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Swan Lake Habitat Rehabilitation and Enhancement Project: Post-Project Monitoring of Water Quality, Sedimentation, Vegetation, Invertebrates, Fish Communities, Fish Movement, and Waterbirds

James E. Garvey
Southern Illinois University Carbondale

John H. Chick
Illinois Natural History Survey

Michael W. Eichholz
Southern Illinois University Carbondale

Greg Conover
U.S. Fish and Wildlife Service

Ronald C. Brooks
Southern Illinois University Carbondale

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Swan Lake Habitat Rehabilitation and Enhancement Project:

Post-Project Monitoring of Water Quality, Sedimentation, Vegetation, Invertebrates, Fish Communities, Fish Movement, and Waterbirds

<u>Principle Investigators</u>:

James E. Garvey[1], John H. Chick[2],
Michael W. Eichholz[3], Greg Conover[4], Ronald C. Brooks[1]

<u>Graduate Research Assistants</u>:

Laura Csoboth[1], Kelly DeGrandchamp[1], Douglas Schultz[1],
Darin Grulkowski[3], Randy Smith[3], Thomas R. Timmerman[2]

<u>Research Assistants</u>:

Chad R. Dolan[2], James T. Lamer[2], Erin Hotchkiss[3], Brett Tevini[3], Colby Wrasse[4],
Nathan Caswell[4], Darrin L. Martin[1]

[1]Fisheries & Illinois Aquaculture Center and Department of Zoology
Southern Illinois University at Carbondale
Carbondale, IL 62901-6511

[2]Great Rivers Field Station
Illinois Natural History Survey
8450 Montclair Ave
Brighton, IL 62002

[3]Cooperative Wildlife Research Lab
Southern Illinois University,
Carbondale, IL 62901-6504

[4]U.S. Fish and Wildlife Service
Carterville Fishery Resources Office
Marion, Illinois

Prepared for the St. Louis District, U.S. Army Corps of Engineers

FINAL REPORT

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SECTION 1: EXECUTIVE SUMMARY

Rivers throughout the world are continually being modified by humans for many uses including navigation, water supply, agriculture, and flood control. One such river region that has felt the marked impact of human activities is the ecologically and economically important confluence area of the Illinois and Mississippi Rivers. This area contains a diverse complex of backwaters, supports recreational fishing and hunting, contributes to downstream river productivity, sustains a commercial fishery, and promotes biological diversity near a major metropolitan area. We were tasked with evaluating the response of a major 1,100-ha Illinois River backwater, Swan Lake (Figure 1) of the Two Rivers National Wildlife Refuge (NWR), to its rehabilitation as an US Army Corps of Engineer's Habitat Rehabilitation and Enhancement Project (HREP) authorized through the Environmental Management Program (EMP) of the Water Resources Development Act (WRDA).

Before European settlement of the confluence region, most of Swan Lake was at a higher elevation than the Illinois River during its base flow, allowing regular spring flooding and summer drainage plus drying (Heitmeyer and Westphall 2007). The advent of agriculture in the region increased sediment loading in the river basin. Further, the construction of Lock and Dam 26 downstream across the Mississippi River in 1938 raised the summer river elevation by about 9 feet, increasing the surface area of Swan Lake by six times its historic extent. Although aquatic (but not terrestrial) biological production within the river area temporarily increased following inundation, lost vegetation diversity, reduced seasonal pulses in primary and secondary production, increased sedimentation, and declining depth rapidly began to compromise the backwater's ecosystem services. By the late 1980s, Swan Lake was identified as a candidate for rehabilitation through the EMP to mitigate these environmental insults.

In the early 1990s, baseline monitoring data were collected in Swan Lake before the HREP so that responses of water quality, vegetation, invertebrates, fish, and waterbirds could be compared to pre-project conditions. Following this monitoring effort, the northernmost portion of the backwater (i.e., Upper Swan Lake plus Fuller Lake) was leveed through the HREP, with management orchestrated by the State of Illinois primarily as a moist-soil unit. The lower portion of Swan Lake was divided by the HREP into two large compartments by a cross levee (Figure 1). These compartments also were isolated from the river by a levee to prevent regular inundation and thus reduce sediment loading. Each compartment (hereafter, Middle Swan [MS] and Lower Swan [LS]) could be connected to the river by a short, narrow (5-m wide) channel with a stop-log, water–control structure (Figure 1). Islands were constructed across the compartments to reduce waves caused by wind across the lake fetch. In LS, deepwater areas were excavated at the base of the river levee to provide fish overwintering habitat. Water levels within MS and LS could be lowered by closing the water-control structures and redirecting water back into the river via a pump within each compartment. Middle Swan and LS are managed under the direction of the Two Rivers NWR.

Following the construction phase of the HREP in summer 2002, flocculent sediments in MS and LS were consolidated by draining the compartments and allowing them to dry. During 2003-2005, MS was again partially dewatered each summer and flooded during fall through spring. In contrast, LS was reconnected to the river via the open water control structure and its associated channel. During summer 2004, a substantive flood occurred, topping the river-side levee and temporarily inundating both MS and LS (see Section 3). Water level management of MS during 2003-2005 was intended to create a wet-dry cycle that promoted moist-soil vegetation for waterfowl. Lower Swan was left “open” (i.e., connected) to the river to encourage fish reproductive activity in spring, fish foraging during summer, and fish resting in winter. In 2006, management of MS remained consistent. However, LS was disconnected from the river in June 2006 and water levels were lowered in the unit until the unit’s pump failed and was removed for repairs.

Water quality, including sediment consolidation and water clarity, were evaluated during 2004-2006. Fish and invertebrate assemblages also were quantified during this 3-year period. Fish movement and waterbirds were monitored during 2004 and 2005. When possible, all sampling was conducted in a manner identical to that implemented in the early 1990s before the HREP. These data were used to evaluate the “success” of the HREP, with some simple expectations:

- Consolidated and compacted lake sediment within MS and LS, creating greater heterogeneity in depth strata and diverse substrate characteristics
- Increased water clarity
- Enhanced submerged and emergent vegetation growth and diversity
- Increased diversity and growth of fish, macroinvertebrates, and waterbirds
- Improved reproductive success of fishes
- Enhanced use of LS by fishes for reproduction, foraging, and wintering

While evaluating the response of Swan Lake to the HREP, other changes within the river region needed to be considered. Asian carp, in particular silver carp (*Hypophthalmichthys molitrix*), became abundant. The interaction of these invasive species with the Swan Lake ecosystem was uncharted territory. Continent-wide waterfowl abundance increased while the HREP was being implemented (see Section 6), which also may have influenced patterns of use. Climatic changes likely also occurred.

Although a primary goal of the HREP was to reduce sediment loading and its negative effects, resuspension of existing sediment and perhaps additional inputs from surrounding tributaries occurred in MS and LS during 2004-2006 (Section 2). Some improvement in sediment depth and water clarity occurred in MS relative to pre-HREP conditions; conversely, LS showed no such positive response. Likely, the regular partial drying of MS facilitated bottom consolidation. The island wind-breaks did not sufficiently reduce mixing of the water column and allow settling to occur. Although all other water quality parameters were biologically acceptable during 2004 through 2006, temperatures in both Swan Lake units were warmer during each summer than in the early 1990s. This pattern may have been climatically driven or more probably related to the fact that the water body was isolated from the thermal influence of the river following the HREP. No submerged or rooted floating vegetation occurred in either MS or LS. The lack of aquatic vegetation in both units was likely due to an absence of a seed bank and tubers, as well as poor water clarity, particularly in LS. In addition, herbivory may have reduced establishment. Emergent vegetation did occur in MS, likely as a function of water level management and improved bottom conditions in this compartment.

Secondary production of macroinvertebrates is one of the most important functions of backwaters for a river catchment because it provides a conduit by which primary production and detrital energy become available to higher trophic levels including birds and fish. Notable improvements in diversity of macroinvertebrates occurred in MS (Section 2), likely as a function of reduced dominance of midge larvae. Abundance of macroinvertebrates during spring appeared to increase in both units (Section 6). These responses were related to improved bottom conditions within MS.

Fish assemblages in MS and LS did not respond positively to the HREP (Sections 2, 5). Most notably, the assemblage of centrarchids present before the HREP were weakly represented. The lack of firm substrate and vegetation negatively affected this group by reducing reproductive success and perhaps foraging. The compartments were still productive for some species including gizzard shad (*Dorosoma cepedianum*), freshwater drum (*Aplodinotus grunniens*), common carp (*Cyprinus carpio*), and emerald shiner (*Notropis atherinoides*). Many of these fishes were small and likely juveniles. Apparent densities of silver or bighead carp (*H. nobilis*) were relatively low in both compartments. No seasonal increases in abundance of any fish species occurred during fall as might be expected if individuals were arriving to avoid adverse river conditions during winter.

A major stumbling block to mitigating the loss of floodplains is ensuring exchange between rivers and their backwaters. In the original design, a compromise was struck by maintaining a seasonal dry-wet cycle in MS and ensuring continuous connectivity in LS. Obviously, the primary goal of bottom consolidation did not occur in LS, placing the management regime of continuous connectivity and relatively constant inundation in question. However, our research did demonstrate that maintaining a conduit between Swan Lake and the river was important

(Sections 3, 4), even though the opening to the backwater was constricted from > 500 m pre-HREP to about 15 m. Trap- and hoop-netting at the LS water control structure revealed no strong seasonal pattern in fish abundance; species composition was similar to that in LS, although more white bass (*Morone chrysops*) were present (Section 3). Species richness was much lower than that occurring in the 1990s (Sheehan et al. 1994). Fish were relatively abundant in nets, with larger fish appearing during spring of each year. This area held drifting invertebrates (see Appendix A) and was dense with young fish, thereby providing foraging opportunities for many species. In addition, the appearance of large fishes each spring suggested that some spawning occurred either within LS or in proximity to the water-control channel in the river.

To refine our understanding of fish use of MS and LS as well as the adjacent river, we conducted a telemetry effort similar to that of Sheehan et al. (1994). Channel catfish, white bass, common carp, and Asian carp were surgically implanted with acoustic transmitters (Section 3). Twenty-five paddlefish also were tagged. We attempted to capture largemouth bass for implantation as well. However, the largemouth bass population that was formerly present in the vicinity of Swan Lake was absent. Stationary, automatic-logging acoustic receivers were placed at the river and lake sides of the water control channel. In addition, receivers were placed about every 10 km along the Illinois River from the confluence to the LaGrange Lock and Dam. Lower Swan was frequented by most telemetered fish, with the highest residency occurring during spring when LS was covered by ice and when temperatures in this backwater were less variable and warmer than the river channel. Paddlefish released into LS rapidly left the backwater through the water control structure and did not return. Asian carp were more frequently found in the Illinois River, and avoided the backwater during the summer months, perhaps because LS exceeded their optimal temperature for growth. These results support those of the monitoring effort, confirming that LS was no longer being used as winter habitat and may only provide refuge when ice cover is present. This area also may provide foraging opportunities when spring productivity is high and temperatures are moderate.

River floodplains and backwater lakes also typically serve as important spawning areas and nursery grounds for fishes. We quantified the production of fish larvae produced in the river and the two lower Swan Lake compartments as well as the drift of larvae between LS and the river through the water control structure (Section 4). The backwater compartments produced at least an order of magnitude higher density of fish larvae than the river. However, the number of larvae drifting into LS was higher than out of the backwater. The flood of 2004 led to a very different species assemblage of larvae in the drift than the drought of 2005; families of fish that depend on flowing water to reproduce were more abundant in the larval assemblage in 2004.

The apparent density of diving and dabbling ducks increased in Swan Lake relative to conditions before the HREP (Section 6). Although this may reflect improved conditions, the continent-wide increase in waterfowl populations also may have contributed. However, ducks were foraging successfully in MS and likely gaining energy for reproduction in spring and migration in the fall. Diets were typically dominated by seeds and plant material rather than by invertebrates. Although spring invertebrate densities increased in MS relative to pre-HREP conditions, the availability of invertebrates may be influenced by their distribution within the flocculent sediments. Perhaps, invertebrates use the deep, unconsolidated sediments as a refuge from predation by birds (see Appendix B).

The monitoring component of the Swan Lake HREP was an ideal example of adaptive management. By comparing responses to expectations we could determine what aspects of the project were successful and evaluate necessary changes:

- The primary goals of consolidating sediments, improving water clarity, and facilitating benthic production were not met, particularly in LS. Both compartments require a regular cycle of drying and compaction to promote benthic production. Further, sediment transport from upland areas adjacent to Swan Lake needs to be curbed (also see Heitmeyer and Westphall 2007), perhaps through improved soil conservation efforts or upstream settling basins. The management goal of maintaining continuous connectivity of LS to the river is obviously not fully compatible with this management need.
- Marginal benthic conditions in MS and LS reduced the success of aquatic life in many ways. Facilitating the ecosystem function and structure of the backwater assemblages hinges not only on maintaining a regular seasonal wet-dry cycle. Other improvements such as reestablishment of vegetation through plantings and increasing depth diversity through dredging (when dry sediments allow) in both compartments should be considered. Given that several species of fishes were absent relative to post HREP, stocking (e.g., largemouth bass) to reestablish recreational opportunities might be considered.
- Although water level management that allows spring inundation of the compartments mimics one component of historical floodplain connectivity with the river channel,

drawing compartments down to reduce water levels, dry sediments, and promote terrestrial vegetation requires (1) shutting water control structures and (2) removing water through pumps. Thus, backwater production in the form of drifting macroinvertebrates and juvenile fishes is entrained within the Swan Lake compartment and subsequently lost to the river. This is clearly incompatible with the life histories of many species of resident river species and may compromise their abundance.

  - The primary goal, of course, is to expose the sediments of both compartments for a sufficient period (e.g., at least 2 years for LS) to compact sediments and prevent their resuspension following inundation. The resulting short-term loss of river productivity will be offset by the future increase in ecosystem services.
  - Following prolonged drying, water- level management of MS and LS should be staggered (i.e., out of phase). While one compartment is experiencing a summer drawdown, the other should remain partially or fully connected to the river to allow seasonal exchange of organisms (Figure 2).
  - Management of water levels at Mel Price Lock and Dam of Pool 26 should consider reducing water depth in the lower Illinois River channel during summer to promote partial drying within connected backwaters (Figure 2).

- Before the HREP, Swan Lake provided winter refuge for fishes that would have likely been incapacitated by the high flow and sub-zero temperatures of the river channel. Sheehan et al. (1994) demonstrated that small-bodied fish are more susceptible due to relatively poor swimming ability in cold water. Deep-water areas (> 3 m) within MS and LS coupled with access during fall through winter (i.e., maintaining an open water control structure) likely would benefit the survival of small-bodied fishes as well as young individuals of larger bodied species. Deep water areas in LS were lost due to sedimentation. We recommend re-establishing deep "holes" in each compartment during times of drying. These areas should be placed in regions of the backwaters in which resuspension of sediments and filling are least likely.

- Habitat rehabilitation projects of this scale and scope are complex; unexpected, surprising results are common. Thus, true adaptive management is required to render these projects successful. Adaptive management mandates sound information from well-designed, consistent, and regular monitoring. Many of the following research recommendations for guiding future management are consistent with the goals of the Navigation Ecosystem Sustainability Program (NESP) of the recently passed (2007) incarnation of WRDA.
  - <u>Water quality and sedimentation</u>. A consistent program of monitoring water clarity and sediment depth within MS and LS needs to be implemented, perhaps on a bi-annual basis. In addition, sediment surveys using hydroacoustics (Figure 3) show some promise for assessing the relative depth of unconsolidated sediments. The design of additional island wind breaks and the addition of tributary sediment traps might be guided by maps of sediment depth and quality in the units.
  - <u>Vegetation and macroinvertebrates</u>. As with water quality, macroinvertebrate and vegetation abundance as well as composition should be monitored on a regular basis. Time-saving techniques for assessing the biomass and distribution of submerged vegetation using hydroacoustics should be explored.
  - <u>Fishes</u>. Many fish species are long-lived and populations will require an extended period to respond to management within Swan Lake. Again, regular monitoring

using consistent methods is necessary. The contribution of the backwater to fish production in the river is not well understood; however, our results on river connectivity confirm that the compartments play an important role in fish reproduction, winter survival, and foraging. Thus, both resident populations in MS and LS as well as those in the adjacent river reach need to be monitored both for abundance as well as basic demographics such as age structure (i.e., reflecting recruitment patterns) and mortality.

- Waterbirds. Like many fishes, waterfowl and other waterbirds are long-lived with complex life histories, for which Swan Lake only may a partial role. Abundance needs to be monitored in a standardized fashion to determine trends in seasonal use (e.g., times of reproduction and migratory staging). In addition, as habitat quality changes with management, the availability and consumption of prey need to be assessed to determine how nutritional ecosystem services change. For example, the importance of macroinvertebrates to waterfowl diets is still unresolved. Is the absence of protein-rich invertebrates in diets of waterbirds due to a behavioral avoidance or a lack of availability of macroinvertebrates in flocculent sediments?
- Ecosystem Services. We evaluated the Swan Lake HREP from a water quality and organismal perspective. However, this backwater system also plays an important role in many related and less directly tangible ecosystem processes including energy transport to the river (e.g., through export versus retention of organic matter, invertebrate secondary production, fish), carbon sequestration, and nutrient cycling (e.g., nitrification versus denitrification). As management of

Swan Lake continues, it would be useful to evaluate its responses from such a process-oriented perspective. In this manner, the contribution of the backwater to adjacent river and terrestrial ecosystems might be quantified, helping guide future HREPs and their management.

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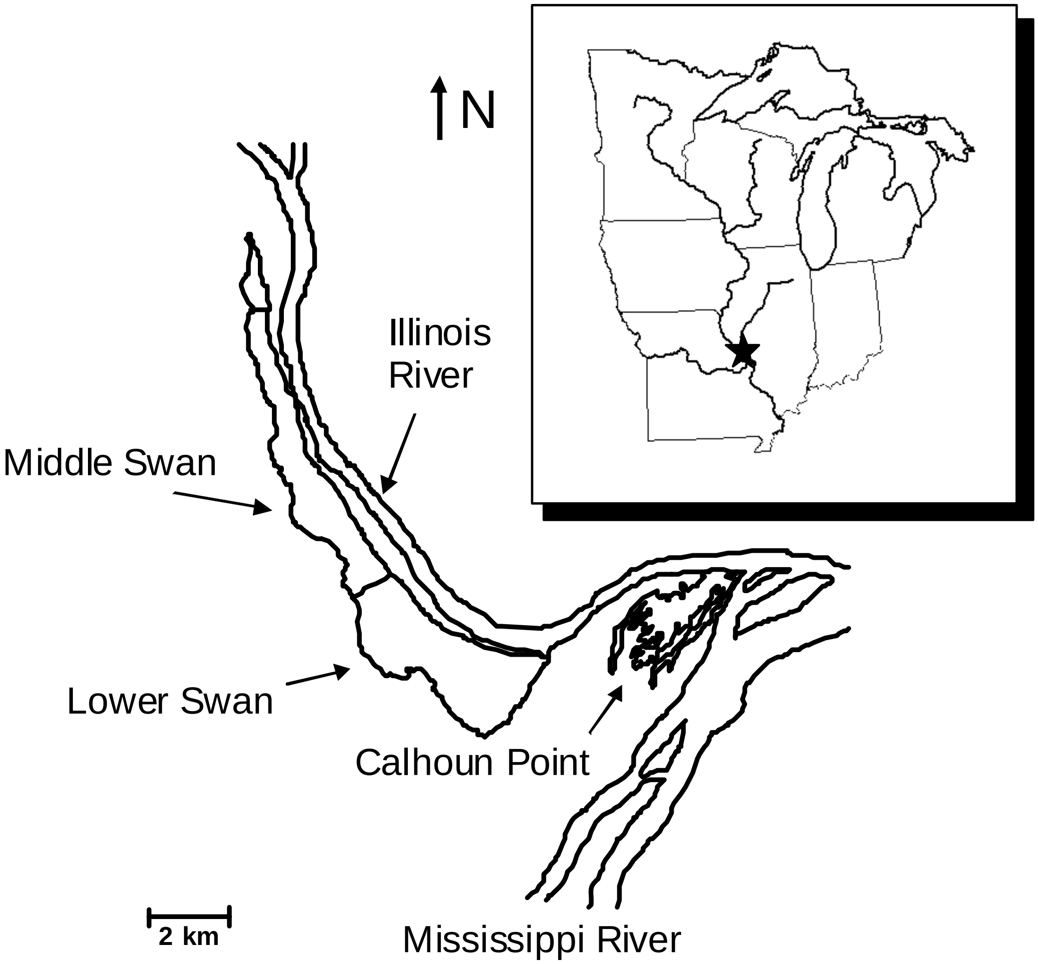


Figure 1. Confluence of the Illinois River and Mississippi River.

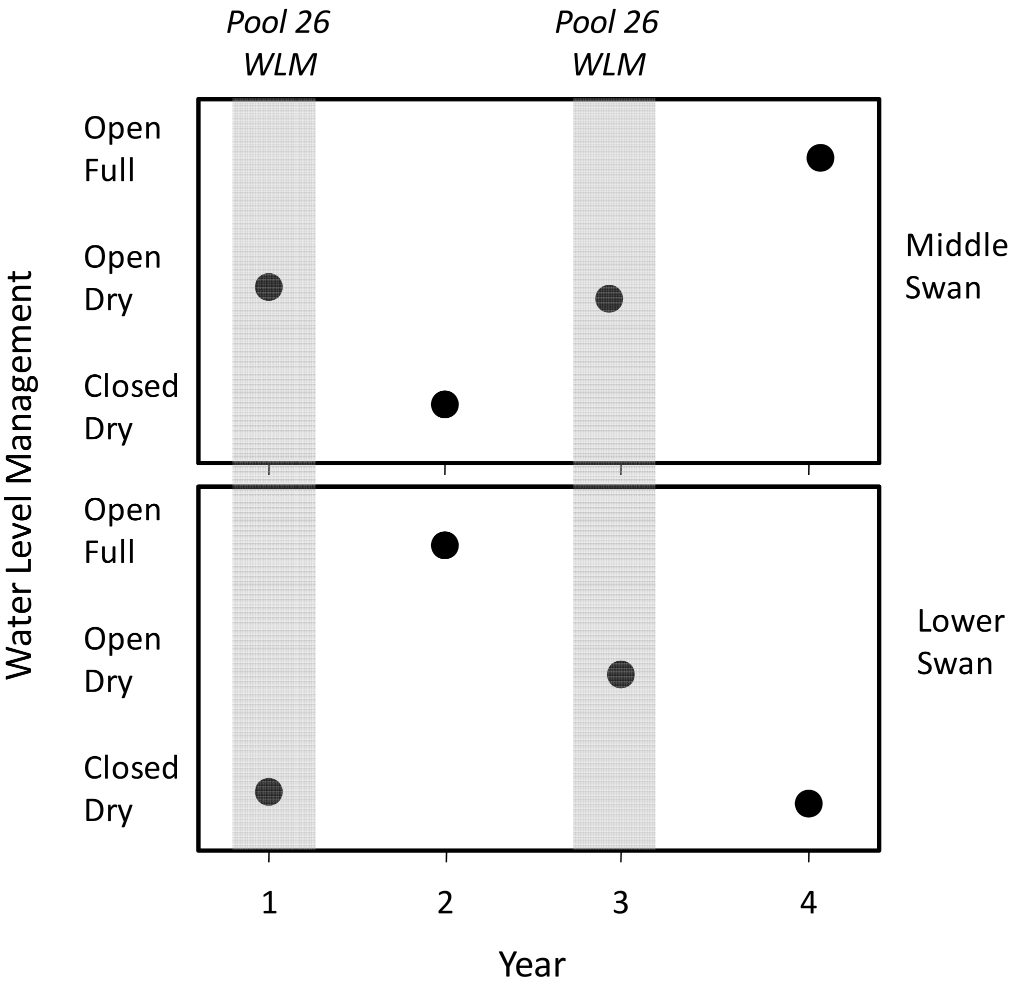


Figure 2. Proposed combined management of water levels within the Middle and Lower Swan compartments by (1) either opening or closing the water control structure within each compartment and (2) raising or lowering the Illinois River channel through management of water released at Mel Price Lock and Dam 26 plus Lock and Dam 25 of the Mississippi River (Pool 26 WLM; see Garvey et al. 2003). "Open Full" condition would occur when the water control structure at the compartment is open and the Illinois River is maintained at full pool. "Open Dry" condition would be achieved by keeping the water control structure open while the Illinois River is lowered temporarily during summer via water-level management at Mel Price. The resulting drop in river elevation relative to Swan Lake would allow water to partially drain from the compartment (see Garvey et al. 2003). "Closed Dry" would occur when the water control structure is closed during spring through fall via stop logs and water fully removed via pumping.

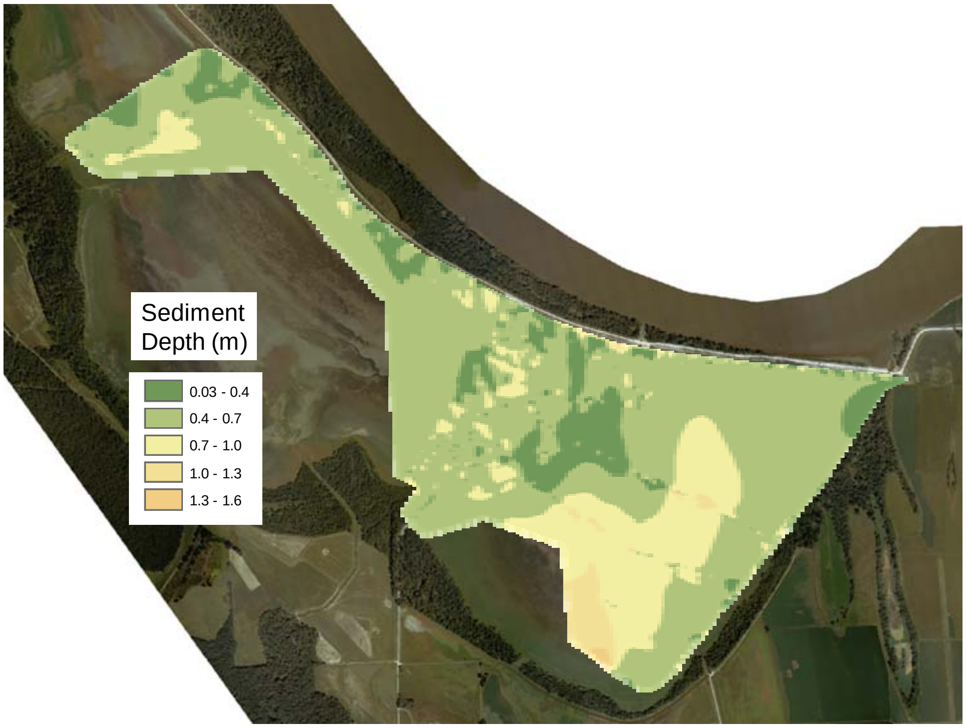


Figure 3. Results of a tentative survey of sediment depths within Lower Swan Lake using a two-frequency Knudsen echosounder during fall 2005. The high frequency transducer (200 kHz) provided an estimate of bottom depths. Simultaneously, a low frequency transducer (28 kHz) penetrated soft sediments. Subtracting the depths generated by the low frequency bathymetry from the high frequency survey provided an estimate of the distribution of soft, unconsolidated sediments. Areas of low sediment depth were near the islands and at the water control structure. See p. 604 for survey of both compartments with a penetrometer.

SECTION 2: MONITORING

Swan Lake Habitat Rehabilitation and Enhancement Project:

Post-Project Ecological Monitoring and Assessment

John H. Chick, Chad R. Dolan, James T. Lamer, and Thomas R. Timmerman

Illinois Natural History Survey

Great Rivers Field Station

8450 Montclair Ave

Brighton, IL 62012

SUMMARY - MONITORING

Swan Lake is a 1,200 ha Illinois River backwater located near the confluence of the Illinois and Mississippi Rivers. Concerns for declining habitat within the lake, principally associated with the accumulation of flocculent sediments, decreased water clarity, and decreased abundance of aquatic vegetation, led to the Swan Lake Habitat Rehabilitation and Enhancement Project implemented by the US Army Corps of Engineers. The main goals of this project (US Army Corps of Engineers 1991) were to: "1) Restore aquatic macrophyte beds and associated invertebrate communities for the benefit of migratory waterfowl, 2) Provide habitat for over winter survival of fish, and 3) Provide habitat for spawning and rearing of fish." Major project attributes designed to achieve these goals included partitioning the lake into three separate units, reducing the connection of the lake with the Illinois River, and the installation of stop-log structures and pumps to allow water level management. The Illinois Natural History Survey's Great Rivers Field Station conducted pre-project monitoring of water quality, sediments, aquatic vegetation, macroinvertebrates, and fishes in Swan Lake from 1992 – 1993 (Theiling et al. 2000). This report details results from post-project monitoring conducted from 2004 to 2006, and direct comparisons of pre- and post-project monitoring to assess ecological goals associated with the habitat project and management of Swan Lake.

Turbidity and Secchi transparency data suggest the habitat project and management strategies used in the middle unit of Swan Lake appear to have led to some increases in water clarity, whereas little improvement was observed in the lower unit of Swan Lake. Water clarity in the middle unit often was at levels low enough (e.g., turbidity ≤ 40 NTU) to allow for the growth of submersed aquatic vegetation. Other water quality parameters remain at suitable levels for most aquatic biota in all three units (lower, middle and upper) of Swan Lake. Our penetrometer results suggest that the habitat project has provided the opportunity to substantially harden lake sediments through draw downs. These data also suggest that draw downs need to be conducted frequently and intensely to obtain desired benefits. Our results suggest that the success of the island groups at reducing wave height and resuspension of sediments was marginal (at best) in the middle unit, and was unsuccessful in the lower unit. Further efforts to reduce wave height likely will be needed to meet project objectives, especially in the lower unit. Our analysis of water depth and lake elevation did not suggest any overall changes in lake depth (either filling in or deepening) across years during post-project monitoring, but we caution that these methods may not be the best strategy to assess project goals for reducing sedimentation in Swan Lake.

To date, submersed and rooted floating vegetation have not re-established in either the lower or middle units of Swan Lake. Further reductions in turbidity and hardening of sediments likely are needed in the lower unit before submersed or rooted floating vegetation can reestablish in this unit. Introduction of seeds and tubers, possibly combined with protection from herbivores, may be needed to reestablish vegetation in both the lower and middle units. The habitat project and management schemes employed in the middle unit were successful in establishing emergent vegetation in the middle unit during 2004 and 2005. Management to promote emergent vegetation to benefit waterfowl should be achievable in both the middle and lower units when additional successful draw downs are conducted.

The only macroinvertebrate goal identified in 1991 Definite Project Report (U.S. Army Corps of Engineers 1991) was to restore macroinvertebrate communities associated with submersed aquatic vegetation. Because submersed aquatic vegetation has not been re-established in the lower and middle units of Swan Lake, this goal was not achieved.

Management of the lower and middle units, specifically draw-downs to compact sediments, appears to have produced some benefits to benthic macroinvertebrates. Diversity increased significantly in both the middle and lower units, whereas there was no significant change in the upper unit. Much of this increase in diversity can be attributed to a decline in the dominance of chironomids.

Comparisons of pre- and post-project monitoring indicate that most of the desired benefits for fishes have not been fully achieved to date. There appears to be notable declines in the use of the lake by several species, including most centrarchids and buffalo. Both species richness and diversity have declined relative to pre-project levels, and there is little evidence of use of the lake for overwinter habitat with the exception of gizzard shad and white bass. We feel the primary reasons for the limited success of the project for fishes are: 1) a need to further improve habitat through draw-downs, especially in the lower unit, 2) the failure to re-establish submersed aquatic vegetation, 3) a need to provide deep-water habitat, and 4) the need for additional time for habitat improvements to translate to changes in the fish community.

INTRODUCTION - MONITORING

Swan Lake is a 1,200 ha Illinois River backwater lake located near the confluence of the Illinois and Mississippi Rivers. The lake was created in 1938 following the construction of Lock and Dam 26 (Mel Price Lock and Dam) on the Mississippi River. In the years following its creation, Swan Lake supported abundant populations of aquatic plants, fishes, and provided valuable habitat for wildlife (Theiling et al. 2000, USACE 1993). After a few decades, however, habitat quality within Swan Lake began to decline as a result of an accumulation of unconsolidated sediments.

The decline of habitat quality in Swan Lake mirrors patterns observed in many backwater lakes on the Mississippi and Illinois rivers. There are two principal factors driving these patterns: first, changes in land use throughout the watershed of the Upper Mississippi River have dramatically increased sedimentation, and second, maintenance of water levels sufficient for navigation has eliminated the summer dry period for floodplain habitats in many areas (Havera and Bellrose 1985, Poff et al. 1997, Theiling 1998, Koel and Sparks 2002). Backwater lakes in these areas are subject to high sediment loading, with little chance for sediments to consolidate and harden during draw downs. For example, Bhomik (1993) reported that many Illinois River backwaters lost from 30-100% of their early 1900's volume by 1993 as a result of increased sedimentation. These changes are especially detrimental to aquatic vegetation because unconsolidated sediments increase turbidity and reduce light penetration, and are poor substrates for rooting. Loss of aquatic vegetation in backwater lakes and other floodplain habitats is a major problem throughout most of the lower half of the Upper Mississippi River Basin (Bellrose et al. 1979, Middleton 2002, Reese and Lubinski 1983, Delong 2005).

To combat sedimentation and loss of aquatic vegetation in backwater lakes, moist soil management projects have been implemented in many areas of the Illinois and Upper Mississippi rivers. These projects usually involve several components, such as building levees and stop-log structures to manage the connection between the backwater lake and river, installation of pumps to manipulate water level, and dredging to remove unconsolidated sediments. These projects allow backwater lakes to be periodically drawn-down, mimicking a more natural water regime and allowing for the consolidation of sediments and promoting the growth of aquatic vegetation, particularly moist-soil vegetation (Havera et al. 1996, Havera and Bellrose 1985). Additionally, reduced connectivity between the lake and river usually greatly reduces the sediment load to the lake. Moist soil management has a proven track record for promoting the growth of moist soil vegetation and restoring habitat for waterfowl. Unfortunately, reduced connectivity between backwater lakes and rivers reduces access and use by fishes and other riverine organisms (Sparks et al. 1998).

Devising management projects and strategies that can restore habitat for both waterfowl and riverine fishes in the Upper Mississippi River System remains a challenge. For example, many riverine fishes use backwater lakes for spawning and nursery habitat, but this requires connectivity with the river. Increased connectivity with the river will increase sediment load to backwater lakes. Additionally, introduced fishes such as common carp and grass carp have substantial negative effects on aquatic vegetation (Bailey 1978, Parkos et al. 2003). Therefore, allowing connectivity for riverine fishes and other organisms can lead to negative effects on habitat restoration in backwater lakes.

The U.S. Army Corp of Engineers' Swan Lake Habitat Rehabilitation and Enhancement Project is one of the few large scale management projects that attempts to restore backwater lake habitat for waterfowl and other wildlife as well as fishes and other riverine organisms. The main goals of the project are to reduce sediment inputs and allow for water level management to promote the growth of aquatic vegetation. To achieve these goals, however, Swan Lake has been sequestered into three distinct units which will allow for adaptive management experiments to identify strategies to meet the sometimes conflicting habitat needs of fish and wildlife:

1) **Upper Swan Lake** - prior to this project, this section was leveed off from the main lake and is currently managed as a moist soil unit.
2) **The Middle Unit** - this unit is leveed off from the lower portion of the lake and from the Illinois River. The unit has been managed to promote the growth of emergent vegetation. A stop-log gate and pump allows the unit to be drawn down either partially or fully. Full draw downs of the middle unit were conducted in 2002, and 2005, and partially drawn downs were conducted in 2003 and 2004.
3) **The Lower Unit** – this unit is leveed off from the Illinois River and a stop-log structure and pump have been installed for water level management. This unit was drawn down in 2002, but remained connected to the river through the stop log structure from 2003 until the summer of 2006 (drawn down after post-project monitoring ended).

The Illinois Natural History Survey was contracted to conduct pre-project monitoring of Swan Lake in 1992, focused on water quality, sediment hardness, sedimentation, aquatic vegetation, aquatic macroinvertebrates, and fishes (Theiling et al. 2000). Our design for post-project monitoring followed the methodology, techniques, and equipment used during pre-project monitoring as closely as possible to allow for direct comparisons of the data collected in the pre- and post-project monitoring periods. This report evaluates effects of the Swan Lake Habitat Rehabilitation and Enhancement Project on these ecological factors. Where appropriate, we draw conclusions regarding the success of the project at meeting goals specified in the 1991 Definite Project Report (U.S. Army Corps of Engineers 1991). The inclusion of Upper Swan Lake is critical to the interpretation of differences between these two periods; management of Upper Swan Lake has remained consistent during this time, allowing this unit to serve as a partial control for temporal effects other than project effects.

WATER QUALITY

Project Goals:

The 1991 Definite Project Report (U.S. Army Corps of Engineers 1991) did not include specific goals or objectives for water quality. Nevertheless, project goal 1: “restore aquatic macrophyte beds and associated invertebrate communities for the benefit of migratory waterfowl,” and associated objectives of reducing sedimentation and resuspension of sediments, has obvious ramifications for water quality. In particular, reducing sedimentation and resuspension of sediments relates to an unstated objective of increasing water clarity. To this end, reductions in turbidity and increases in Secchi transparency would be patterns consistent with the stated goals and objectives of the project. Patterns in other water quality parameters needed to be monitored to be sure that unforeseen adverse conditions had not arisen as an artifact of the project components or other factors.

METHODS

Pre-project activities - Water quality monitoring included bi-monthly (i.e., every other week) transect sampling (five sites per transect) within each of the three Swan Lake units. For both the fixed sites and transects, a suite of basic LTRMP water quality parameters were collected including water depth (cm), dissolved oxygen ($mg \cdot l^{-1}$), water temperature (C), Secchi transparency (cm), turbidity (ntu), and conductivity ($\mu s \cdot cm^{-1}$).

Post-project monitoring - We repeated the bi-monthly sampling at the three transects established in each unit during pre-project monitoring. Three additional transects were established in the lower and middle units, and two additional transects were established in the upper unit, and five sites were sampled for each transect on a monthly basis. For transect samples, we measured basic LTRMP water quality parameters (water depth (cm), dissolved oxygen ($mg\cdot l^{-1}$), water temperature (C), Secchi transparency (cm), turbidity (ntu), and conductivity ($\mu s\cdot cm^{-1}$)). Sampling gear included a sounding pole with a 20-cm diameter shoe, a Secchi disk, a YSI 85 salinity, conductivity, dissolved oxygen, and temperature meter. Water samples (250 ml) were collected at each transect site and brought back to the lab for turbidity measurements using a Hach Turbidimeter Model 2100P. Where appropriate, we followed LTRMP water quality field methods and procedures which Soballe and Fischer (2004) describe in detail. Total sampling efforts yielded 14 transect samples (70 sites) per month when all sites and transects were accessible. We were not always able to collect to full suite of monthly samples because lake level and ice cover occasionally limited accessibility (Table WQ1).

Analyses:

For inferences to project effects on water quality variables, comparisons of the three original transects between pre and post-project monitoring are appropriate. Analysis of variance (ANOVA) was used to compare turbidity and Secchi transparency between the pre- and post-project periods. Separate analyses were conducted for each unit of Swan Lake, and the model include period, month, and the interaction between period and month. To conform to assumptions of normality and homoscedasticity, turbidity and Secchi data were square-root transformed based on qualitative examination of residuals prior to final analyses. Monthly means and standard error from the three original transects were calculated for temperature, dissolved oxygen, and conductivity, to allow qualitative comparisons between pre- and post-project monitoring. Finally, monthly means of all water quality parameters were calculated from all transects sampled during post-project monitoring to depict temporal trends during this period.

RESULTS AND DISCUSSION

Depth:

There was a brief flooding event in June 2004, and high water during from December 2004 to February 2005 (Figure WQ1). From spring 2005 till the end of the project, no further high water events occurred and drought conditions were present in the region. Draw downs are evident beginning in June for the middle and upper units. Aside from these draw down periods, depth in the middle unit was consistently deeper than either the lower or upper units. Active draw downs in the middle and upper units have important implications for interpretation of other water quality factors. During draw downs, only sites in the deepest portion of the lake can be sampled, often in areas that infrequently have the opportunity to dry and consolidate sediments. As a result, monthly means for water quality parameters, especially turbidity, secchi, and conductivity, likely will be an artifact of sampling being restricted to these areas. As a result we would expect to see greater turbidity readings and shallower Secchi transparencys in the middle and upper units during draw downs.

Water Temperature:

Across the post-project monitoring period, there was little difference in temperature among the three units (Figure WQ2). Data from the three transects sampled in both pre- and post-project monitoring suggest that summer temperature was warmer during post-project monitoring (Figure WQ3). Because this trend was present in all three units there is little reason to suspect that this pattern is related to any aspects of the habitat project.

Dissolved Oxygen:

Little evidence of consistent differences in dissolved oxygen levels among the three units was found during post-project monitoring (Figure WQ4). Monthly mean dissolved oxygen dipped below 5 $mg \cdot l^{-1}$ a few times in the upper and middle units, but never fell below 2 $mg \cdot l^{-1}$. Some of these episodes correspond with lake draw downs and likely should not be cause for concern. In the lower and middle units, dissolved oxygen was lower during post-project monitoring relative to pre-project monitoring, but levels were not low enough to raise any biological concern, nor are they likely to be a function of the habitat project (Figure WQ5). Given that water temperature was higher during post-project monitoring, lower dissolved oxygen levels would be expected during this period. In the spring and summer of 2006, lower oxygen levels in the middle unit corresponded with an algal bloom we observed during sampling events.

Turbidity:

There is some evidence of reductions of turbidity related to the habitat project and management actions in the middle unit of Swan Lake. There were no significant differences in turbidity between pre- and post-project monitoring in the upper unit (Table WQ2), though turbidity tended to be greater during post-project monitoring (Figure WQ6). Across all months, turbidity was significantly greater during post-project monitoring in the lower unit, although this pattern was reversed for June and July, contributing to a significant interaction between period and month (Table WQ2, Figure WQ6). In contrast, turbidity was significantly reduced during post-project monitoring in the middle unit, and the interaction between period and month was not significant (Table WQ2, Figure WQ6). Given that this pattern contrasts with those in the upper and lower units, this is evidence that reduction in turbidity can be attributed to the habitat project and the specific management scheme employed in the middle unit. Comparing monthly means across the entire post-project period, turbidity was consistently greater in the lower unit compared with the middle and upper units (Figure WQ7). Furthermore, turbidity in the middle unit fell below 40 NTU, a level LTRMP data suggests is low enough to allow growth of submersed aquatic vegetation (Dr. Yao Yin, USGS Upper Midwest Environmental Sciences Center, personal communication) on several occasions. Reversals and high variability in this pattern occurred during draw downs in 2004 and 2005.

Secchi:

Patterns in Secchi transparency also suggest some success in the goals of the habitat project and management strategies employed in the middle unit for increasing water clarity. Secchi transparency decreased significantly between pre- and post-project monitoring in both the lower and upper units of Swan Lake (Table WQ3, Figure WQ8). In contrast, the trend was for deeper Secchi transparency during post-project monitoring in the middle unit, though these differences were not significant (Table WQ3, Figure WQ8). This suggests that the habitat project and management strategies employed in the middle unit likely prevented the decreased water clarity observed in the lower and upper units, and appears to be trending toward improved water clarity in the middle unit. Across the entire post-project monitoring period, middle unit Secchi transparency was consistently deeper relative to the lower unit (Figure WQ9).

Conductivity:

Differences in conductivity among units during post-project monitoring were not great or consistent (Figure WQ10), and there were no dramatic or consistent differences between pre- and post-project monitoring in any of the three units (Figure WQ11). Across the whole post-project monitoring period, there was an overall increase in conductivity in all three units beginning in the spring of 2005 (Figure WQ10). This period coincides with drought-like conditions in the region, which likely influenced this pattern. Increases in conductivity were also seen in other backwater lakes sampled through the LTRMP program (unpublished data from Pool 26) further suggesting a regional cause such as drought.

CONCLUSIONS

The habitat project and management strategies used in the middle unit appear to have led to some increases in water clarity, based on the patterns observed in turbidity and Secchi transparency. Other water quality parameters remain at suitable levels for most aquatic biota. Water clarity in the middle unit often was at levels low enough to allow for the growth of submersed aquatic vegetation, a conclusion supported by enclosure/exclosure experiments we have conducted in this unit that demonstrate submersed aquatic vegetation can grow when protected from common carp and herbivores (red-eared sliders and grass carp, unpublished data).

Table WQ1. Number of samples collected each month from all transects in the lower, middle and upper units of Swan Lake for each water quality parameter.

Unit	Year	Month	Depth	Water Temperature	Dissolved Oxygen	Turbidity	Secchi Transparency	Conductivity
Lower	2004	4	30	29	30	30	30	29
Lower	2004	5	24	24	24	24	24	24
Lower	2004	6	25	25	25	25	25	25
Lower	2004	7	25	24	25	20	25	25
Lower	2004	8	25	25	25	25	25	25
Lower	2004	9	45	45	45	42	45	45
Lower	2004	10	25	25	25	25	25	25
Lower	2004	11	25	25	25	25	25	25
Lower	2004	0	0	0	0	0	0	0
Lower	2005	1	5	5	5	5	5	5
Lower	2005	2	25	25	25	25	25	25
Lower	2005	3	25	25	25	25	25	25
Lower	2005	4	25	25	25	25	25	25
Lower	2005	5	25	25	25	25	25	25
Lower	2005	6	25	25	25	25	25	25
Lower	2005	7	24	25	25	25	20	25
Lower	2005	8	25	25	25	25	25	25
Lower	2005	9	25	25	25	25	25	25
Lower	2005	10	25	25	25	25	25	25
Lower	2005	11	25	25	25	25	25	25
Lower	2005	12	20	20	20	19	20	20
Lower	2006	1	25	25	25	25	25	25
Lower	2006	2	5	5	5	5	5	5
Lower	2006	3	10	10	10	4	10	10
Lower	2006	4	10	10	10	10	10	10
Lower	2006	5	10	10	10	10	10	10
Lower	2006	6	5	5	5	5	5	5
Lower	2006	7	0	0	0	0	0	0
Lower	2006	8	0	0	0	0	0	0
Lower	2006	9	0	0	0	0	0	0

Table 1 continued.

