments also, with abundance differing over time. Although species richness in pine savanna buffers declined initially due to the understory disturbance from treatment harvesting, most species were more abundant in pine savanna buffers by the second year post-treatment. Three general responses were observed, as typified by Brown-headed Nuthatch, Indigo Bunting, and Red-eyed Vireo. Species that nest and forage in the pine canopy (e.g., Brown-headed Nuthatch), responded positively to the removal of hardwoods in the first year post-treatment and their abundance remained high in the savanna treatment. Species that nest in the understory of the pine forest (e.g., Indigo Bunting), responded positively in the second year post-treatment, as understory structure increased with the canopy thinning, and their abundance remained high. Species that use hardwoods and tend to avoid open savanna and edge habitat (e.g., Red-eyed Vireo) declined overall, but more dramatically where hardwoods were removed (i.e., in pine savanna buffers).

Figure 6.3. Influence of margin treatment on bird usage.

Despite severe drought that limited restoration of the hydrology of Carolina bay wetlands, several species of birds responded positively to bay restoration activities. Wetland birds, which had not used the bays prior to restoration, were recorded whenever water was ponded at numbers at least as great as in undisturbed bays. Additionally, upland birds of the shrub-scrub community, a group of considerable conservation concern, responded positively to management of the upland pine savanna buffers surrounding bays.

Literature Cited

- Barton, C. 2003. The Carolina Bay Restoration Project: Status Report 2000-2002. Westinghouse Savannah River Company. Aiken, SC. ESH-ERR-2003-00352. 61 pp.
- Barton, C., and J. Singer. 2001. Carolina Bay Restoration: SRS Wetland Mitigation Bank Documentation. Center for Forested Wetlands Research, Charleston, SC. 104 pp.
- Boileau, M. G., and B. E. Taylor. 1994. Chance events, habitat age, and the genetic structure of pond populations. *Archiv für Hydrobiologie* 132: 191-202.
- Cross, W. H. 1955. Anisopteran Odonata of the Savannah River Plant, South Carolina. *Journal of the Elisha Mitchell Scientific Society* 71: 9-17.
- DeBiase, A. E., and B. E. Taylor. New records of Anostraca and Conchostraca in South Carolina. *The Southeastern Naturalist*. (Accepted pending revision; revision submitted)
- DeBiase, A. E., and B. E. Taylor. Freshwater microcrustacean assemblages: diversity and differentiation on a hydrologic gradient in South Carolina. *Freshwater Biology*. (In review)
- Dietz, S. E., D. P. Batzer, B. E. Taylor, and A. E. DeBiase. 2001. Invertebrates of twenty ditched Carolina bays scheduled for restoration. In K. J. Hatcher, editor, *Proceedings of the 2001 Georgia Water Resources Conference*, pp. 321-324. Institute of Ecology, University of Georgia, Athens, Georgia.
- Dietz, S.E. 2001. Variability of invertebrate communities in seasonally flooded, woodland ponds. M.S. thesis, University of Georgia, Athens, Georgia.
- Gibbons, J. W. and R. D. Semlitsch. 1991. *Guide to the Reptiles and Amphibians of the Savannah River Site*. The University of Georgia Press, Athens, GA, USA.
- Kirkman, L.K., R.F. Lide, G.R. Wein, and R.R. Sharitz. 1996. Vegetation changes and land-use legacies of depression wetlands of the western coastal plain of South Carolina 1951-1992. *Wetlands*. 16:564-576.
- Kondratieff, B. C., and C. J. Pyott. 1987. The Anisoptera of the Savannah River Plant, South Carolina, United States: thirty years later. *Odonatologica* 16: 9-23.
- Leeper, D. A., and B. E. Taylor. 1998. Abundance, biomass and production of aquatic invertebrates in Rainbow Bay, a temporary wetland in South Carolina, USA. *Archiv für Hydrobiologie* 143: 335-362.
- Mahoney, D. L., M. A. Mort, and B. E. Taylor. 1990. Species richness of calanoid copepods, cladocerans and other branchiopods in Carolina bay temporary ponds. *The American Midland Naturalist* 123: 244-258.

- Medland, V. L., and B. E. Taylor. 2001. Strategies of emergence from diapause for cyclopoid copepods in a temporary pond. *Archiv für Hydrobiologie* 150: 329-349.
- McClure, M. A. 1994. Aquatic insects of five clay-based Carolina bays in South Carolina. M.S. thesis, Clemson University, Clemson, South Carolina.
- Pechmann, J. H. K., D. E. Scott, R. D. Semlitsch, J. P. Caldwell, and J. W. Gibbons. 1991. Declining amphibian populations: the problem of separating human impacts from natural fluctuations. *Science* 253:892-895.
- Philippi, T. E., P. M. Dixon, and B. E. Taylor. 1998. Detecting trends in species composition. *Ecological Applications* 8: 300-308.
- Riekerk, H. 1989. Influence of silvicultural practices on the hydrology of pine flatwoods in Florida. *Water Resources Research*. 25: 713-719.
- Schalles, J. F, and D. J. Shure. 1989. Hydrology, community structure, and productivity patterns of a dystrophic Carolina bay wetland. *Ecological Monographs* 59: 365-385.
- Schalles, J. F., R. R. Sharitz, J. W. Gibbons, G. J. Leversee and J. N. Knox. 1989. *Carolina bays of the Savannah River Plant, Aiken, South Carolina*. National Environmental Research Park Program Publication, Savannah River Ecology Laboratory, Aiken, South Carolina. SRO-NERP-18. 70 p.
- Sun, G, H. Reikerk and V. Kornhak. 2000. Ground-water table rise after forest harvesting on cypress-pine flatwoods in Florida. *Wetlands*. 20: 101-120.
- Taylor, B. E., and D. L. Mahoney. 1990. Zooplankton in Rainbow Bay, a Carolina bay pond: population dynamics in a temporary habitat. *Freshwater Biology* 24: 597-612.
- Taylor, B. E., G. A. Wyngaard, and D. L. Mahoney. 1990. Hatching of *Diaptomus stagnalis* eggs from a temporary pond after a prolonged dry period. *Archiv für Hydrobiologie* 117: 217-278.
- U.S. Department of Energy, Savannah River Operations Office. 1997. Memorandum of Agreement for the Savannah River Site Wetland Mitigation Bank. DE-MA090-96SR18577. Savannah River Operations Office, Aiken, SC. 33p.

Appendix I

Photographs Outlining Project Progress

**SRS Carolina Bays
1951 Condition**

Pre-Restoration Condition

Bay 5128

0.1 0 0.1 0.2 Miles

Carolina Bay Restoration – Timber Harvest

Pre-Harvest

During Harvest

During Harvest

After Harvest

Carolina Bay Harvest, 2000-2001

**Mixed Pine/Hardwood, Bay
5071**

**Open Canopy Pine Savanna, Bay
126**

Forest Planting

Herbaceous Planting

Hydrologic Disturbance

Drainage Ditch

Active Drain

Ditch Repair

Before and After Harvest

Bay 5190

Bay 5128

Herbaceous Plots

5204

126

Bay Burning

131

Data Logger and Piezometer Nest (5184)

Throughfall Collector (5011)

Hydrology Monitoring Equipment

Sample Well (5135)

Seedling Growth

4 yr Tupelo

4 yr Cypress

Status Summer 2005

Bay 124

Bay 5128