Unit	Year	Month	Depth	Water Temperature	Dissolved Oxygen	Turbidity	Secchi Transparency	Conductivity
Middle	2004	4	30	30	30	30	30	30
Middle	2004	5	25	25	25	25	25	25
Middle	2004	6	25	25	25	25	25	25
Middle	2004	7	25	25	25	25	25	25
Middle	2004	8	20	13	13	13	13	13
Middle	2004	9	45	21	21	21	15	21
Middle	2004	10	5	3	3	3	3	3
Middle	2004	11	45	42	42	42	42	41
Middle	2004	12	5	5	5	5	5	5
Middle	2005	1	5	5	5	5	5	5
Middle	2005	2	25	25	25	25	25	25
Middle	2005	3	25	25	25	25	25	25
Middle	2005	4	29	29	29	29	29	29
Middle	2005	5	25	25	25	25	25	25
Middle	2005	6	25	24	25	25	25	25
Middle	2005	7	25	3	3	3	0	3
Middle	2005	8	25	3	3	3	3	3
Middle	2005	9	25	18	18	18	18	18
Middle	2005	10	25	25	25	25	25	25
Middle	2005	11	25	25	25	24	25	25
Middle	2005	12	13	14	14	14	14	14
Middle	2006	1	25	25	25	25	25	25
Middle	2006	2	5	5	5	5	5	5
Middle	2006	3	10	10	10	9	10	10
Middle	2006	4	10	10	10	10	10	10
Middle	2006	5	10	10	10	10	10	10
Middle	2006	6	10	10	10	5	10	10
Middle	2006	7	5	5	5	5	5	5
Middle	2006	8	10	10	10	10	10	10
Middle	2006	9	10	10	10	10	10	10

Table 1 continued.

Unit	Year	Month	Depth	Water Temperature	Dissolved Oxygen	Turbidity	Secchi Transparency	Conductivity
Upper	2004	4	20	20	20	20	20	20
Upper	2004	5	15	12	12	12	12	12
Upper	2004	6	25	21	21	19	21	21
Upper	2004	7	20	20	20	20	20	20
Upper	2004	8	5	3	3	3	3	3
Upper	2004	9	35	20	20	20	20	20
Upper	2004	10	20	20	20	20	20	20
Upper	2004	11	0	0	0	0	0	0
Upper	2004	12	0	0	0	0	0	0
Upper	2005	1	5	5	5	5	5	5
Upper	2005	2	20	20	20	20	20	20
Upper	2005	3	20	18	18	18	18	18
Upper	2005	4	20	20	20	20	20	20
Upper	2005	5	20	20	20	20	20	20
Upper	2005	6	17	11	11	13	13	11
Upper	2005	7	20	2	2	2	2	2
Upper	2005	8	20	1	1	1	1	1
Upper	2005	9	20	15	15	15	15	15
Upper	2005	10	20	20	20	20	20	20
Upper	2005	11	0	0	0	0	0	0
Upper	2005	12	0	0	0	0	0	0
Upper	2006	1	15	15	15	15	15	15
Upper	2006	2	5	5	5	5	5	5
Upper	2006	3	10	10	10	10	10	10
Upper	2006	4	10	10	10	10	10	10
Upper	2006	5	10	10	10	10	10	10
Upper	2006	6	0	0	0	0	0	0
Upper	2006	7	0	0	0	0	0	0
Upper	2006	8	0	0	0	0	0	0
Upper	2006	9	0	0	0	0	0	0

Table WQ2. Analysis of variance results testing for differences in turbidity between project periods (pre- versus post-project) and months. Separate analyses were conducted for each unit of Swan Lake.

Source	DF	Mean Square	F-value	P
		Lower Unit $R^2 = 0.516$		
Model	19	5.79	2.14	0.023
Period	1	21.26	7.73	0.008
Month	9	3.22	1.19	0.141
Period*Month	9	6.89	2.54	0.022
Error	38	2.71		
		Middle Unit $R^2 = 0.644$		
Model	19	16.26	3.72	<0.001
Period	1	19.59	4.48	0.041
Month	9	18.45	4.22	<0.001
Period*Month	9	4.77	1.09	0.393
Error	39	4.38		
		Upper Unit $R^2 = 0.264$		
Model	9	5.77	0.80	0.622
Period	1	21.04	2.92	0.103
Month	4	4.27	0.59	0.673
Period*Month	4	2.89	0.40	0.806
Error	20	7.23		

Table WQ3. Analysis of variance results testing for differences in Secchi transparency between project periods (pre- versus post-project) and months. Separate analyses were conducted for each unit of Swan Lake.

Source	DF	Mean Square	F-value	P
		Lower Unit $R^2 = 0.646$		
Model	19	0.85	3.55	<0.001
Period	1	6.63	27.66	<0.001
Month	9	0.93	3.86	0.002
Period*Month	9	0.51	2.15	0.049
Error	37	2.71		
		Middle Unit $R^2 = 0.676$		
Model	19	2.76	4.28	<0.001
Period	1	1.23	1.91	0.175
Month	9	3.90	6.05	<0.001
Period*Month	9	0.74	1.14	0.358
Error	39	0.65		
		Upper Unit $R^2 = 0.546$		
Model	9	5.17	2.67	0.033
Period	1	21.49	11.10	0.003
Month	4	3.99	2.06	0.124
Period*Month	4	1.89	0.97	0.444
Error	20	1.94		

Figure WQ1. Monthly mean depth (± standard error) from all transects sampled in the three units of Swan Lake during post-project monitoring.

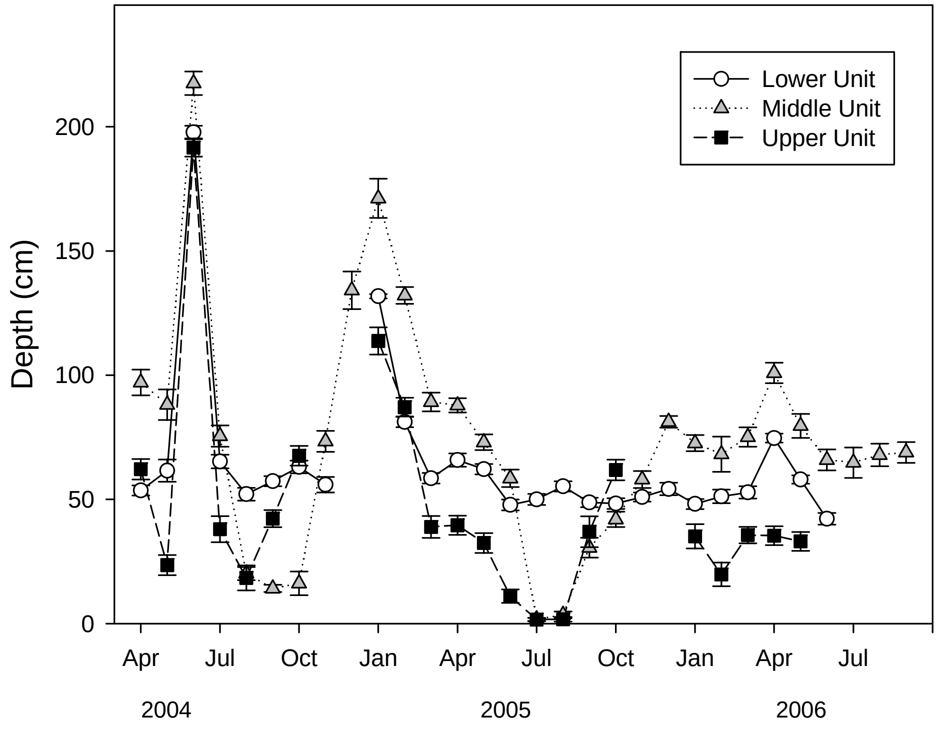


Figure WQ2. Monthly mean water temperature (± standard error) from all transects sampled in the three units of Swan Lake during post-project monitoring.

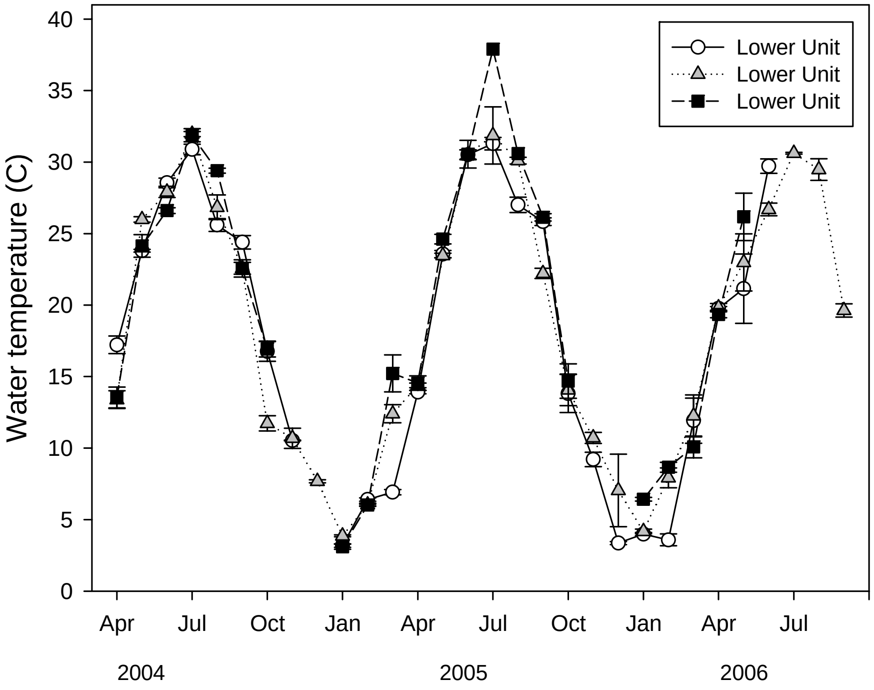


Figure WQ3. Monthly mean water temperature (± standard error) from the three transects sampled during both pre- and post-project monitoring of Swan Lake.

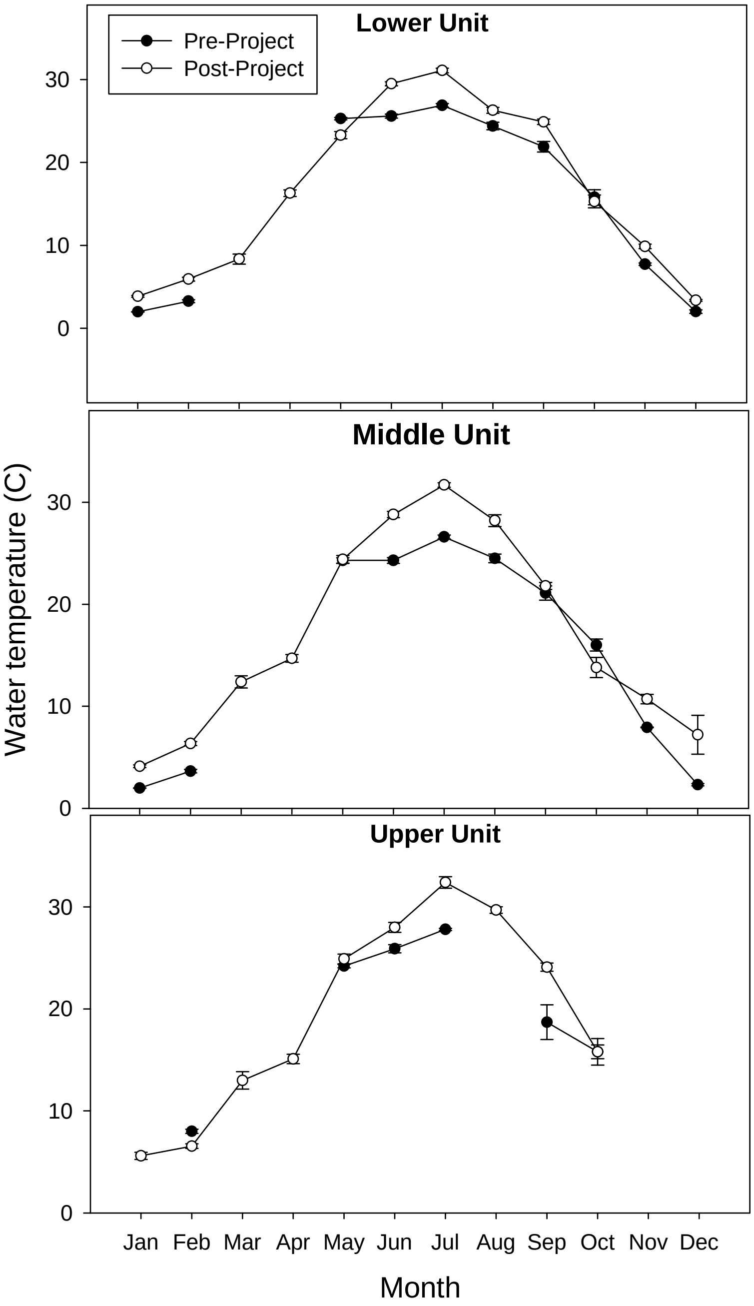


Figure WQ4. Monthly mean dissolved oxygen (± standard error) from all transects sampled in the three units of Swan Lake during post-project monitoring. The horizontal line indicates 5 $mg \cdot l^{-1}$.

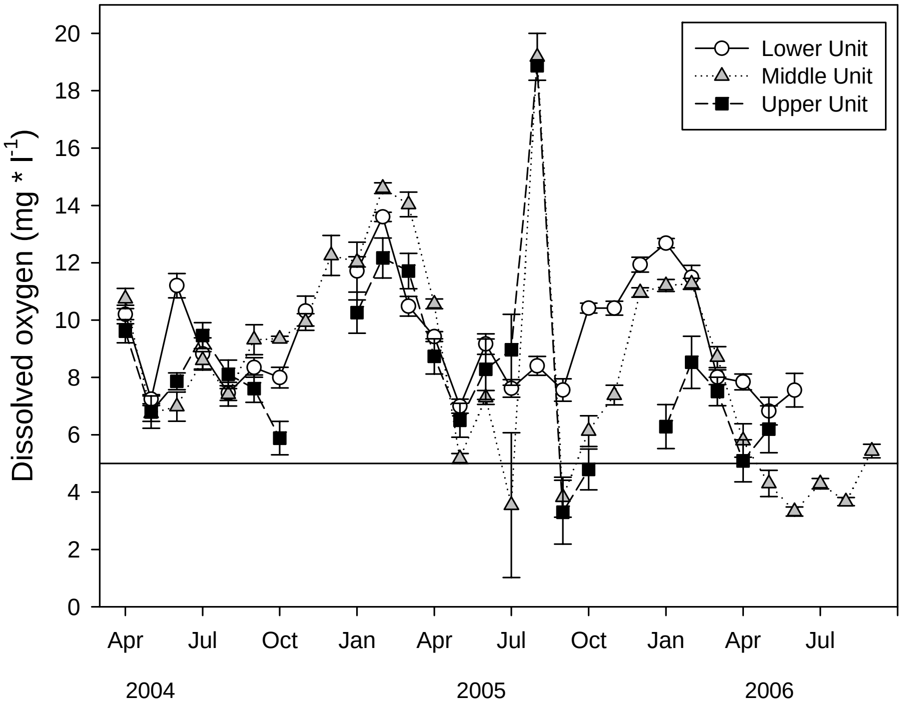


Figure WQ5. Monthly mean dissolved oxygen (± standard error) from the three transects sampled during both pre- and post-project monitoring of Swan Lake.

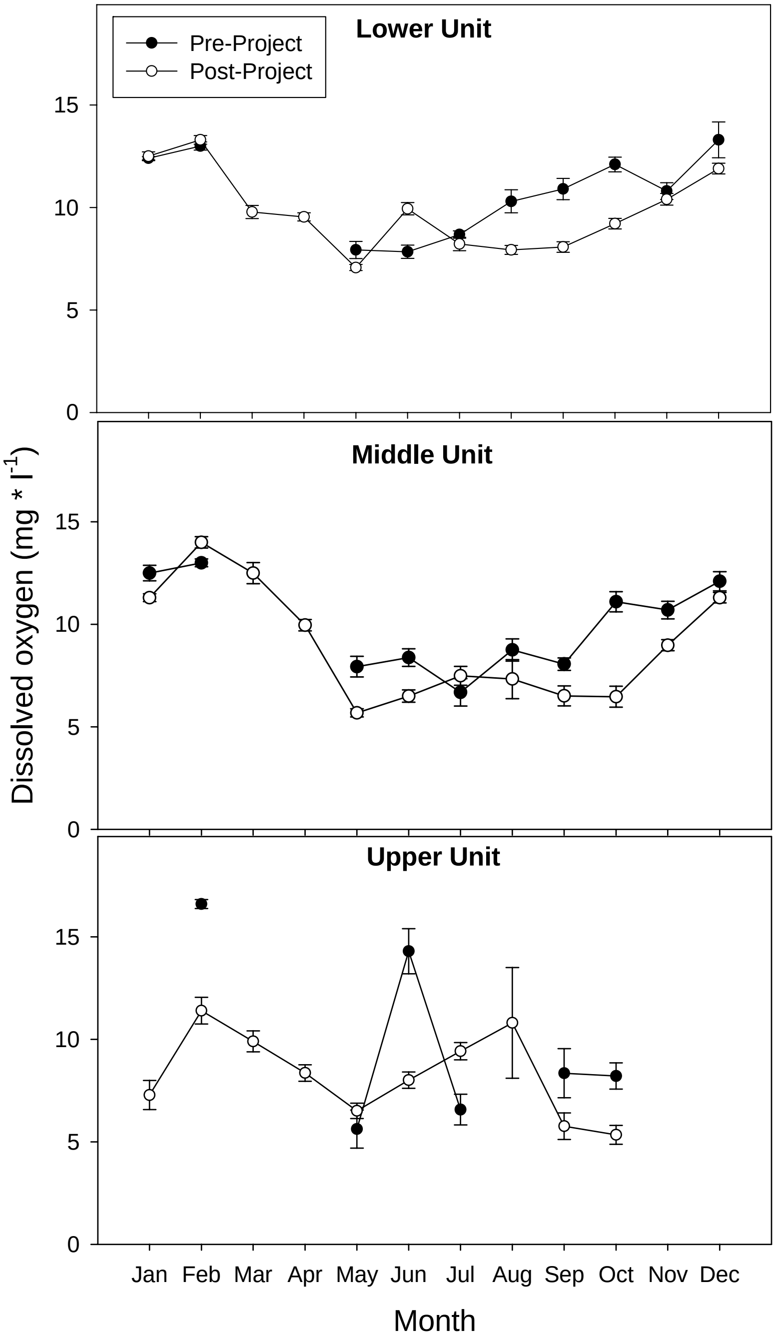


Figure WQ6. Monthly mean turbidity (± standard error) from the three transects sampled during both pre- and post-project monitoring of Swan Lake.

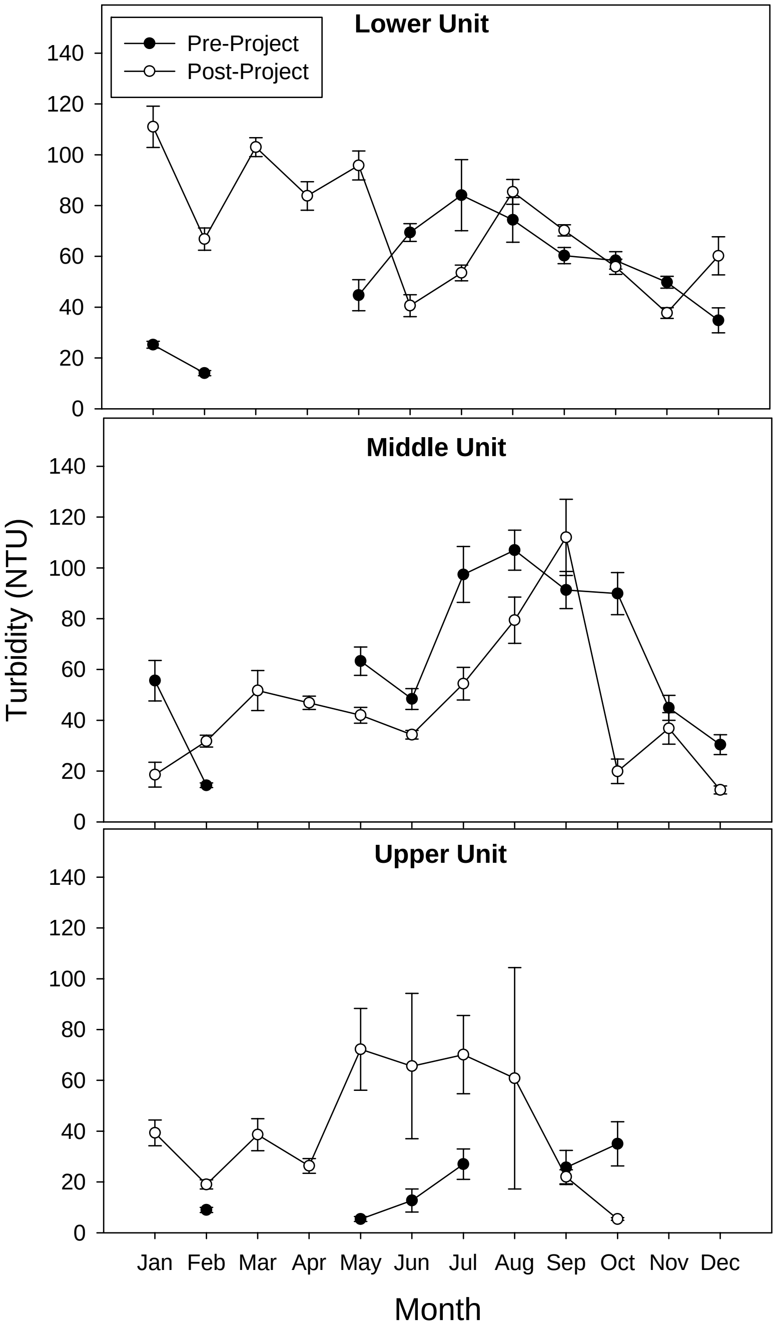


Figure WQ7. Monthly mean turbidity (± standard error) from all transects sampled in the three units of Swan Lake during post-project monitoring. The horizontal line indicates turbidity = 40 NTU.

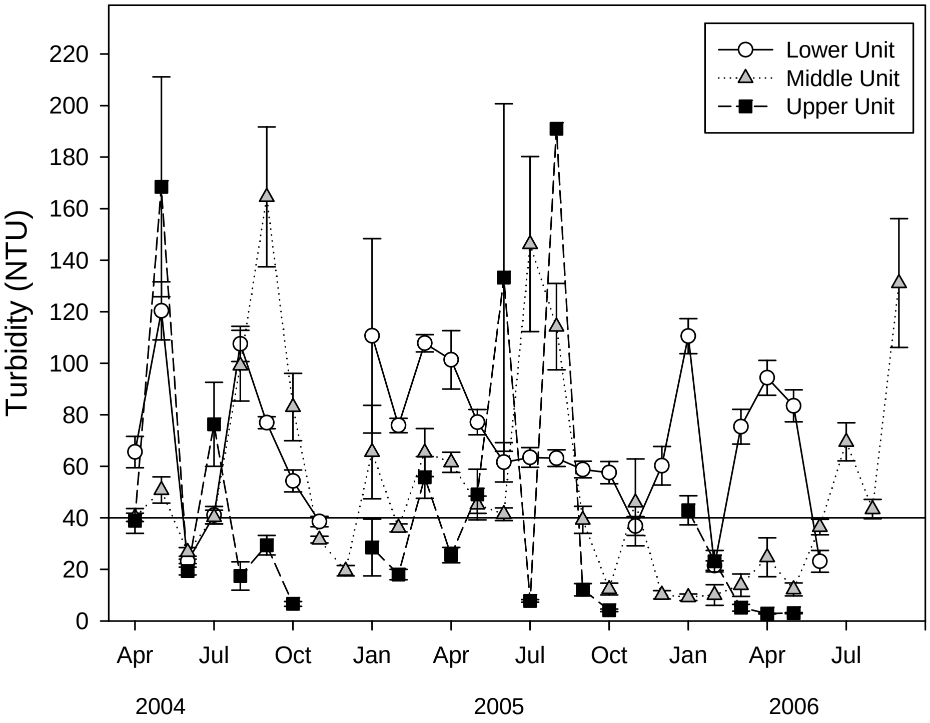


Figure WQ8. Monthly mean Secchi transparency (± standard error) from the three transects sampled during both pre- and post-project monitoring of Swan Lake.

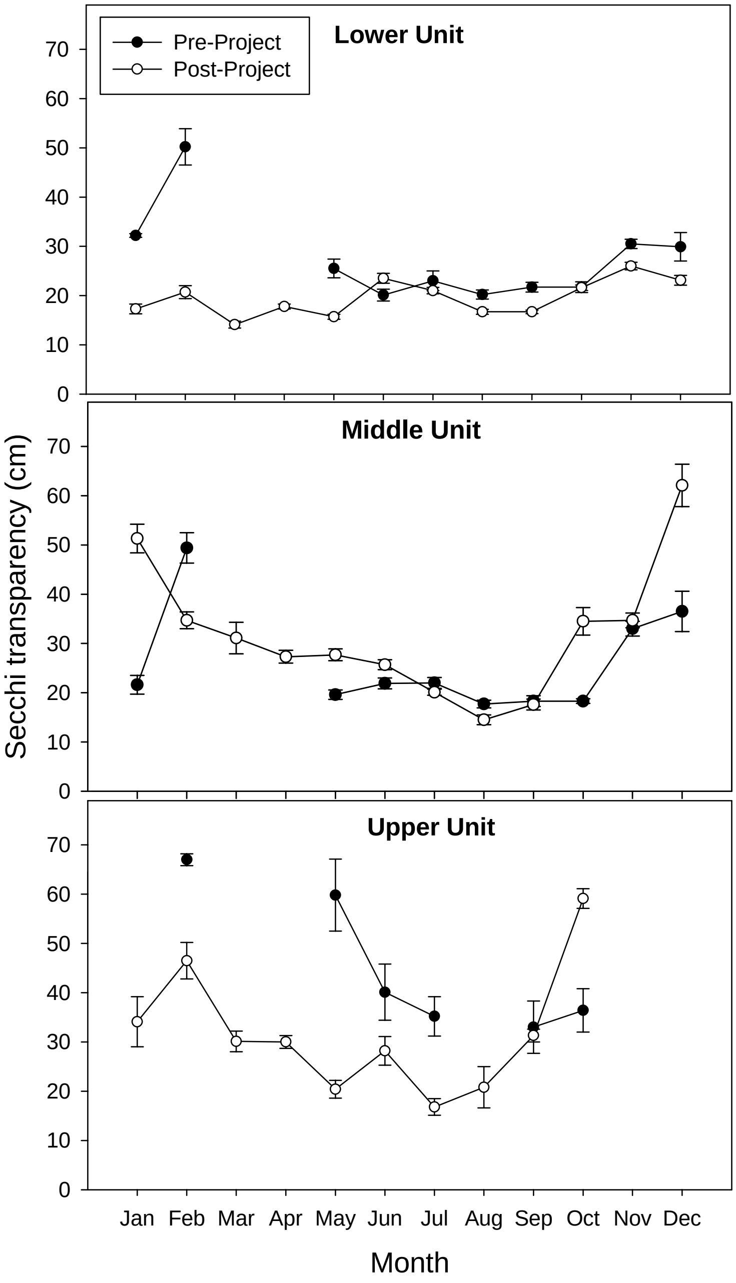


Figure WQ9. Monthly mean Secchi transparency (± standard error) from all transects sampled in the three units of Swan Lake during post-project monitoring.

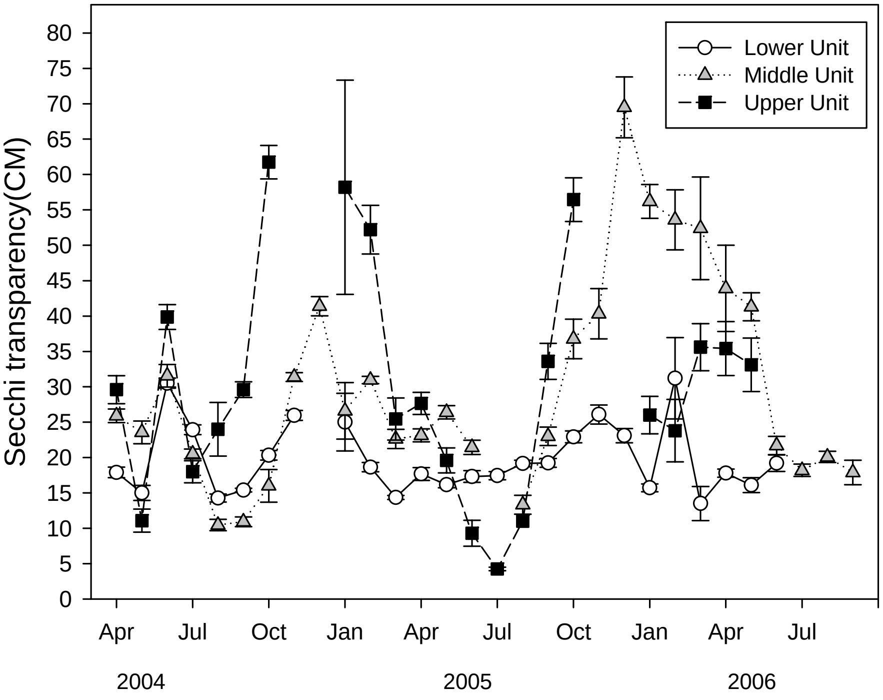


Figure WQ10. Monthly mean conductivity (± standard error) from all transects sampled in the three units of Swan Lake during post-project monitoring.

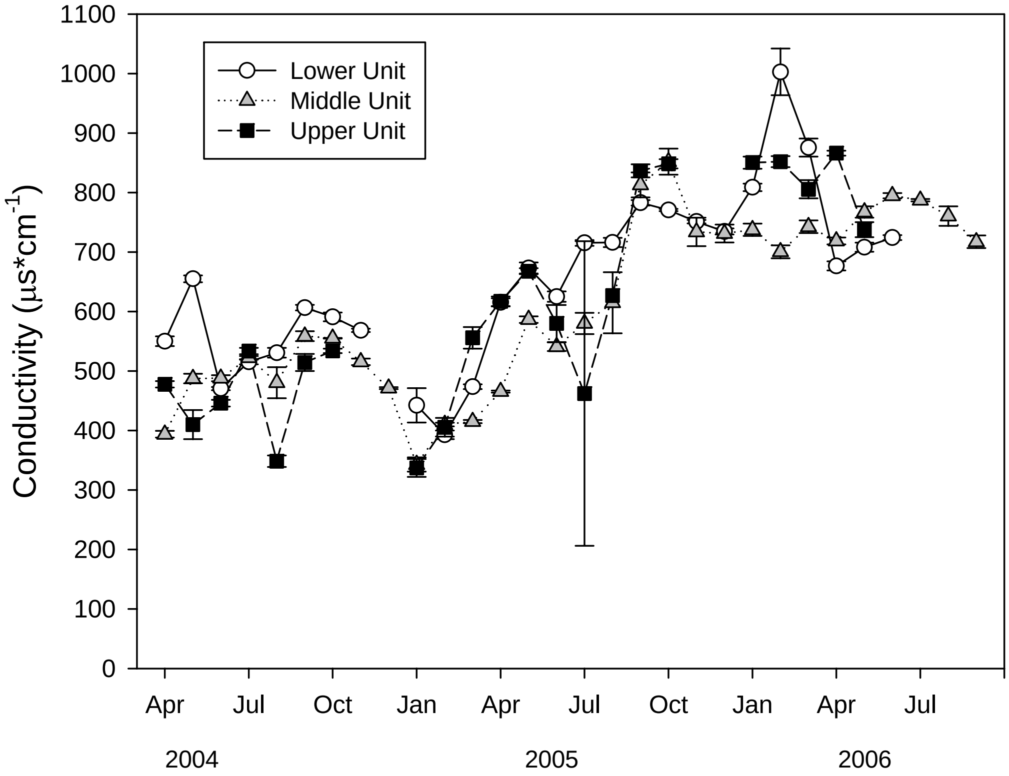
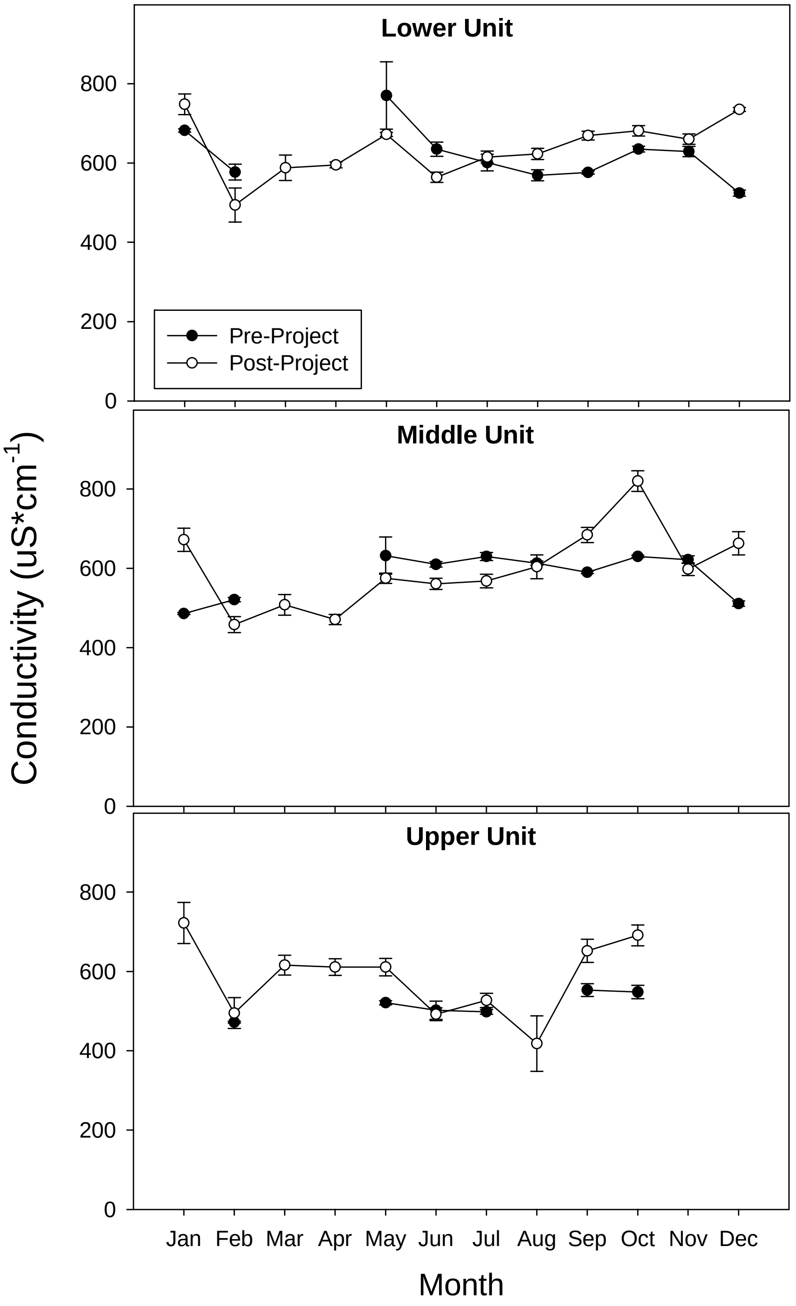


Figure WQ11. Monthly mean conductivity (± standard error) from the three transects sampled during both pre- and post-project monitoring of Swan Lake.



SEDIMENT HARDNESS

Project Goals:

Associated with the project goal of restoring aquatic macrophyte beds and associated invertebrate communities was the objective of providing the ability to solidify the lake bottom (U.S. Army Corps of Engineers 1991). Reducing flocculent sediments and hardening sediments is critical to aquatic and moist soil vegetation both for rooting substrate, and in water clarity because flocculent sediments are easily resuspended by wind and wave action. Toward this end, installation of stop-log structures and pumps allow for water management including periodic draw downs.

METHODS

Pre-project monitoring of sediment hardness was conducted with a sediment penetrometer at 14 sites in the lower and middle units (conditions were identical during the pre-project period), and 15 sites within upper Swan Lake. At each site, three to 15 replicate measures were taken. Penetration measurements (cm; 1-3 per site) were made with a sediment penetrometer constructed from aluminum conduit mounted vertically on a base plate that prevents the device from sinking into the sediment. A 2-cm diameter pipe equipped with a hook to suspend a five-pound weight, slid inside a 3-cm diameter pipe, and the depth the pipe penetrates the sediment is the measure of sediment hardness. Identical methods were used during post-project monitoring, and sediment hardness was measured at all random sites sampled for fish, macroinvertebrates, and aquatic vegetation, yielding a much greater sample size.

Analyses:

Analysis of Variance (ANOVA) was used to test for differences in penetrometer depth among units, period (pre- vs post-project) and the interaction of unit and period. Single degree of freedom contrasts (F-tests) were used to compare penetrometer depth between periods within each unit. Based on qualitative analysis of model residuals, penetrometer depth data was log transformed ($\log_{10}$ depth+1) to comply with ANOVA assumptions of normality and homoscedasticity. To compare the distribution of sediment hardness with units, we divided data into 10 cm depth groups and calculated the frequency of occurrence for each group.

RESULTS AND DISCUSSION

The habitat project and draw downs conducted in the middle unit were successful in hardening the sediments, whereas the less frequent draw down schedule in the lower unit was ineffective in meeting this goal. Although differences in hardness between pre- and post-project monitoring were not significant across all units, the significant interaction between unit and period indicates differences between periods occurred in some units (Table SH1). Single degree of freedom contrasts revealed that sediments were significantly harder in the post-project period in the middle ($F_{1,\,1036} = 8.47$; $P = 0.004$), significantly softer in the post-project period in the lower unit ($F_{1,\,1036} = 4.96$; $P = 0.026$), but did not differ significantly ($F_{1,\,1036} = 1.23$; $P = 0.291$)

between periods in the upper unit (Figure SH1, SH2). Additionally, sediment hardness differed significantly among units across both pre- and post-project monitoring (Table SH1).

Given that sediments became softer between pre- and post-project monitoring in the lower unit and did not change in the upper unit, there can be little doubt that the firmer sediments observed in the middle unit can be attributed to the habitat project and management employed in this unit. The cumulative effects of the 2004 and 2005 draw downs are obvious in the distribution of the data, but it is somewhat surprising how quickly the beneficial effects of the 2005 draw down were lost in 2006 when no draw down was conducted (Figure SH3).

CONCLUSIONS

Our penetrometer results suggest that the habitat project has provided the opportunity to substantially harden lake sediments through draw downs. These data also suggest that draw downs need to be conducted frequently and intensely to obtain desired benefits. The single draw down conducted in the lower unit was unsuccessful in hardening sediments. Additionally, although the cumulative effects of the 2004 and 2005 draw downs already showed signs of degrading slightly in 2006 when no draw down was conducted.

Table SH1. Analysis of variance results comparing penetrometer depth (a measure of sediment hardness) between the pre- and post-project periods and among the lower, middle, and upper units of Swan Lake. Model $R^2 = 0.405$.

Source	DF	Mean Square	F Value	P
Model	5	17.46	140.93	<0.001
Period	1	0.004	0.04	0.851
Unit	2	9.31	75.10	<0.001
Period*Unit	2	0.90	7.25	<0.001
Error	1036	0.12		

Figure SH1. Mean (± standard error) penetrometer depth from pre- and post-project monitoring in the lower, middle and upper units of Swan Lake.

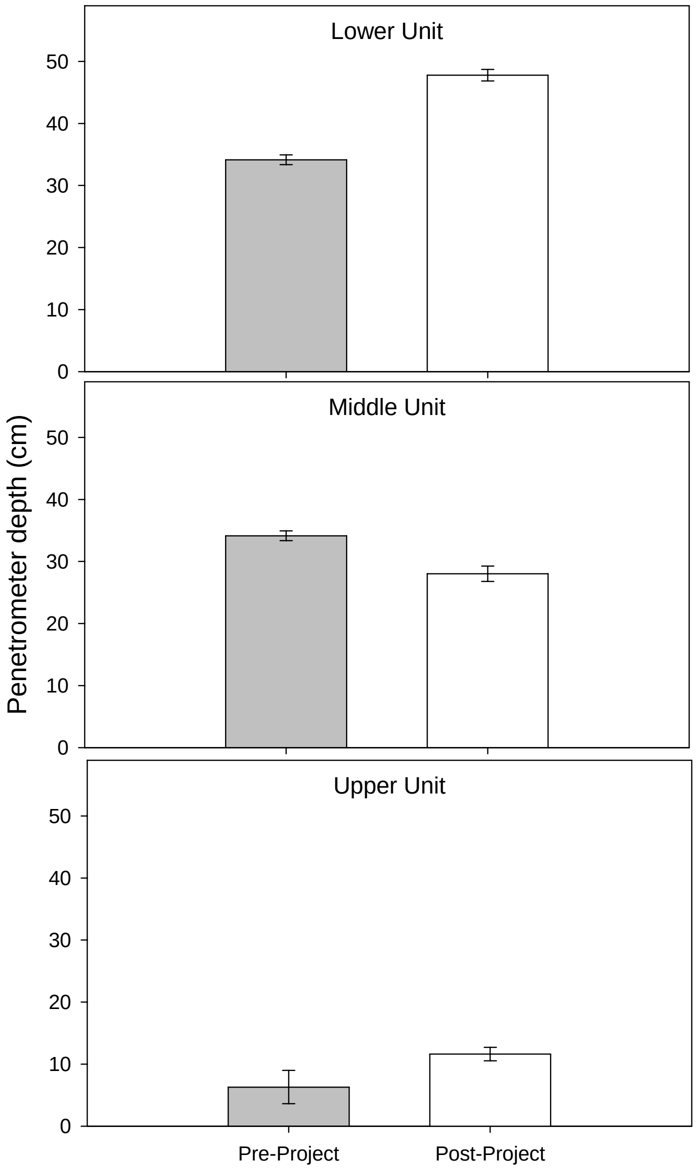


Figure SH2. Frequency of occurrence of sites among 10 cm groupings of penetrometer depth measured during pre- and post-project monitoring in the lower, middle and upper units of Swan Lake.

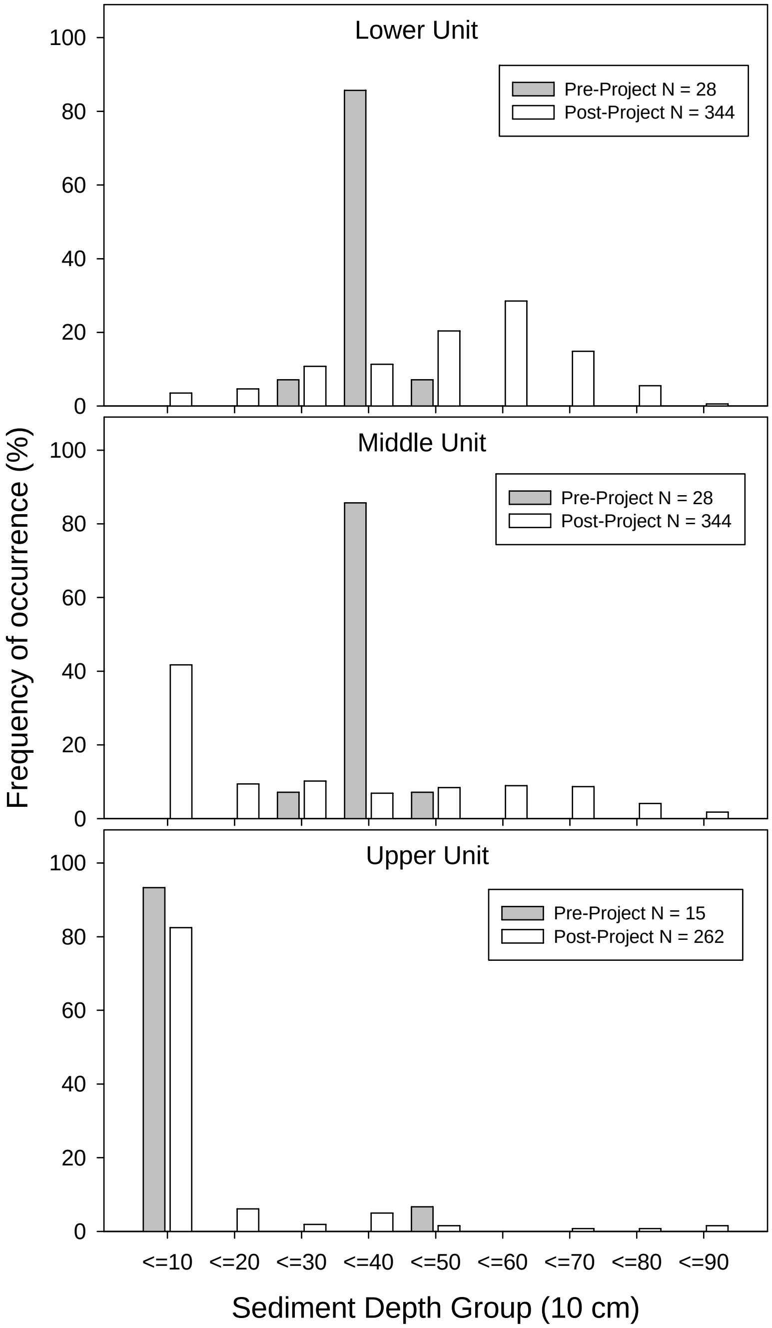
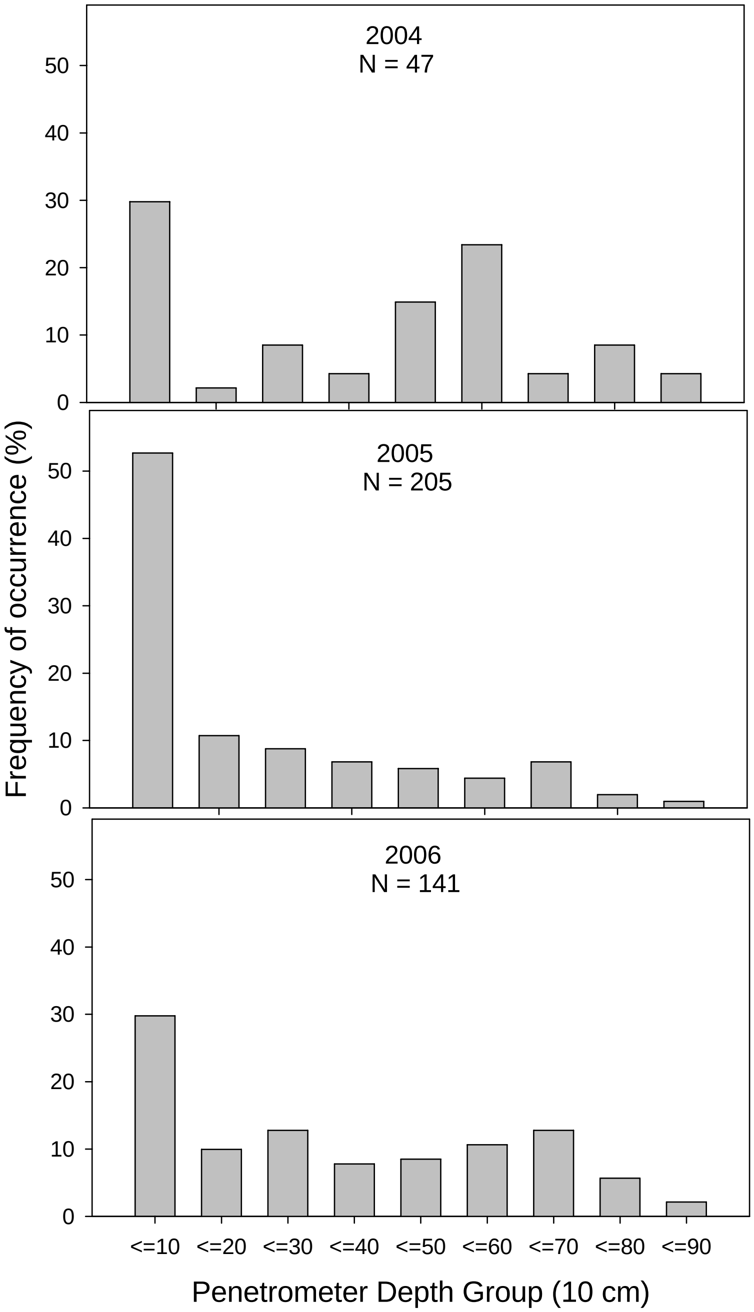


Figure SH3. Frequency of occurrence of sites among 10 cm groupings of penetrometer depth measured during post-project monitoring (2004, 2005, 2006) in the middle unit of Swan Lake.



SEDIMENTATION AND ISLAND EFFECTS ON SEDIMENT RESUSPENSION

Project Goals:

Under the major goal of restoring aquatic macrophyte beds and associated invertebrate communities was the objective of providing wave control (U.S. Army Corps of Engineers 1991). Reduction in wave height provides better opportunity for restoring aquatic macrophyte beds primarily by reducing the resuspension of lake sediments. To achieve this objective, island groups were constructed in the lower and middle units of Swan Lake. Reductions in wave height relative to wind speed, and reduced sediment resuspension are primary measures to assess the effectiveness of the constructed island groups. Another major objective of the project was substantial reductions in future sedimentation. Partial assessment of this goal can be made by examining the relationship between water depth and elevation through time.

METHODS

Wind speed ($m \cdot sec^{-1}$), wave height (cm), and depth (cm) were made at all monthly and bimonthly transect sites. Wind speed was measured using a Kestrel 2000 anemometer and water depth and wave height were measured using a sounding pole with a 20 cm diameter shoe. Wind and wave height were assessed using identical methods during pre- and post-project monitoring.

Sediment deposition studies were conducted in the lower unit to assess island effects on sediment resuspension. Sediments were collected with deposition samplers consisting of four PVC-pipe chambers held vertically in a steel frame. The samplers were built to the specifications recommended by Hakanson and Jansson (1983). A single sampler was placed upstream and downstream of the island group for two-week periods from July-September. In 2004, samplers were placed in the identical locations sampled during pre-project monitoring, whereas in 2005 samplers were placed in areas suspected to have more compact sediment to increase the likelihood of detecting localized changes in sediment resuspension. Samples were transferred from the deposition samplers into leak-proof containers, transported to the laboratory, and allowed to settle overnight while refrigerated. Samples were then decanted to remove overlying water, weighed, dried at 105° C to a constant weight to determine soil moisture content (g), and burned at 550° C to estimate organic content (g) of the settled sediments. Gross sedimentation (g dry wt $\cdot$ m^{-2} $\cdot$ day^{-1}) was calculated as the average of the four replicate samples collected at each sample station.

Analyses:

Wind speed and wave height data measured from the bimonthly sampling at the three original water quality transects was used to assess island effects on wave height. We used analysis of covariance to test for differences in the wave height – wind speed relationship between pre- and post-project monitoring. Separate ANCOVA tests were run for each unit, with models that included the main effects of period, wind direction, the interaction between period and wind direction, and wind speed as a covariate. Interaction terms involving the wind speed covariate were not significant and were eliminated from the final models. Based on qualitative assessment of residual patterns, wave height data were square root transformed to comply with ANCOVA assumptions of normality and homoscedasticity. We used ANOVA to test for differences in gross sedimentation rate between period and site. Preliminary analysis revealed no differences in gross sedimentation rate between 2004 and 2005, so data for the two years were combined for these analyses. Qualitative examination of residuals suggested no need for data transformation. Analysis of covariance was used to test for differences in the depth – elevation relationship among years for each of the three units with data from all transect samples collected during the three years of post-project monitoring. Qualitative examination of residuals suggested no need for data transformation.

RESULTS AND DISCUSSION

Island Effects on Wave Height:

Effects of the island groups at reducing wave height are marginal and only apparent in the middle unit. Wave height only differed significantly between the pre- and post-project period in the middle unit (Table SI1). Wave height was significantly related to wind speed in both the lower and middle units, but the ANCOVA model was not significant for the upper unit, likely because of small sample size. The relationship between wave height and wind speed during post-project monitoring was shifted to the right (i.e. smaller wave height at a given wind speed) in the middle unit (Figure SI1). Nevertheless, the magnitude of this reduction in wave height was only around 2 cm so it is difficult to judge the ecological significance of this reduction. Wave heights exceeding 7 to 10 cm were still observed in the middle and lower units during post-project monitoring, whereas wave heights above 5 cm were never observed in the upper unit during either pre- or post-project monitoring under similar wind speeds (Figure SI1).

Sediment Resuspension:

We found no evidence of reduced resuspension of sediments either upstream or downstream of the island group in the lower unit of Swan Lake. The ANOVA model was not significant (Table SI2), and there was little suggestion of reductions in gross sedimentation rate at either location during post-project monitoring (Figure SI2). Two factors likely contributing to the lack of success in achieving this objective are: 1) the failure of the lower unit island group to reduce wave height (see above), and 2) the relative ineffectiveness of the single draw down at firming sediments in the lower unit (see Sediment Hardness section), especially in the center of the lake. Without success in these two areas, it was very unlikely that we would see reductions in the resuspension of sediments in this unit.

Depth-Elevation Relationship:

Examination of the relationship between depth and lake elevation revealed little suggestion of dramatic changes in depth, either from filling in with sediments or deepening through scouring, among years during post-project monitoring. The relationship between depth and elevation was linear for all three units (Figure SI3) and significant in all but the upper unit (Table SI3). Neither year, nor the interaction between year and elevation was significant for any unit, and there was no evidence of a shift in the water depth – lake elevation relationship in any unit (Figure SI3). Although the ANCOVA models explained a great amount of variance in the water depth – lake elevation relationship, it is unclear whether these data are sufficient to draw conclusions about overall sedimentation in Swan Lake. The root means square error for water depth was 4.25 cm in the lower unit, and 8.49 cm in the middle unit so only a relatively large shift in the water depth – lake elevation relationship likely would be detected using these methods.

CONCLUSIONS

Our results suggest that the success of the island groups at reducing wave height and resuspension of sediments was marginal (at best) in the middle unit, and unsuccessful in the lower unit. Further efforts to reduce wave height likely will be needed to meet project objectives, especially in the lower unit. Our analysis of water depth and lake elevation did not suggest any overall changes in lake depth (either filling in or deepening) across years during post-project monitoring, but we caution that these methods may not be the best strategy to assess project goals for reducing sedimentation in Swan Lake.

Table SI1. Analysis of covariance results testing for differences in the wave height-wind speed relationship among project periods (pre- and post-project monitoring) in the lower, middle and upper units of Swan Lake.

Source	DF	Means Square	F-Value	*P*
		Lower Unit (R^2 = 0.738)		
Model	4	7.64	35.98	< 0.001
Period	1	0.01	0.01	0.925
Wind Direction	1	0.05	0.21	0.646
Period * Wind Direction	1	0.41	1.94	0.170
Wind	1	29.47	138.82	<0.001
Error	51	0.21		
		Middle Unit (R^2 = 0.641)		
Model	4	6.18	24.13	<0.001
Period	1	1.20	4.68	0.035
Wind Direction	1	0.21	0.81	0.373
Period * Wind Direction	1	0.33	1.29	0.261
Wind	1	21.07	82.23	<0.001
Error	54	0.26		
		Upper Unit (R^2 = 0.060)		
Model	4	0.10	0.10	0.829
Period	1	0.02	0.05	0.817
Wind Direction	1	0.07	0.27	0.609
Period * Wind Direction	1	0.02	0.06	0.814
Wind	1	0.19	0.70	0.413
Error	23	0.27		

Table SI2. Analysis of variance results testing for differences in gross sedimentation (g dry wt · m^{-2} · day^{-1}) between period (pre- and post-project monitoring) and location (upstream and downstream of island groups) in the lower unit of Swan Lake.

Source	DF	Mean Square	F-Value	*P*
Model	3	52142.40	0.73	0.547
Period	1	114721.17	1.61	0.222
Site	1	2439.82	0.03	0.855
Period*Site	1	39266.23	0.55	0.468
Error	16	71143.23		

Table SI3. Analysis of covariance results testing for differences in the water depth-lake elevation relationship among years during post-project monitoring of the lower, middle and upper units of Swan Lake.

Source	DF	Mean Square	F-Value	P
		Lower Unit (R^2 = 0.99)		
Model	5	21466.35	1187.78	<0.001
Year	2	10.33	0.57	0.567
Elevation	1	1117.03	61.81	<0.001
Year*Elevation	2	10.42	0.58	0.564
Error	73	18.07		
		Middle Unit (R^2 = 98)		
Model	5	49552.15	687.00	<0.001
Year	2	1.89	0.03	0.974
Elevation	1	1993.24	27.63	<0.001
Year*Elevation	2	1.88	0.03	0.974
Error	88	72.13		
		Upper Unit (R^2 = 0.93)		
Model	6	20557.95	84.22	<0.001
Year	1	618.15	2.53	0.120
Elevation	1	875.04	3.58	0.067
Year*Elevation	2	310.33	1.27	0.292
Error	43	244.09		

Figure SI1. The relationship between wave height and wind speed between pre- and post-project monitoring of the lower, middle, and upper units of Swan Lake.

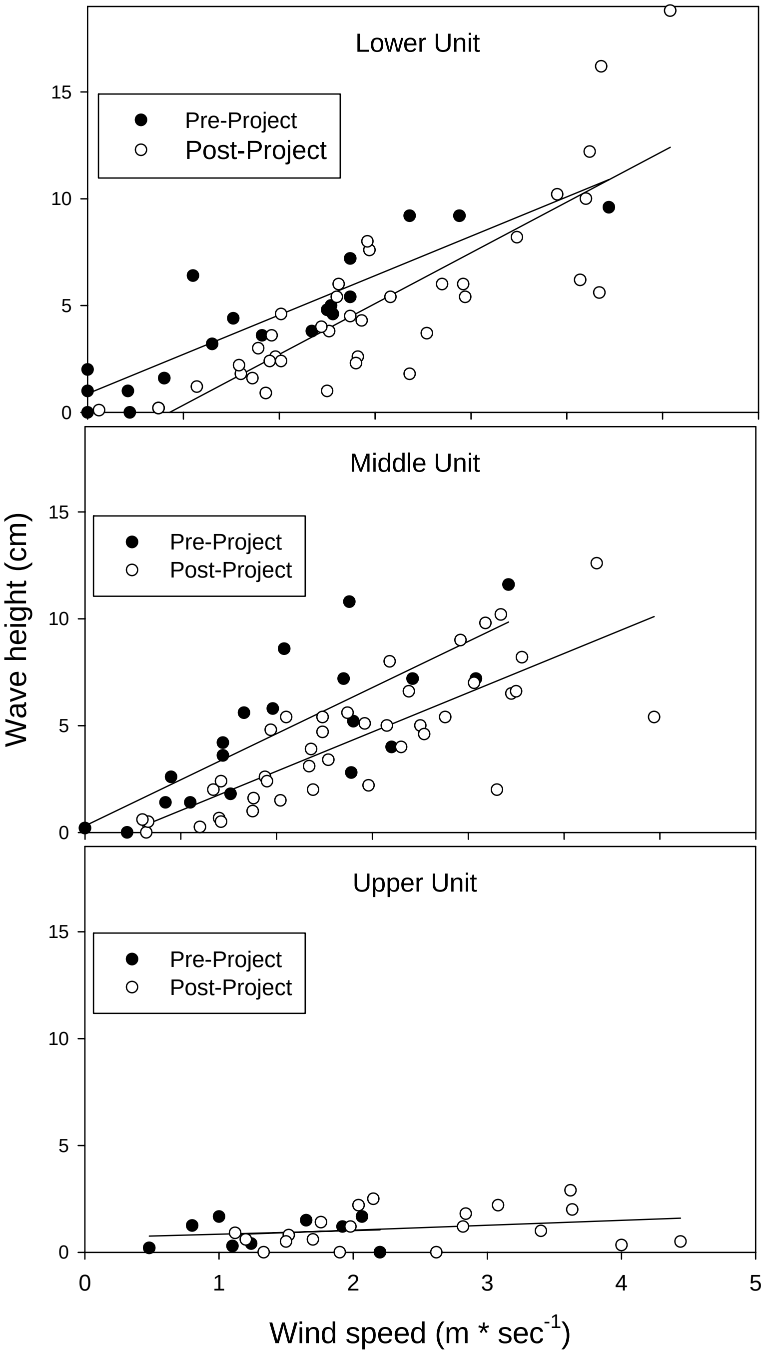


Figure SI2. Mean (± standard error) gross sedimentation rate sampled at one location upstream and one location downstream of the island groups in the lower unit of Swan Lake during pre- and post-project monitoring.

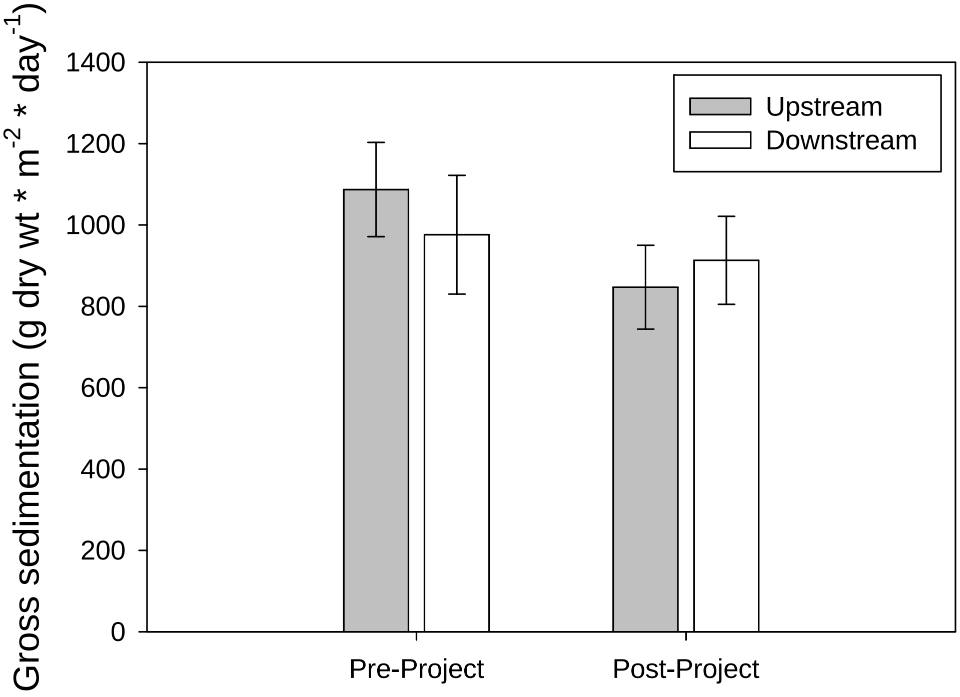
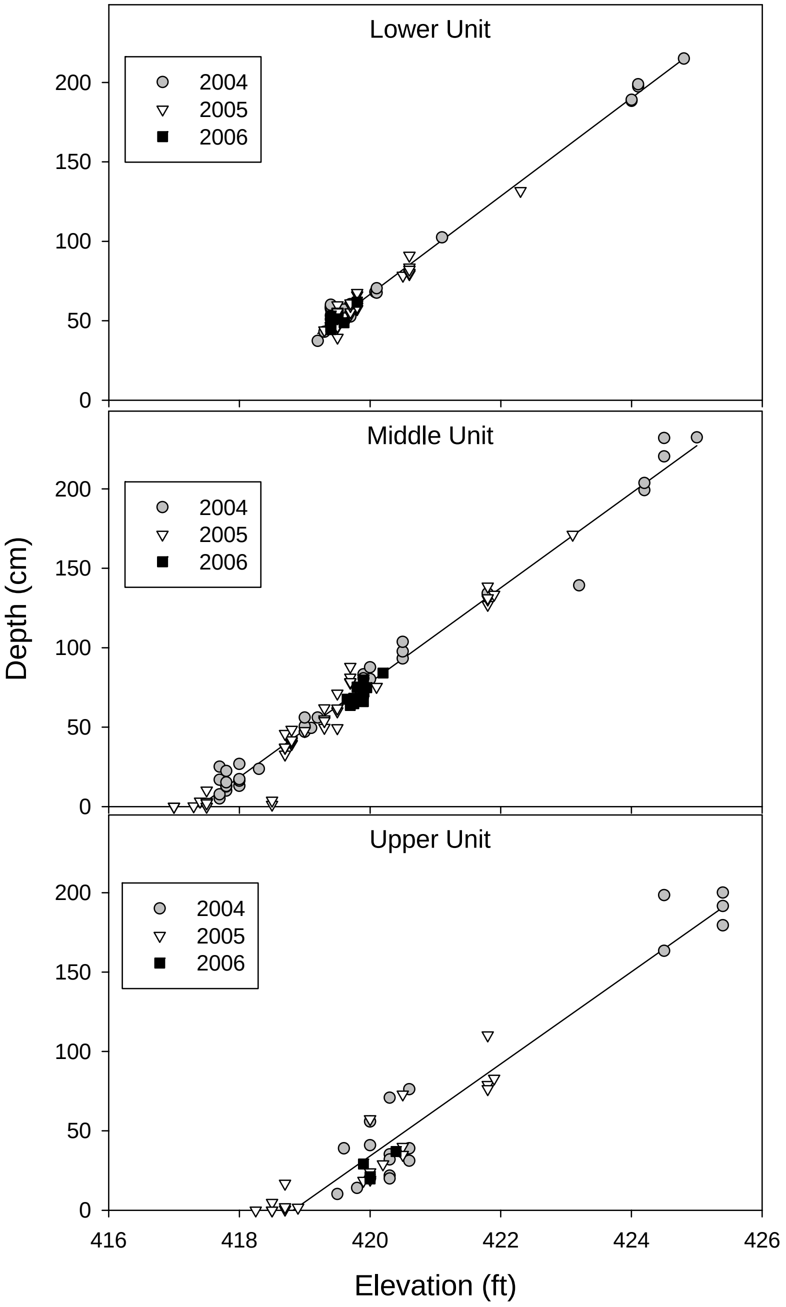


Figure SI3. Relationships of water depth to lake elevation in the lower, middle and upper units of Swan Lake during the three years of post-project monitoring.



AQUATIC VEGETATION

Project Goals:

A major goal of the Swan Lake Habitat Rehabilitation and Enhancement project was to restore aquatic macrophyte beds (US Army Corps of Engineers 1991). Establishment of aquatic vegetation can reduce sediment resuspension and turbidity (via root consolidation) as well as provide habitat for macroinvertebrates, fish, and waterfowl. To achieve this goal, Swan Lake was divided into three distinct units and stop-log gates and pumps were installed to allow manipulation of water levels. Draw-downs were implemented during the summer dry-period to mimic the historic, more natural, hydrologic regime. Flocculent sediments were hardened by exposing the lake bottom to the open air. The effectiveness of these actions was evaluated through comparisons of aquatic vegetation data collected in Swan Lake during the both the pre- and post-project periods.

METHODS

Methods for sampling aquatic vegetation differed between pre- and post-project periods. Abundant aquatic vegetation beds were known to exist in the lower and upper units during the pre-project; thus, a transect sampling-method (sites were sampled at 15 m intervals) was employed to assess the status of aquatic vegetation. Vegetation was completely devoid from all but the upper unit by the onset of the post-project. Therefore, we were unable to employ the transect-sampling methods used in pre-project monitoring and employed random sampling to detect reestablishment of aquatic vegetation throughout Swan Lake. Aquatic vegetation was sampled at 75 random sites (i.e., lower=30, middle=30, upper=15) across all units both early (May/June) and late (August/September) in the growing season in 2004 and 2005. In 2006, late season vegetation sampling was only conducted in the middle unit because of site access problems (upper and lower units were drawn down). Sampling was conducted by both visual observation and via a long-handled, incremented rake according to LTRMP protocols (Yin et al. 2000). A 0.25 m^2 quadrat sampler (i.e., a ¾" PVC-frame) was randomly tossed three times at each site in order to assess surface coverage of aquatic vegetation. Identification of specimens observed at the site or collected on the rake was performed in the field; however, vouchers were taken and transported to the laboratory for confirmation.

Analyses:

Frequency of occurrence (=number of occurrences/number of sites sampled x 100) was calculated for each vegetation species. Average surface cover ((sum of % cover for each quadrat)/3 x 100) was calculated for species sampled within the boundaries of the vegetation quadrats. Quantitative comparisons of vegetation measures between project periods were not possible due to differences in sampling methodology; transect sampling was used in established submersed vegetation beds during pre-project monitoring and these transects could not be re-established during post-project monitoring.

RESULTS AND DISCUSSION

Neither submersed aquatic vegetation nor rooted floating vegetation was re-established in either the lower or middle units. No specimens from either of these groups were found at any of the random sites used to assess aquatic vegetation. Across all random and transect sites (i.e., all aquatic vegetation, water quality, macroinvertebrate, and fish sampling sites) visited for the entire post-project monitoring period, only two specimens of American lotus (*Nelumbo lutea*) were observed in the middle unit and one specimen was observed in the lower unit. In contrast, sago pondweed (S*tuckenia pectinatus*), coontail (*Ceratophyllum demersum*), southern waternymph (*Najas quadalupensis*), floating primrose (*Ludwigia peploides*), leafy pondweed (*Potamogeton foliosus*), horned pondweed (*Zannichellia palustris*) and grassleaf mudplantain (*Heteranthera dubia*) were sampled in the lower unit during pre-project monitoring (Theiling et al. 2000). Submersed and rooted floating vegetation remained established in upper unit, with similar species present as in pre-project monitoring (Table AV1). Caging experiments conducted in the middle unit of Swan Lake (unpublished data of authors) have demonstrated that introduced sago pondweed (*Stuckenia pectinatus*) can grow in this unit when protected from common carp and herbivores (red-eared sliders and grass carp). Additionally, although not encountered in our random sampling, sparse growth of leafy pondweed, longleaf pondweed (*Potamogeton nodosus*), southern waternymph, and Carolina mosquitofern (*Azolla caroliniana*) were observed in a single cover within the middle unit. This suggests a program of seed and tuber introduction, possibly combined with protection from common carp and herbivores, might lead to successful reestablishment of submersed and rooted floating vegetation in the middle unit. Further hardening of sediments will be needed in the lower unit before aquatic vegetation can be restored.

Draw downs conducted in the middle unit were successful in promoting the growth of emergent vegetation in 2004 and 2005, whereas the lack of a draw down in the middle unit during 2006 prevent establishment of emergent vegetation. In total, 18 taxa of emergent vegetation were encountered in the middle unit (Table AV1). Most species were only encountered in samples collected late in the growing season. The five most frequently encountered species, and those with the greatest surface coverage, were redroot flatsedge (*Cyperus erythrorhizos*), Amazon sprangletop (*Leptochloa panicoides*), rough cocklebur (*Xanthium strumarium*), curlytop knotweed (*Polygonum lapathifolium*), and three varieties of native millet (*Echinochloa* spp.). Redroot flatsedge achieved the greatest frequency of occurrence of any species, occurring at over 60% of all vegetation sites in the middle unit during the late growing season 2005, also achieving the greatest surface coverage during this time (Figure AV1, AV2).

CONCLUSIONS

To date, submersed and rooted floating vegetation have not re-established in either the lower or middle units of Swan Lake. Further reductions in turbidity and hardening of sediments likely are needed in the lower unit before submersed or rooted floating vegetation can reestablish in this unit. Introduction of seeds and tubers, possibly combined with protection from herbivores, may be needed to reestablish vegetation in both the lower and middle units. The habitat project and management schemes employed in the middle unit were successful in establishing emergent vegetation in the middle unit during 2004 and 2005. Management to promote emergent vegetation to benefit waterfowl should be achievable in both the middle and lower units when additional successful draw downs are conducted.

Table AV1. Frequency of occurrence (%) and mean surface coverage (%) for aquatic vegetation species found during post-project monitoring (2004-2006) of the middle unit of Swan Lake.

Common Name	Scientific Name	Frequency of Occurrence	Mean Coverage	Coverage Standard Error
	Non-rooted Floating Vegetation			
Common duckweed	*Lemna minor*	11.73	0.019	0.008
Greater duckweed	*Spirodela polyrrhiza*	4.47	0.007	0.005
	Rooted Floating Vegetation			
American lotus	*Nelumbo lutea*	1.12	0.00	0.00
	Emergent Vegetation			
Redroot flatsedge	*Cyperus erythrorhizos*	15.64	3.359	0.884
Amazon sprangletop	*Leptochloa panicoides*	11.73	1.421	0.576
Millet	*Echinochloa spp.**	11.73	0.205	0.123
Rough cocklebur	*Xanthium strumarium*	7.82	0.618	0.281
Curlytop knotweed	*Polygonum lapathifolium*	6.70	0.054	0.035
Nodding beggartick	*Bidens cernua*	6.70	0.417	0.267
Valley redstem	*Ammannia coccinea*	5.59	0.205	0.092
Rice cutgrass	*Leersia oryzoides*	5.59	0.050	0.033
Indian lovegrass	*Eragrostis pilosa*	5.03	1.719	0.694
Roundfruit hedgehyssop	*Gratiola virginiana*	2.79	0.106	0.054
Black willow	*Salix nigra*	2.23	0.00	0.00
Coast cockspur grass	*Echinochloa walteri*	1.68	0.00	0.00
Arumleaf arrowhead	*Sagittaria cuneata*	1.68	0.073	0.060
Disk waterhyssop	*Bacopa rotundifolia*	1.12	0.024	0.019
Whitestar	*Ipomoea lacunosa*	1.12	0.006	0.004
Broadleaf arrowhead	*Sagittaria latifolia*	1.12	0.108	0.108
Yellow nutsedge	*Cyperus esculentus*	1.12	0.004	0.004
Tall amaranth	*Amaranthus rudis*	0.56	0.002	0.002
Eastern cottonwood	*Populus deltoides*	0.56	0.002	0.002
Prickly fanpetals	*Sida spinosa*	0.56	0.002	0.002
Fall panicgrass	*Panicum dichotomiflorum*	0.56	0.00	0.00
Witchgrass	*Panicum capillare*	0.56	0.00	0.00
Pennsylvania smartweed	*Polygonum pensylvanicum*	0.56	0.00	0.00
Filamentous algae		2.23	0.112	0.112

**Echinochloa* spp. Includes three varieties of native millet that are very difficult to differentiate: *E. crus-galli, E. muricata microstachya, and E. muricata muricata*.

Table AV2. Frequency of occurrence (%) and mean surface coverage (%) for aquatic vegetation species found during post-project monitoring (2004-2006) of the upper unit of Swan Lake.

Common Name	Scientific Name	Frequency of Occurrence	Mean Coverage	Coverage Standard Error
	Non-rooted Floating Vegetation			
Common duckweed	*Lemna minor*	58.75	1.031	0.450
Greater duckweed	*Spirodela polyrrhiza*	37.50	0.444	0.223
Columbian watermeal	*Wolffia columbiana*	2.50	0.042	0.042
	Rooted Floating Vegetating			
Floating primrose	*Ludwigia peploides*	30.00	2.296	0.908
	Submersed Aquatic Vegetation			
Sago pondweed	*Stuckenia pectinata*	31.25	0.471	0.471
Southern waternymph	*Najas guadalupensis*	15.00	0.021	0.021
Coontail	*Ceratophyllum demersum*	10.00	0.00	0.00
Horned pondweed	*Zannichellia palustris*	8.75	0.642	0.604
Leafy pondweed	*Potamogeton foliosus*	2.50	0.00	0.00
	Emergent Vegetation			
Millet	*Echinochloa spp.*	26.25	1.733	0.784
Japanese millet	*Echinochloa esculenta*	13.75	3.429	1.778
Curlytop knotweed	*Polygonum lapathifolium*	8.75	0.021	0.021
Amazon sprangletop	*Leptochloa panicoides*	8.75	0.392	0.255
Common buttonbush	*Cephalanthus occidentalis*	7.50	0.00	0.00
Redroot flatsedge	*Cyperus erythrorhizos*	6.25	1.846	1.177
Rice cutgrass	*Leersia oryzoides*	5.00	0.317	0.258
Tall amaranth	*Amaranthus rudis*	5.00	0.050	0.050
Pennsylvania smartweed	*Polygonum pensylvanicum*	3.75	0.00	0.00
Coast cockspur grass	*Echinochloa walteri*	3.75	0.004	0.004
Halberdleaf rosemallow	*Hibiscus laevis*	3.75	0.008	0.008
Eastern swampprivet	*Forestiera acuminata*	3.75	0.00	0.00
Unidentified smartweed	*Polygonum* spp.	3.75	0.00	0.00
Black willow	*Salix nigra*	1.25	0.021	0.021
Whitestar	*Ipomoea lacunosa*	1.25	0.00	0.00
Horsetail paspalum	*Paspalum fluitans*	1.25	0.00	0.00
Filamentous algae		35.00	5.404	1.989

Figure AV1. Frequency of occurrence for the five most common taxa of emergent vegetation from post-project monitoring, late-season sampling, in the middle unit of Swan Lake. *Cyperus* is redroot flatsedge, *Leptochloa* is Amazon sprangletop, *Xanthium* is rough cocklebur, *Polygonum* is curlytop knotweed, and *Echinochloa* includes several species of native millet.

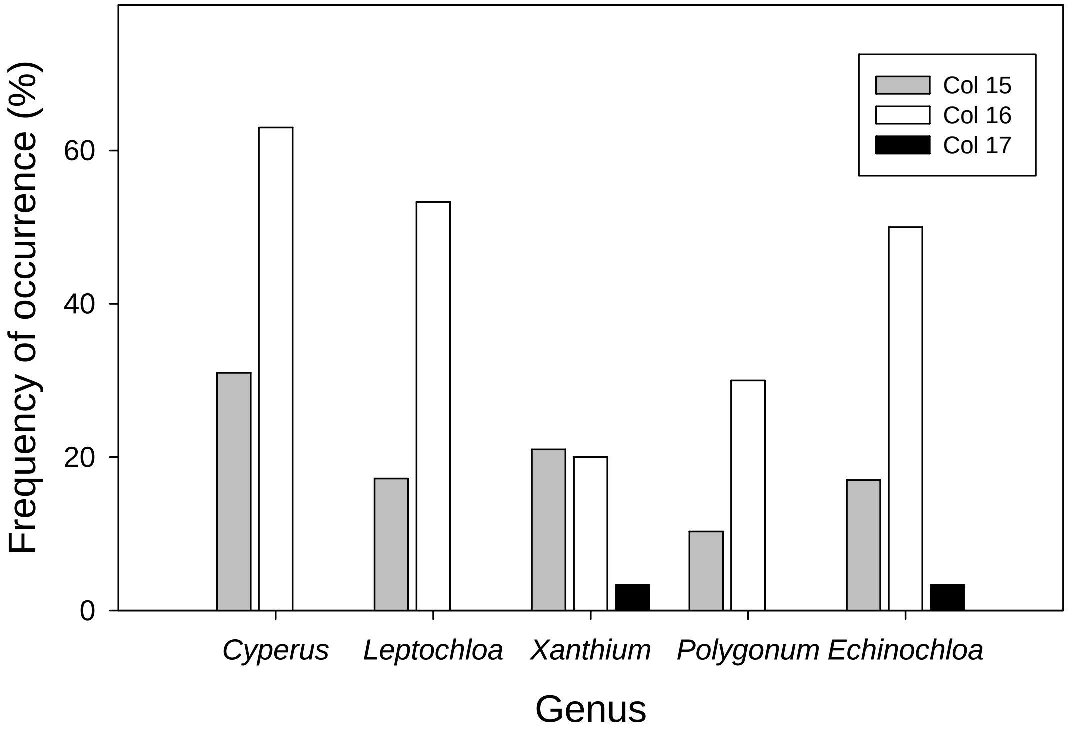
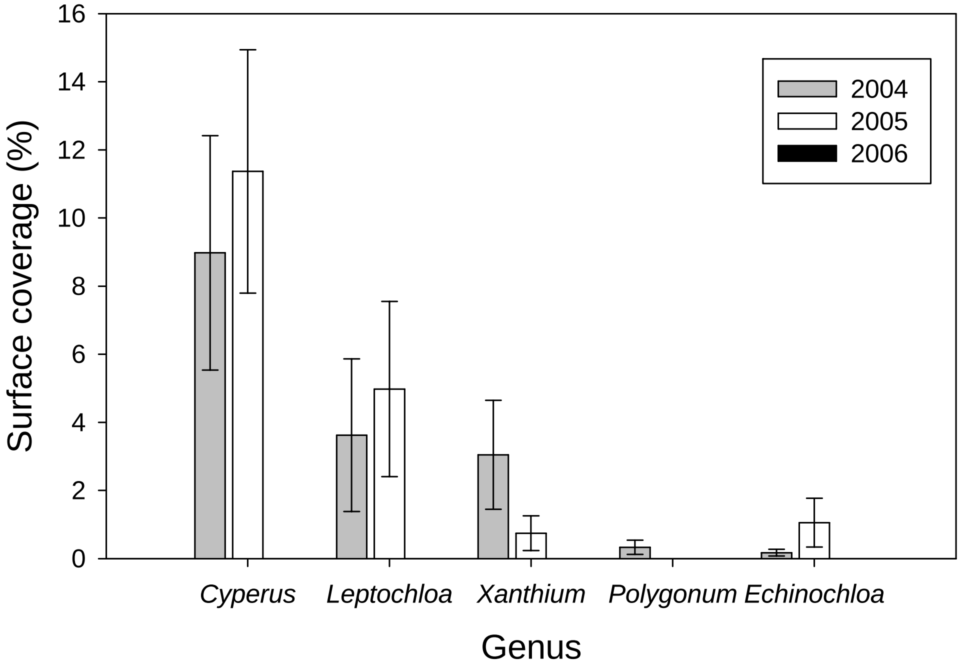


Figure AV2. Mean surface coverage (± standard error) for the five most common taxa of emergent vegetation from post-project monitoring, late-season sampling, in the middle unit of Swan Lake. *Cyperus* is redroot flatsedge, *Leptochloa* is Amazon sprangletop, *Xanthium* is rough cocklebur, *Polygonum* is curlytop knotweed, and *Echinochloa* includes several species of native millet.



MACROINVERTEBRATES

Project Goals:

The 1991 Definite Project Report (U.S. Army Corps of Engineers 1991) included no specific goals for benthic macroinvertebrates, though there was a goal to restore the macroinvertebrate communities associated with submersed aquatic vegetation (SAV). Because SAV has not been re-established in Swan Lake, this goal has not been met and is not further addressed in this chapter. Analysis of pre-project monitoring revealed that the upper unit of Swan Lake had a more diverse community of benthic macroinvertebrates relative to the lower or middle units (Theiling et al. 2000). A logical goal for the project would be to increase the richness and diversity of benthic macroinvertebrates in the lower and middle units of Swan Lake through hardening sediments and the establishment of aquatic vegetation (emergent and moist soil).

METHODS

Macroinvertebrate samples were collected during the months of May and July, 2004 and 2005, at 20 random sites each in the upper, middle, and lower units with a standard ponar grab (524 cm^2). Benthos samples were washed through a U.S. No. 35 Sieve (0.5 mm) and preserved in 10% buffered formalin. These methods were identical to those used for pre-project monitoring in 1992. Samples were stored at the laboratory until sampling in all units was complete, and then transported to Southern Illinois University for processing. Taxa were classified at least to family (to a lower classification when possible), enumerated, and total length (mm) was measured for intact specimens.

Analyses:

We analyzed catch per unit effort (numbers) of benthic macroinvertebrates from standard ponar samples. A combination of univariate and multivariate techniques were used to test for differences between pre- and post-project monitoring. Analysis of variance was used to test for differences in taxon richness and taxon diversity from between project periods and units. We used the Shannon index as a measure of species diversity:

$$H' = -\sum_{i=1}^{s} p_i \ln p_i$$

Where:

S = total number of taxa

p_i = the relative abundance of each taxon i (number of individuals in taxon i / the total number of individuals for all taxa).

We used analysis of similarity to test for differences in community structure between pre- and post-project samples for each unit. Analysis of similarity (ANOSIM) is a multivariate corollary to ANOVA used to test for differences among groups of samples (Clarke and Warwick 2001). A similarity matrix (Bray Curtis) is computed yielding a single measure of similarity between all pairs of samples. From this matrix, ANOSIM computes a test statistic, R, based on the differences in rank similarities within and between groups. Values of R close to 1 indicate strong differences between groups, whereas values of R close to zero indicate weak differences among groups. A p-value is calculated through a random permutation procedure, and is the proportion of times a random combination of similarity values yields a greater R than the original similarity matrix. For these analyses, p-values were based on 10,000 random permutations of each similarity matrix. We limited the number of taxa analyzed through ANOSIM to those that occurred in at least 5% of all samples for each gear type.

RESULTS AND DISCUSSION

A total of 15,677 macroinvertebrates were captured comprising 21 taxa (Table M1). Chironomids were the dominant taxon in both pre-project and post-project monitoring, and exhibited the greatest change in CPUE between these two periods. The abundance of chironomids decreased dramatically in the lower and middle unit of Swan Lake, whereas abundance increased in the upper unit. Several taxon were collected during post-project monitoring that were not collected during pre-project monitoring, including oligochaetes, and several mayfly taxa (Table M1).

Management of the lower and middle units appears to have increased taxon diversity of macroinvertebrates, whereas taxon richness increases cannot be attributed to management practices. Species richness increased significantly in all three units from pre-project to post-project (Table M2, Figure M1). Therefore, this increase appears to be the result of some factor common to all three units rather than a result of the management of the lower and middle unit. Shannon diversity increased significantly in the lower and middle units, but did not increase significantly in the upper unit (Table M2, Figure M1). Therefore, the increase in diversity appears to be related to the management of the lower and middle unit. It is very likely that the decreased CPUE of chironomids in these two units had the greatest influence on taxon diversity.

Community structure of macroinvertebrates varied substantially between pre- and post-project monitoring in all three units, but the changes in the upper unit were very different from those in the lower and middle units. Shifts in community structure in the upper unit between the pre- and post-project monitoring were significant ($R = 0.440$; $P < 0.001$) and were driven by increases in the CPUE of chironomids, ceratopogonids, and hirudinea (Figure M2). Pre- and post-project community structure also varied significantly in the middle ($R = 0.178$; $P = 0.0270$) and lower ($R = 0.590$; $P < 0.001$) units. For both of these units, however, CPUE of chironomids and ceratopogonids decreased substantially from pre- to post-project monitoring (Figure M2). In general, macroinvertebrate community structure in the lower and middle units were more similar to the community structure in the upper unit in the post-project monitoring than in the pre-project monitoring. This is a positive result that can be attributed to management of these two units, specifically in the firming of sediments associated with draw-downs.

CONCLUSIONS

The only macroinvertebrate goal identified in 1991 Definite Project Report (U.S. Army Corps of Engineers 1991) was to restore macroinvertebrate communities associated with submersed aquatic vegetation. Because submersed aquatic vegetation has not been re-established in the lower and middle unites of Swan Lake, this goal was not achieved. Management of the lower and middle units, specifically draw-downs to compact sediments, appears to have produced some benefits to benthic macroinvertebrates. Diversity increased significantly in both the middle and lower units, whereas there was no significant change in the upper unit. Much of this increase in diversity can be attributed to a decline in the dominance of chironomids. Timmermann (2007) demonstrated another effect of draw-downs on benthic macroinvertebrates, noting an increase in the proportion of invertebrate biomass present in the top 10 cm of sediment where draw-downs successfully hardened sediments (Appendix A). This finding appears to have important consequences to benthic fishes and may also be relevant to waterfowl.

Table M1. Total number of individuals captured for each taxon in each unit for pre-projects (Pre) and post-project (Post) monitoring. Pre-project monitoring was conducted in 1992 and post-project monitoring was conducted in 2004 and 2005.

		Upper		Middle		Lower	
Taxa	**Common name**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
Mollusca	molluscs						
Gastropoda	Snails	76	30	0	10	0	0
Sphaeriidae	fingernail clams	0	142	0	45	0	121
Annelida	segmented worms						
Oligochaeta	earth worms	0	599	0	273	0	232
Hirudinea	Leech	208	180	6	88	0	38
Arthropoda	arthropods						
Arachnida	spiders and mites						
Tetragnathidae	stretch spiders	0	2	0	0	0	0
Hydrachnida	Water mites	7	0	0	2	0	0
Insecta	insects						
Ephemperoptera	mayflies						
Ephemeridae	mayfly	0	0	0	0	0	1
Coenagrionidae	mayfly	0	2	0	0	0	0
Baetidae	mayfly	0	2	0	0	0	0
Caenidae	mayfly	0	14	0	0	0	0
Odonata	dragonflies, damselflies						
Libellulidae	dragonfly	1	9	0	0	0	0
Trichoptera	caddisflies						
Leptoceridae	caddis fly	0	4	0	0	0	0
Heteroptera	aquatic bugs						
Coryxidae	Water boatmen	33	225	16	22	1	15
Mesoveliidae	Water striders	0	5	0	0	0	0
Coleoptera	beetles						
Dytiscidae	Water beetles	0	2	0	0	0	0
Heteroceridae	subaquatic beetles	0	2	0	0	0	0
Hydrophilidae	Water beetles	0	9	0	10	0	0
Diptera	flies and midges						
Chironomidae	non-biting midges	320	2707	2549	2049	3484	1855
Ceratopogonidae	Biting midges	9	94	79	4	73	9
Crustacea	crustaceans						
Isopoda	sow bugs	3	3	0	3	0	0
Amphipoda	Scuds	0	3	0	1	0	0

Table M2. ANOVA results for taxon richness and diversity of macroinvertebrates collected during pre- and post-project monitoring.

Source	**DF**	**Mean Square**	**F**	**P**
		Taxon Richness		
Model	5	22.692	16.77	<0.001
Error	151	1.353		
Unit	2	19.261	14.24	<0.001
Period	1	57.550	42.54	<0.001
Unit*Period	2	1.173	0.87	0.422
Contrasts				
Period – Lower	1	17.816	13.17	<0.001
Period – Middle	1	10.833	8.01	0.005
Period – Upper	1	32.633	24.12	<0.001
		Shannon Diversity		
Model	5	1.528	15.27	<0.001
Error	151	0.100		
Unit	2	2.106	21.04	<0.001
Period	1	3.736	37.33	<0.001
Unit*Period	2	0.551	5.51	0.005
Contrasts				
Period – Lower	1	2.926	29.23	<0.001
Period – Middle	1	1.730	17.29	<0.001
Period – Upper	1	0.092	0.92	0.339

Figure M1. Taxon richness diversity for macroinvertebrates collected in the upper, middle, and lower units of Swan Lake during pre- (1992) and post-project (2004 – 2005) monitoring.

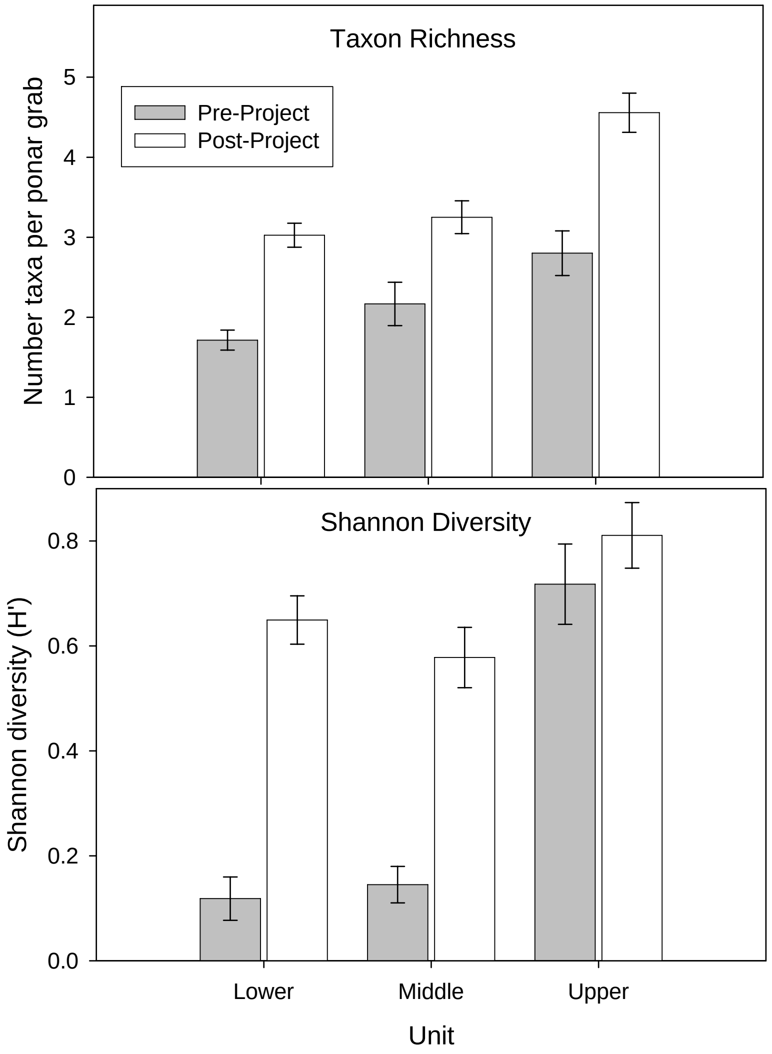
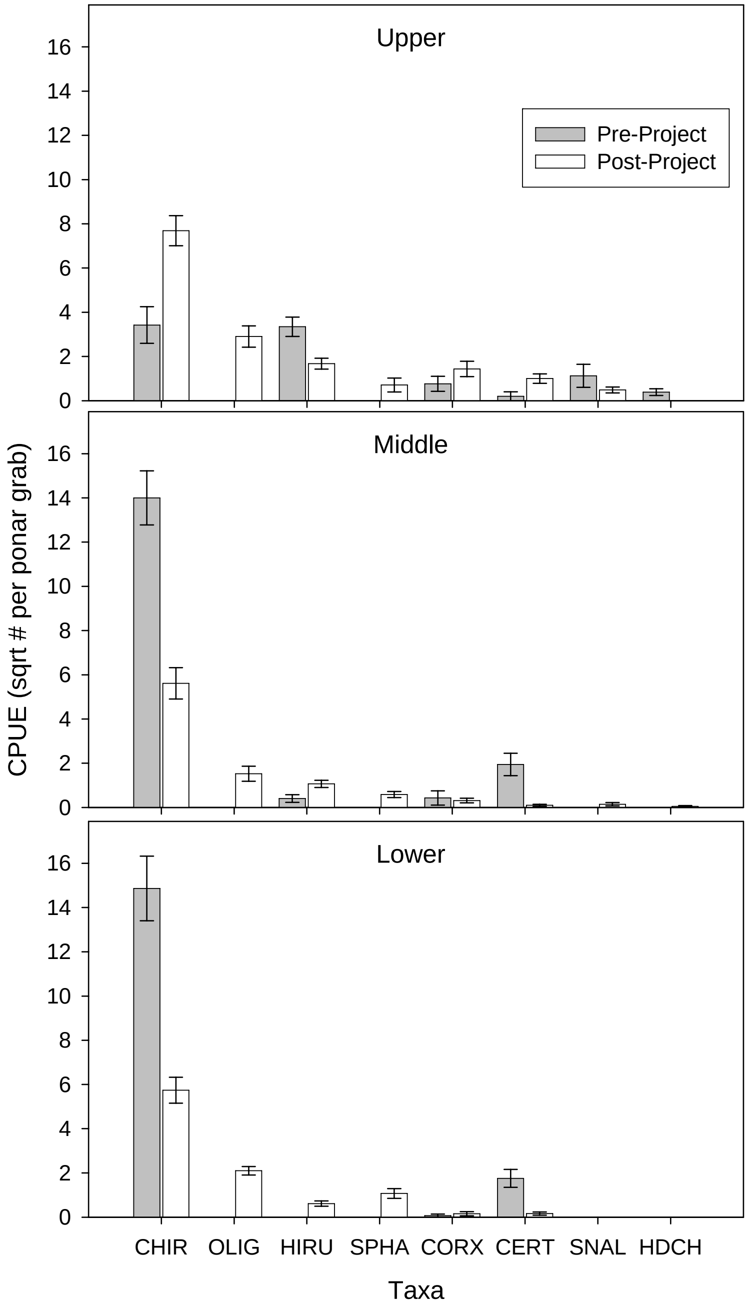


Figure M2. Catch-per-unit-effort of macroinvertebrates collected from the upper, middle, and lower units of Swan Lake during pre- (1992) and post-project (2004 – 2005) monitoring.



FISH

Project Goals:

Swan Lake encompasses a significant portion of the available spawning, rearing, and overwinter habitat for the lower Illinois River and Pool 26 of the Mississippi River (U.S. Army Corps of Engineers 1991). The project goals specific to fishes identified in the 1991 Definite Project Report (U.S. Army Corps of Engineers 1991) include providing habitat for over winter survival of fish, and habitat for spawning and rearing of fish.

METHODS

Small, intermediate, and large size classes of fish were sampled in both summer (June-September) and winter (November-March) from 2004 to 2006 at 20 random sites (same as macroinvertebrate sites) in each unit using three gears. At each site, samples were taken with mini-fyke nets (for small size classes; Wisconsin-style fyke nets; 15’ X 2’ lead, two 2.2’ X 4’ frames, two 2’ diameter hoops, one throat with a 2” opening, and 1/8” ace-style, green-dipped mesh), fyke nets (for intermediate size classes; Wisconsin-style fyke nets; 50’ X 4.5’ lead, two 3’ X 6’ frames, six 3’ diameter hoops, two throats - the front throat is 20 meshes X 40 meshes around and the back throat is 28 meshes X 32 meshes around, 3/4” bar mesh of #12 bonded nylon treated with net coat), and trammel nets (for large size classes; 300’ long, outer panels of 14” bar mesh and an inner panel of 3” bar mesh of multifilament nylon). Leads of mini-fyke and fyke nets were tied together and fished in tandem in the lower and middle units. A site was deemed suitable for sampling if depth was greater than or equal to 40 cm (the depth necessary to submerge the throats of fyke nets). Low water conditions in the upper unit, due to draw-downs for aquatic nuisance species/aquatic vegetation management, prevented fish sampling in this compartment of Swan Lake. Tandem mini-fyke and fyke nets were fished for 24 h, and trammel nets were fished for 1 h. Tandem mini-fyke nets were only fished during the summer period when YOY fishes were expected to be present in the lake. These methods were identical to those used in pre-project monitoring conducted in 1992 with the exception of the time that trammel nets were fished. During pre-project monitoring, trammel nets were fished for 24 h. We limited trammel net sets to 1 h during post-project monitoring because of the establishment of Asian carp in Swan Lake. We felt it likely that 24 h trammel net sets would lead to extremely large catches of Asian carp and unacceptable levels of gear destruction.

Large fishes were measured to total length (TL; mm) weighed (g) and released. Fishes greater than 200 mm TL received a left-pelvic fin clip. Many small fishes (TL < 100 mm; i.e., the primary catch in mini-fyke samples) were transferred to sample jars, preserved in 10% formalin, and transported to the lab for processing. Standard water quality parameters (depth, temperature, DO, conductivity, Secchi depth, turbidity, sediment hardness) were measured at all fish sites.

Analyses:

We analyzed catch per unit effort in numbers for tandem fyke and tandem mini-fyke samples because small individuals were not weighed with sufficient precision during pre-project monitoring. Trammel net data were converted to relative abundance (biomass), because of the difference in fishing time between pre- and post-project monitoring. A combination of univariate and multivariate techniques were used to test for differences between pre- and post-project monitoring. Analysis of variance was used to test for differences in species richness and species diversity from tandem fyke data between project periods and units. We chose only to analyze tandem fyke data for richness and diversity, because this gear captures a greater number of species relative to tandem mini-fykes and trammel nets. Separate analyses were run for summer and winter samples. We used the Shannon index as a measure of species diversity:

$$H' = -\sum_{i=1}^{s} p_i \ln p_i$$

Where:

S = total number of species

p_i = the relative abundance of each species i (number of individuals in species i / the total number of individuals for all species).

We used analysis of similarity to test for differences in community structure between pre- and post-project samples. Analysis of similarity (ANOSIM) is a multivariate corollary to ANOVA used to test for differences among groups of samples (Clarke and Warwick 2001). A similarity matrix (Bray Curtis) is computed yielding a single measure of similarity between all pairs of samples. From this matrix, ANOSIM computes a test statistic, R, based on the differences in rank similarities within and between groups. Values of R close to 1 indicate strong differences between groups, whereas values of R close to zero indicate weak differences among groups. A p-value is calculated through a random permutation procedure, and is the proportion of times a random combination of similarity values yields a greater R than the original similarity matrix. For these analyses, p-values were based on 10,000 random permutations of each similarity matrix. We limited the number of species analyzed through ANOSIM to those that occurred in at least 5% of all samples for each gear type. For tandem mini-fyke nets, we further limited our analyses to young-of-the-year (YOY) fishes (determined through length criteria) and fishes of small adult size.

Separate ANOSIM tests were conducted for each unit, season, and gear type, yielding a total of ten tests. Because of this high number of tests and the fact that analysis of similarity is a relatively powerful statistical test (Clarke and Warwick 2001), we used a Bonferroni adjustment with $P \leq 0.005$ indicating statistical significance. For tandem fyke and trammel net data, we also tested for differences between seasons for each unit within both pre- and post-project monitoring periods. This yielded eight tests for seasonal effects, so we adjusted to $P \leq 0.006$ for statistical significance.

RESULTS AND DISCUSSION

A total of 36 and 43 species of fish were captured in the pre-project and post-project monitoring periods respectively (Table F1). The five species captured in the greatest abundance during pre-project monitoring were gizzard shad, bluegill, freshwater drum, carp, and emerald shiner. During post-project monitoring, neither bluegill nor carp were among the top five species captured in the greatest abundance, whereas orangespotted sunfish and silver band shiner were in the top five. Common carp have been declining in abundance since the 1993 flood throughout the UMRS (Chick et al. 2005). Bluegill are one of several species, including black crappie, white crappie, bigmouth buffalo, black buffalo, smallmouth buffalo, and river carpsucker, displaying decreases in total catch from the pre-project monitoring to the post-project monitoring (Table F1). Other species, including Asian carp, orangespotted sunfish, green sunfish, emerald shiner, silverband shiner, and mosquitofish, were captured in far greater numbers during post-project monitoring. Note that when comparing the total number of fish captured, it is important to consider that pre-project monitoring consists of one year of data, whereas post-project monitoring consists of two to three years of data.

Despite the fact that more species were captured across the whole post-project monitoring period, tandem fyke net samples during post-project monitoring had lower species richness and diversity on average compared to pre-project monitoring. Mean species richness was significantly greater during pre-project monitoring during the summer for both the lower and middle units (Table F2, Figure F1). Similar patterns were observed in the summer for diversity, though the statistical model was not significant. During the winter, both species richness and diversity were significantly greater for pre-project monitoring relative to post-project monitoring, with differences especially pronounced in the middle unit (Table F2, Figure F1).

Tandem Fyke net samples yielded little evidence of a consistent shift in community structure between the pre- and post-project periods for either unit (Figure F2, F3). Community structure in the lower unit did not differ significantly between pre- and post-project monitoring for either the summer ($R = 0.002$; $P = 0.465$) or winter ($R = -0.041$; $P = 0.707$). Similarly, there was no significant shift in community structure for the middle unit between pre- and post project monitoring in the summer ($R = -0.039$; $P = 0.65$) or winter ($R = -0.086$; $P = 0.844$). Although mean CPUE of several species was greater during pre-project monitoring, high variance among samples likely resulted in the lack of any consistent shifts in community structure between project monitoring periods (Figure F2, F3).

During both the pre- and post-project monitoring, tandem fyke samples displayed substantial shifts between seasons in community structure (Figures F2, F3). Shifts in community structure between seasons were stronger during post-project monitoring ($R = 0.245$; $P \leq 0.001$)

compared to pre-project monitoring (R = 0.101; P = 0.025) for the lower unit. During pre-project monitoring, there was a general decline from the summer to the winter for most species, whereas there were increases in CPUE of gizzard shad and white bass in the winter during post-project monitoring suggesting some use of this unit as a winter refuge for these species (Figure F2). Shifts between seasons were also stronger in the middle unit during post-project monitoring (R = 0.411; P ≤ 0.001) compared to pre-project monitoring (R = 0.115; P = 0.045), but species patterns differed greatly from the lower unit. During pre-project monitoring, increased CPUE for black crappie, white crappie, and bluegill suggested use of the middle unit as a winter refuge (Figure F3; Theiling et al. 2000). During post-project monitoring, only gizzard shad and white bass showed increases in CPUE during the winter, suggesting much of the winter refuge value of this unit has been reduced for centrarchids (Figure F3).

Differences in the relative abundance (mass) of fishes from trammel nets between pre- and post-project monitoring were most obvious when comparing winter use of the Swan Lake (Figure F4, F5). During the summer, there were no significant community-level shifts between pre- and post-project monitoring for either the lower (R = -0.013; P = 0.521) or middle units (R = 0.165; P = 0.05). During the winter, significant community-level shifts occurred in the lower unit (R = 0.349; P ≤ 0.001), and shifts in the middle unit were also more pronounced, though not significant (R = 0.262; P ≤ 0.009). In the lower unit, relative abundance of bigmouth buffalo and black buffalo were greater during pre-project monitoring, whereas relative abundance of common carp and bighead carp were greater during post-project sampling (Figure F4). Similar patterns were also observed in the middle unit (Figure F5). Overall, both units appear to be used far less as a winter refuge for bigmouth and black buffalo compared to the pre-project monitoring. Community-level shifts between seasons in relative abundance were not significant in either pre- (R = 0.067; P = 0.084) or post-project (R = 0.041; P = 0.023) monitoring in the lower unit, nor were shifts significant in either the pre- (R = 0.182; P = 0.021) or post-project (R = 0.044; P = 0.132) monitoring in the middle unit.

There were notable differences in the use of Swan Lake by YOY fishes and fishes of small adult size (Figures F6, F7). Community structure from tandem mini-fyke nets differed significantly between pre- and post-project monitoring in the lower unit (R = 0.616; P < 0.001). Catch-per-unit-effort of YOY fishes in the lower unit was substantially greater during pre-project monitoring, compared to post-project monitoring, for freshwater drum, bluegill, carp, and buffalo spp. (Figure F6). On the other hand, CPUE of emerald shiner and orangespotted sunfish were greater during post-project monitoring, compared to pre-project monitoring, in the lower unit (Figure F7). Overall, there was no significant community structure shifts between pre- and post-project monitoring for the middle unit (R = 0.127; P = 0.006), though CPUE of emerald shiner, orangespotted sunfish, mosquitofish, and silverband shiner were all greater during post-project monitoring (Figure F7). In terms of the use of the middle unit by YOY fishes, this is a positive result. Only the CPUE of bluegill dropped substantially between the pre- and post-project monitoring, suggesting that the middle unit has maintained its value as a nursery area to a greater extent than the lower unit (Figure F6).

CONCLUSIONS

Comparisons of pre- and post-project monitoring indicate that most of the desired benefits for fishes have not been fully achieved to date. There appears to be notable declines in the use of the lake by several species, including most centrarchids and buffalo. Both species richness and diversity have declined relative to pre-project levels, and there is little evidence of use of the lake for overwinter habitat with the exception of gizzard shad and white bass. We feel the primary reasons for the limited success of the project for fishes are: 1) a need to further improve habitat through draw-downs, especially in the lower unit, 2) the failure to re-establish submersed aquatic vegetation, 3) a need to provide deep-water habitat, and 4) the need for additional time for habitat improvements to translate to changes in the fish community.

The most critical aspect of improving habitat in Swan Lake is to harden lake sediments. Firm sediments provide better rooting substrate for macrophytes, reduce turbidity associated with wind and wave resuspension, and keep a greater proportion of benthic macroinvertebrates in the top 10 cm of sediments where they are more vulnerable to benthic fishes (see appendix A). From this perspective, the middle unit of Swan Lake is currently in better condition than the lower unit. For fishes, this improved habitat appears to have translated to improved foraging for benthivorous fishes (appendix A) and greater use of the middle unit by YOY fishes. If submersed aquatic vegetation can be re-established in Swan Lake, this will yield further changes in the fish community.

The decline in fish use of Swan Lake for overwinter habitat, especially in the middle unit, is a disappointment. There are at least three possible explanations for this result: 1) reduced overwinter use might be a result of reduced connectivity with the Illinois River (i.e., reduced size of the connection), 2) a lack of deep-water habitat in either unit could explain this result, or 3) reduced use of the lake for overwinter habitat could result from climate change. This last explanation appears to be the weakest. Although climatic warming is a viable possibility, temperature in Pool 26 of the Mississippi river dropped below 1° C during the winter of 2004 and 2005 (see online LTRMP WQ data - http://www.umesc.usgs.gov). Although we cannot dismiss the possibility that decreased connectivity with the river influenced our findings, it is somewhat strange that this would influence certain species, but not others (i.e., gizzard shad and white bass). Therefore, we feel the overall lack of deep-water habitat in Swan Lake is the most likely factor influencing our findings. Although attempts were made to provide deep water habitat, these areas have rapidly filled in with sediments. This suggests that efforts to create deep-water habitat will be most successful when made after sufficient hardening of lake sediments through draw-downs. Efforts to create deep-water habitat should not be made until sediment hardening has been verified.

Finally, we caution that it is difficult to predict the time that is necessary to detect fish community responses to habitat improvements. For example, the increases in abundance of orangespotted sunfish and green sunfish post-project is not surprising given the fact that both species are described as pioneering species that take advantage of areas where the fish community has been disrupted (Pflieger 1975). Abundance of both of these species often declines as other centrarchids become established. Therefore, the elevated abundance of these species could be an indication that the fish community is still adjusting to the habitat changes made to this lake.

Table F1. Total number of fishes captured in the lower and middle units of Swan Lake during pre- and post-project monitoring.

			Summer				Winter			
			Lower		Middle		Lower		Middle	
Scientific Name	Common Name	Code	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Lepisosteidae										
Lepisosteus platostomus	shortnose gar	SNGR	112	222	47	214	18	24	84	28
Lepisosteus oculatus	spotted gar	STGR	3	0	0	5	0	0	2	1
Amiidae										
Amia calva	bowfin	BWFN	2	9	15	29	0	1	11	31
Clupeidae										
Alosa chrysochloris	skipjack herring	SJHR	16	8	10	8	0	0	0	0
Dorosoma cepedianum	gizzard shad	GZSD	2431	12439	10110	29523	113	803	103	1778
Dorosoma petenense	threadfin shad	TFSD	2	4	12	0	0	0	0	3
Hiodontidae										
Hiodon tergisus	mooneye	MNEY	0	0	0	0	3	0	0	0
Cyprinidae										
Hypophthalmichthys nobilis	bighead carp	BHCP	1	40	0	92	0	24	0	47
Hypophthalmichthys molitrix	silver carp	SVCP	0	22	0	4	0	4	0	28
Ctenopharyngodon idella	grass carp	GSCP	0	0	0	1	0	2	0	26
Carassius auratus	goldfish	GDFH	5	2	1	2	2	1	3	3
Cyprinus carpio	common carp	CARP	713	124	303	670	371	326	231	254
Notemigonus crysoleucas	golden shiner	GDSN	0	1	0	1	0	0	0	0
Macrhybopsis storeriana	silver chub	SVCB	1	0	2	0	0	0	0	0
Cyprinella spiloptera	spotfin shiner	SFSN	0	0	0	1	0	0	0	0
Pimephales promelas	fathead minnow	FHMW	0	0	6	0	0	0	0	0
Pimephales notatus	bluntnose minnow	BNMW	0	1	0	2	0	0	0	0
Pimephales vigilax	bullhead minnow	BHMW	0	2	1	1	0	0	0	0
Notropis atherinoides	emerald shiner	ERSN	454	12357	396	10132	0	0	0	0
Notropis shumardi	silverband shiner	SBSN	0	1	0	1762	0	0	0	0
Notropis stramineus	sand shiner	SNSN	0	0	0	1	0	0	0	0
Notropis buchanani	ghost shiner	GTSN	1	0	0	0	0	0	0	0
Catostomidae										
Ictiobus cyprinellus	bigmouth buffalo	BMBF	87	9	32	20	93	2	145	40
Ictiobus bubalus	smallmouth buffalo	SMBF	2	0	3	1	6	1	3	2
Ictiobus niger	black buffalo	BKBF	19	9	30	9	58	2	65	7

Table F1 Continued

			Summer				Winter			
			Lower		Middle		Lower		Middle	
Scientific Name	Common Name	Code	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Ictiobus spp.	YOY buffalo	YOYbuff	40	9	146	504	0	2	0	1
Carpiodes carpio	river carpsucker	RVCS	4	0	5	1	9	1	6	2
Catostomus commersonii	white sucker	WTSK	0	0	0	1	0	0	0	0
Ictaluridae										
Ictalurus punctatus	channel catfish	CNCF	28	75	13	54	3	1	30	3
Ameiurus natalis	yellow bullhead	YLBH	11	2	1	28	5	4	16	6
Ameiurus melas	black bullhead	BKBH	1	2	16	53	0	5	6	3
Ameiurus nebulosus	brown bullhead	BNBH	0	0	0	16	0	0	0	1
Noturus gyrinus	tadpole madtom	TPMT	1	0	0	0	0	0	0	0
Poeciliidae										
Gambusia affinis	western mosquitofish	MQTF	23	41	5	975	0	0	0	0
Atherinidae										
Labidesthes sicculus	brook silverside	BKSS	2	2	0	7	0	0	0	0
Percichthyidae										
Morone chrysops	white bass	WTBS	48	196	35	162	35	168	27	39
Morone mississippiensis	yellow bass	YWBS	3	2	1	9	3	13	9	9
Centrarchidae										
Pomoxis nigromaculatus	black crappie	BKCP	43	61	288	458	29	31	328	109
Pomoxis annularis	white crappie	WTCP	39	31	54	96	36	25	292	23
Micropterus salmoides	largemouth bass	LMBS	1	2	0	1	1	1	2	0
Lepomis gulosus	warmouth	WRMH	0	2	0	89	0	0	0	0
Lepomis cyanellus	green sunfish	GNSF	1	14	2	21	0	6	2	1
Lepomis macrochirus	bluegill	BLGL	448	61	1953	596	36	23	231	19
Lepomis humilis	orangespotted sunfish	OSSF	1	1224	45	10921	1	14	0	2
Percidae										
Stizostedion canadense	sauger	SGER	0	0	0	1	0	0	0	0
Etheostoma asprigene	mud darter	MDDR	1	1	0	3	0	0	0	0
Sciaenidae										
Aplodinotus grunniens	freshwater drum	FWDM	792	269	1000	1930	51	26	142	27

Table F2. ANOVA results from analysis of species richness and Shannon diversity index from tandem fyke samples.

Species Richness Summer – $R^2 = 0.08$				
Source	DF	Mean Square	F	P
Model	3	10.876	3.24	0.025
Error	121	3.360		
Unit	1	7.703	2.29	0.133
Period	1	20.638	6.14	0.015
Period*Unit	1	0.218	0.06	0.799
Species Richness Winter – $R^2 = 0.25$				
Model	3	53.071	10.53	< 0.001
Error	96	5.039		
Unit	1	24.885	4.94	0.029
Period	1	113.662	23.15	< 0.001
Period*Unit	1	49.603	9.84	0.002
Shannon Diversity Summer – $R^2 = 0.04$				
Model	3	0.286	1.75	0.16
Error	121	0.163		
Unit	1	0.069	0.42	0.516
Period	1	0.744	4.55	0.035
Period*Unit	1	0.009	0.05	0.820
Shannon Diversity Winter – $R^2 = 0.22$				
Model	3	2.088	9.28	< 0.001
Error	96	0.225		
Unit	1	0.014	0.06	0.801
Period	1	5.64	25.04	< 0.001
Period*Unit	1	0.758	3.37	0.069

Figure F1. Mean species richness and Shannon diversity index from tandem fyke samples in the lower unit of Swan Lake for pre-project (1992 – 1993) and post-project (2004-2006) monitoring periods. Error bars are one standard error.

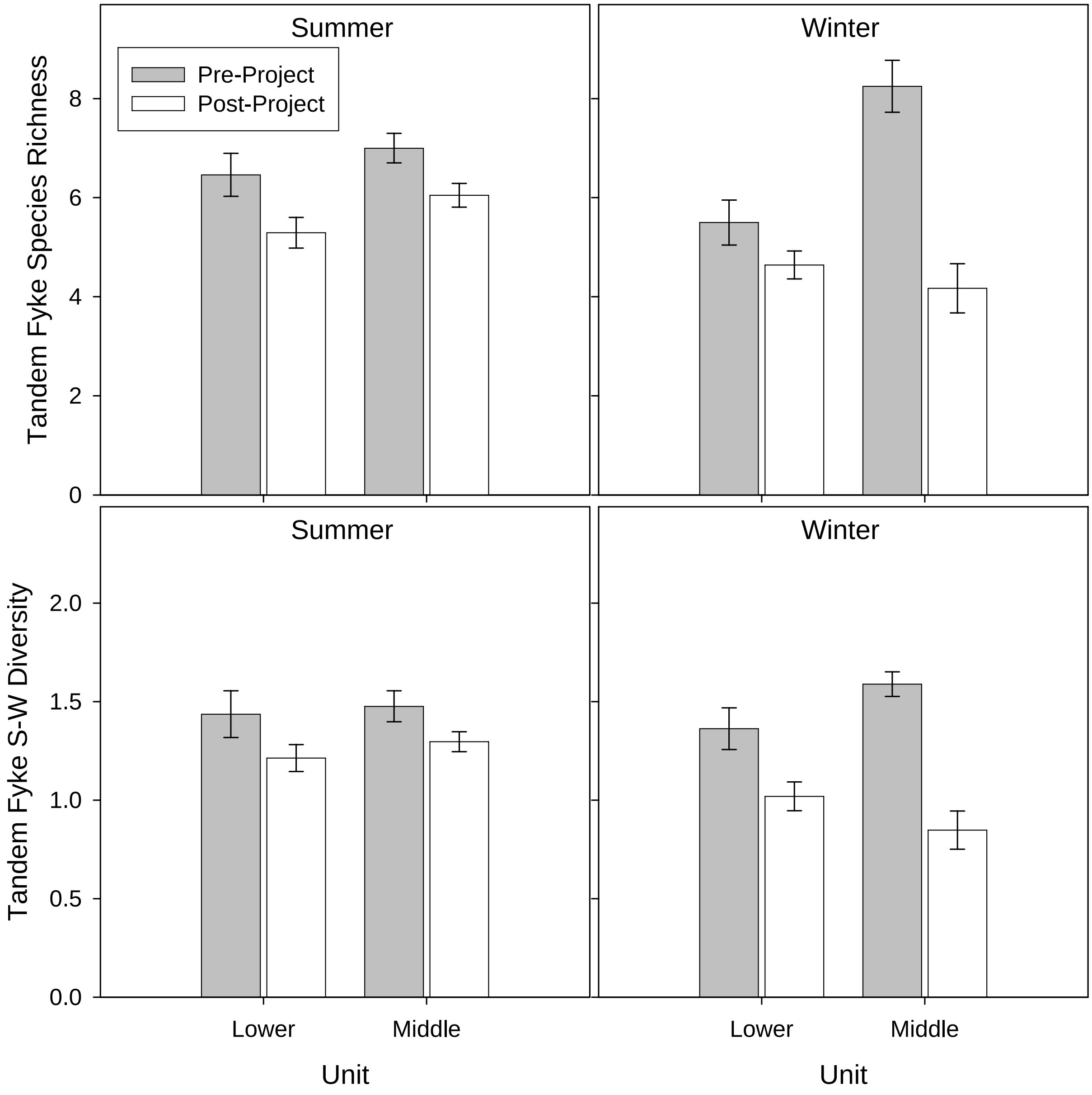


Figure F2. Mean catch-per-unit-effort (CPUE) from tandem fyke nets sets in the lower unit of Swan Lake for pre-project (1992 – 1993) and post-project (2004-2006) monitoring periods. See Table F1 for species codes. Error bars are one standard error.

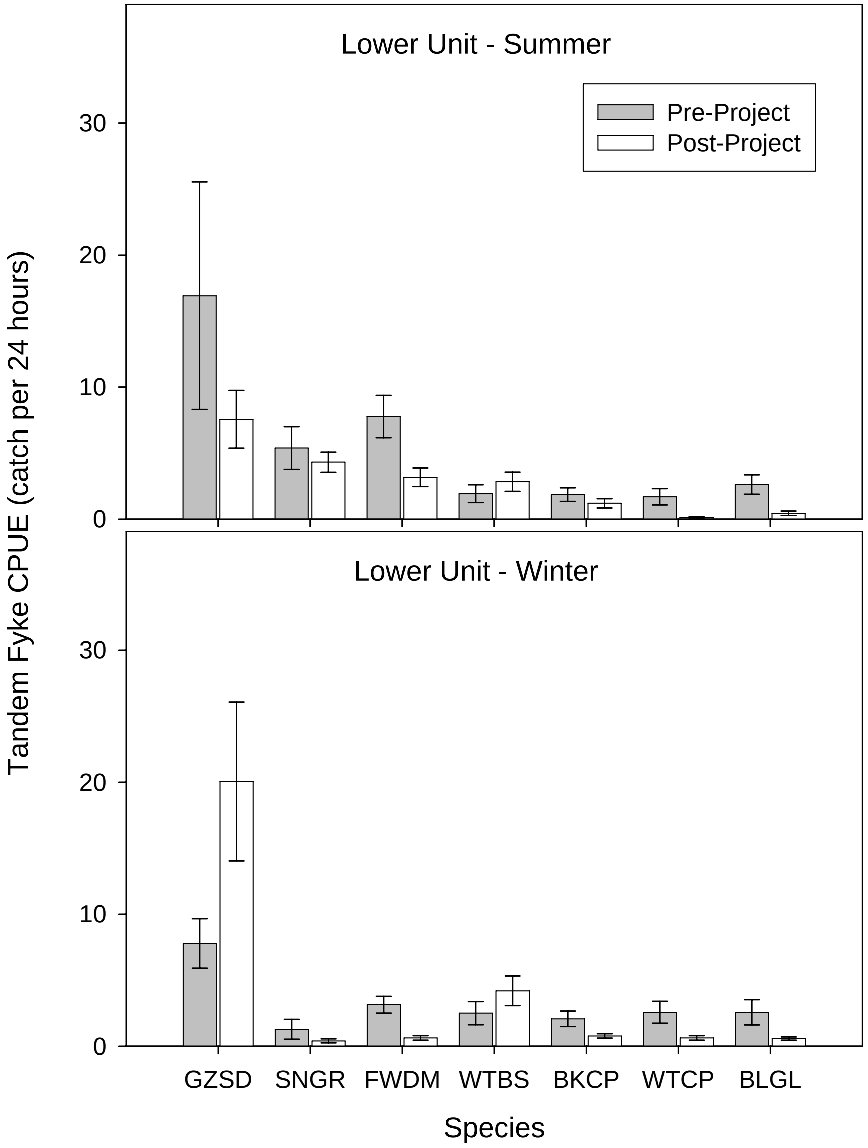

Lower Unit - Summer
Pre-Project
Post-Project
Lower Unit - Winter
Tandem Fyke CPUE (catch per 24 hours)
0
10
20
30
GZSD
SNGR
FWDM
WTBS
BKCP
WTCP
BLGL
Species

Figure F3. Mean catch-per-unit-effort (CPUE) from tandem fyke nets sets in the middle unit of Swan Lake for pre-project (1992 – 1993) and post-project (2004-2006) monitoring periods. See Table F1 for species codes. Error bars are one standard error.

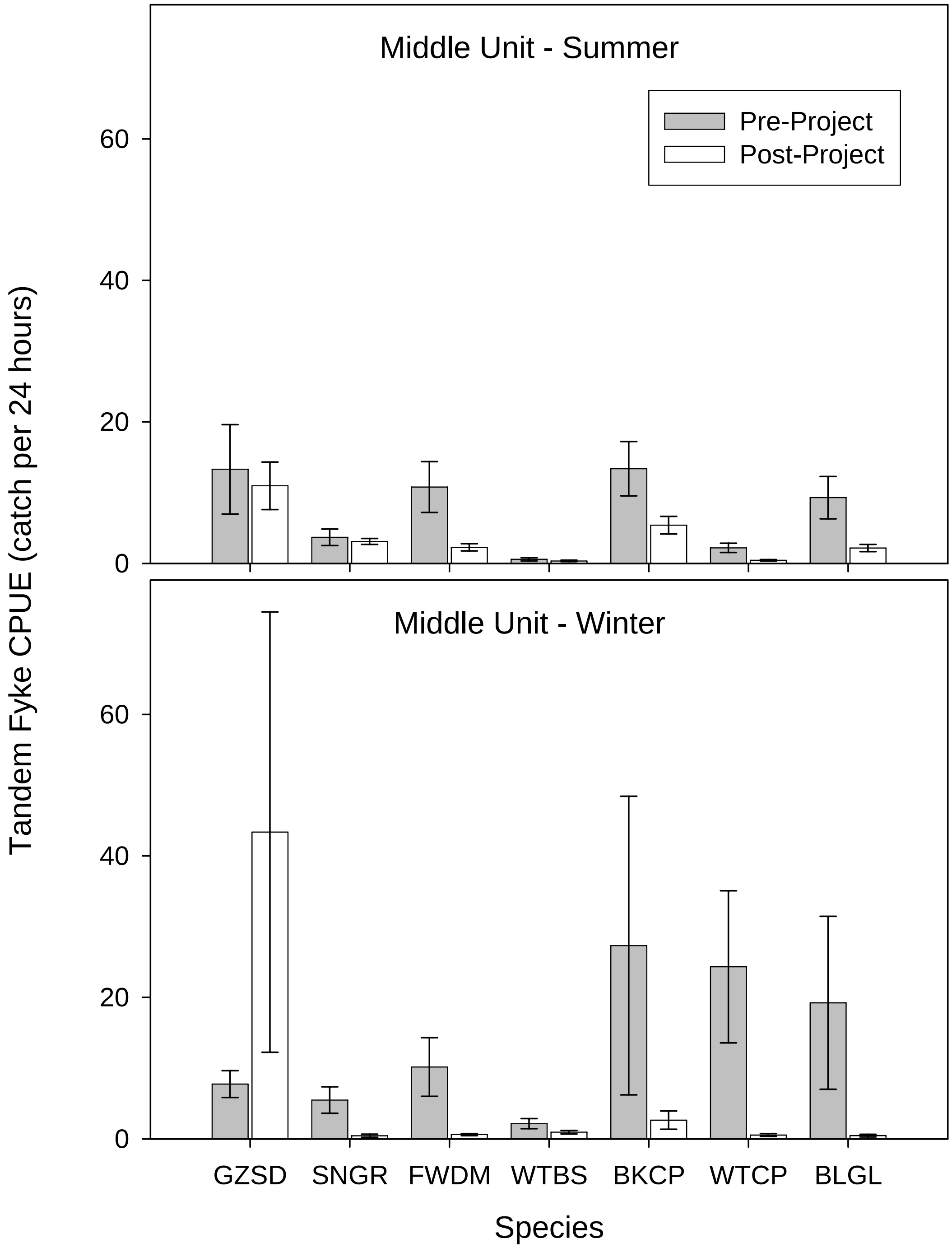


Figure F4. Mean relative abundance from trammel nets sets in the lower unit of Swan Lake for pre-project (1992 – 1993) and post-project (2004-2006) monitoring periods. See Table F1 for species codes. Error bars are one standard error.

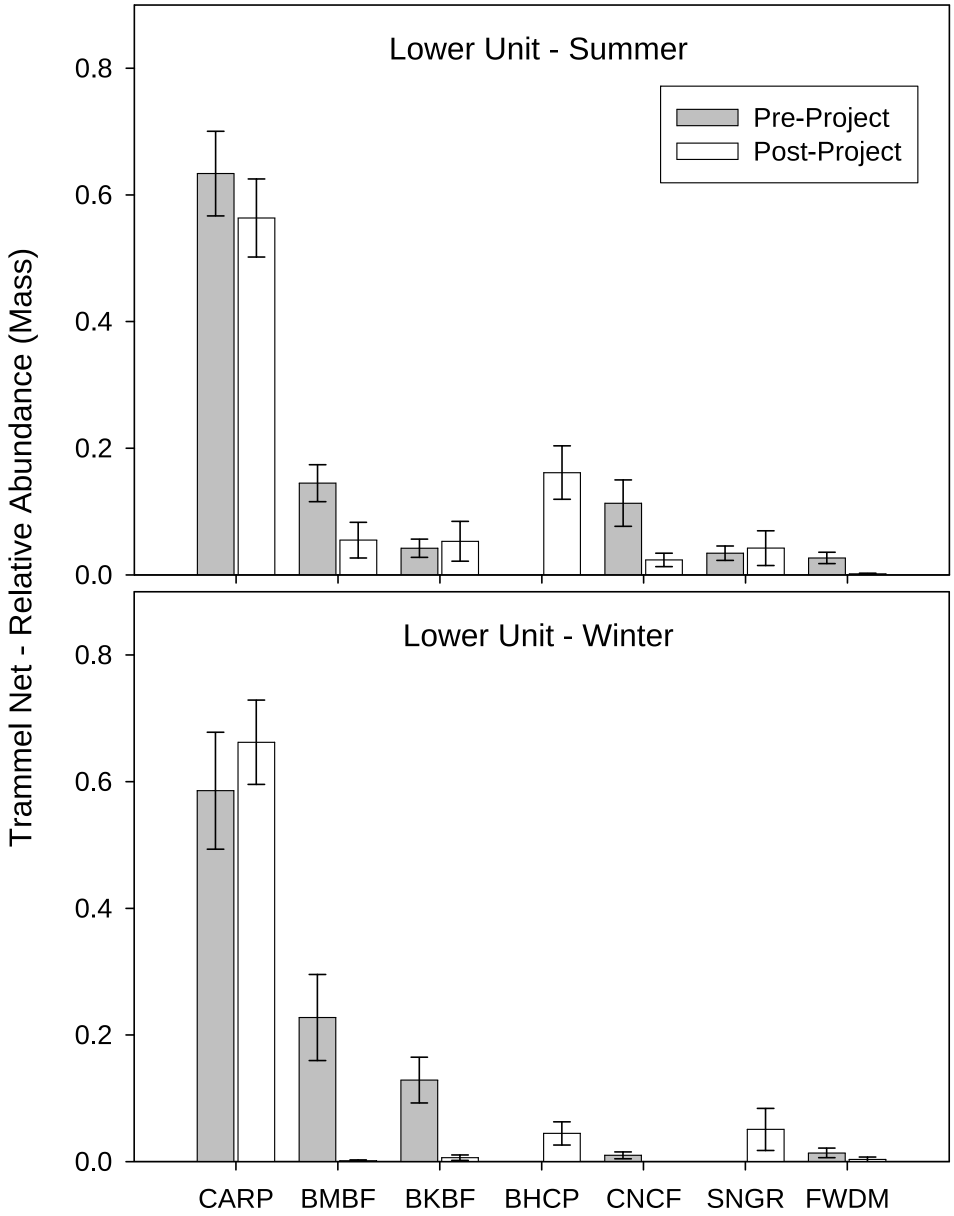


Figure F5. Mean relative abundance from trammel nets sets in the middle unit of Swan Lake for pre-project (1992 – 1993) and post-project (2004-2006) monitoring periods. See Table F1 for species codes. Error bars are one standard error.

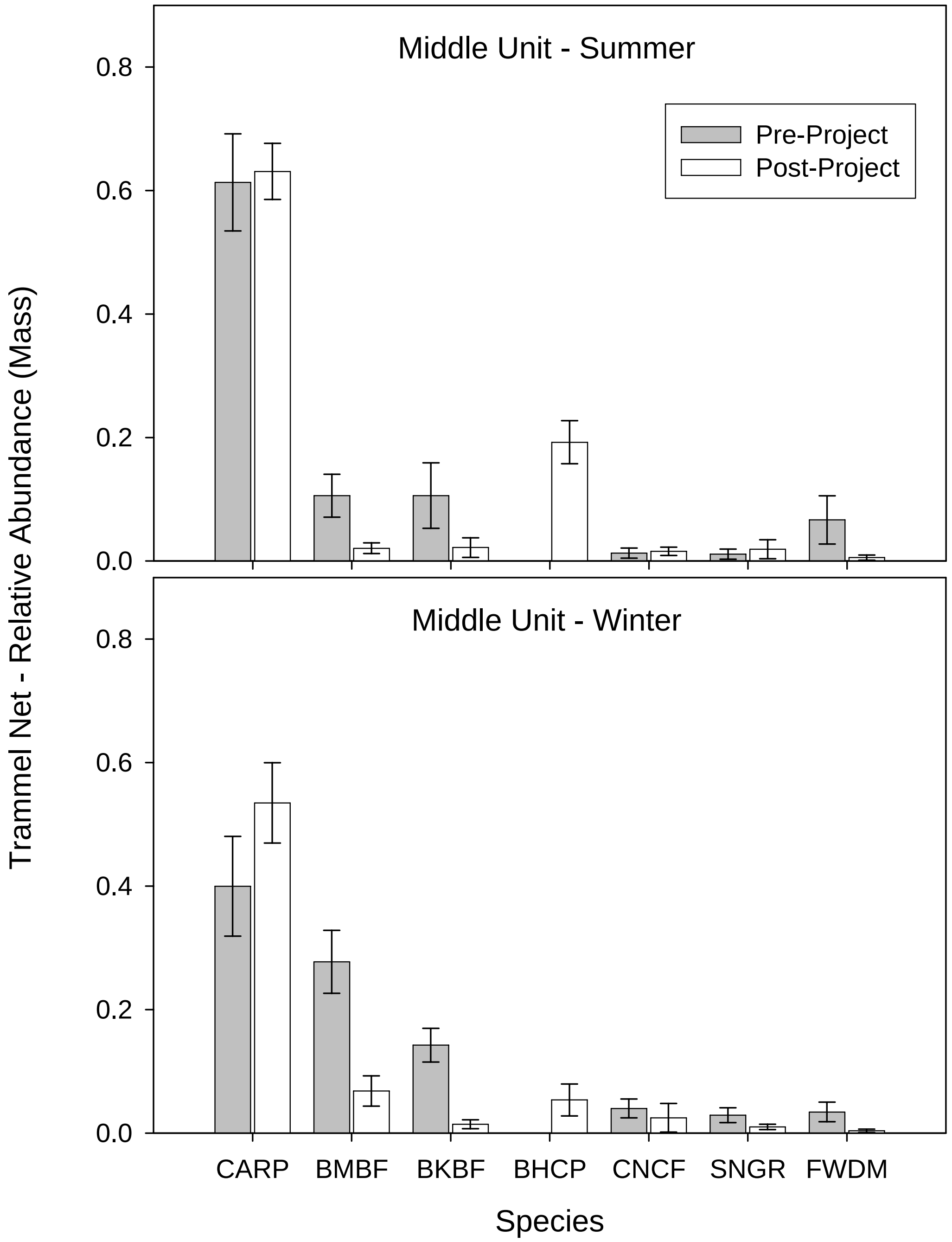


Figure F6. Mean catch-per-unit-effort (CPUE) for young-of-the-year fishes captured in tandem mini-fyke nets sets in Swan Lake for pre-project (1992 – 1993) and post-project (2004-2006) monitoring periods. See Table F1 for species codes. Error bars are one standard error.

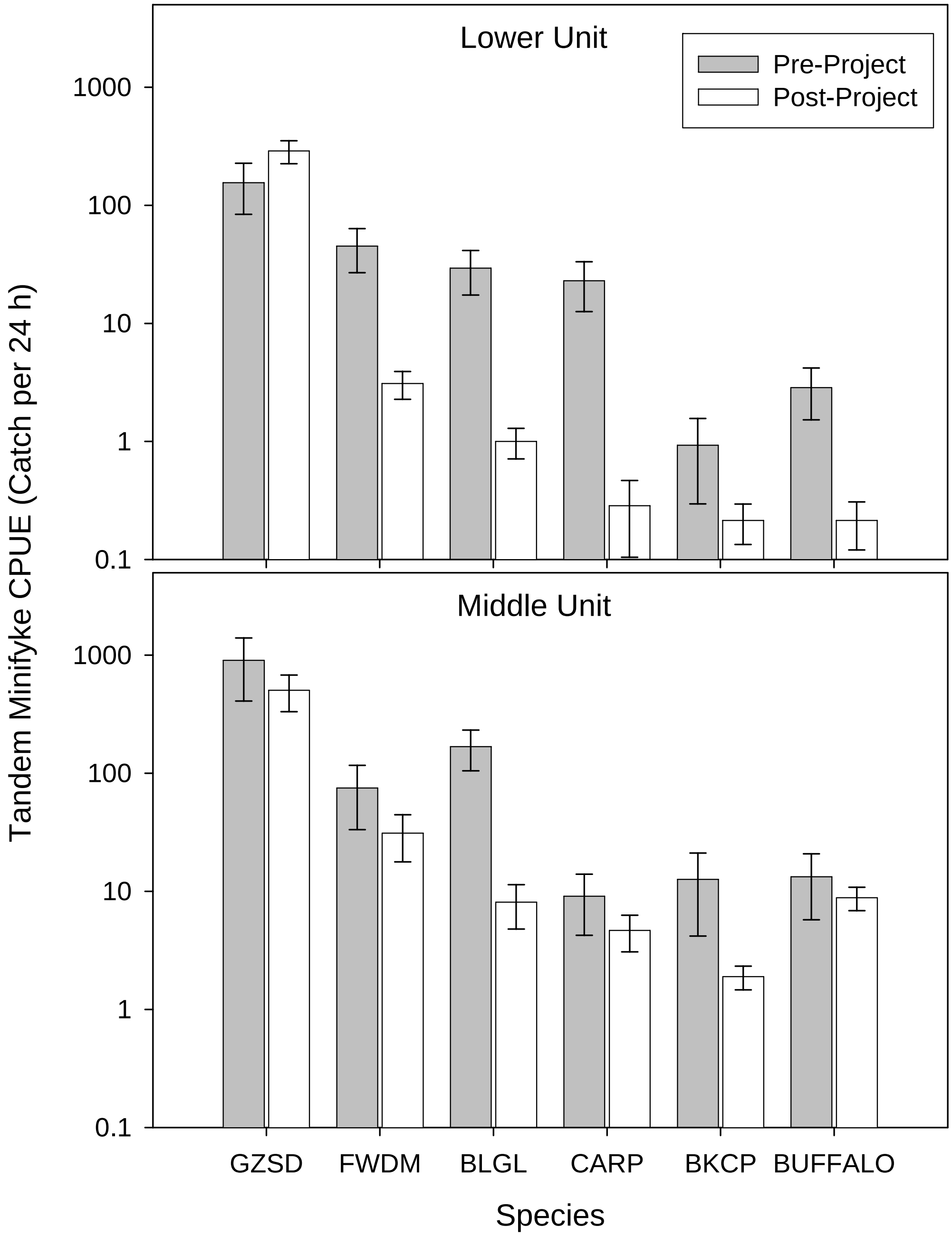
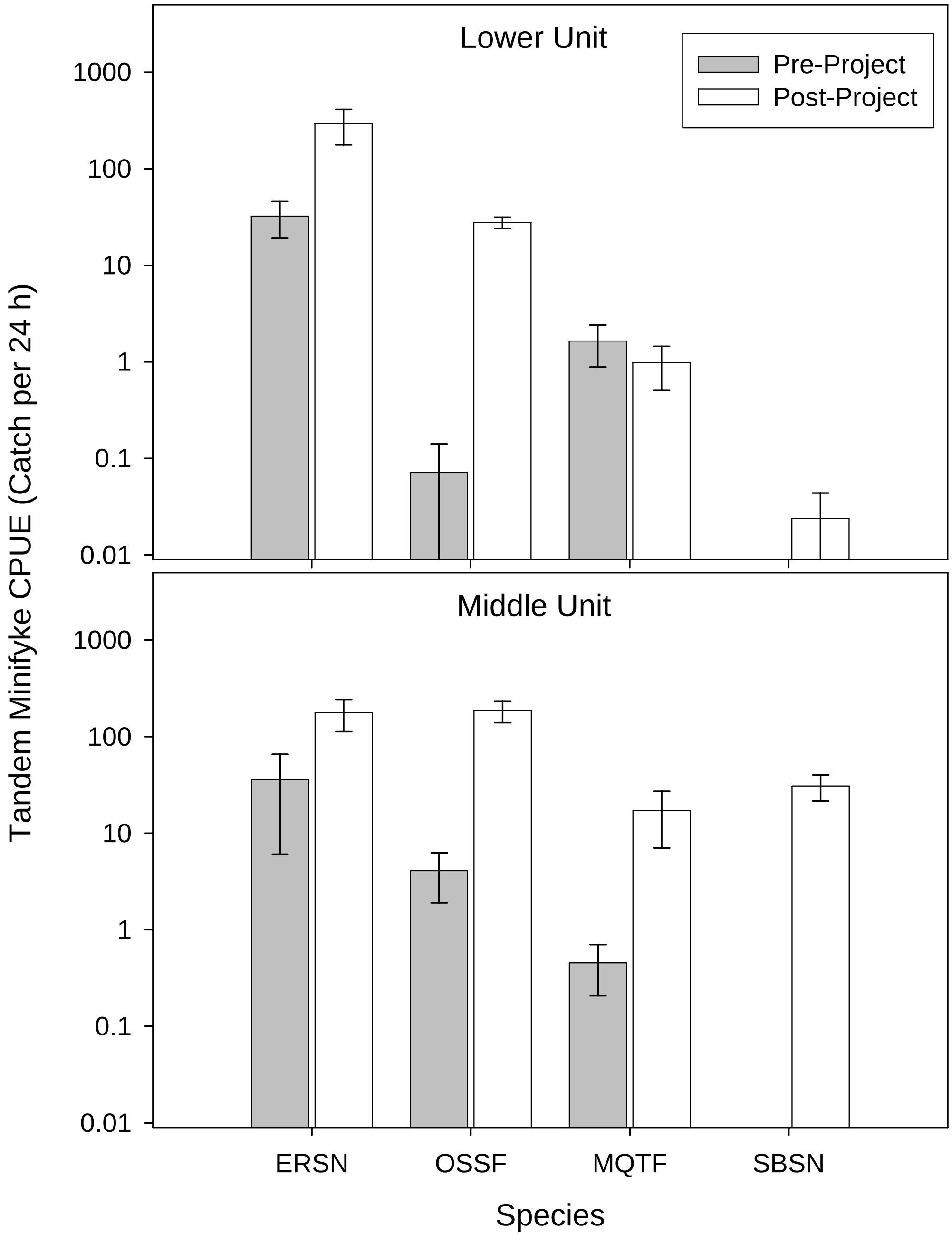


Figure F7. Mean catch-per-unit-effort (CPUE) for species of small adult size captured in tandem mini-fyke nets sets in Swan Lake for pre-project (1992 – 1993) and post-project (2004-2006) monitoring periods. See Table F1 for species codes. Error bars are one standard error.



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SECTION 3: FISH HABITAT USE AND MOVEMENT

Swan Lake Habitat Rehabilitation and Enhancement Project:

Post-Project Ecological Monitoring and Assessment

James E. Garvey, Douglas Schultz, Laura Csoboth, and Kelly DeGrandchamp

Fisheries & Illinois Aquaculture Center and Department of Zoology
Southern Illinois University at Carbondale
Carbondale, IL 62901-6511

FISH MOVEMENT

Summary

Anthropogenic degradation of large inland rivers has substantially reduced or impaired backwater habitats for native fishes. Swan Lake, an Illinois River backwater, was isolated from the river and compartmentalized to reduce sedimentation and dredged to create winter habitat. We quantified seasonal movement of native and exotic species through a water control structure between the shallow (0.5 m) lower compartment of the backwater and the river using directional trap nets and ultrasonic telemetry. Trap net catch rates of all species were highest during spring and summer ($P < 0.05$), and consistently dominated by juveniles. Ultrasonically tagged common carp *Cyprinus carpio* residency within the backwater was highest during winter ice cover ($P < 0.0001$). Telemetered Asian carps *Hypophthalmichthys spp.* moved between the lake and the river most during night; backwater residency was highest during spring months ($P < 0.0001$) and positively associated with water depth ($P < 0.0001$). The connected lower compartment provides poor winter habitat but likely supports reproduction for many river fishes. The backwater may be benefiting Asian carp recruitment as foraging and nursery habitat, particularly during spring.

Introduction

River backwater and floodplain areas are important habitats for most river fishes by providing lentic habitats essential for reproduction and development (Patton and Hubert 1993; Raibley et al. 1995), foraging (Fisher et al. 2001), and overwintering success (Bodensteiner and Lewis 1992; Knights et al. 1995; Raibley et al. 1997). Hence, species richness and diversity are typically higher in backwater habitats relative to main channels (Patton and Hubert 1993; Pinder et al. 1997). Loss and degradation of backwater habitats have reduced riverine fish communities (Karr et al. 1985; Nielsen et al. 1986).

Backwaters directly influence recruitment dynamics by affecting reproductive and overwintering success. Backwaters are typically warmer than their main-channel counterparts (Carter et al. 1985) and provide diverse lentic-like habitat in lotic environments essential for spawning and larval development (Carter et al. 1985; Nielsen et al. 1986; Patton and Hubert 1993; Raibley et al. 1995). These habitats also provide thermal refuge from the temperature extremes found in main-channel habitats (Raibley et al. 1997; Dettmers et al. 2001). Declining main-channel temperatures stimulate large movements of river fishes into warmer, stable backwater overwintering sites (Greenbank 1956; Sheehan et al. 1990; Raibley et al. 1997; Brown et al. 2001). Therefore, winter mortality of river fishes can be reduced (Bodensteiner and Lewis 1992; Fullerton et al. 2000; Braaten and Guy 2004) given suitable backwater habitat is available for winter use. Tolerance of prolonged cold temperature exposure differs greatly among species (Sheehan et al. 1990). Winter movement and backwater use by river fishes also increases with river discharge and ice break-up (Greenbank 1956; Brown et al. 2000; Brown et al. 2001).

Sedimentation has degraded or isolated many backwaters. Fish migration is reduced by intensified siltation in impounded rivers (Nielsen et al. 1986). Havera and Bellrose (1985) estimated over the next 100 years, sedimentation will reduce the volume of Illinois River Valley bottomland lakes, restraining their capacity to conduct floodwaters and diminishing their biological and recreational values. Consequently, sedimentation has reduced the number of potential large-river backwaters available for fish use and has limited the maximum production, use, and overall value of others (Brown and Coon 1994; Raibley et al. 1997; Sheehan and Rasmussen 1999). River fish abundance has declined accordingly (Jackson and Starret 1959; Karr et al. 1985; Sheehan and Rasmussen 1999; Raborn et al. 2001).

Recent attempts at deterring sedimentation in large-river systems have been made, and several backwater restoration projects have either been completed or are currently underway. Since the implementation of Habitat Rehabilitation and Enhancement Projects (HREP) in the Upper Mississippi River System in 1985, approximately 40 projects have restored floodplain habitat on nearly 67,000 acres (U.S. Army Corps of Engineers 2004). The number of projects is anticipated to increase in the future, and improved fisheries resulting from HREPs have been documented (Gent et al. 1995; O'Hara et al. 2001).

We explored the interaction between a HREP-restored major backwater system, Swan Lake, and the Illinois River. The primary goal of this restoration was to reduce sedimentation, while secondary effects, specifically improved winter habitat, were also sought (U.S. Army Corps of Engineers 1993). Pre-HREP research determined Swan Lake to have the most volatile and least favorable temperatures for winter fish use of local backwaters previously studied (Sheehan et al. 1990; Sheehan et al. 1994). Therefore, our objectives were to 1) assess seasonal movement of both native fishes and recent invaders between the lake and the river, 2) determine the abiotic influences on movement and residency within the lake, and 3) provide a comprehensive evaluation of this restoration project.

Methods

Study site

Swan Lake is a 1,100-ha Illinois River backwater that comprises a majority of the available backwater habitat near the confluence of the Illinois and Mississippi rivers. Swan Lake parallels the Illinois River in Calhoun County, Illinois beginning near Illinois river-kilometer (IRK) 19 and ending at IRK 8. The lake was isolated from the Illinois River in 1993 by a levee and compartmentalized into three impoundments. The Fuller Lake/upper Swan Lake complex is managed exclusively for waterfowl by the Illinois Department of Natural Resources. Middle and lower Swan Lake are managed by the Two Rivers National Wildlife Refuge, and are the only two units with the potential for implementing fisheries-specific management objectives.

Since HREP completion, middle Swan Lake, which is the deeper of the two, has been isolated almost continuously from the Illinois River and is managed for waterfowl using water-level manipulations to promote vegetative growth and to consolidate sediments. Post-HREP lower Swan Lake has remained continuously connected to the Illinois River through a water control structure approximately 5.1 m wide for fish availability. The original backwater-river interface was a 400-m wide sand bar. Water flow direction through the water control structure changes frequently within a 24-h period and velocities vary greatly. Lower Swan Lake is windswept and turbid; vegetation is absent and mean lake depth is 0.5 m when the river stage is 127.9 m. (Garvey et al. 2004). We focused specifically on lower Swan Lake; however, middle Swan Lake may be important habitat for river fishes if accessible.

Environmental Effects

Seasons were apportioned into 3-month intervals: fall consisted of 1 September through 30 November, and so on because changes in temperature and water levels affect movement (Greenbank 1956; Sheehan et al. 1990). Seasons were treated as independent among years. A Doppler velocity meter (Model 6526-51 Starflow Ultrasonic Doppler Flow Meter; Unidata America, Lake Oswego, Oregon, USA) quantified and recorded mean water velocity (m/s), depth (m), and temperature (°C) every 30 minutes within the most upriver column of the water control structure. Supplemental temperature data were collected with loggers (1-h interval; ± 0.3 °C accuracy; 8-bit Minilog-TR, Vemco Ltd., Nova Scotia, Canada) placed at 0.5-m depth intervals in lower Swan Lake and in the main channel of the Illinois River. All analyses were completed using the Statistical Analysis System (SAS) software v. 8.2. Mean daily temperatures across all loggers were determined within the lake and the river. Mean daily temperature within the Illinois River was subtracted from mean daily temperature within lower Swan Lake to determine daily differential temperature. A paired t-test (TTEST procedure; SAS 1999) was used to compare seasonal differences in mean daily temperatures between the lake and the river.

Trap Nets

Trap nets (wing design; 19-mm bar mesh, 15.2-m leads) were typically set on both sides of the water control structure for 24-h periods (Figure 1) during October 2003 through May 2005 (Schultz 2005). Nets were fished weekly during fall and spring and were fished on alternating weeks during winter and summer to quantify directional movement between the Illinois River and lower Swan Lake. Daily catch-per-unit-effort (CPUE) was calculated for 24-h net sets and $\log_{10}(x + 1)$ transformed to better meet the assumptions of normality. Daily CPUEs for gizzard shad *Dorosoma cepedianum*, freshwater drum *Aplodinotus grunniens*, and white bass *Morone chrysops* were compared among seasons using a two-way analysis of variance (ANOVA) for repeated measures under the maximum likelihood model (MIXED procedure; SAS 1999). Variation in weekly CPUE was partitioned by season, direction, and the interaction term. Pairwise comparisons were made post hoc using Tukey-Kramer adjusted least-squared means.

We used manual forward stepwise regression (GLM procedure; SAS 1999) to model relationships between species-specific CPUEs and measured environmental effects (Tabachnick and Fidell 2001). Effects in the model included season, year, and corresponding mean daily values of depth at the water control structure, velocity through the structure, and differential temperature between lower Swan Lake and the Illinois River. Size distributions of freshwater drum, gizzard shad, and white bass were compared among seasons using Kolmogorov-Smirnov (KS) pairwise comparisons (NPAR1WAY procedure; SAS 1999). A significance level α of 0.05 was established a priori for all tests; however, a Bonferroni-corrected α of 0.007 was used to determine significance for comparisons of length-frequency distributions among seasons.

Telemetry

To more accurately describe movement periodicity, movement frequency, and residency times between the Illinois River and lower Swan Lake, bighead carp *Hypophthalmichthys nobilis*, channel catfish, common carp, and silver carp *H. molitrix* were surgically implanted as per Winter (1996) with ultrasonic transmitters (Vemco Ltd., Nova Scotia, Canada) between November 2003 and April 2005 (Schultz 2005). Fishes were captured using multiple gears. Scales were removed from the area to be incised on carps. Transmitters implanted into channel catfish were internally attached (Siegwarth and Pitlo 1999) to prevent trans-intestinal expulsion. Fishes were anesthetized with compressed carbon dioxide; anesthetic water was buffered with sodium bicarbonate and compressed oxygen. Incisions were closed with 2/0 (bighead and silver carp) or 3/0 (common carp and channel catfish) braided nylon monofilament sutures, and individuals were released only after balance and orientation had been regained. All fishes were captured and released either within lower Swan Lake or within 6.4 IRK of lower Swan Lake.

Fish movement between the Illinois River and lower Swan Lake was logged by two omni-directional stationary ultrasonic receivers (Model VR2, Vemco Ltd., Nova Scotia, Canada); one was oriented in the channel/river interface and the other was in the channel/lake interface (Figure 1). Range of these receivers was approximately 150 m and minor overlap in the channel occurred (D. Schultz, unpublished data). Fish were also manually located by boat. Observations from stationary receivers were combined into 30-min intervals for each tagged individual, and fish passage was determined for each respective interval based on location (lower Swan Lake or the Illinois River) (Schultz 2005). Residency time (% h/week) within lower Swan Lake was determined based on fish passage (Schultz 2005). Residency data were compared against tracking data to verify accuracy. All analyses were restricted to the minimum transmitter life specified by the manufacturer. Furthermore, observations within the data set were limited only to those individuals which either entered or exited lower Swan Lake through the water control structure. Therefore, fish that did not pass through the structure were also excluded from all analyses, regardless of whether they remained in the lake or in the river for the life of their respective transmitters.

The total number of entrances and exits per individual were summarized by date into total movement between lower Swan Lake and the Illinois River to index activity. Activity (total daily N entrances/exits per tagged fish) was tested directionally within seasons using a t-test (TTEST procedure; SAS 1999). Total activity (daily N entrances + daily N exits) was tested as a function of season, species, and their respective interaction using a two-way ANOVA of repeated measures (MIXED procedure; SAS 1999); individuals were nested within species as the repeated measure, and the correlation structure of the model was specified as autoregressive [ar(1)]. Post hoc comparisons were implemented using Tukey-Kramer adjusted least-squared means. Family-specific trap net CPUE and species-specific passage data were compared using correlations (CORR procedure; SAS 1999).

We used BIC model selection (REG procedure; SAS 1999) to determine which environmental effects influenced species-specific passage through the structure. BIC-selected effects were then tested using multiple regression (REG procedure; SAS 1999). Independent model effects were the 24-h change in depth and the daily means of depth at the water control structure, velocity of flow through the structure, differential temperature between the lake and the river, and river temperature. Logistic regression (LOGISTIC procedure; SAS 1999) was used to detect species-specific diel patterns in passage through the structure; time of day was grouped into four 6-h intervals.

Residency times were acrsine square-root transformed and compared using a two-way ANOVA for repeated measures (MIXED procedure; SAS 1999); species, season, and their interaction term were the independent variables, and correlation structure of the model was specified as autoregressive [ar(1)]. Individuals were nested within species as the repeated subject. Post hoc comparisons were implemented using Tukey-Kramer adjusted least-squared means. We used BIC model selection to determine which environmental effects influenced species-specific residency within lower Swan Lake, and linear multiple regression was performed on the BIC-selected effects. Model effects were weekly means of depth at the water control structure, velocity of flow through the structure, differential temperature between the lake and the river, and change in depth.

Results

Environmental Effects

Mean daily temperatures within lower Swan Lake ranged from 0.39 to 31.6 °C and from -0.2 to 31.9 °C in the Illinois River during the course of the study. Differential temperature (lower Swan Lake minus Illinois River) ranged from -7.40 to 8.34 °C (Figure 2). Lower Swan Lake was warmer than the Illinois River during most seasons (Table 1). Ice covered lower Swan Lake completely from mid-January through mid-February during 2004, and nearly completely from mid-January through February during 2005 (Figure 2). Mean daily depth in the water control structure ranged from 1.18 to 6.29 m (Figure 3). Substantial flooding occurred during the spring and summer of 2004; moderate flooding occurred during winter 2004-2005 (Figure 3). Mean daily flow velocities through the structure ranged from 0.05 to 1.99 m/s and were highest during the 2004 flood pulse; velocities substantially exceeded 2.00 m/s, but these observations were lost because the sensor was dislodged as velocities approached 2.00 m/s (Schultz 2005).

Trap Nets

A total of 110 net nights captured 36 species comprising 11 families. Catches were dominated primarily by clupeids and sciaenids throughout the study, but moronids and cyprinids were also captured in substantial quantities during various seasons (Schultz 2005). Respectively, species captured in highest total abundances were gizzard shad (percent total catch; 55.6%), freshwater drum (28.8%), white bass (9.4%), and common carp (3.1%).

Gizzard shad, freshwater drum, and white bass comprised approximately 94% of the total catch. Pooled catch rates were highest and most diverse during spring and summer (Schultz 2005). There was no statistical interaction between season and direction as it related to species-specific CPUE ($P > 0.05$). Gizzard shad catch rates did not differ among seasons ($F = 0.89$; df = 6, 94; $P > 0.05$) (Figure 4). Catch rates of freshwater drum were generally highest during spring and summer ($F = 3.53$; df = 6, 94; $P < 0.01$), and more freshwater drum were caught entering lower Swan Lake than exiting it ($F = 10.63$; df = 1, 94; $P < 0.01$). Conversely, white bass CPUE was highest during fall 2004 ($F = 3.78$; df = 6, 94; $P < 0.01$). Variation in gizzard shad and freshwater drum catch rates (directions combined) was strongly associated with depth (Table 2).

Length frequencies varied among seasons for freshwater drum (asymptotic KS statistic [KSa] ≥ 0.81; df ≥ 134; $P < 0.007$) (Figure 5), gizzard shad (KSa ≥ 2.32; df ≥ 619; $P < 0.007$), and white bass (KSa ≥ 2.77; df ≥ 323; $P < 0.007$). Although length-frequency distributions were dominated by sub-adult fishes during all seasons and the statistical differences were driven in large part by CPUE of these fishes, sizes of fishes caught during spring were consistently larger than those observed during other seasons.

Telemetry

We surgically implanted 50 bighead carp, 31 common carp, 31 channel catfish, and 50 silver carp with ultrasonic transmitters between November 2003 and April 2005 (Table 3). Largemouth bass *Micropterus salmoides* were a target species for this study, but the local population has declined (Schultz 2005). No mortalities were apparent; however, one silver carp was harvested (K. DeGrandchamp, SIUC, unpublished data). Asian carp demonstrated the ability to move ~ 320 km in 30 d (K. DeGrandchamp, SIUC, unpublished data). Two channel catfish traveled in excess of 60 km during this study, one of which covered this distance in only 8 weeks (Schultz 2005). Not all fish passed through the lower Swan Lake water control structure, and were therefore excluded from analyses (Table 4). Additionally, some fishes were actively located in middle Swan Lake and the Mississippi River during (Table 4) (Schultz 2005). This behavior was not incorporated into analyses.

Stationary receivers logged approximately 1.1 million observations during November 2003 through August 2005 at the lower Swan Lake water control structure; most occurred during prolonged residency within the connecting channel by carps. A species-specific t-test determined passage through the structure did not differ directionally within or among seasons for any of the tagged species ($P > 0.05$). Asian carps were the only fishes to exhibit diel differences in movement. Bighead carp traversed the structure during the final 6-h of a standard 24-h day almost twice as frequently than during other 6-h intervals (logistic regression: $\chi^2 = 13.13$; df = 3; $P < 0.01$). Conversely, silver carp moved through the structure less frequently during the afternoon than during other intervals, which were relatively equal amounts ($\chi^2 = 11.38$; df = 3; $P < 0.01$). Common carp and channel catfish did not exhibit diel differences in passage ($P > 0.05$). The number of passages through the structure differed among species (2-way ANOVA: $F = 15.79$; df = 7, 256; $P < 0.0001$) and seasons ($F = 10.56$; df = 4, 85; $P < 0.0001$), and a significant interaction existed between season and species (*F* = 8.07; df = 19, 256; $P < 0.0001$) (Figure 6). Species-specific passage data were not significantly correlated with family-specific trap net CPUE ($P > 0.05$). Bighead carp passage through the structure was negatively influenced by river temperature (multiple regression: $F = 4.41$; $r^2 = 0.06$; $P < 0.05$). Conversely, channel catfish traversed the structure more when river temperatures and differential temperatures were warmer ($F = 4.88$; $r^2 = 0.11$; $P = 0.01$). Regression models were not significant for common or silver carps ($P > 0.05$).

Ultrasonically-tagged fishes used lower Swan Lake differently during different times of the year. Residency within lower Swan Lake was highest during spring months for bighead (2-way ANOVA: $F = 4.68$; df = 17, 302; $P < 0.0001$) and silver ($F = 3.95$; df = 17, 241; $P < 0.0001$) carps (Figure 7). Conversely, common carp resided in lower Swan Lake more during January and February than during other months ($F = 3.65$; df = 18, 167; $P < 0.0001$), and this seemed closely associated with peaks in differential temperature. Channel catfish residency within lower Swan Lake did not differ among months ($F = 1.20$; df = 18, 104; $P > 0.05$).

Residency within lower Swan Lake was affected by several environmental effects across species. Bighead (multiple regression: $F = 11.74$; $r^2 = 0.14$; $P = 0.001$) and silver carp ($F = 11.07$; $r^2 = 0.25$; $P < 0.0001$) residency within lower Swan Lake was positively associated with depth. Channel catfish use of the lake was positively related to velocity through the structure and river temperature ($F = 125.04$; $r^2 = 0.78$; $P < 0.0001$). There was no statistical relationship between common carp residency and measured environmental effects ($F = 1.39$; $r^2 = 0.02$; $P > 0.05$).

Discussion

Sedimentation of large rivers has reduced available backwater habitats for river fishes (Knights et al. 1995; Raibley et al. 1997; Sheehan and Rassmussen 1999) and has resulted in population declines of several species (Jackson and Starret 1959; Karr et al. 1985; Sheehan and Rassmussen 1999). Consequently, the current challenge is to restore habitats necessary for sustainable animal populations, and to quantitatively evaluate how rehabilitation projects affect animal communities. Multi-dimensional evaluations such as this one can further improve the fiscal and biological efficiency of future habitat restoration efforts.

Winter habitat use

Large-river backwaters are important to winter survival of river fishes (Bodensteiner and Lewis 1992; Knights et al. 1995), and backwater use by fishes tends to increase during winter (Pitlo 1992; Gent et al. 1995; Knights et al. 1995; Raibley et al. 1997) Although we anticipated substantial migrations into lower Swan Lake for winter use, our results contradict previous reports which suggested Swan Lake serves as valuable winter habitat for river fishes (Sheehan et al. 1990; Sheehan et al. 1994). No pulse of directional movement into lower Swan Lake with implications for prolonged winter residency occurred. However, intermittent increases in trap net catch rates as well as temporary increases in residency times of some telemetered fishes were indicative of moderate thermal selection associated with backwater-river temperature dynamics during winter.

Aquatic habitats are structured by gradients of temperature, light, dissolved oxygen, flow, water chemistry, substrate, vegetation, and physical structure. Additionally, wind-induced mixing affects thermal dynamics (Horne and Goldman 1994). The microhabitat structure resulting from interactions of these factors influences the movement patterns and distributions of fishes (Prophet et al. 1989; Baldwin et al. 2002; Paukert and Willis 2002; Barwick et al. 2004). Applied to lower Swan Lake, mean daily temperatures varied greatly, and thermal stratification and stability were limited to infrequent periods of ice cover and flooding. Strong prevailing winds combined with air temperatures substantially warmer or colder than shallow water temperatures rapidly and dramatically alter lake temperatures. Therefore, the lake was strongly influenced by air temperatures via wind action.

Temperature has been one of the most consistent environmental influences available to fishes throughout their evolutionary process (Coutant 1987). From a physiological perspective, stable or gradually changing temperatures promote enzyme activity and reduce metabolic demands in fishes (Becker et al. 1997), and temperature directly or indirectly influences habitat use (Bevelhimer 1996; Braaten and Guy 1999; Baldwin et al. 2002; Paukert and Willis 2002; Barwick et al. 2004), food consumption (Salvatore et al. 1987; Braaten and Guy 1999), and activity in fishes (Dwyer and Kramer 1975; Braaten and Guy 1999; Power and van den Heuvel 1999). Therefore, fishes select and use the most physiologically efficient habitats available, and habitat selection, use, and movement often correspond with temperature (Coutant 1987).

Lower Swan Lake is unsuitable habitat river fishes as primary overwintering habitat because extreme temperature volatility makes physiological acclimation and preferred temperature selection difficult. The relatively absent largemouth bass population within the lake may be most indicative of these effects (Schultz 2005); largemouth bass year-class strength is strongly influenced by first-winter survival of young-of-year (YOY) fish (Fullerton et al. 2000). However, ice cover and flooding promoted thermal stability during this study by reducing wind-induced mixing of the water column, and fish use of the lake increased accordingly. Ice formation is a primary constituent in determining shallow backwater use by river fishes during winter (Pitlo 1992; Knights et al. 1995; Raibley et al. 1997), and the ability of a backwater to form and sustain ice cover during winter should be considered when assessing its value to fishes during winter as well as its potential for reclamation. Therefore, the Swan Lake HREP was ineffective at providing suitable winter habitat for fishes in the form of lower Swan Lake.

Reproduction

Our findings support previous work indicating Swan Lake provides habitat important to recruitment dynamics of Illinois River fishes (Sheehan et al. 1990; Sheehan et al. 1994). Trap net CPUE was generally highest during spring and summer. Catches during all seasons were dominated by sub-adult fishes; however, the increased frequency of larger, more diverse fishes in trap nets was unique to spring and summer catches. Likely, adult fishes move to the lake as reproductive habitat to the degree appropriate to their respective life-history characteristics.

Floodplain inundation is vital to species relying on backwaters as optimum spawning or nursery habitat (Patton and Hubert 1993). In this study inundation also promoted thermal stability, increasing backwater value. During spring 2005 when flooding did not occur; temperatures within the lake were often equal to or less than those observed in the river, and were more variable than those observed during the 2004 flood event. Hence, flooding not only buffers the effects of wind-induced mixing in shallow backwaters and promotes a thermally-stable environment during winter, but has a similar effect during other seasons. Consequently, adult fish use of backwaters during spring is strongly influenced by the flood pulse, although secondary effects, such as thermal variability during years when a flood pulse does not occur, appear to be as important in determining fish use as inundation.

In addition to the primary uses of the lake by native fishes as reproductive habitat, exotic species may be increasing their reproductive potential by using backwaters. Lower Swan Lake is highly productive, and yields invertebrate densities substantially higher than those found in main channel of the Illinois River (L. Csoboth, SIUC, unpublished data). Asian carps graze heavily on zooplankton and grow rapidly (Fuller et al. 1999; Williamson and Garvey 2005). Asian carps are likely foraging in lower Swan Lake and increasing their reproductive capabilities; this was supported by increased residency within the lake during spring. Asian carp using backwaters during winter and pre-flood spring can potentially maintain reproductive requirements or elevate their reproductive potential while not compromising growth. Fish activity is inversely related to gonad development (Koch and Wieser 1983), and resources are often repartitioned from somatic growth and metabolic demands to reproduction (Dygert 1990). Furthermore, most Asian carps captured in trap nets were YOY fishes as they moved between the lake and the river during the summer and the fall of 2004 (Schultz 2005), and suggests this system enhances reproduction via improved hatching success and nursery habitat.

HREP effects

Constriction of the lower Swan Lake-Illinois River connection constituted a 99% reduction in water exchange rate. Furthermore, areas of Swan Lake previously used by telemetered fishes during pre-HREP research (specifically, middle Swan Lake) (Sheehan et al. 1994) currently are only accessible via substantial flood events or intermittent connections through the middle-lower Swan Lake sluice gate or the middle Swan Lake water control structure; these gates are typically closed.

Installation of the water control structure has greatly altered rates of water exchange between the lake and the river, creating a unique and permanent backwater-river interface. Closure of the natural opening has altered the hydrological regime of the lake itself. Proportional to the river, water levels within the lake currently rise and fall at a slower pace during flood events than historically. This appears to have had negative impacts on the fish community. Highly variable flow velocities through the structure contributed substantially to fish exchange rates between the lake and the river, particularly during floods when river stage increased or decreased rapidly and observed velocities through the structure peaked (Schultz 2005). Furthermore, telemetered fishes often oriented themselves near the channel connecting the lake and the river, and invertebrate and larval fish exchange through the structure is high (L. Csoboth, SIUC, unpublished data). Thus, fishes were likely foraging in this area. Flow velocity through the structure, whether independently or as an interaction with other environmental effects, is a significant constituent of seasonal fish exchange and habitat use between lower Swan Lake and the Illinois River, and should be considered a separate component of the system based on its unique hydrological properties and the biological consequences.

Management Recommendations

The division of Swan Lake into two units has created two unique habitats. Isolation of middle Swan Lake from the Illinois River has had a negative impact on the river fishery by excluding fish from previously available deeper habitat; this has forced fish to negotiate less favorable conditions in lower Swan Lake, particularly during winter. Consequently, fish movement between lower Swan Lake and the Illinois River has decreased because the lake no longer maintains its historic thermal and hydrologic properties. Telemetered fishes accessed and used middle Swan Lake during this study. Therefore, consideration should be given towards assessing the potential value of middle Swan Lake as overwintering and reproductive habitat for Illinois River fishes. Pending a comparison of both studies, consideration should be given towards managing middle Swan Lake for fish and managing lower Swan Lake as a moist soil unit.

Since the HREP initiation, populations of non-sport fishes increased in both middle and lower Swan lake while populations of sport fishes decreased (Garvey et al. 2004; Schultz 2005). Large-bodied non-sport fishes, such as common carp, are known to increase turbidity by re-suspending sediments (Drenner et al. 1997), and negatively affect vegetative growth in aquatic systems as a result; Asian carps likely have similar effects. Therefore, both units should be opened to commercial fishing when navigable; promoting commercial harvest within them would complement restoration efforts.

River restoration projects have benefited fisheries in several systems (Gent et al. 1995; O'Hara et al. 2001). However, the Swan Lake HREP has had consequential effects on the sport fishery, particularly centrarchids (Schultz 2005). Future restoration efforts should be directed towards identifying, improving, and maintaining deep-water habitat conducive to winter survival and providing adequate spawning habitat during years inundation does not occur.

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Table 1. Results of seasonal paired t-tests comparing mean daily temperatures within lower Swan Lake against mean daily temperatures within the Illinois River from October 2003 through August 2005. Lower Swan Lake was warmer than the Illinois River during most seasons.

Year	Season	df	*t*	*P*
2003	Fall*	42	2.41	0.02
	Winter	90	6.63	<0.0001
2004	Spring	91	6.48	<0.0001
	Summer	91	2.36	0.02
	Fall*	90	2.23	0.03
	Winter	89	6.14	<0.0001
2005	Spring	91	5.91	<0.0001
	Summer*	34	1.85	0.07

*Indicates the Illinois River was warmer than lower Swan Lake.

Table 2. Multiple regression results for log-transformed 24-h trap net catch-per-unit-effort (CPUE) at the lower Swan Lake water control structure between October 2003 and May 2005. CV refers to the coefficient of variation. Only white bass exhibited a significant interaction among model effects.

CPUE	Effect	*F*	df	CV	r^2	*P*
Gizzard shad	Model	5.22	3, 83	39.73	0.16	0.002
	Year	3.88	2			0.02
	Depth	6.55	1			0.01
Freshwater drum	Model	3.89	4, 82	55.94	0.16	0.006
	Season	4.92	3			0.003
	Depth	3.96	1			0.05
White bass	Model	3.94	7, 102	73.6	0.2	0.002
	Year*season	3.54	7			0.002

Table 3. Summary of fishes surgically implanted with ultrasonic transmitters from November 2003 through April 2005.

Month	Year	Species*	Tag Type	Tag Life (d)	Capture/ Release**	Mean TL (mm)	Mean W (g)	*n*
Nov	2003	CCF	V13	230	LSL	416	760	1
					ILR	565	1,933	3
		CMC	V13	230	LSL	658	4,063	8
					ILR	567	2,790	7
Mar	2004	BHC	V16	641	LSL	764	5,300	1
					ILR	781	5,938	24
		CCF	V13	230	LSL	515	1,548	2
					ILR	449	894	3
		CMC	V13	230	LSL	734	4,022	1
		SLC	V16	641	ILR	758	4,963	4
Apr	2004	CCF	V13	230	LSL	579	2,338	6
					ILR	578	2,357	1
		SLC	V16	641	ILR	694	4,364	14
		SLC	V16	641	ILR	742	4,900	7
Oct	2004	BHC	V16	641	LSL	765	5,050	8
					ILR	767	5,271	7
		CCF	V13	230	LSL	536	1,450	7
					ILR	486	1,187	8
		CMC	V13	230	LSL	662	4,377	8
					ILR	536	2,205	7
		SLC	V16	641	ILR	763	5,040	15
Mar	2005	BHC	V16	366	LSL	793	5,860	5
					ILR	767	5,700	5
Apr	2005	SLC	V16	366	ILR	768	5,920	10

*BHC = Bighead carp, CCF = Channel catfish, CMC = Common carp, SLC = Silver carp.
**LSL = Lower Swan Lake, ILR = Illinois River.

Table 4. Proportions of ultrasonically tagged fishes located either actively or passively within various water bodies during November 2003 – August 2005, and proportion of fish captured and released in either the Illinois River or lower Swan Lake that did not pass through the lower Swan Lake water control structure during the minimum transmitter life. Fishes not passing through the structure were excluded from analyses.

		Located manually or passively (%)**					Did not pass thorugh structure (%)**	
Species*	N	ILR	LSL	MSL	MSR	UK	ILR	LSL
BHC	50	78	34	2	2	2	32	12
CCF	31	77	65	10	3	10	23	23
CMC	31	68	71	13	0	6	10	10
SLC	50	80	22	8	2	2	62	0

* BHC = bighead carp, CCF = channel catfish, CMC = common carp, SLC = silver carp

** ILR = Illinois River, LSL = Lower Swan Lake, MSL = Middle Swan Lake, MSR = Mississippi River, UK = Unknown

List of Figures.

Letters indicate significant differences among seasons. Gizzard shad and white bass showed similar trends in length-frequency distributions, particularly the higher frequency of adults during spring.

Figure 6. Sample sizes (lines) and differences in mean + 1 SE movement (bars) through the lower Swan Lake water control structure for ultrasonically tagged fishes monitored with stationary receivers. All data were standardized by the daily number of tagged fish that passed through the structure (dashed line) and were restricted to the minimum transmitter life specified by the manufacturer. Letters indicate Tukey-Kramer adjusted differences among seasons.

Figure 7. Mean + 1 SE monthly Asian carp residency time (bars) within lower Swan Lake and mean monthly depth (lines) at the lower Swan Lake water control structure (left column). Mean + 1 SE monthly common carp and channel catfish residency time (bars) within lower Swan Lake and mean monthly differential temperature between lower Swan Lake and the Illinois River (lake minus river; lines; right column). Only those abiotic factors found to be associated with residency time are plotted on graphs.

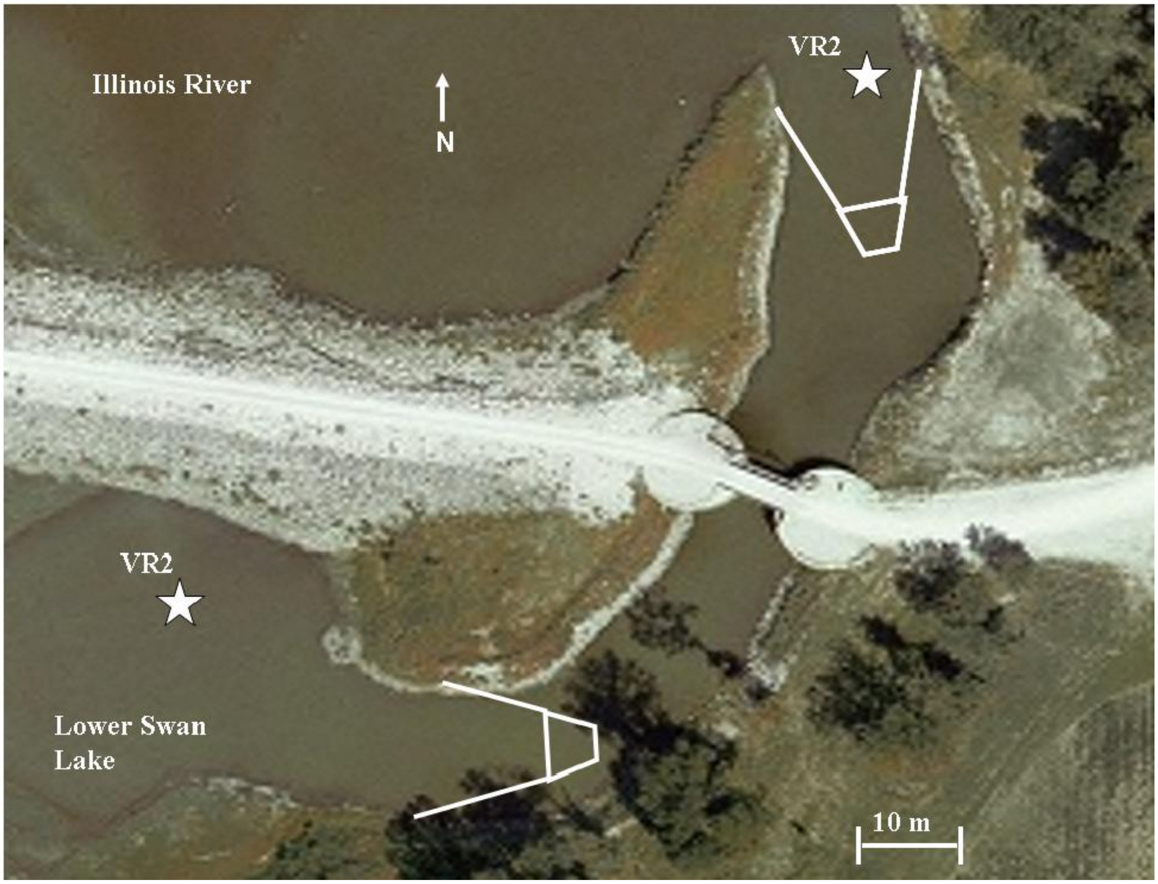


Figure 1.

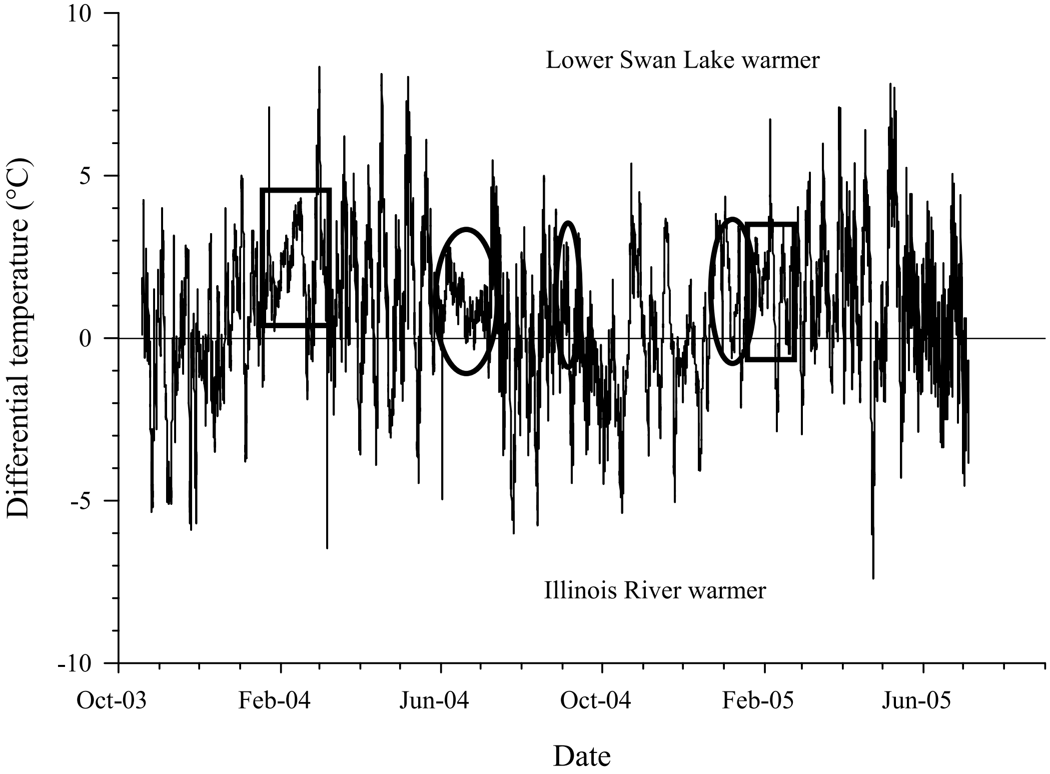
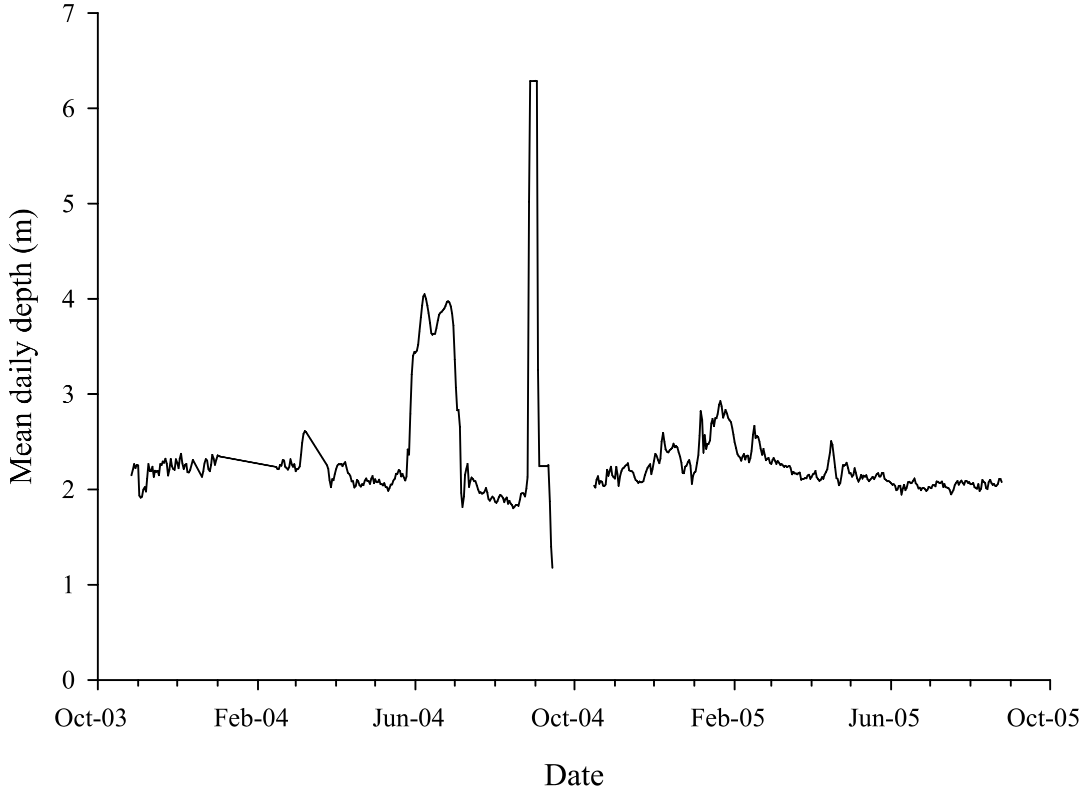

10
5
0
-5
-10
Differential temperature (°C)
Lower Swan Lake warmer
Illinois River warmer
Oct-03
Feb-04
Jun-04
Oct-04
Feb-05
Jun-05
Date

Figure 2.



Mean daily depth (m)
7
6
5
4
3
2
1
0
Oct-03
Feb-04
Jun-04
Oct-04
Feb-05
Jun-05
Oct-05
Date

Figure 3.

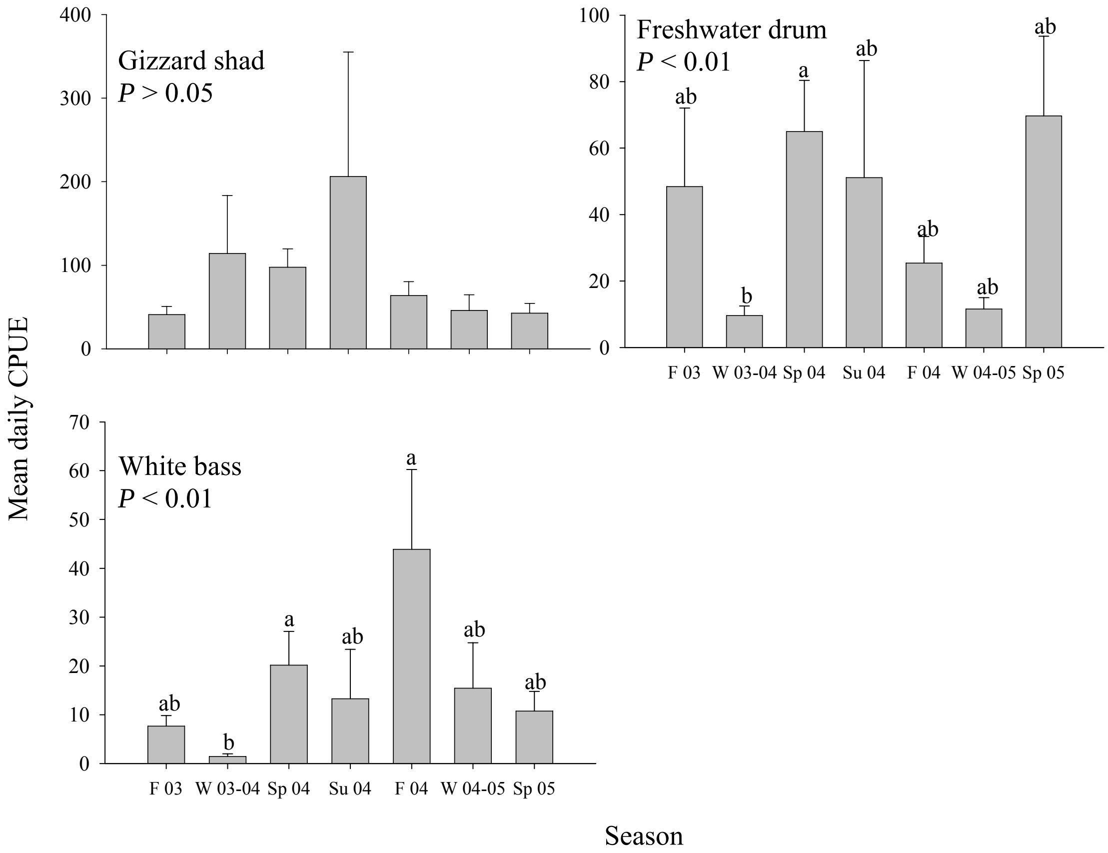

Gizzard shad
P > 0.05
400
300
200
100
0
Freshwater drum
P < 0.01
100
80
60
40
20
0
ab
b
a
ab
ab
ab
ab
F 03
W 03-04
Sp 04
Su 04
F 04
W 04-05
Sp 05
White bass
P < 0.01
70
60
50
40
30
20
10
0
ab
b
a
ab
a
ab
ab
F 03
W 03-04
Sp 04
Su 04
F 04
W 04-05
Sp 05
Mean daily CPUE
Season

Figure 4.

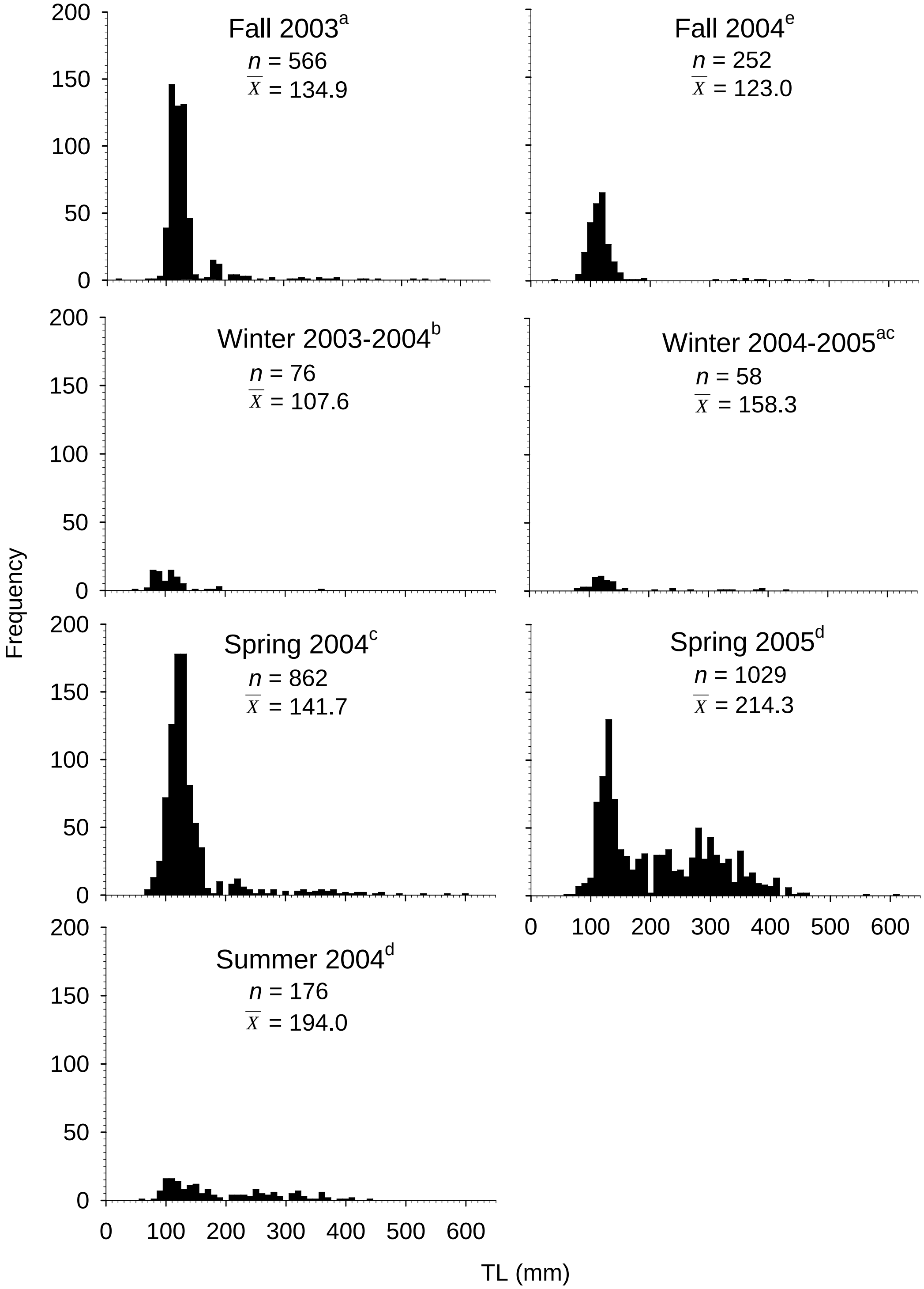

Fall 2003^a
n = 566
X̄ = 134.9
Fall 2004^e
n = 252
X̄ = 123.0
Winter 2003-2004^b
n = 76
X̄ = 107.6
Winter 2004-2005^ac
n = 58
X̄ = 158.3
Spring 2004^c
n = 862
X̄ = 141.7
Spring 2005^d
n = 1029
X̄ = 214.3
Summer 2004^d
n = 176
X̄ = 194.0
Frequency
0
50
100
150
200
0
100
200
300
400
500
600
TL (mm)

Figure 5.

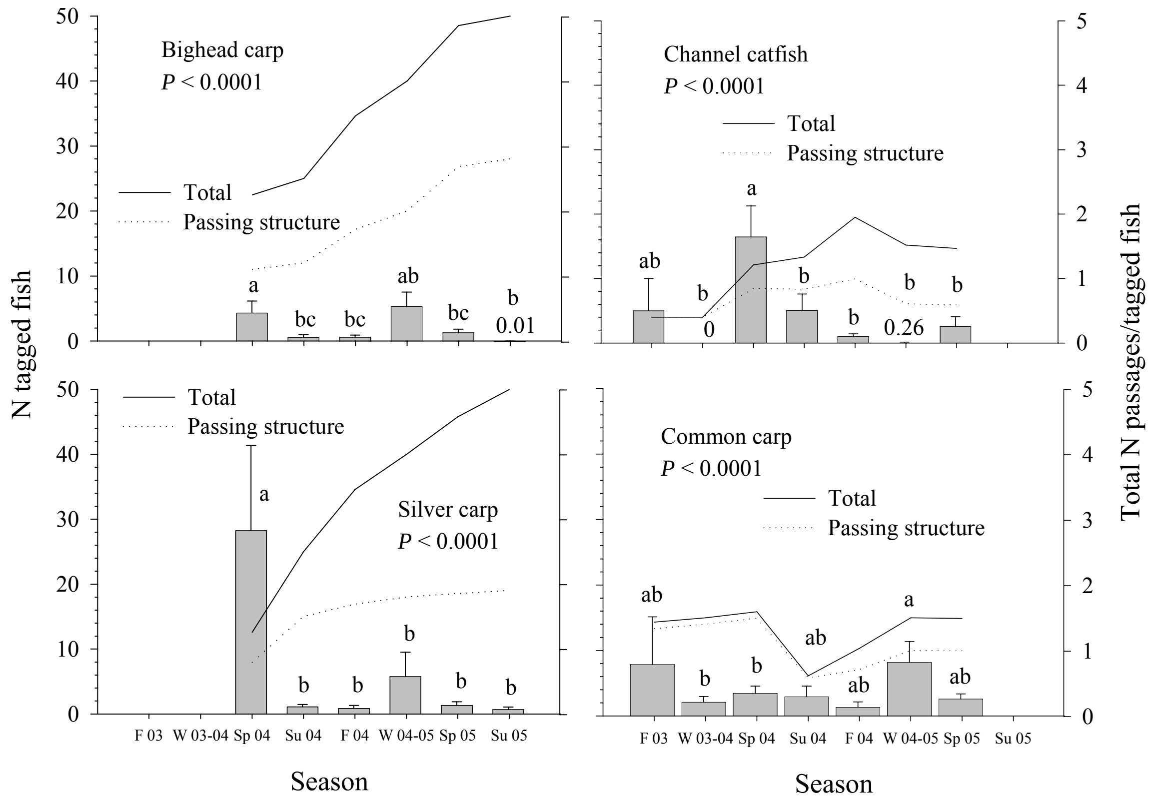

Bighead carp
P < 0.0001
Channel catfish
P < 0.0001
Silver carp
P < 0.0001
Common carp
P < 0.0001
Total
Passing structure
N tagged fish
Total N passages/tagged fish
Season
F 03
W 03-04
Sp 04
Su 04
F 04
W 04-05
Sp 05
Su 05
0
10
20
30
40
50
0
1
2
3
4
5
a
bc
bc
ab
bc
b
0.01
ab
b
0
a
b
b
0.26
b
b
a
b
b
b
b
b
ab
b
b
ab
ab
a
ab

Figure 6.

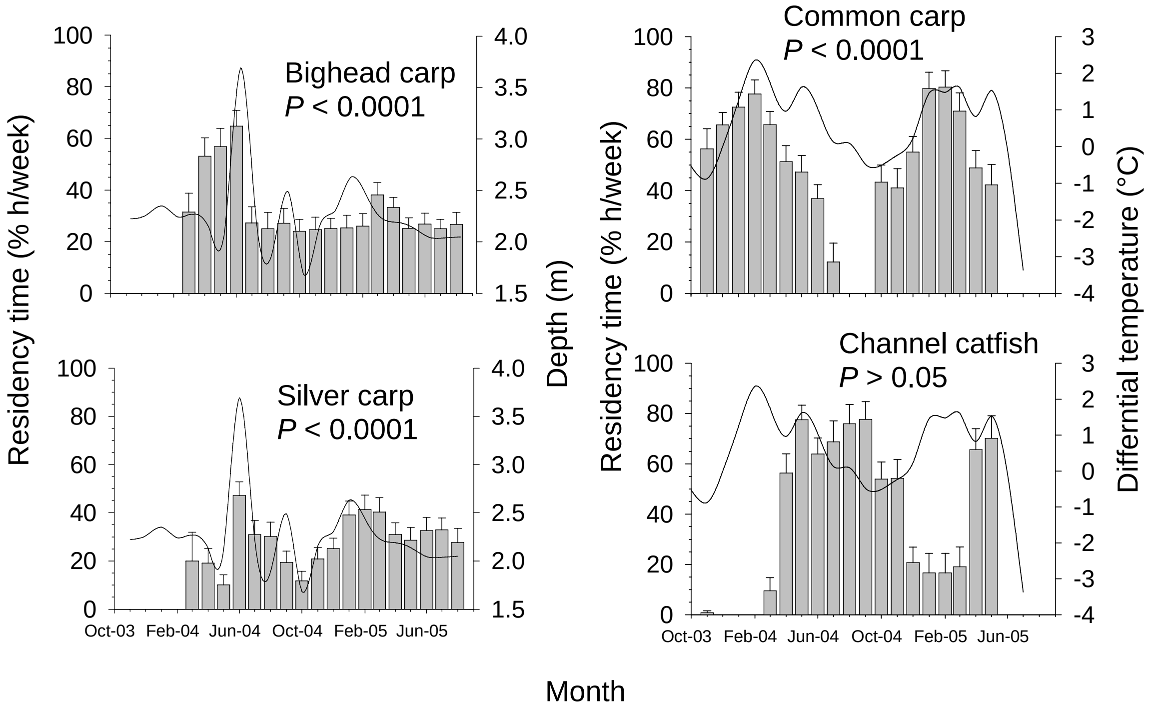
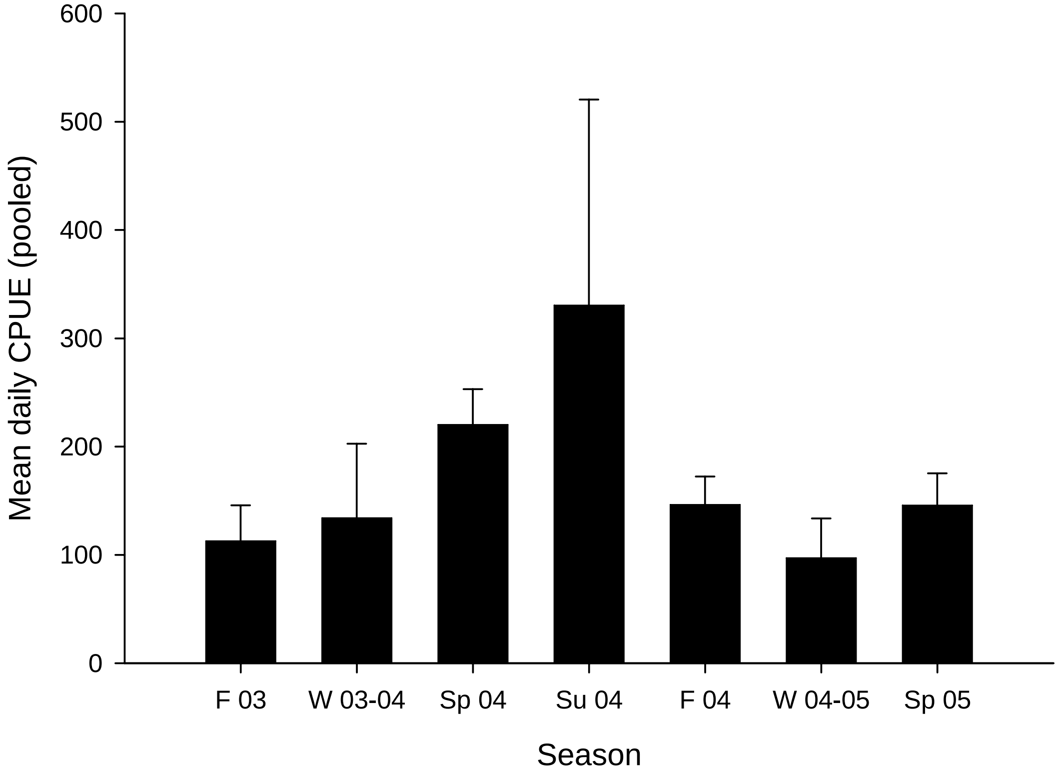

Bighead carp
P < 0.0001
Silver carp
P < 0.0001
Common carp
P < 0.0001
Channel catfish
P > 0.05
Residency time (% h/week)
Depth (m)
Differential temperature (°C)
100
80
60
40
20
0
4.0
3.5
3.0
2.5
2.0
1.5
3
2
1
0
-1
-2
-3
-4
Oct-03
Feb-04
Jun-04
Oct-04
Feb-05
Jun-05
Month

Figure 7.

Appendix 1. Species composition of seasonal trap net catches at the lower Swan Lake water control structure. *Ictiobus spp.* includes *I. niger*, *I. cyprinellus*, and *I. bubalus*.

		Proportion of seasonal catch (%)			
Family	Species	Fall	Winter	Spring	Summer
Amiidae	*Amia calva*	0.18	0.00	0.15	0.06
Anquillidae	*Anguilla rostrata*	0.00	0.00	0.02	0.00
Catastomidae	*Ictiobus spp.*	1.45	0.93	1.83	0.19
	Carpiodes carpio	0.33	0.34	1.54	0.19
	Moxostoma macrolepidotum	0.04	0.05	0.29	0.19
	Catostomus commersoni	0.04	0.00	0.02	0.00
Centrarchidae	*Pomoxis nigromaculatus*	0.55	0.73	1.32	2.51
	Lepomis macrochirus	0.07	0.05	0.39	0.00
	L. cyanellus	0.04	0.00	0.15	0.00
	L. humillis	0.00	0.00	0.02	0.00
	L. gulosus	0.00	0.00	0.07	0.00
	P. annularis	0.25	0.00	0.10	0.00
Clupeidae	*Dorosoma cepedianum*	41.08	79.31	46.78	72.93
	Alosa chrysochloris	0.25	0.00	0.00	0.38
	D. petenense	0.58	0.24	0.00	0.00
Cyprinidae	*Hypophthalmichthys nobilis*	0.18	1.66	0.24	0.57
	Cyprinus carpio	2.73	2.14	4.69	0.82
	Carassius auratus	0.11	0.00	1.91	0.13
	Ctenopharyngodon idella	0.07	0.19	0.10	0.00
	H. molitrix	0.22	2.58	0.20	6.66
Ictaluridae	*Ameiurus nebulosus*	0.04	0.00	0.07	0.06
	A. melas	0.11	0.00	0.12	0.00
	Ictalurus furcatus	0.00	0.00	0.02	0.00
	I. punctatus	0.36	0.15	1.22	0.75
	Pylodictis olivaris	0.00	0.00	0.00	0.06
	A. natalis	0.36	0.05	0.59	0.06
Lepisosteidae	*Lepisosteus platostomus*	0.15	0.00	0.73	0.31
Perchichthyidae	*Morone chrysops*	19.48	4.24	7.82	2.89
	M. americana	0.04	0.05	0.05	0.00
	M. mississippiensis	1.56	0.78	0.46	0.13
Percidae	*Sander canadense*	0.00	0.00	0.00	0.06
Sciaenidae	*Aplodinotus grunniens*	29.73	6.52	46.19	11.06
Total *n*		2751	2054	4094	1592



Appendix 2. Mean daily CPUE of pooled trap net catches at the lower Swan Lake water control structure. CPUE did not differ among seasons.

Appendix 3. Species composition of seasonal stop log catches within the lower Swan Lake water control structure. *Ictiobus spp.* includes *I. niger*, *I. cyprinellus*, and *I. bubalus*.

		Proportion of seasonal catch (%)			
Family	Species	Spring	Summer	Fall	Winter
Amiidae	*Amia calva*	0.01	0.51	0.01	0.00
Catastomidae	*Ictiobus spp.*	0.46	0.51	0.06	0.69
	Carpiodes carpio	0.13	0.00	0.01	2.53
	Moxostoma macrolepidotum	0.01	0.00	0.00	0.00
	Catostomus commersoni	0.01	0.00	0.00	0.00
Centrarchidae	*Pomoxis nigromaculatus*	0.21	10.15	0.10	0.46
	Lepomis macrochirus	0.10	0.00	0.03	0.46
	L. cyanellus	0.03	0.00	0.01	0.00
	Micropterus salmoides	0.00	0.00	0.00	0.23
	L. humillis	0.01	0.51	0.00	0.00
	L. gulosus	0.01	0.00	0.00	0.00
	P. annularis	0.00	1.02	0.01	0.00
Clupeidae	*Dorosoma cepedianum*	65.70	42.13	82.46	57.83
	Alosa chrysochloris	0.00	1.52	1.21	0.00
	D. petenense	0.00	0.00	0.46	0.00
Cyprinidae	*Hypophthalmichthys nobilis*	0.19	3.05	0.01	2.30
	Cyprinus carpio	0.49	2.54	0.18	0.92
	Notropis antherinoides	0.01	0.00	0.00	0.00
	Carassius auratus	0.09	3.55	0.01	0.00
	Ctenopharyngodon idella	0.00	0.51	0.00	0.00
	H. molitrix	0.32	1.02	0.97	5.53
Hiodontidae	*Hiodon tergisus*	0.00	0.00	0.00	0.23
Ictaluridae	*Ameiurus nebulosus*	0.06	1.02	0.00	0.23
	A. melas	0.17	0.00	0.00	0.00
	Ictalurus punctatus	1.83	2.03	0.42	1.84
	A. natalis	0.19	0.00	0.01	0.23
Lepisosteidae	*Lepisosteus platostomus*	0.64	0.51	0.07	1.15
Perchichthyidae	*Morone chrysops*	1.37	7.61	3.49	1.61
	M. americana	0.04	0.00	0.06	0.23
	M. mississippiensis	0.13	1.52	0.13	0.23
Polyodontidae	*Polyodon spathula*	0.01	0.00	0.00	0.00
Sciaenidae	*Aplodinotus grunniens*	27.76	20.30	10.24	23.27
Total *n*		6982	197	6682	434

CHANNEL USE BY FISHES

SUMMARY

Recent efforts to reduce sediment loading in large-river backwaters have relied on inexpensive low-maintenance structures that promote river-backwater connectivity. How these structures influence ichthyofauna is not understood. We quantified the distribution of fishes through a stop-log structure installed during the Swan Lake habitat restoration project on the lower Illinois River. Modified hoop nets were systematically fished simultaneously in a two-dimensional 3 x 3 grid to capture fishes as they immigrated into the lake through the structure. Catches were dominated by gizzard shad *Dorosoma cepedianum* and freshwater drum *Aplodinotus grunniens*. Adult freshwater drum, adult common carp *Cyprinus carpio*, and juvenile silver carp *Hypopthalmichthys molitrix* exhibited an affinity for the lower two-thirds of the water column. Lengths of freshwater drum increased with depth, particularly during the spring. Common carp catch rates were highest during spring, whereas those for silver carp were highest during fall and winter. Closure of the bottom two-thirds of the lower Swan Lake water control structure could reduce immigration of exotic and non-desirable species into the backwater; their recruitment associated with the lake could be affected accordingly. The water control structure does not appear to have negatively affected lateral fish movement. However, habitat quality and availability likely determine seasonal patterns in backwater use. In systems where exotic species are abundant, lateral connectivity may need to be assessed from a cost-benefit perspective, specifically as it describes maintaining habitat availability for indigenous fishes versus limiting habitat availability for introduced species.

Introduction

Large-river backwaters and floodplains are critical for various life stages of many river fishes (Bodensteiner and Lewis 1992; Patton and Hubert 1993; Knights et al. 1995; Raibley et al. 1995; Raibley et al. 1997). Consequently, species richness and diversity are typically higher in these habitats relative to main channels (Patton and Hubert 1993; Pinder et al. 1997; Koel 2004). However, ever-increasing anthropogenic modifications to large rivers and their associated watersheds have degraded backwaters and reduced their connectivity to the main stem, resulting in population declines of native species with specific life history demands and promoting more tolerant generalist species, particularly exotics (Karr et al. 1985; Knights et al. 1995; Koel and Sparks 2002).

A substantial number of backwaters have been completely disconnected from their rivers to facilitate navigation and agriculture. Increased agricultural production on floodplains has increased sediment loading and deposition into river systems, particularly backwaters. Sediment deposition is especially prominent in the agriculturally dominated Illinois River Valley (Bellrose et al. 1983; Havera and Bellrose 1985; Ickes et al. 2005). Consequently, ichthyofauna use of floodplains for foraging, reproduction, and winter survival has been compromised (Carter et al. 1985; Sheehan and Rasmussen 1999; Ickes et al. 2005). Much focus over the last 20 years has been placed on reducing sedimentation in large-river systems, particularly in the Upper Mississippi River (UMR) system, and restoring backwater habitat quality while maintaining lateral connectivity; fish use is often a low priority of these projects. Restorations have been accomplished primarily through Habitat Rehabilitation and Enhancement Projects (HREP), which focus on restoring floodplain habitat within the UMR watershed.

Since initiation in 1985, approximately 40 HREPs have improved habitat on nearly 27,000 hectares (U.S. Army Corps of Engineers [USACE] 2004). These projects often include the installation of inexpensive low-maintenance water control structures (also called stop-log structures or gates; see Ickes et al. 2005 for details) engineered to facilitate water level management, support connectivity, and reduce sediment loading (T. George and G. Lee, USACE, personal communication). The openings within the structures can be manipulated to further influence connectivity and ultimately fish accessibility.

Literature describing HREP effects on lateral fish movement is limited. Restorations have improved localized fish populations (O'Hara et al. 2001) and season-specific uses (Gent et al. 1995); however, these studies did not specifically review effects of connectivity on temporal immigration. Ickes et al. (2005) noted structures associated with backwater restoration projects aimed at mitigating sediment loading could be limiting seasonal fish passage. They also suggested literature describing fish passage through such structures as well as methods to limit movements of destructive species, such as common carp *Cyprinus carpio*, silver carp *Hypopthalmichthys molitrix*, and bighead carp *H. nobilis*, were also needed. Common carp have detrimental effects on water quality and aquatic macrophyte growth (Drenner et al. 1997), and Asian carp likely have similar effects by re-suspending sediments. Therefore, the objectives of this study were to 1) quantify the vertical and horizontal orientation of fishes as they entered a recently HREP-restored backwater through a water control structure with implications for reducing immigration of antagonistic species, and 2) assess the effects of the recently installed structure on lateral connectivity.

Methods

Study site

The Illinois River (ILR) enters UMR Pool 26 at UMR-kilometer 354 near Grafton, IL (Figure 1). Swan Lake is a 1,100-ha ILR backwater that parallels the ILR in Calhoun County, Illinois beginning near ILR-kilometer 19 and ending at ILR-kilometer 8. The backwater comprises the majority of permanently available floodplain habitat near the ILR and UMR confluence. The lake, which is a substantial portion of the Two Rivers National Wildlife Refuge (NWR), was isolated from the Illinois River in 1993 and compartmentalized into three impoundments via levee erections as part of the Swan Lake HREP; the natural connection between the lake and the ILR was ~ 400 m. The lower compartment of post-HREP Swan Lake (lower Swan Lake [LSL]; 550-ha) has remained continuously connected to the ILR for fish availability through an in-expensive water control structure. The structure regulates flow through four distinct 1.25-m wide columns, and facilitates partial and complete isolation and water level manipulation using stop-logs (25 cm x 25 cm square logs ~ 1-m long) placed within the columns. Structure design is typical of those used in other restoration projects (T. George, USACE, personal communication).

A rip-rap-lined channel ~12-m wide supported by a concrete base currently connects LSL and the ILR through the structure. At average river stage 127.9 m, mean depth in the structure is approximately 2.2 m and mean lake depth is 0.5 m. Water flow direction through the water control structure changes frequently within a 24-h period and velocity is highly variable. Flow direction and velocity are affected by watershed inputs into the lake, river stage, and localized commercial and recreational traffic. LSL is often windswept and turbid; sediments are flocculent and aquatic vegetation is absent (Garvey et al. 2004).

Stop-log nets

Rectangular wooden frames were constructed from 25 cm x 25 cm timbers to sample fishes at variable horizontal and vertical positions within the water control structure. Frames were placed into their respective structural columns two weeks prior to sampling initiation to soak and lose their buoyancy. Modified hoop nets (19-mm bar mesh; five 0.9-m fiberglass hoops with throats on the first and third hoops; 1.0-m wide x 1.2-m high rectangular metal frame net opening) were attached to the frames by outlining the sides and the bottom of the frame openings with galvanized steel C-channel. Nets were slid into the C-channel from the top, lowered to their respective sampling depth, and stretched tight horizontally (Figure 2). Nets were systematically fished simultaneously at three depths (surface, intermediate, and bottom) in the three adjacent most-downriver columns (Figure 3) over a 24-h period during March 2004 through May 2005 to capture fishes as they entered the lake through the structure. Fish exiting the lake were not sampled. Nets were fished weekly during spring and fall and bi-monthly during winter and summer when environmental conditions permitted. Fishes captured were identified to species, measured to the nearest millimeter (total length [TL]), and weighed (g). When more than 100 freshwater drum *Aplodinotus grunniens* or white bass *Morone chrysops* <

200 mm TL were present in a single catch, a representative sub-sample of fish < 200 mm TL were measured and the rest were counted; all fish $\geq$ 200 mm TL were measured. Gizzard shad *Dorosoma cepedianum* were sub-sampled similarly, but independently of length. Within the data set, measured fish were placed into 10-mm length bins, and bin-specific lengths were assigned to individuals not measured in the field based on proportion of occurrence within the measured sub-sample. Nets that did not fish effectively over a 24-h period (i.e. anchor slippage resulted in collapse past the first hoop) were not included in analyses.

Statistical Analyses

All statistical analyses were performed using Statistical Analysis System (SAS) software v. 8.2. Daily catch rates were calculated for 24-h net sets and were quantified to characterize pooled and species-specific densities and size distributions of fishes entering the lake with respect to vertical and horizontal orientation. Daily CPUE was $\log_{10}(x + 1)$ transformed to normalize the data. Data collected during summer 2004 were excluded from all analyses because highly variable water levels resulted in sporadic sampling. Daily CPUE (seasons combined) was compared on both pooled and species-specific levels among respective horizontal and vertical positions using a two-way ANOVA for repeated measures (MIXED procedure; SAS 1999). Daily CPUEs were compared within seasons in a similar manner; seasons among years were tested independently of each other. Species-specific ANOVAs were limited to gizzard shad, freshwater drum, and white bass. Variation in CPUE was partitioned among depths, columns, and depth*column interaction for all ANOVA models. Post hoc pairwise comparisons were implemented using CONTRAST statements. Mean daily catch rates of bighead, common, and silver carp among seasons were calculated, but were not tested statistically due to their high variability.

Seasonal length-frequency distributions of freshwater drum and gizzard shad were tested among horizontal and vertical positions using Kolmogorov-Smirnov (KS) pairwise comparisons (NPAR1WAY procedure; SAS 1999); white bass were excluded from length-frequency analyses because too few fish were captured during most seasons. A significance level α of 0.05 was established a priori for all tests except for KS comparisons, where a Bonferroni-corrected α of 0.017 was used to determine significance.

Results

A total of 99 stop-log net nights were fished between March 2004 and May 2005. However, only 6 sets occurred during summer 2004; these sets were removed from analyses, yielding a revised total of 93 net nights (Table 1). Taxa collected consisted of 34 species comprising 11 families (Table 2). Gizzard shad (percent total catch; 73.0%), freshwater drum (19.3%), and white bass (2.5%) were most abundant.

Using two-way ANOVAs, no overall or seasonal differences existed among columns ($P > 0.05$) or depths ($P > 0.05$) of passage for pooled catches, nor did a statistical interaction exist between depth and column on pooled CPUE ($P > 0.05$). Catch rates did not differ among columns overall or seasonally for gizzard shad, freshwater drum, or white bass ($P > 0.05$). However, overall catch rates of freshwater drum were higher at the middle and bottom depths than at the surface ($F = 8.59$; df = 2; $P < 0.001$) (Figure 4). Seasonally, this same difference existed only during spring 2004 ($F = 4.58$; df = 2, 22; $P < 0.05$), although the general trend was higher catch rates in the lower two-thirds of the water column during all seasons. No significant interactions occurred between depth and column of passage for overall species-specific catches ($P > 0.05$), but a significant interaction between depth and column occurred during winter 2004-2005 for catches of freshwater drum ($F = 46.58$; df = 2; $P < 0.01$) (Figure 5). Bighead carp CPUE was generally random among depths; immigration into the backwater was highest during winter (Table 3). Conversely, catch rates of common and silver carp were typically highest at middle and lower depths, and silver carp showed a general affinity for channel edge (Table 3, column C; see Figure 3). Common carp CPUE was highest during spring, whereas silver carp CPUE peaked during fall and winter. Other common taxa not analyzed due to consistently low catch rates, such as catastomids and ictalurids, were collected at all three depths and in all three columns.

Analyses of length-frequency distributions using KS pairwise comparisons determined lengths of fishes captured differed primarily among depths; differences in lengths among columns were marginal, and therefore were not reported. Freshwater drum captured at the bottom of the structure were typically longer than those captured at the surface ($P < 0.0001$) or the middle ($P < 0.0001$) depths (Table 4). Gizzard shad were generally shorter in surface sets during winter 2004 and during spring 2005 than at the other two depths ($P < 0.017$). Conversely, longer gizzard shad were caught at the surface than at middle and bottom depths during spring and fall 2004 ($P < 0.001$). In general, both freshwater drum and gizzard shad length-frequency distributions were multi-modal, consisting of an abundant group of juvenile fishes and a less abundant group of adult fishes; this trend was most prominent during spring. Common carp captured were usually > 400 mm TL, whereas the majority of bighead and silver carp captured were < 300 mm TL.

Discussion

Immigration and exclusion

Fish passage through artificial structures in large rivers has been evaluated on several levels (Odeh 2000). However, these assessments specifically addressed linear (upstream and downstream) navigation of dams by migratory species, and therefore have limited applicability to lateral floodplain connectivity through water control structures such as those installed at Swan Lake and elsewhere. A potential management application of stop-log structures is reducing immigration of certain species into a backwater. Determent, exclusion, and fish passage systems using bioacoustics, physical barriers, and other forms of barriers and structures have been evaluated for several taxa with variable success rates (Thompson and Rahel 1998; Odeh 2000; Josephson et al. 2001; Winter and Van Densen 2001; Clarkson 2004; Taylor et al. 2005). Generally, gears are designed to prevent the spread of a species into other areas. This study, however, focused on limiting habitat availability based on the two-dimensional distribution of fishes as they passed through a water control structure. Using these structures, partial exclusion can potentially be accomplished using stop logs selectively placed within the water control structure. However, stop-log barrier effectiveness will hinge largely on fish behavior.

Fish movement into LSL seems unaffected by the stop-log structure. Gizzard shad and bighead carp passed randomly through the structure, showing little preference for depth or column. Conversely, freshwater drum was the only species that exhibited statistical selection for passage depth; freshwater drum generally entered the lake through the bottom 1.5 m of the structure. Differences in gizzard shad length-frequency distributions among depths were driven primarily by seasonal size-specific catch rates, not size-selective vertical distribution. For both species, lengths were most variable during fall and winter 2004. Increased variability in the length-frequency distributions of freshwater drum and gizzard shad during these two seasons can be attributed to infrequent use of LSL as winter habitat, particularly by adults. However, longer fish were consistently captured during spring and this was associated with reproduction; larval tows within LSL during 2004 and 2005 verified reproductive success (L. Csoboth, SIUC, unpublished data).

Catch rates of carps were highly variable and were typified by seasonal increases in abundance. Similar to freshwater drum, common and silver carp were captured most often in the lower two-thirds of the water column. Silver carp CPUE, which was dominated primarily by juveniles, was highest during fall and winter, whereas common carp CPUE was highest during spring and comprised entirely of adults. Similar to silver carp, bighead carp CPUE also peaked during winter and consisted primarily of juveniles. Juvenile bighead and silver carp may be seeking thermal refugia within backwaters during winter in a manner similar to those described for other species (Bodensteiner and Lewis 1992; Knights et al. 1995; Raibley et al. 1997). In contrast, common carp were likely using the backwater for reproductive purposes; this was similar to results using other gears (Schultz 2006). Therefore, the immigration of juvenile Asian carp and adult common carp into backwaters through structures similar to those at LSL could be reduced by closing off the bottom two-thirds of the structure, and may similarly reduce immigration of native freshwater drum. In theory, effective reductions in immigration could affect recruitment dynamics by decreasing winter survival of young-of-year Asian carp and freshwater drum in a fashion similar to other species (McCollum et al. 2003), and may also reduce the spawning stock density of adult common carp and freshwater drum present in the lake during spring. Reductions in the number of adults within the lake could promote improvements in habitat quality, such as reduced turbidity (Drenner et al. 1997) and increased macrophyte growth.

Effects of partial closure on immigration will depend largely on fish behavior, i.e. will an individual's affinity for depth override its impulse to enter a backwater. Partial closure effectiveness would likely be minimal during the spring flood pulse and during winter immigration for winter survival; backwater use during both seasons influence the recruitment dynamics of many river fishes (Bodensteiner and Lewis 1992; Patton and Hubert 1993; Knights et al. 1995; Raibley et al. 1995; Raibley et al. 1997), and fishes have an instinctive desire to enter backwaters for these reasons. Therefore, we recommend that the effectiveness of partial closure on immigration of fishes into backwaters be evaluated to determine its applicability. Ultimately, the cost-benefit ratio of maintaining complete connectivity for native species versus partial or complete closure to exclude exotic species needs to be evaluated when determining management objectives.

Lateral connectivity

Maintaining lateral connectivity in our river systems is a vital component to the life histories of our native riverine fishes (Sheehan and Rassmussen 1999; Ickes et al.

2005); conversely, connectivity supports exotic species in a similar manner. The structures installed at backwaters throughout the UMR create permanent connections between backwaters and their rivers, thereby allowing continuous exchange of biomass and nutrients. These structures also mimic historical hydrology when managed passively (Ickes et al. 2005), although water exchange rates are substantially reduced, particularly during floods, and may in turn negatively influence passage of some species (Ickes et al. 2005; Schultz 2006).

Schultz (2006) surmised the Swan Lake HREP was largely ineffective at restoring habitat for fishes within LSL, particularly deep-water winter habitat and aquatic vegetation, due to its vertical homogeneity, substantial fetch, and thermal instability. For example, the historically abundant largemouth bass *Micropterus salmoides* population within Swan Lake (Sheehan et al. 1990; Sheehan et al. 1994) had declined; high winter mortality of young-of-year fish (McCollum et al. 2003) resulting from poor winter habitat, not limited fish access resulting from structure installation, may be the primary cause of recession (Schultz 2006). Therefore, the lake constitutes marginal habitat overall, and our results may not be representative of the potential benefits structure installations may have in other systems, particularly for species heavily dependent on backwaters, such as centrarchids.

Ickes et al. (2005) suggested stop-log structures might preclude some large-bodied species, benthic species, and species with highly developed electroreceptors, such as catfish (New 1999) and paddlefish *Polyodon spathula* (Gurgens et al. 2000), from entering floodplains. The stop-log structure appears to have had minimal impacts on the immigration patterns of large-bodied cyprinids, catastomids, and moronids common to

Swan Lake during the pre-HREP era (Sheehan et al. 1990; Sheehan et al. 1994) as well as newly introduced species. Channel catfish passed through the structure regularly during all seasons (Schultz 2006), and paddlefish were willing to navigate structures as well (D. Schultz, unpublished data), although their use of the lake has diminished substantially relative to historical records (R. Maher, IDNR, unpublished data). Fish passage seems largely driven by habitat quality and its interaction with hydrology (Koel 2004; Schultz 2006). Therefore, in backwaters where habitat is more favorable for fishes, these structures are, in all likelihood, highly beneficial to the river fishery by ensuring connectivity.

Conclusions and Future Considerations

This was an exploratory study; no other literature exists describing the distributions of freshwater fishes through this type of structure. Hence, comparative research evaluating the potential for and the effectiveness of reducing immigration of non-desirable ichthyofauna through these structures is still needed. Specifically, these evaluations should consider habitat quality as a functional component of connectivity. Nets used during this evaluation were 19-mm bar mesh; these nets may have underestimated immigration when fished with velocities exceeding 1.0 m/s (cod end being most up-flow) because debris accumulated on the cod end, forcing the anchors holding the nets horizontal to slip and causing partial collapse of the net. Larger mesh sizes would likely reduce these effects, but would in turn bias results based on gear size-selectivity. Therefore, future efforts must address specific study objectives as they pertain to fish size before study initiation.

Regarding HREP effects and lateral connectivity, we cannot emphasize enough the importance pre-HREP data collection has towards assessing restoration effects on the fishery. Koel (2004) suggested using pools with high native species richness as relative references for more degraded systems, such as the Illinois River. However, such comparisons would be modest at best relative to a thorough before-after comparison of the same fish community. With this in mind, data describing the specific importance of these habitats to fishes, particularly species in decline, need to be collected. More consideration during HREP planning with implications for research needs to be directed towards fishes, specifically the role of backwaters to their life histories, and the direct effects HREPs have on associated river fisheries.

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Table 1. Distribution of seasonal sampling effort among depths and columns at the lower Swan Lake water control structure, Calhoun County, Illinois from March 2004 through May 2005.

		Total N sets				
Column	Depth	Spring 2004	Fall 2004	Winter 2004	Spring 2005	Total
A	1	4	3	2	3	12
	2	3	2	1	3	9
	3	3	3	1	3	10
B	1	3	3	1	3	10
	2	5	3	2	3	13
	3	3	2	1	3	9
C	1	3	2	1	3	9
	2	3	3	1	2	9
	3	4	3	2	3	12
Total		31	24	12	26	93

Table 2. Seasonal species composition (%) of seasonal stop log catches within the lower Swan Lake water control structure. *Ictiobus spp.* includes *I. niger*, *I. cyprinellus*, and *I. bubalus*.

		Proportion of seasonal catch (%)			
Family	Species	Spring 2004	Fall 2004	Winter 2004	Spring 2005
Amiidae	*Amia calva*	0.03	0.01	0.00	0.00
Catastomidae	*Carpiodes carpio*	0.07	0.01	2.53	0.17
	Catostomus commersoni	0.00	0.00	0.00	0.02
	Ictiobus spp.	0.34	0.06	0.69	0.54
	Moxostoma macrolepidotum	0.00	0.00	0.00	0.02
Centrarchidae	*Lepomis cyanellus*	0.00	0.01	0.00	0.05
	L. gulosus	0.03	0.00	0.00	0.00
	L. humillis	0.03	0.00	0.00	0.00
	L. macrochirus	0.17	0.03	0.46	0.05
	Micropterus salmoides	0.00	0.00	0.23	0.00
	Pomoxis annularis	0.00	0.01	0.00	0.00
	P. nigromaculatus	0.41	0.10	0.46	0.07
Clupeidae	*Alosa chrysochloris*	0.00	1.21	0.00	0.00
	Dorosoma cepedianum	58.33	82.46	57.83	70.94
	D. petenense	0.00	0.46	0.00	0.00
Cyprinidae	*Carassius auratus*	0.14	0.01	0.00	0.05
	Ctenopharyngodon idella	0.00	0.00	0.00	0.00
	Cyprinus carpio	0.65	0.18	0.92	0.37
	Hypophthalmichthys molitrix	0.34	0.97	5.53	0.29
	H. nobilis	0.17	0.01	2.30	0.20
	Notropis atherinoides	0.03	0.00	0.00	0.00
Hiodontidae	*Hiodon tergisus*	0.00	0.00	0.23	0.00
Ictaluridae	*Ameiurus melas*	0.00	0.00	0.00	0.29
	A. natalis	0.31	0.01	0.23	0.10
	A. nebulosus	0.14	0.00	0.23	0.00
	Ictalurus punctatus	2.13	0.42	1.84	1.62
Lepisosteidae	*Lepisosteus platostomus*	1.38	0.07	1.15	0.12
Moronidae	*Morone americana*	0.07	0.06	0.23	0.02
	M. chrysops	1.00	3.49	1.61	1.64
	M. mississippiensis	0.14	0.13	0.23	0.12
Polyodontidae	*Polyodon spathula*	0.00	0.00	0.00	0.02
Sciaenidae	*Aplodinotus grunniens*	34.06	10.24	23.27	23.27
Total *n*		2,904	6,682	434	4,078

Table 3. Mean daily catch rates of bighead carp *Hypopthalmichthys nobilis*, common carp *Cyprinus carpio*, and silver carp *H. molitrix* captured at varying depths and columns within the lower Swan Lake water control structure.

			Mean Daily CPUE		
Location	Season	Label	Bighead carp	Common carp	Silver carp
Depth	Spring 2004	1	0.39 (0.26)	0.42 (0.28)	0.59 (0.42)
		2	0.21 (0.21)	1.24 (0.66)	0.73 (0.41)
		3	0.47 (0.47)	2.11 (0.89)	0.62 (0.32)
	Fall 2004	1	0.24 (0.24)	0.26 (0.26)	1.09 (1.09)
		2	0.00 (0.00)	0.73 (0.52)	5.60 (3.83)
		3	0.00 (0.00)	1.96 (0.76)	8.60 (7.75)
	Winter 2004	1	1.46 (0.94)	0.00 (0.00)	0.47 (0.47)
		2	0.99 (0.57)	1.48 (0.93)	9.38 (4.30)
		3	2.52 (1.26)	0.52 (0.52)	1.50 (1.50)
	Spring 2005	1	1.11 (0.88)	0.65 (0.46)	0.89 (0.49)
		2	0.49 (0.49)	1.21 (0.81)	0.70 (0.70)
		3	0.23 (0.23)	1.51 (0.43)	1.06 (0.46)
Column	Spring 2004	A	0.63 (0.32)	1.53 (0.67)	0.59 (0.42)
		B	0.43 (0.43)	1.02 (0.71)	0.73 (0.41)
		C	0.00 (0.00)	1.24 (0.69)	0.62 (0.32)
	Fall 2004	A	0.00 (0.00)	1.20 (0.52)	1.34 (1.09)
		B	0.24 (0.24)	0.49 (0.49)	4.68 (3.86)
		C	0.00 (0.00)	1.26 (0.76)	9.28 (7.70)
	Winter 2004	A	1.46 (0.94)	0.00 (0.00)	0.93 (0.54)
		B	1.49 (0.50)	1.48 (0.93)	4.85 (4.20)
		C	2.03 (1.41)	0.53 (0.53)	5.58 (3.86)
	Spring 2005	A	0.44 (0.44)	1.06 (0.47)	1.06 (0.46)
		B	1.11 (0.88)	0.86 (0.65)	0.84 (0.63)
		C	0.26 (0.26)	1.49 (0.62)	0.76 (0.54)

Table 4. Seasonal length frequency statistics for freshwater drum *Aplodinotus grunniens*, gizzard shad *Dorosoma cepedianum*, and white bass *Morone chrysops* captured at varying depths within the lower Swan Lake water control structure. Refer to Figure 3 for clarification of depth. SE = standard error of the mean, CV = coefficient of variation.

			Statistic				
Species	Season	Depth	Mean	SE	Median	CV	N
Freshwater drum	Spring 2004	1	132.9	3.9	120.0	31.4	116
		2	132.8	2.4	120.0	35.0	375
		3	145.8	2.8	120.0	42.8	498
	Fall 2004	1	122.4	18.7	100.0	70.1	21
		2	116.9	2.3	110.0	32.3	263
		3	118.9	1.8	110.0	30.2	400
	Winter 2004	1	167.5	30.2	120.0	62.4	12
		2	174.8	19.4	130.0	53.3	23
		3	160.2	10.5	130.0	53.2	66
	Spring 2005	1	134.5	3.5	120.0	34.8	182
		2	128.7	3.0	120.0	31.6	186
		3	155.1	2.5	130.0	38.7	581
Gizzard shad	Spring 2004	1	203.8	3.9	190.0	35.9	360
		2	184.7	2.4	160.0	33.9	678
		3	190.1	2.1	170.0	28.9	656
	Fall 2004	1	133.2	2.3	100.0	54.1	1,000
		2	119.6	0.9	100.0	42.5	3,057
		3	123.0	1.5	100.0	45.8	1,453
	Winter 2004	1	150.1	7.4	110.0	50.0	102
		2	210.9	10.7	240.0	45.8	82
		3	168.8	11.8	100.0	57.4	67
	Spring 2005	1	199.1	1.6	210.0	31.6	1,478
		2	206.0	2.6	220.0	34.8	759
		3	221.8	2.6	230.0	30.3	656

Figure 1. Location of the study site, lower Swan Lake, Calhoun County, Illinois.

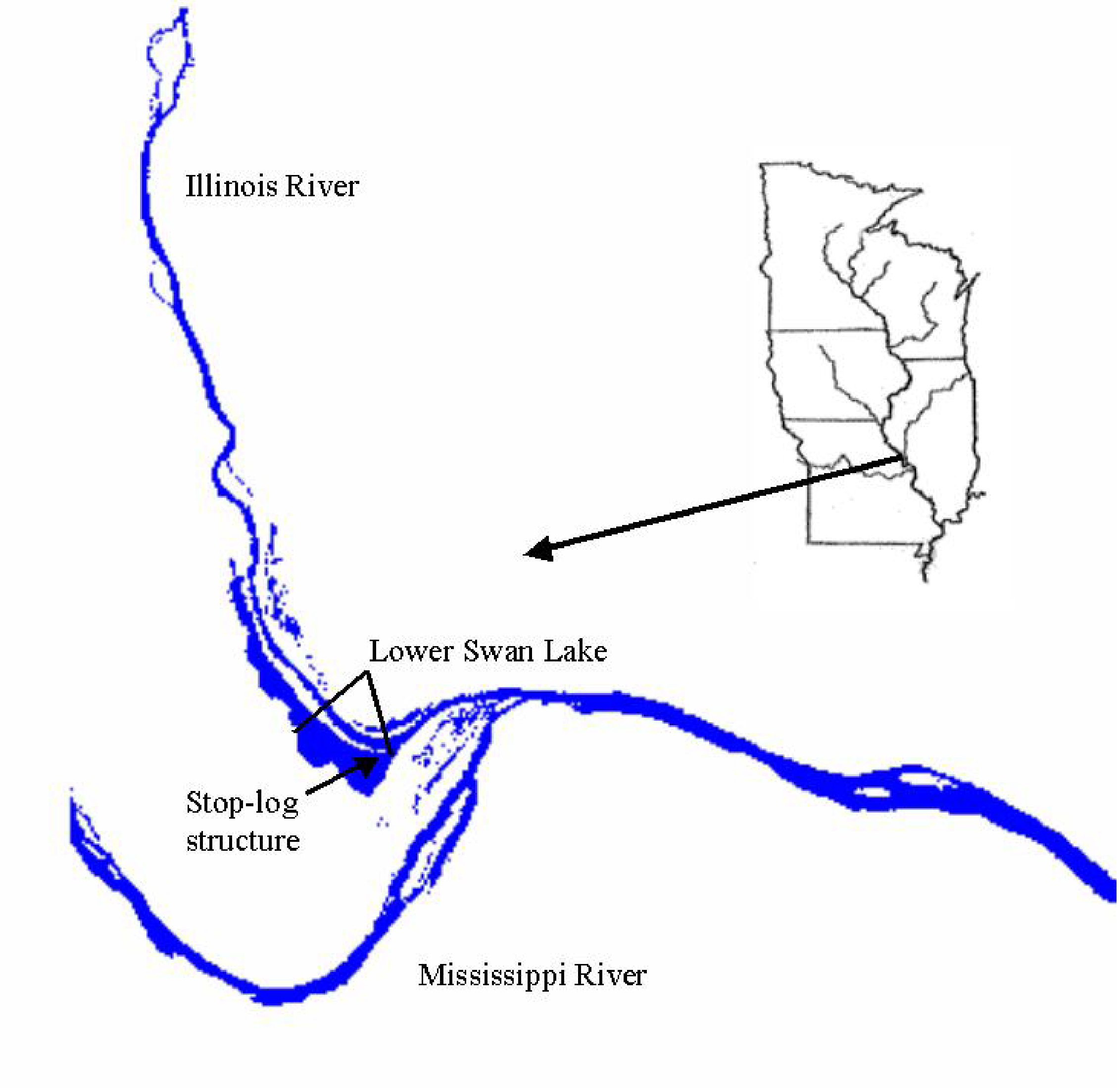
Figure 2. A side view of methods and equipment utilized to fish stop log nets horizontally in the lower Swan Lake water control structure as they captured fish entering lower Swan Lake. The cable was attached to the steel bulkheads by welding chain links to 5 cm x 5 cm angle iron in 0.3-m intervals and then welding the angle iron to the bulkheads vertically above the water line.

Figure 3. Wooden frames (hanging above the water) used to fish stop-log nets within the lower Swan Lake water control structure and the sampling grid used to fish nets on successive trips. "Set 1" refers to the depth-column configuration of nets set during the first sampling trip. The fourth sampling trip would be a repetition of "Set 1". Frames with nets attached were lowered to their respective positions, and nets were fished for 24-h periods.

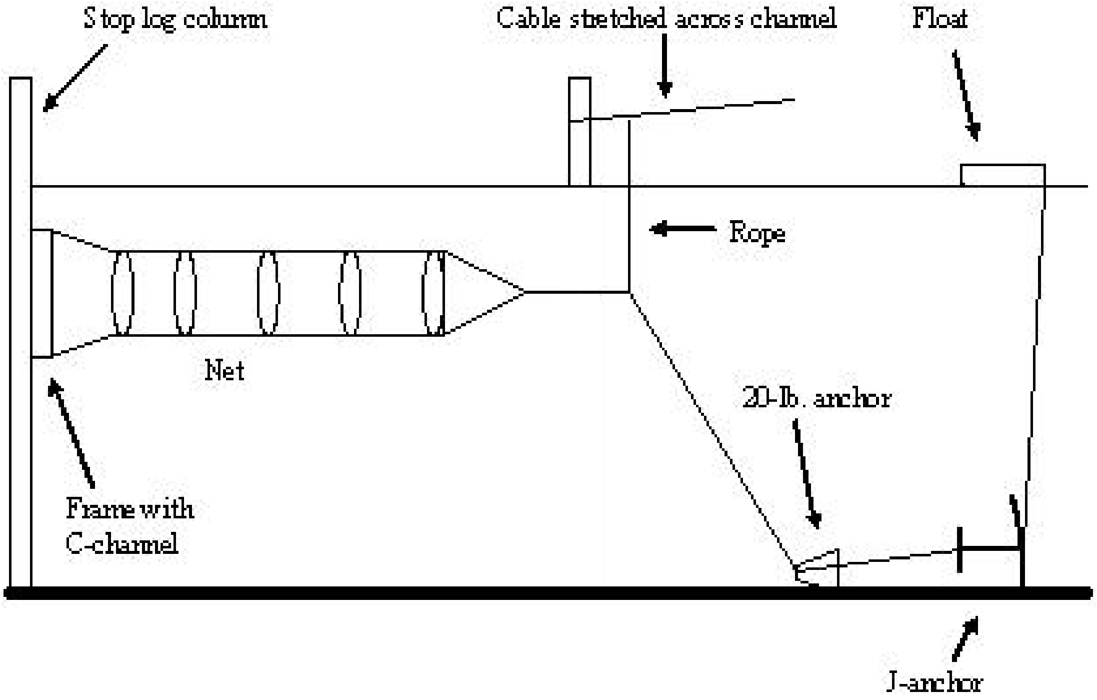
Figure 4. Species-specific mean + 1 se daily CPUE (seasons combined) for fishes captured in stop log nets as they entered lower Swan Lake through different columns and at varying depths within the structure. No species exhibited differences among columns of passage. Only freshwater drum *Aplodinotus grunniens* catch rates differed among depths; Tukey-Kramer adjusted differences are indicated by letters. FWD = freshwater drum, GZS = gizzard shad *Dorosoma cepedianum*, and WHB = white bass *Morone chrysops*.

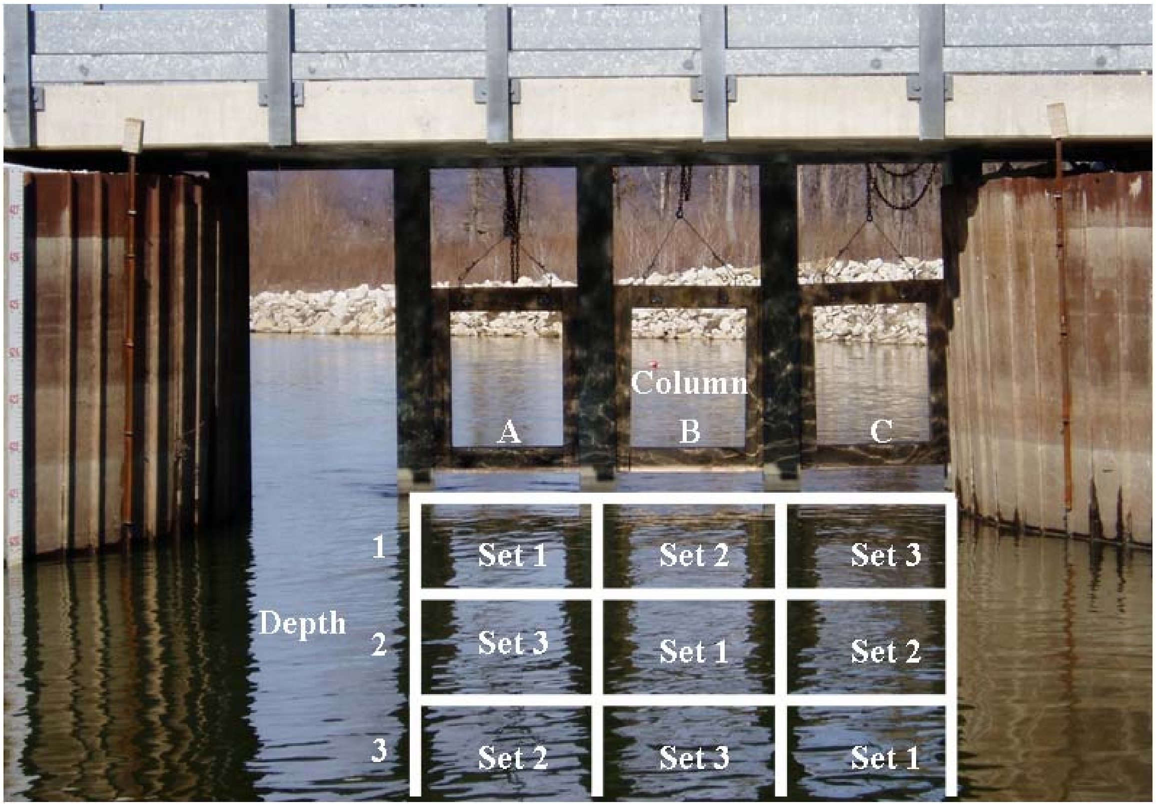
Figure 5. Mean + 1 se daily CPUE of freshwater drum *Aplodinotus grunniens* captured in stop log nets during winter 2004. A significant interaction existed among depths and columns (block; refer to Figure 3).

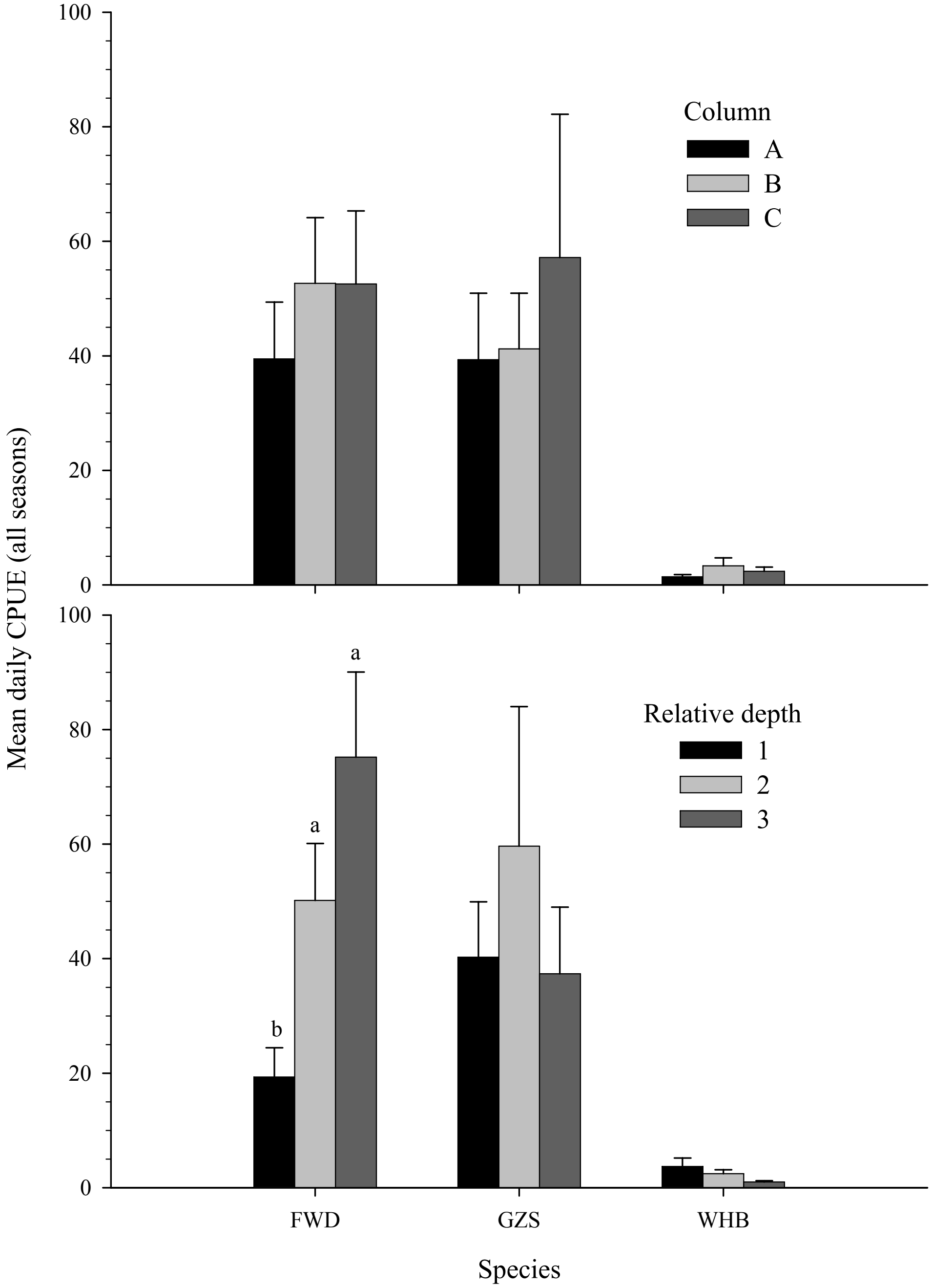
Figure 1: Study site



← Illinois River Lower Swan Lake →




Column
A
B
C
Depth
1
2
3
Set 1
Set 2
Set 3
Set 3
Set 1
Set 2
Set 2
Set 3
Set 1


100
80
60
40
20
0
Column
A
B
C
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80
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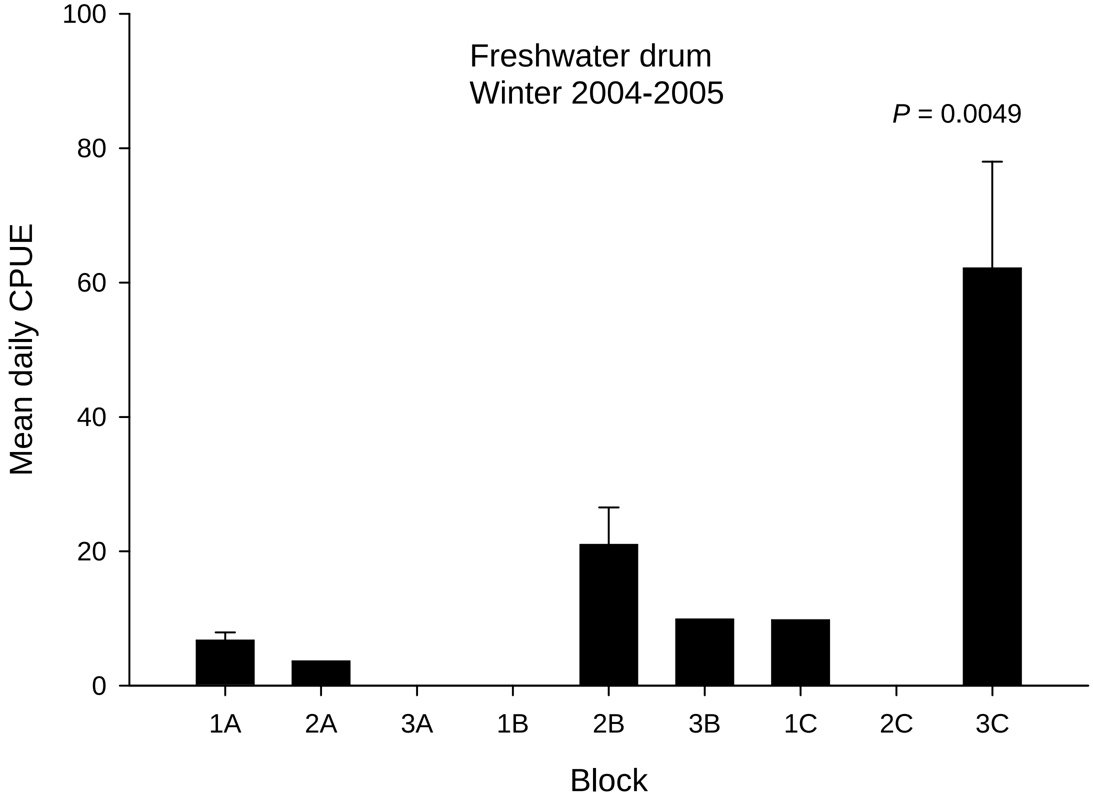
40
20
0
a
a
b
Relative depth
1
2
3
Mean daily CPUE (all seasons)
FWD
GZS
WHB
Species


Freshwater drum
Winter 2004-2005
P = 0.0049
Mean daily CPUE
0
20
40
60
80
100
1A
2A
3A
1B
2B
3B
1C
2C
3C
Block

ASIAN CARP HABITAT USE & MOVEMENT

SUMMARY

We evaluated habitat use and movements of fifty adult bighead carp *Hypophthalmichthys nobilis* and fifty silver carp *H. molitrix* using ultrasonic telemetry during spring through summer 2004 and 2005 to gain insight into conditions that facilitate their establishment, persistence, and dispersal in the lower Illinois River (river kilometers 0-130). Movement and habitat use were monitored with stationary receivers and boat-mounted tracking. The relative availability of main channel, island side channel, channel border, and connected backwater was quantified to determine selection; discriminant function analysis evaluated changes in physical characteristics within each habitat category. A flood pulse occurred in spring through early summer 2004 but not in 2005. Movement rates (km/week) of both species were positively correlated with flow but not temperature. Including data from stationary receivers greatly increased estimates of daily movement. During low summer flow, both species typically selected channel borders and avoided the main channel and backwaters. Both species rarely occupied depths > 4 m, regardless of abiotic conditions. Flood pulses appear to trigger dispersal, while habitat use is only specific during low summer flow. Thus, preventing movement (e.g., dispersal barriers) should be especially vigilant during late-winter/spring flooding and controlled removal (e.g., harvest) should be directed toward selected habitats during summer.

INTRODUCTION

Many successful invading fishes possess life history traits of r-selected species, generally exhibiting rapid growth rates, short generation times, exceptional dispersal capabilities, high reproductive output early in life, high density in their native range, and broad environmental tolerance (Ehrlich 1984; Lodge 1993). These opportunistic characteristics allow populations to become dense soon after establishing (Lodge 1993; Williamson 1996; McMahon 2002). The ability of invasive exotics to disperse and then establish in novel locations is particularly problematic in rivers due to the broad range and high connectivity among these systems (Junk et al. 1989).

Two river-dwelling Asian fishes, bighead carp *Hypophthalmichthys nobilis* and silver carp *H. molitrix* became established in the Mississippi River basin in the early 1980s (Freeze & Henderson 1982; Costa-Pierce 1992). In the late 1990s, these species expanded into the connected Illinois River system and their density has increased (Koel et al. 2000; Chick and Pegg 2000; Conover et al. 2006). Clearly, the connection between the Mississippi River and Illinois River plus the apparently high dispersal potential of these species facilitate their expansion. Given that the Illinois River is connected to Lake Michigan via a shipping canal, there is great need to understand factors influencing the ability for these species to move into novel areas and become established.

Movement and habitat selection by adult carps should provide insight into the conditions that facilitate their establishment, persistence, and dispersal. We used ultrasonic telemetry to quantify movement and habitat selection of both species in an area of a well-established Asian carp assemblage: the lower reach of the Illinois River and one of its major backwaters, the 1,100 ha Swan Lake. We expected that habitat use in both the river and the adjacent backwater would be nonrandom, reflecting selection. Movement from selected areas would be seasonal and perhaps related to spawning. In their native waters, a sharp rise in stage and current velocity has been associated with spawning migrations, and shortly thereafter spawning (Krykhtin and Gorbach 1981; Abdusamadov 1987). In the LaGrange Pool of the Illinois River, movement of bighead carp appeared to increase with increased flow (Peters et al. 2006). However, it remains unclear about the relative contribution of river stage and temperature (or a combination) to movement and how this differs between bighead and silver carp.

The objectives of our telemetry effort were to (1) determine whether temperature or flow was related to bighead and silver carp movement in the lower Illinois River (i.e., an index of dispersal potential), (2) identify gross habitat categories (hereafter macrohabitats) that bighead and silver carp avoid or select, and (3) identify abiotic characteristics at fish locations (i.e., microhabitats) within selected macrohabitats.

METHODS

<u>Study Site</u>. The low-gradient, lower Illinois River extends between the La Grange Lock and Dam (River kilometer, Rkm 130) at Beardstown, IL to the confluence of the Mississippi River (Rkm 0) at Grafton, IL. Despite a century of alterations due to dredging, water diversion from Lake Michigan, channelization, and levee construction, the river still retains an annual, albeit flashy flood pulse (Karr et al. 1985; Sparks 1995).

Swan Lake is the major (1,100 ha) backwater of the Illinois River-Mississippi River confluence and an important source of secondary and ultimately fish production. To reduce its sedimentation, this backwater was separated into three management compartments by levees to form Lower Swan Lake, Middle Swan Lake, and Upper Swan Lake (Figure 1). Lower Swan Lake comprises nearly half the area, and is the only compartment continuously connected with the Illinois River that subsequently allows access to Asian carps and other fishes. Middle Swan Lake becomes accessible to fishes during flooding.

We classified the lower 41 km of the Illinois River, including Swan Lake (Figure 1), into four macrohabitat categories: main channel; channel border; island side channel; and backwater (i.e., mostly Swan Lake). The proportion of available habitat was derived using digital raster graphic (DRG) topographic maps obtained from the Illinois Natural Resources Geospatial Clearinghouse (http://www.isgs.uiuc.edu/nsdihome/), ArcMap 9.2 and US Army Corps of Engineers navigation maps (http://www2.mvr.usace.army.mil/NIC2/ilwwcharts.cfm). Areas of the river with a depth less than 5.0 m at normal pool (NP) were classified as channel border; areas equal to and greater than 5.0 m NP were classified as main channel; areas between the channel border and islands classified as island side channel; and Swan Lake (Lower and Middle) comprised the backwater habitat.

<u>Fish Collection and Transmitter Implantation</u>. Fish to be tracked with telemetry were collected either from the lower Illinois River near Swan Lake (N=86) or from

Lower Swan Lake (N=14) (typically rkms 0-10; Figure 1). Asian carps are notoriously difficult to sample (Williamson and Garvey 2005). Thus, a combination of gears was used. Drifting and dead-set trammel nets (experimental nets of 51 mm, 76 mm, 102 mm and 89 mm bar mesh panels; 3.7 m, 4 m, 4.3 m and 4 m outer wall respectively; 91.4 m in length) were primarily used, but hoop nets (38 mm bar mesh, 1-m diameter fiberglass hoops), trap nets, commercial fishers, electrofishing, and fish jumping into the boat also were sources of fish (see Figure 1 for distribution of capture). During March through April 2004, twenty-five bighead carp and twenty-five silver carp were collected. During September 2004, an additional fifteen fish from each species were caught. During March 2005, another ten individuals within each species were sampled (50 total each species).

After capture, surgery and implantation of transmitters were conducted following the guidelines of Summerfelt and Smith (1990). Each fish was placed in a holding tank with buffered (sodium bicarbonate) river water; carbon dioxide (CO_2) gas was diffused into the tank for anesthetization. Fish were measured (total length, TL, mm) and weighed (kg). River water was circulated over their gills. Before surgical incision, scales were removed from the ventral left area of each fish, posterior to the pelvic fin and anterior to the anus. After the removal of scales, the area was disinfected with betadine. For silver carp, the incision was made further dorsally than bighead carp to account for the displacement of the body cavity due to the well-developed keel.

Ultrasonic transmitters (69 kHz; 10 g in water and 65-mm long; < 2% body weight; V16, Vemco Ltd., Nova Scotia, Canada) for remote individual identification were implanted during surgery. Each transmitter was pulse-coded, which provided unique identification numbers. Transmitters implanted into fish in the March through

April 2004 had a minimum life expectancy of 570 days. The remaining 50 transmitters had a minimum life expectancy of 366 days. Each transmitter was tested for recognition before its use with a portable hydrophone and receiver (Vemco Model VR60). Immediately following surgery and implantation, each fish was placed in a recovery tank supplemented with oxygen and was released at the capture site after regaining buoyancy and swimming independently (Figure 1). We allowed implanted fish 2 weeks at large to recover before logging their movements with telemetry (Winter 1996).

Mobile Tracking. To quantify movement and macro- and microhabitat selection in the lower 41 km of the river (see Figure 1), fish were tracked by boat with a portable omnidirectional hydrophone and receiver. During both years, tagged fish were tracked monthly during April through August (period of purported spawning; but see DeGrandchamp et al. in press) starting upstream at Rkm 130 and progressing toward Rkm 0 and the lower and middle compartments of Swan Lake.

When detected with the omnidirectional hydrophone, each fish location was determined by drifting the boat toward the fish until the signal strength was the same at all directions with a directional hydrophone. Each fish location was georeferenced and microhabitat variables including depth (m), water temperature (ºC at 1 m depth; YSI Model 85 Dissolved Oxygen Meter; Yellow Springs Instruments, Yellow Springs, Ohio, USA), dissolved oxygen (DO; mg/L at 1 m depth; YSI 85), and water velocity (m/s at 1 m depth; Flo-Mate Model 2000 Velocity Meter; Marsh-McBirney, Inc., Fredrick, Maryland, USA) were quantified. Sediment was sampled at each site using a petite ponar grab (2.4 L volume; Wildlife Supply Company, Buffalo, New York, USA) and classified as predominately clay, silt, sand, gravel, or organic material (see Bain 1999).

Stationary Receivers. To enhance our movement data, stationary receivers (Vemco VR2 Single Channel Monitoring Receiver) also logged fish movement. In March 2004, two stationary receivers were mounted underwater at each side of the Lower Swan Lake channel (Figure 1), primarily to document movements in and out of Swan Lake. These receivers also continuously quantified main-channel passage past this location. In November 2004, additional receivers were affixed to navigation buoys and placed in the main channel of the Illinois River adjacent to the main navigation channel. Buoys were deployed at the following river kilometers (approximately every 16 kilometers): Rkm 3.2, 22.7, 36.4, 50.7, 67.4, 84.5, and 100.4. All were checked and downloaded every three weeks.

Statistical Analysis. For all analyses, spring was defined as March through May and summer as June through August. All statistical analyses were conducted with SAS 9.1 (SAS Institute 1996). Critical alpha for all tests was set at 0.05.

All geographic coordinates of fish locations were mapped using ArcMap 9.2; distances between successive individual fish locations were calculated. Fish that were detected either by mobile, boat-mounted tracking or by stationary receivers at least once during a season (about 90 days) were included in movement analysis. Distance between locations was measured as the shortest linear distance through water between successive fish locations, and therefore probably underestimated the actual distance traveled by individual fish.

Daily rate of movement (km/day) was quantified for individual fish during the spring and summer of both years, and a mean daily rate of movement was quantified for each species. The deployment of additional stationary receivers in late 2004 increased our detection rate, which in turn increased the precision of weekly and daily movement estimates in 2005. We compared rates with mobile tracking only to those from mobile tracking plus stationary receivers. To further determine how augmenting manual tracking with stationary receivers affected daily movement rates, a truncated 32-km section of river encompassing the three stationary receivers in the lower river was selected to determine the movement rate in 2005. This fine-scale daily rate of movement was based on the passage of 20 fish (12 bighead carp, 8 silver carp) during April through June 2005. Directionality of movement was quantified by determining the proportion of locations of each individual in Swan Lake as well as above and below its release point.

To determine how river conditions affected movement of bighead and silver carp, we quantified weekly rates (i.e., mean km/week) moved for each species in the spring and summer of 2004 and 2005. Again, we did this with mobile tracking only and mobile tracking plus stationary receiver locations. Mean daily water temperature was obtained at Rkm 34.6 with a submerged temperature logger (Vemco Minilog Temperature Data Logger). Daily river stage was obtained from the United States Geological Survey river gage at Rkm 34.6. Pearson correlation was used to test the linear relationship between mean weekly movement estimates and mean weekly river stage and temperature for both species during both years.

We calculated habitat selection in the lower 41-km using the individual fish as the sampling unit (Otis and White 1999). Selection within each species, season (spring and summer), and year was quantified separately. To determine how fish were distributed among macrohabitat categories, a likelihood chi-square test was conducted to test the null hypothesis $H_0(1)$: fish locations of each species were uniformly distributed across habitats (e.g., if all four habitats were equally abundant, 25% of fish would be in each). To test selection by each individual for habitat types, we generated $H_0(2)$: the proportion of habitat used by individuals was equal to the proportion of habitat available.

To test the first null hypothesis, we used the equation presented by Manly et al. (2002) if: u_{ij} = the amount of habitat type i used by fish j; u_{i+} = the amount of type i habitat used by all fish; u_{+j} = the total amount of habitat units used by fish j; and u_{++} = the total number of habitat units used by all fish, then the log-likelihood test statistic is

$$X^2 = 2\sum_{j=1}^{n}\sum_{i=1}^{I} u_{ij} \ln \left(u_{ij} / (E(u_{ij}))\right)$$

where $E\,(u_{ij}) = u_{i\,+}u_{+\,j}\,u_{++}$. If the resulting value, with $(I - 1)\,(n - 1)$ degrees of freedom (where I = number of habitat categories and n = number of fish), is large compared to the chi-square distribution, then there is evidence that fish are not uniformly distributed across habitats (Manly et al. 2002).

To determine whether individual fish were selecting or avoiding specific habitat types (second null hypothesis), Manly et al. (2002) proposed the use of the same log-likelihood test statistic, but now $E\,(u_{ij}) = \pi_i\, u_{+\,j}$, with π_i = the proportion of the population of available units in habitat type i. In this case, selection or avoidance is established if the chi-square is large with $n(I - 1)$ degrees of freedom. A p-value determined whether each fish was selective in its habitat choice.

A selection ratio was used to determine the selected habitat type. Because we were interested in the population as a whole, with species per season per year as a population, Manly et al. (2002) estimated the ratio to be $\hat{W}_i = (u_{i+}) / (\pi_j u_{++})$, which is a measure of the proportion of habitat used over that available ($\hat{W}_i$. > 1 is selection; $\hat{W}_i < 1$ avoidance; $\hat{W}_i = 1$ neutral). We calculated Bonferonni 95% confidence intervals around each mean selection ratio to determine whether it encompassed the neutral selection value of 1 (Thomas and Taylor 1990). Pearson correlation tested whether bighead and silver carp habitat selection ratios were linearly correlated.

To determine how the microhabitat (i.e., point of fish location) characteristics chosen by fish within each of the four macrohabitats changed through time, discriminant function analysis (DFA) was conducted on four microhabitat variables (depth, velocity, temperature, and dissolved oxygen) for combined species data. DFA was conducted to account for the different river stage conditions present among years, with the relatively high water of spring-summer 2004 (flood year) and the relatively low water of 2005 during the same period. Only observations that included a value for all four macrohabitat variables were included in the analysis (N=386 observations). Groups were defined by year (2004 or 2005), season (spring or summer), and macrohabitat type (backwater, main channel, channel border, or island side channel). Four discriminant functions were generated for the DFA; the first two were retained in each analysis because they accounted for most of the variance. A structure matrix, where the correlation between each variable and each discriminant function determined the differences between macrohabitat types, ranked habitat variables.

RESULTS

Fish and Physical Conditions. Mean TL and wet weight of tagged bighead carp were 774 mm ± 6 (SE; range 665-856 mm) and 5,657 g ± 159 (range 3200-9500 g). Mean length and weight of tagged silver carp were 740 mm ± 13 (range 538-954 mm) and 5,024 g ± 264 (range 1800-8250 g). High river stages occurred during spring through summer 2004 (mean monthly temperature ^{o}C: April 12.3, May 18.8, June 23.5, July 26.1, August 26.5, September 24.7; Figure 2); low water occurred during this time in 2005 (mean monthly temperature ^{o}C: April 15.4, May 19.4, June 27.0, July 29.6, August 28.6, September 26.9; Figure 2). For those fish tagged in spring 2004, the median detection of tags was 11 months, with about 20% of fish being detected through the study period. The majority (80%) of fish tagged in fall 2004 or spring 2005 were detected through the remainder of our tracking effort.

Movement. In 2004, 25 bighead carp and 21 silver carp were located and thus used to generate movement data. In 2005, 35 bighead carp and 32 silver carp were used. In addition to mobile tracking, 733 locations logged by stationary receivers were used for analysis (2004: bighead carp 235 locations, silver carp 169 locations; 2005: bighead carp 190 locations, silver carp 139 locations).

Across all years and seasons, bighead carp and silver carp were more frequently located upstream of where they were released (mean proportion detections ± 1 SD; bighead carp: upstream = 0.50 ± 0.40, downstream = 0.27 ± 0.35, Swan Lake = 0.23 ±

0.35; silver carp: upstream = 0.60 ± 0.37, downstream = 0.21 ± 0.29, Swan Lake = 0.18 ± 0.33), which is not surprising given that a much greater distance of river was monitored upstream of fish releases (see Figure 1). For the entire 130-km study reach and mobile tracking only, daily rate of movement for bighead carp was similar between 2004 and 2005 (Table 1); with the addition of reach-wide stationary receivers in fall 2004, this estimate increased to 3.6 km/day by 2005 (Table 1). With mobile tracking only, silver carp moved at similar rates between 2004 and 2005 (Table 1). Again, adding stationary receivers increased our estimated rate of movement of this species (Table 1). Including VR2s with mobile tracking also increased our ability to detect total movement of both species, with the maximum distance moved for bighead carp and silver carp being 462 km and 411 km, respectively (Table 1). Evaluating the performance of VR2s solely in the truncated 32-km section in April through June 2005 generated the highest estimates of daily movement (Table 1). Bighead carp in this partial stretch of river moved an average of 6.8 km/day, while silver carp moved 10.6 km/day (Table 1).

Using mobile tracking data only, we found no relationships between abiotic factors and weekly movement for either species. Combining stationary receiver data with mobile tracking revealed that weekly movement (km/week) of bighead carp in 2004 was positively, linearly correlated with river stage, being highest in early summer, ($r = 0.63$; $p = 0.02$; Figure 2), but unrelated to temperature ($r = -0.074$; $p = 0.81$; Figure 2). In 2005, movement of bighead carp was again positively correlated with river stage, being highest in April ($r = 0.62$; $p = 0.042$; Figure 2), and negatively correlated with temperature ($r = -0.59$; $p = 0.06$; Figure 2). Weekly movement estimates for silver carp in 2004 were not correlated with river stage ($r = 0.28$; $p = 0.40$; Figure 2), but were negatively correlated with temperature (r = -0.65; p = 0.030; Figure 2). In 2005, silver carp movement was positively correlated with river stage, again being highest in April (r = 0.75; p = 0.013; Figure 2), but unrelated to temperature (r = -0.47; p = 0.17; Figure 2).

Habitat. Macrohabitat categories in the lower 41-km reach were 28.7% main channel, 41.1% channel border, 7% island side channel and 23% backwater. Data for 35 silver carp and 45 bighead carp were used for habitat use analyses. The remaining fish were never located with mobile tracking. Mobile tracking resulted in 538 locations used for analysis (2004: bighead carp, 179 locations; silver carp 109 locations; 2005: bighead carp, 142 locations; silver carp, 108 locations).

Bighead carp were not uniformly distributed across macrohabitats (test of $H_0$1), except during summer 2005. The second null hypothesis was rejected; bighead carp did select habitat (Table 2). The Bonferroni confidence intervals around the selection ratios for bighead carp in spring 2004 reflected neutral selection among habitats (Figure 3). Conversely, during summer 2004, bighead carp avoided backwater and main channel habitat (Figure 3). Bighead carp avoided main channel habitat in spring 2005 (Figure 3). Bighead carp selected for channel border habitat, and selected against backwater and main channel habitat in summer 2005 (Figure 3).

Silver carp were not uniformly distributed among habitats across all seasons except for spring 2005 ($H_0$1). Fish selected habitats differently among seasons ($H_0$2; Table 2). The Bonferroni confidence intervals around the selection ratios for spring and summer 2004 did not indicate true selection or avoidance for any habitat type (Figure 3). Silver carp avoided main channel habitat and selected for channel border habitat in spring

2005 (Figure 3). Silver carp avoided both backwater and main channel habitat in summer 2005 (Figure 3).

Macrohabitat selection was similar between species; habitat selection ratios were positively correlated between the species (r = 0.60, p=0.01). Consequently, a discriminant function analysis (DFA) combining data between species was justified. For species combined within each macrohabitat type, the DFA correctly classified point-of-location, microhabitat data selected by both species 89.5% of the time within backwater; 76.2% within main channel; 75.1% within channel border; and 82.0% within island side channel. The first discriminant function (F1) ranked depth and velocity as explaining 45.9% of the variance; the second discriminant function (F2) ranked temperature and dissolved oxygen as explaining 38.3% of the variance (84.2% cumulative variance). Summer habitat types were associated with higher temperatures plus lower DO concentrations, while spring microhabitat types were associated with cooler temperatures plus higher DO concentrations (Figure 4; Table 3). Microhabitat within backwaters was shallow with low velocities in both years (Figure 4; Table 3). Channel border and island side channel habitat types of each season were clustered, indicating characteristics of these macrohabitat types were similar (Figure 4; Table 3). Selected microhabitats within channel borders were slightly shallower and had slower velocities than island side channels during spring and summer of both years (Figure 4; Table 3). Main channel microhabitat was consistently deeper with higher water velocities, relative to other habitat types, in both years (Figure 4; Table 3). The proximity of centroids for each group (macrohabitat type, season, and year; Figure 4) indicates that similar microhabitats within each macrohabitat type were used despite markedly different river stages between 2004 and 2005 (see Figure 2).

DISCUSSION

Water levels during spring through summer 2004 were more typical of historical conditions in the Illinois River. In 2005 flooding occurred in late winter (DeGrandchamp et al. in press), and was subsiding by the time we began sampling that year. These conditions allowed us to bracket the movement and habitat selection of both Asian carps under two different environmental scenarios that are likely important to their life histories.

Movement. Movement has two components: directed movement away from the point of capture (i.e. dispersal) and active movement within the area of release. Both kinds of movement varied in intensity among seasons. We captured and tagged the majority of silver carp and bighead carp near Swan Lake in the lower Illinois River. However, individuals moved at least 130 kilometers upstream to the LaGrange Lock and Dam and as far as 80 km downstream into the Mississippi River (Garvey and DeGrandchamp unpublished data). The total extent and rates of movement were similar to those of bighead carp in the LaGrange Pool of the Illinois River, where movement of 23 individuals averaged 1.7 km/d (Peters et al. 2006). The dispersal rate and capacity of bighead and silver carp are comparable to the range and rate of movement of native Mississippi River species including paddlefish (Zigler et al. 2003), lake sturgeon (Knights et al. 2002), and pallid sturgeon (Hurley et al. 1987; Garvey et al. 2007), leading to comparable North American distributions.

River stage should play an important role in the life history of bighead and silver carp. We predicted that movement would peak when river stage was rising. Despite the high water during late spring/early summer 2004 and the low water during the same period in 2005, movement was positively correlated with relatively high river stage within each year. Peak movement was earlier in 2005 when temperatures were still cool in April, corresponding with a receding early winter flood (see DeGrandchamp et al. in press for hydrographs). Thus, an annual rise in river stage may serve as a cue for movement, which is consistent with reports from native waters in Asia (Krykhtin and Gorbach 1981; Abdusamadov 1987).

How temperature affects both short- and long-range movement was less clear. Temperature was negatively correlated with movement for bighead carp in 2004 and silver carp 2005, suggesting that both species move less when their growth optimum of 26ºC is exceeded during summer (Verigin et al. 1978; Krykhtin and Gorbach 1981; Abdusamadov 1987; Jennings 1988). These species are warm-water spawners (≥17ºC). Because fish moved long distances early and at cool (<17ºC) temperatures several months before the purported spawning period in 2005, it appears that peak movement is more closely linked to river stage, regardless of temperature and its importance to reproduction. Indeed spawning did not appear to occur during 2005, likely due to the lack of congruence between the flood pulse and warm temperature (DeGrandchamp et al. in press).

<u>Habitat Selection</u>. The similar patterns of habitat selection between bighead and silver carp in the lower reach of the Illinois River suggest that these species must co-exist

by partitioning resources other than space. Both species seem to have similar reproductive requirements in rivers (e.g., high flow, unimpeded river; see DeGrandchamp et al. in press) and their offspring likely share similar zooplankton resources (J.E. Garvey and A. Lohmeyer, SIUC, unpublished data). However, adults occupy different ecological feeding niches, with bighead carp being zooplanktivorous and silver carp consuming smaller particles such as phytoplankton and fine particulate organic matter (Etnier and Starnes 1993; Laird and Page 1996; Pflieger 1997; Fuller et al. 1999; Williamson and Garvey 2005; Sampson 2005). Thus, these two fishes may coexist in space by consuming different prey.

Tracking demonstrated that adults of both species have specific habitat requirements because individuals did not distribute themselves uniformly across macrohabitats and actively selected and avoided particular macrohabitats among seasons. Both species typically avoided the main channel and only used it in proportion to its abundance during high flow (e.g., the spring-summer 2004 flood) – a period when the channel may be energetically expensive to occupy due to swimming costs. One hypothesis for this pattern revolves around food availability, because the main channel has especially high densities of zooplankton and likely particulate organic matter during high flow (Goodrich 1999; Dettmers et al. 2001, Csoboth 2006). Also, given that adults were moving long distances during high flow, presence in the main channel may have been related to increased local movements among macrohabitats and dispersal from the reach.

During low water, the avoidance of the main channel by adults may have been due to low food availability and perhaps avoidance of frequent barge traffic, which can induce mortality through propeller injuries when water levels are low. Avoidance of backwater macrohabitat by both species, notably during the summer 2005 drought, again may have been related to poor food availability with low water in this habitat. Also, the dominant backwater, Swan Lake, was > 5^{o}C warmer than the river during this time and may have exceeded temperatures ideal for growth (Schultz 2006).

Different river conditions between years produced a wide range of point-of-fish microhabitat conditions within each predefined macrohabitat type, yet both species occupied the same specific microhabitats (i.e., physical conditions) each year. Thus, identifying the particular suite of physical conditions (e.g., low flow, shallow water, and proximity to shore) may also be useful toward directing sampling and control efforts within the larger macrohabitat categories (e.g., side channel borders during summer).

Management Implications. Combining the fixed receivers with our manual, mobile tracking greatly improved our understanding of the great distances rapidly traveled during both years, particularly when flow increased. If managers want to improve detection rates (i.e., increase precision) and better predict dispersal potential, then maintaining the existing stationary receivers and expanding them into uninvaded river reaches would be judicious. Because individuals are capable of much long-range movement, strategies to impede their upstream dispersal, such as the Chicago Sanitary and Ship Canal electrified barrier (Moy 2005) may be justified. Bubble and sound barriers also may deter these fishes from moving further north in the river system (FishPro, Inc. 2004). These barriers would be at greatest risk of being breeched during high flow in spring, regardless of temperature, and would require high vigilance during those times.

Although our research suggests that stationary receivers are necessary for assessing long-range movements as a function of environmental conditions, mobile tracking is necessary for understanding habitat selection and patterns of activity at local scales (e.g., movement among habitats). Quantifying habitat selection is critical for predicting the impact and spread of these and other aquatic invasive species. Targeting Asian carps for harvest within selected macrohabitats at selected areas of establishment such as the lower Illinois River and Swan Lake (e.g., near the channel border in water < 4 m during low summer flow) may aid in greatly decreasing biomass of these species and subsequently inhibiting population growth and dispersal potential.

Currently, management efforts have been aimed at containing Asian carp and preventing further dispersal using barriers (Kolar et al 2005; Conover et al. 2006). Although this research supports the idea that dispersal is not random through time and might be effectively stopped during spring flooding by barriers, it also suggests that management efforts designed to target adults for removal from specific locations also is a viable option that requires further exploration.

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Table 1. Daily rate of movement and total range of movement for bighead and silver carp in the lower Illinois River and Swan Lake during spring through summer 2004 and 2005. Movement was quantified using mobile tracking across the 130 km reach during both years (Mobile only) and using a combination of mobile tracking and stationary receivers (Mobile + Stationary). To determine the impact of stationary receivers on our estimates, we quantified movement in a 32-km reach of the lower Illinois River during 2005 with stationary receivers only (Stationary only).

Species	**Year**	**Reach**	**Mean km/day (±SE)**	**Max km/day**	**Max total km**
Bighead carp	2004	Mobile only	0.21 (±0.05)	4.3	89
		Mobile + Stationary	0.35 (±0.07)	10.0	172
	2005	Mobile only	0.20 (±0.05)	2.1	197
		Mobile + Stationary	3.60 (±0.75)	64.0	462
		Stationary only	6.83 (±1.75)	16.2	60
Silver carp	2004	Mobile only	0.27 (±0.05)	3.3	105
		Mobile + Stationary	0.31 (±0.05)	3.4	112
	2005	Mobile only	0.38 (±0.15)	5.8	219
		Mobile + Stationary	3.18 (±0.71)	64.0	411
		Stationary only	10.61 (±2.65)	29.7	30

Table 2. Likelihood chi-square statistics testing the distribution of Asian carps across gross habitat (i.e., macrohabitat) types ($H_0(1)$), and selection or avoidance of a macrohabitat type ($H_0(2)$) for Asian carps in the lower Illinois River. *n.s.* = $p > 0.05$. Macrohabitat types were main channel, channel border, island side channel, and backwater.

		Distribution ($H_0(1)$)			Selection ($H_0(2)$)		
ear	Season	χ^2	df	p-value	χ^2	df	p-value
2004	Spring	134.1	63	0.001	145.9	66	0.001
	ummer	72.2	39	0.001	106.5	42	0.001
2005	Spring	121.4	66	0.001	163.5	69	0.001
	ummer	48.2	36	*n.s.*	101.5	39	0.001
2004	Spring	60.7	42	0.03	66.8	45	0.02
	ummer	49.5	33	0.03	51.5	36	0.04
2005	Spring	49.1	45	*n.s.*	65.8	48	0.04
	ummer	51.8	33	0.02	109.8	36	0.001

Table 3. Microhabitat (i.e., point of location) features [depth (m), velocity (m/s), temperature (ºC), dissolved oxygen (mg/L)] used by bighead and silver carp in the lower Illinois River and Swan Lake across all fish locations during spring and summer 2004 and 2005.

Species	**Season**	**Habitat Variable**	**Minimum**	**Maximum**	**Mean**
Bighead carp	Spring	Depth	0.5	13.7	4.0
		Velocity	0.0	0.7	0.2
		Temperature	5.6	25.7	16.1
		Dissolved oxygen	3.4	19.1	9.9
	Summer	Depth	0.9	8.5	4.1
		Velocity	0.0	1.0	0.2
		Temperature	22.8	31.6	27.0
		Dissolved oxygen	2.3	13.5	6.0
Silver carp	Spring	Depth	0.5	8.3	3.8
		Velocity	0.0	0.6	0.2
		Temperature	5.9	26.5	17.7
		Dissolved oxygen	3.4	18.5	9.0
	Summer	Depth	0.8	9.1	3.9
		Velocity	0.0	1.2	0.2
		Temperature	21.7	32.0	27.1
		Dissolved oxygen	2.2	13.5	6.4

FIGURE CAPTIONS

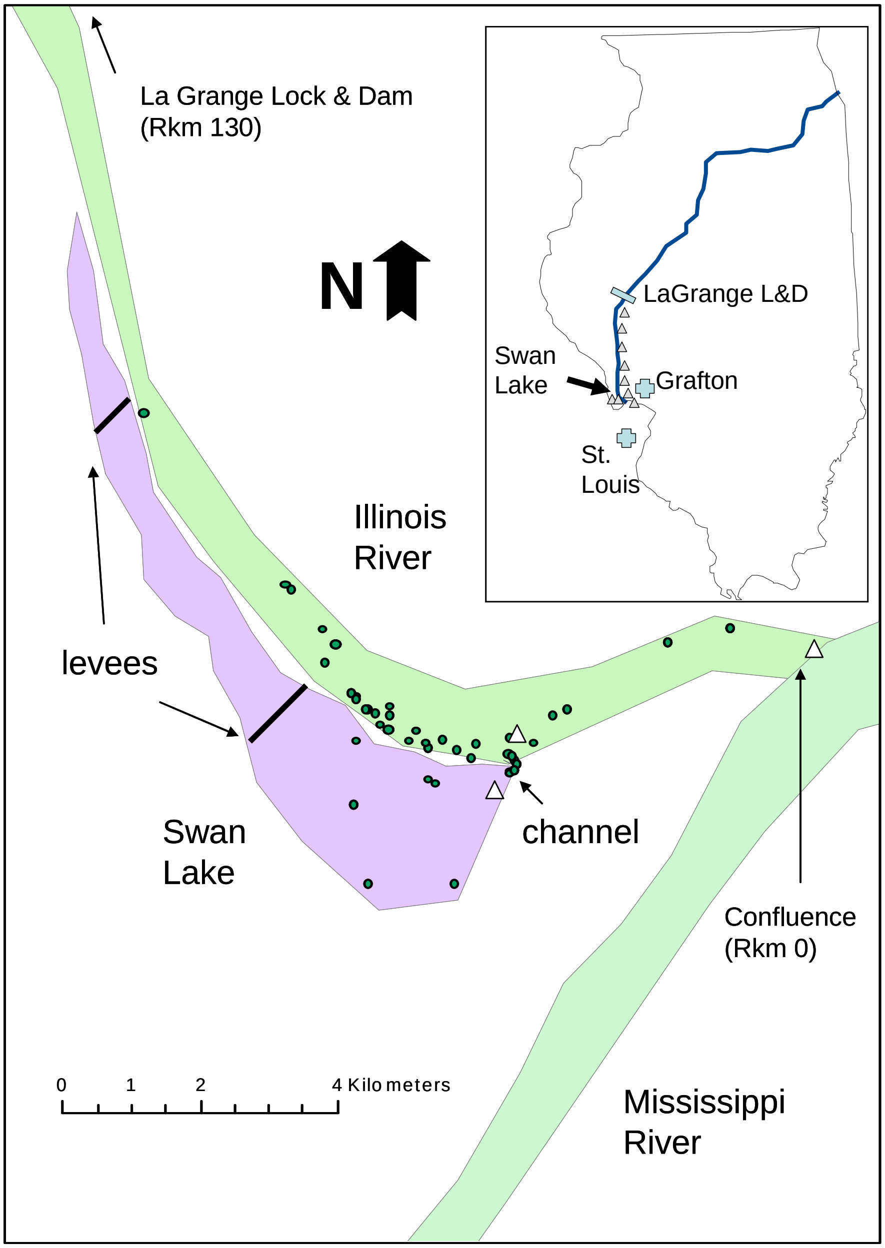
Figure 1. Map of lower Illinois River and the associated Swan Lake backwater where bighead carp and silver carp were captured, implanted with ultrasonic transmitters, and released (overlapping points) during 2004 through 2005. Triangles are locations of stationary ultrasonic receivers.

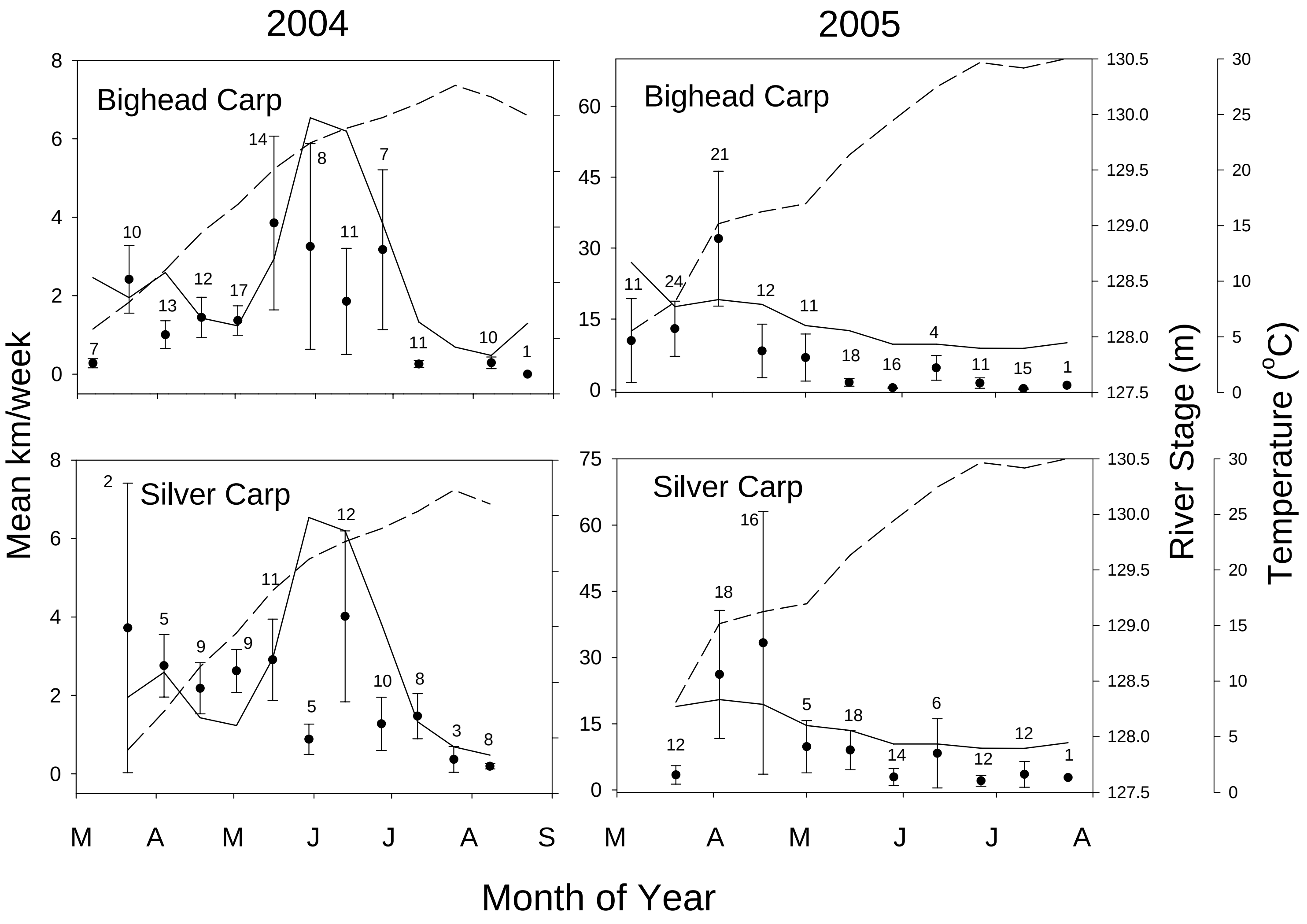
Figure 2. Mean weekly river stage (m, solid line), mean weekly water temperature (^{o}C, dashed line), and mean ± 1 SE rate of movement of bighead carp and silver carp in the lower Illinois River and Swan Lake during 2004 through 2005. Numbers at the top of error bars indicate tagged fish sample size for each period.

Figure 3. Mean habitat (BW = backwater, MC =main channel, CB = channel border, ISC = island side channel) selection ratios by bighead carp and silver carp in the lower Illinois River spring and summer 2004 and 2005. Dotted line at Wi=1 equals no selection. Points above line indicate selection for each habitat type (selection); points below line indicate avoidance of habitat (avoidance). Error bars are ± 95% Bonferroni confidence intervals. (Bighead carp: Spring 2004, N = 22 individuals. Summer 2004, N = 14. Spring 2005, N = 23. Summer 2005, N = 13). (Silver carp: Spring 2004, N = 15 individuals. Summer 2004, N = 12. Spring 2005, N = 16. Summer 2005, N = 12).

Figure 4. Discriminant function ordination of four macrohabitat types in the lower Illinois River for the spring (SP) and summer (SU) of 2004 (04) and 2005 (05): backwater (BW; grey), channel

border (CB; striped), island side channel (ISC; white), and main channel (MC; black). Points indicate group centroids; mean N=26 observations per centroid. N=386 observations.




2004
2005
Bighead Carp
Silver Carp
Mean km/week
River Stage (m)
Temperature (°C)
Month of Year
0
2
4
6
8
0
15
30
45
60
75
127.5
128.0
128.5
129.0
129.5
130.0
130.5
0
5
10
15
20
25
30
M
A
M
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J
A
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24
21
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16
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16
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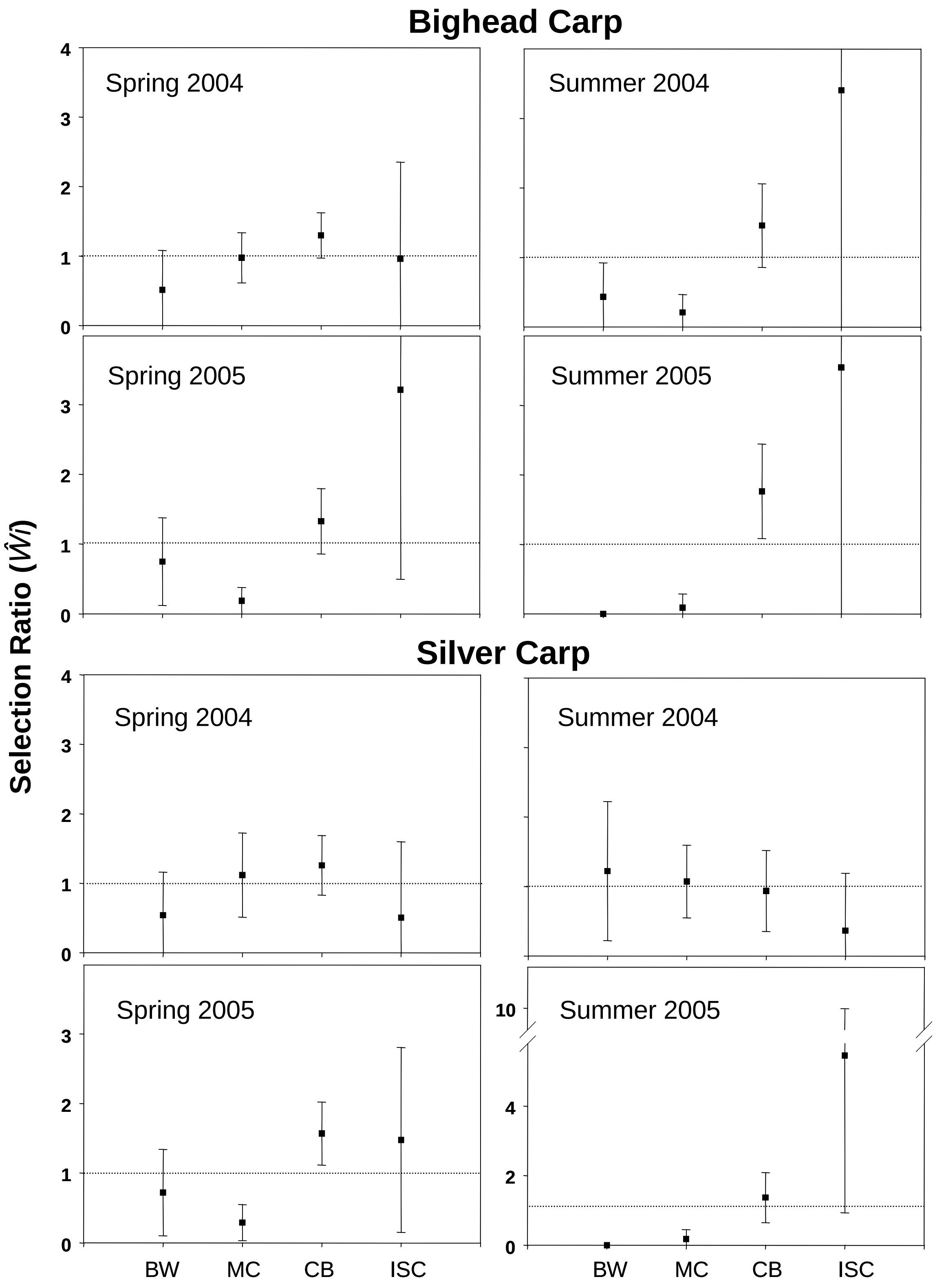
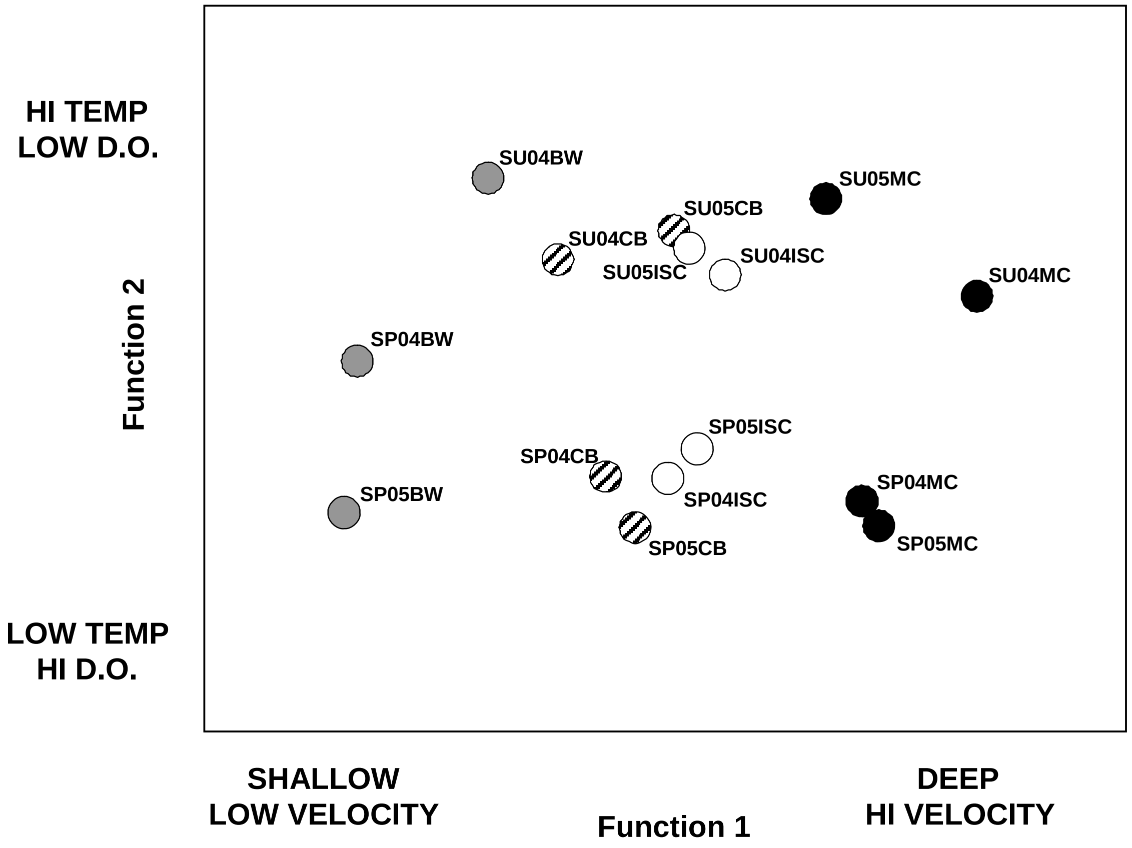
18
14
6
12
12
1


Bighead Carp
Spring 2004
Summer 2004
Spring 2005
Summer 2005
Silver Carp
Spring 2004
Summer 2004
Spring 2005
Summer 2005
Selection Ratio (Ŵi)
0
1
2
3
4
10
BW
MC
CB
ISC


HI TEMP
LOW D.O.
LOW TEMP
HI D.O.
Function 2
SU04BW
SU05MC
SU05CB
SU04CB
SU05ISC
SU04ISC
SU04MC
SP04BW
SP05ISC
SP04CB
SP04MC
SP04ISC
SP05BW
SP05CB
SP05MC
SHALLOW
LOW VELOCITY
Function 1
DEEP
HI VELOCITY

Section 4

FISH REPRODUCTION: DRIFT

SUMMARY

Exchange of larval fishes between a river's main channel and its floodplain backwaters is compromised by sedimentation. Restoration projects to reduce sediment loading are being implemented in large rivers of the midwestern United States to curb backwater habitat loss and restore backwater-river connectivity. During 2004 and 2005, drift nets were set bi-directionally within a constructed channel between the Illinois River and an adjacent, 1,100-ha restored backwater, Swan Lake, to investigate the interplay between life history strategies and lateral drift on a diel and seasonal basis. Ambient larval density and species composition within the river and backwater also were quantified. Drift was positively correlated with water velocity during 2004, and an estimated 32.3 million larvae drifted at the surface of the channel into Swan Lake. In the absence of a flood in 2005, the density and composition of the larval fish assemblage in Swan Lake and the Illinois River appeared to drive larval drift timing, magnitude, and composition. Swan Lake's restoration has maintained some river connectivity and lateral drift functionality for resident fishes but its relationship to natural, connected river-backwater systems remains unknown.

Introduction

Propensity to drift in streams and rivers is an evolutionary response in aquatic organisms (Eckblad et al. 1984; Kennedy and Vinyard 1997). In fishes, drift is usually restricted to early life stages, such as seasonal larval fish drift in lotic systems and largely influences spatiotemporal patterns in larval density (Holland 1986). Drift in riverine larval fishes is typically downstream, but also can transport larvae laterally into slackwater habitat, where these fish may be exchanged between river and off-channel habitats (Humphries et al. 1999). This exchange has been hypothesized due to ichthyoplankton (hereafter termed larvae) density differences among slackwater and river habitats (Sheaffer and Nickum 1986; Brown and Coon 1994; King 2004) and due to downstream larval drift from tributaries to main channel areas (Eckblad et al. 1984; Muth and Schmulbach 1984). However, the extent of lateral larval drift into and out of backwaters relative to downstream drift in rivers is currently undocumented and may be valuable in determining origins and destinations of larval fishes.

We define lateral drift as the movement of organisms between a river's main channel and its adjacent floodplain waters. When connected to the river, these backwaters provide a lentic-lotic gradient whereby fish and other aquatic organisms find spawning grounds, structural refuge, food resources, and overwinter habitat (Junk et al. 1989). Slack-water areas have been recognized as a critical component in the early life history of fishes (Holland 1986), but the biological interplay these habitats have with the main channel is largely unknown. These areas may be a productive source of age-0 fishes given the higher larval densities downstream of backwater outflow (Sheaffer and Nickum 1986) and large migrations of juveniles to the main channel from backwaters (Molls 1999). However, the contribution of river larvae to backwaters may also be significant and a vital process bringing riverine spawned larvae to productive nurseries (de Graaf et al. 1999). Due to the nature of these backwater-river confluences, with periods of inflow to the backwater, outflow to the river, and stagnant waters following high water events, larval exchange is likely complex (Brown and Coon 1994). These flow patterns at the confluence are apt to drive the ability of backwaters to function as nursery habitat for larval and juvenile fishes (Brown and Coon 1994).

Most investigators have focused on patterns of downstream drift, but not in the context of how timing and behavior may shape trends in lateral exchange. Fish spawning behavior dictates temporal patterns in larval drift, where drift duration increases with the duration of spawning (Reichard et al. 2001), and rising water temperatures and spring flood events that cue spawning often result in peak larval drift densities (Carter et al. 1986; de Graaf et al.1999; Auer and Baker 2002). Depending on the species, larval metamorphosis into exogenously feeding fish can either initiate or cease drift behavior (Carter et al. 1986; Dudley and Platania 2000), or have no influence on the species' presence or absence in the drift (Auer and Baker 2002). Downstream drift probably allows access to nursery habitats with amenable growing conditions, and scatters the cohort, which may reduce competition for food and space as well as reduce conspicuousness to predators (Bardonnet 2001). Thus, the interplay of abiotic and biotic factors determines the timing, duration, and taxa of fish drifting, but their influence on lateral exchange is unknown.

Larvae have diurnal, nocturnal, or crepuscular drift patterns that depend on species developmental stage (Gale and Mohr 1978; Muth and Schmulbach 1984; Gadomski and Barfoot 1998). These photokinetic responses may change in some species as larvae age (Bulkowski and Meade 1983), possibly a result of ontogenetic diet changes or susceptibility to or avoidance of predation. However, diel patterns in drift are contradictory, where fish species of the same life stage exhibit significant nocturnal drift in some water bodies and diurnally drift in others (Muth and Schmulbach 1984; Smith and Brown 2002).

Management decisions aimed at improving nursery and spawning habitats and survival of early life stages of fish may fall short due to gaps in our understanding of lateral larval drift timing and behavior. All riverine fishes exhibit certain life history strategies based on their relative dependence on flowing water. Some are specialized for riverine environments and adapted for downstream drift, others require flowing water for a portion of their life history where habitat changes likely entail lateral movement; and finally generalists reside mostly in lentic areas and should be less prone to entering the drift. We investigated the interplay of life history strategies with lateral drift dynamics on a diel and seasonal basis within a restored backwater lake, which enabled us to investigate the role of backwaters as larval fish nurseries in large rivers and to address the influence of habitat alterations on larval fish assemblages among backwater lakes and river reaches. Specifically, we (1) quantified bidirectional drift between the restored backwater and its adjacent river both among seasons and within days and (2) related drift patterns to abiotic conditions (e.g., temperature, flow) and ambient larval densities in the backwater and river.

Methods

Study Area

Swan Lake, a 1,100-ha Illinois River (ILR) backwater located between ILR kilometer 8 and 21, is vitally important for fishes of the Illinois and Mississippi Rivers (USACE 1991). It is the largest connected backwater lake for more than 100 km on the Illinois River and within 30 km upstream or downstream on the Mississippi River and was rapidly losing connectivity due to sedimentation. A Habitat Rehabilitation and Enhancement Project (HREP), through the federal Environmental Management Program (EMP), was initiated to improve connectivity and increase habitat heterogeneity (USACE 1991). The downstream portion of the backwater (Lower Swan [LS], 567 ha) was managed to be continuously connected to the ILR through a water control structure. Historically, Swan Lake was connected to the ILR through a 0.5-km wide opening at LS, and during flood events, had multiple lateral connections with the river. Restoration of the backwater complex restricted the river connection to the width of a stop-log water control structure, about 5-m wide, which was nested within a rip-rap lined channel (ca. 50 m long x 12 m wide). The backwater's natural levee with the river was heightened to stabilize water levels for wetland plant growth and to give managers more control over water level fluctuations. During normal pool stage, the stop-log structure at LS is the only avenue through which larval fish movement between the backwater and river occurs, making the stop-log structure the focal point of our study site.

Fish Sampling

To quantify bidirectional larval drift between LS and the ILR, three conical drift nets (0.5 m x 2 m, 500-µm mesh) were attached to a floating, anchored PVC frame and fished during late March through July 2004, and March through August 2005 (Figure 1). We sampled larvae for 15 minutes at the surface (approximately one-third channel depth) every two weeks on the lake-side of the LS stop-log structure (Figure 1). During each sampling event, nets were set either facing the lake or the channel and then the direction was changed. Two directional net sets, one sampling larvae potentially moving into LS and one set sampling larvae leaving LS, were conducted at dawn, mid-day, dusk, and mid-night within 36 hours. During flooding, sampling frequency was increased to weekly, but fewer night sets occurred.

At each sampling time (e.g., dawn, mid-day, dusk, mid-night), surface water temperature (^oC) and dissolved oxygen (mg/L [YSI Model 52 Dissolved Oxygen Meter; Yellow Springs Instruments, Yellow Springs, Ohio, USA]), secchi depth (cm), and surface water velocity (m/s [Flo-Mate Model 2000, Marsh McBirney, Inc., Frederick, Maryland, USA]) and average wind speed and direction (km/h [Kestrel 1000, NFS – Radiation Protection Systems, Inc., Groton, Connecticut, USA]) were quantified. A Doppler bi-directional velocity meter (Model 6526-51 Starflow; Unidata America, Lake Oswega, Oregon, USA), anchored to the bottom of the water control structure, recorded temperature (^oC), depth (mm), and mean velocity (mm/s) twice an hour. River stage data from Hardin, Illinois were obtained from the U.S. Army Corps of Engineers. No Swan Lake depth data were collected during September through November 2004, and no velocity data were collected during October 2004 through April 2005.

Upon completion of each drift net set, contents were flushed into the cod end and preserved in 95% ethanol. Each sample was split to approximately 200 fish using a Folsom plankton splitter (Aquatic Research Instruments, Hope, Idaho, USA). All age-0 fish were counted, identified to the lowest possible taxon, typically genus, and classified to a developmental stage (i.e., yolk-sac, larval, juvenile) using descriptions and keys in Soin and Sukhanova (1972), Auer (1982), Murty et al. (1986), Tweb et al. (1990) and voucher specimens from Southern Illinois University's Fluid Vertebrate Collection (B. Burr, Carbondale, Illinois, USA) and Colorado State University's Larval Fish Laboratory (D. Snyder, Fort Collins, Colorado, USA). For each net, a subsample of ten fish from each taxon and stage identified was measured (total length [TL]; mm) using Scion Image® software, which was calibrated to 0.5 mm, or metric calipers (*N* = 10 fish per taxon per life stage per net).

Larval densities also were quantified in LS and the ILR to characterize the taxa and abundance within each site and compare larval densities to drift. We used paired, bow-mounted ichthyoplankton nets (0.5 m diameter x 2 m long, 500-µm mesh) to sample these habitats on the same dates as drift nets were set. Four randomly stratified chosen transects within LS and two randomly chosen transects in the ILR (river kilometer 8.0) were sampled weekly. We stratified backwater transects into two inshore and two offshore tows, and river tows were conducted within 1 km upstream and 1 km downstream of the LS-ILR confluence.

At each transect, nets were towed at the surface for 5 minutes at a constant speed, with a calibrated mechanical flow meter (Model 2030R, General Oceanics, Inc., Miami, Florida, USA) mounted in the mouth of one net to standardize sampling effort (i.e., volume sampled). Inshore backwater tows followed the shoreline; offshore backwater tows were straight transects; and river tows were straight transects conducted perpendicular to flow direction. River tows started at the main channel border, continuing across to the opposite main channel border. If 5 minutes had not passed, the direction was reversed with nets still in the water, and sampling continued until 5 minutes was reached. Tow samples were preserved, processed, and identified as with drift net samples.

Habitat Use Classes

Fish collected were grouped by family into one of three generalized classes to better explain trends observed between years and treatment groups (Galat and Zweimuller 2001). The classes were as follows: fluvial specialists, fluvial dependent, and macrohabitat generalists.

Fluvial specialists are fish that inhabit streams and rivers throughout their entire life and rarely enter floodplain habitats (Galat and Zweimuller 2001). Fluvial dependent species regularly use lentic backwater or reservoir habitats, but certain life history traits depend on lotic environments (Galat and Zweimuller 2001). These species are typically broadcast spawners, where developing eggs and larvae are semi-boyant and passively drift in wind-induced or downstream currents (Holland 1986). Adult fluvial dependent fishes also may make lateral migrations into slow-flowing lentic areas to spawn—activities likely corresponding with increasing temperatures and rising water levels (Junk et al. 1989). Macrohabitat generalists include species commonly found in reservoirs and off-channel habitats that do not depend on lotic systems (Galat and Zweimuller 2001). When these fishes use the river, it is typically as a corridor to move among backwaters (Junk et al. 1989; Dettmers et al. 2001). Spawning usually occurs in off-channel habitats and offspring generally do not leave this habitat until the juvenile stage (Holland 1986). We grouped families based on Galat and Zweimuller (2001; taxonomy from Nelson [1994]), and the only deviation from their groupings was Sciaenidae, which we classified as fluvial dependent based on life history descriptions and strong positive correlations between young-of-year catch and flood pulses from Dettmers et al. (2001) and Koel and Sparks (2002), respectively.

Data Analysis

We investigated directional (i.e., into vs. out of backwater), diel, seasonal, and between-year patterns in drift data, as well as correlations with abiotic factors and ambient larval densities within each habitat. Larval drift reflected larval exchange between the backwater and river and was standardized as a rate (fish per minute). Some fish were caught in nets set opposite to the direction of flowing water when velocities were 0.1 m/s or greater (e.g., frame positioned to catch fish drifting out of LS while water flowed into LS at 0.11 m/s). These fish were not considered to be drifting because they had to actively swim into the net mouth to become entrained there (Scheidegger and Bain 1995) and were removed from data sets before analyzing. Larval tow data at each site were calculated as fish per m^3. Total larval production from tow data was calculated as the sum of weekly densities during each sampling season. Similarly, we summed the rates of larvae drifting both into and out of the channel each year to determine gross movement through time. All data were $\log_{10}(x+1)$ transformed to meet assumptions of normality.

Two-way repeated measures ANOVA (proc MIXED, SAS Institute 1999) tested for differences among treatments sampled over time (Hurlbert 1984): (1) bi-weekly drift catch rates into and out of the backwater, (2) bi-weekly drift catch rates at dawn, day, dusk, and night, and (3) weekly mean densities of tow data between sites were compared.

We also determined overall trends within and between years for larval densities. Non-zero catch rates of drift data for each direction and year were regressed (multiple regression, proc REG, SAS Institute 1999) against environmental parameters collected from the Doppler device corresponding to the time and date of drift (e.g., depth, temperature, velocity), and for significant relationships, environmental data were used to estimate the total amount of larval drift for each season. A one-way ANOVA design tested how larval density and catch rates differed between years. To control for experimentwise error rates, we used a Tukey-Kramer post-hoc test (Sokal and Rohlf 1995). Relationships between larval production and lateral drift were determined using linear regression on untransformed data.

Sizes of fish may have differed between tow sites, drift direction, year, or among diel time points. Proportions of fish per 1-mm total length group were used to compare treatments.

Kolmogorov-Smirnov tests compared (1) size distribution of fish drifting into versus out of LS in each year, (2) total size structure of fish drifting during 2004 and 2005, (3) size structure of fish collected from tows, and (4) between year differences for LS and ILR tow data. Multiple pairwise comparisons determined size structure differences among the times of day sampled (e.g., dawn vs. night, dawn vs. day, dawn vs. dusk, etc.), with Bonferroni adjusted alpha values (alpha = 0.05/6).

Results

Abiotic Patterns

Larval production and lateral exchange were related to the different flood regimes that took place on the lower Illinois River in 2004 and 2005. A moderate flood pulse occurred in 2004, where water levels in the lower Illinois River were above flood stage for approximately five weeks during June (Figure 2). During the 2005 sampling season, water levels remained at or below normal pool level (i.e., 128 m). Water temperatures in the river gradually rose and fell during 2004, peaking in late July at 28 ^{o}C, but varied more during 2005 and rose over 30 ^{o}C twice that season. Lower Swan mimicked the river, although water temperatures were higher and more variable due to shallower water levels (Figure 2).

Water depth and bi-directional velocity in the narrow channel separating LS from the ILR varied markedly between years. Aside from the 2004 flood pulse which increased the channel depth to 3.98 m, channel depth remained above 2 m for most of 2004 and averaged 2.04 m after May 2005 (Figure 3). Movement of water through the LS water control structure frequently changed direction during each day, and velocities varied greatly (Figure 3). This movement was likely influenced by many factors including river stage, river velocity, wind, and barge traffic.

On average water was moving into LS, with 86% of differential velocity values being positive (i.e., into the lake; Figure 3). Mean channel velocities were typically less than 1 m/s during the two sampling seasons, with only three percent of data points greater than or equal to this amount. In 2004, velocities flowing into LS peaked at 2.2 m/s during the rise in flood waters, dropped to near zero at flood crest, and flowed out at peak velocities of 1.5 m/s during the rapid recession of flood waters. Near-zero velocities occurred during low water periods, particularly those occurring during summer 2005, where differential velocity values were tightly positioned around zero (Figure 3).

Seasonal Patterns

Fish occurred in drift nets during 1 May through mid-July 2004, and for a more protracted duration of 26 March through late-August 2005 (Figure 4). Total production of larvae, garnered from summing tow data, began in the ILR during mid-April each year, whereas production in LS began during late-March 2004 and occurred three weeks later the following year (Figure 5). A major pulse of fish drifted into LS during late May 2004 as flood waters rose (Figure 4); a synchronous pulse of larvae occurred in tows during June 2004 during floodwater inundation (Figure 5). During 2005, a smaller, discrete peak of larvae drifted between LS and the ILR, which was 3 weeks later and at less than half the magnitude (Figure 4). In 2005, larvae peaked 2 weeks later in the ILR than in LS (Figure 5).

Mean rates (#/minute) of fish drifting into LS during 2004 varied complexly through time; differences over time and an interaction between drift direction and time occurred (Figure 4; all $P < 0.01$; direction: $F_{1,11} = 18.53$; week: $F_{1,11} = 7.67$; direction*week: $F_{1,11} = 8.2$). Ingress and egress of ichthyoplankton were similar in 2005 (Figure 4; $F_{1,11} = 3.8$, $P = 0.10$). The peak rate of larval movement was higher in 2004 (47.6 larvae/minute) than in 2005 (15.8 larvae/minute; Figure 4). However, mean daily drift rates of larvae by direction did not differ between years (two-way ANOVA: $F_{3,132} = 1.6$, $P = 0.21$), with means of 4.2 fish/minute and 0.5 fish/minute, respectively. The non-significant between-year result was due to high variance within year and similar rates of larvae drifting out of the backwater between years at 0.4 fish/minute each year (Figure 4).

Larval densities in LS were 10 times higher than in the ILR during both years (Figure 5; two-way repeated measures ANOVA: $P < 0.01$; 2004: $F_{1,20} = 66.97$; 2005: $F_{1,18} = 109.78$), although they differed over time ($P < 0.01$; 2004: $F_{21,20} = 23.27$; 2005: $F_{22,18} = 15.11$) and also interacted ($P < 0.01$; 2004: $F_{1,21} = 4.43$; 2005: $F_{1,22} = 3.74$). Separate one-way ANOVAs of larval densities averaged across weeks in 2004 ($P< 0.01$, 2004: $F_{1,138} = 9.56$) and 2005 ($P< 0.01$, 2004: $F_{1,122} = 22.84$) confirmed that densities were higher in LS. Total larval production estimates summed across sampling dates were similar between years in the ILR with 41.5 and 31.0 larvae/m^3, and LS with 434.8 and 530.5 larvae/m^3, during 2004 and 2005, respectively.

Both LS and the ILR had spatially homogeneous larval distributions, with no differences in LS between inshore and offshore (2004: $t_{1,20} = -2.08$, $P = 0.05$; 2005: $t_{1,18} = -0.66$, $P = 0.52$) or in the ILR between upstream and downstream stratified transects (2004: $t_{1,21} = 0.94$, $P = 0.36$; 2005: $t_{1,22} = -0.27$, $P = 0.79$).

No diel patterns in catch rates occurred during 2004 (Table 1; $F_{3,11} = 0.1$, $P > 0.05$), though a diel drift pattern occurred during 2005, where more fish larvae drifted at night than during other times of day ($F_{3,35} = 15.9$, $P = 0.01$).

Grouping families by habitat use class revealed macrohabitat generalists being abundant in tows during both years, while drift taxa differed between years (Table 2, Figure 6). Fluvial dependents were most abundant drifting into Lower Swan during 2004, with sciaenids, clupeids, and cyprinids representing 90% of the fish exchange and each occurring equally (Table 2, Figure 6). Temporal drift patterns varied by habitat use class, with fluvial dependent taxa peaking in a discrete pulse during rising flood waters. Furthermore, fluvial dependent taxa were most abundant in ILR tow samples during 2004, which were five times higher than densities of those taxa in LS. Macrohabitat generalists, comprised mostly of clupeids, also drifted in peak rates during the rising flood waters, beginning in May before the pulse and continuing for two months. Macrohabitat generalists were collected in the drift during 2004 and 2005 without much directional difference, making these taxa some of the only that drifted out of LS (Figure 6).

If LS was contributing macrohabitat generalists to the ILR, we might expect downstream larval composition to differ from the upstream ILR site. This did not occur. Conversely, the ILR did appear to influence drift into LS in 2004 because taxa present in tows in the ILR were comparable to those drifting into LS during this year (Figure 6). During the non-flood year, drift rates and density of fluvial dependents were lower than macrohabitat generalists (Table 2, Figure 6). No fluvial specialists were sampled during either year.

Drift rates were correlated with abiotic and biotic factors. Non-zero catch rates of fish drifting into the backwater during 2004 correlated positively with velocity, but not temperature or channel depth (multiple regression model: adj. $r^2 = 0.92$, $F_{3,13} = 61.83$, $P < 0.01$; Velocity: $t = 13.19$, $P < 0.01$). Using the regression model and continuous velocity data, we extrapolated catch rates to the seasonal duration of drift (i.e., 1 May through 15 July) and channel volume. An estimated 32.3 million fish drifted into LS at the surface during the 2004 sampling season, estimated using a regression equation (mean catch = [33.86 * velocity] – 2.91) and summing mean catch across the channel as predicted by velocity across days. Drift catch rates into LS were weakly positively correlated with ILR tow-derived densities during 2004 ($P = 0.02$, $r^2 = 0.36$). During 2005, abiotic variables were unrelated to drift. However, drift into LS was positively linearly related to ILR larval densities ($P < 0.01$, $r^2 = 0.78$), while LS larval tow-derived densities likely influenced catch rates of drift to the river ($P < 0.01$, $r^2 = 0.94$). Therefore, abiotic factors combined with larval density appeared to influence 2004 larval drift and density to affect 2005 drift.

Size Structure

Larval sizes differed between years and among sites. Similarly sized fish drifted into and out of LS during 2004, although fish drifting out were slightly larger. During 2005, larger fish drifted into the backwater than out (Table 3, Figure 7). However, there were no differences in size distribution of larvae caught in tows upstream and downstream of LS. Higher river and channel velocities during the flood pulse did not entrain larger fish into LS, and, regardless of net direction (i.e., in versus out), larger fish occurred in drift nets during 2005 (KSa = 8.01, $P < 0.01$). Despite this change in size structure, larval sizes remained larger in the backwater than the ILR regardless of year (Table 3, Figure 7). Furthermore, both backwater and river yielded larger larval sizes during 2005 than 2004 (LS: KSa = 2.53, $P < 0.01$; ILR: KSa = 10.55, $P < 0.01$).

Lengths varied with time of day. Multiple pair-wise Kolmogorov-Smirnov comparisons revealed a propensity for larger fish to drift at night (Table 3). In 2004, fish were similarly sized during dawn, day, and dusk (Table 3, Figure 8). Stronger diel patterns occurred during 2005, with night catch collecting the largest fish and dusk, dawn, and day catches each sampling progressively smaller fish (Table 3). Regardless of year, ichthyoplankton drifting at dawn, day, and dusk were predominantly less than 8 mm TL, with cumulative percent frequencies between 74 - 80% for each time period in 2004, and between 56 – 82% in 2005. Almost 70% of larvae during night catch were 8 mm or greater (Figure 8).

Discussion

Seasonal lateral larval fish drift occurred at the restored connection between LS and the lower ILR, with patterns shaped by the annual flood pulse and the fishes' early life history strategies. The flood pulse concept specifies that coupling increasing water levels with temperatures cues spawning and yields high recruitment of riverine fishes (Junk et al. 1989; Harvey 1987). In our study, fish production differed between LS and the ILR. Within the backwater and river, larval densities but not species composition were similar between two physically contrasting seasons, a flood and a non-flood year. Abiotic conditions apparently induced a change in ambient family densities between years, which may have influenced between-year variation in magnitude, direction, and composition of lateral larval exchange. During 2004, the large pulse of larvae drifting into LS was timed to the rising flood waters, and exchange was temporally isolated to that flood event (Reichard et al. 2001). In contrast, peak drift rates in 2005 were less than half that of 2004. Only during 2005 were ambient larval densities positively correlated to drift rates, demonstrating a strong biotic influence in the absence of the spring flood.

Abiotic cues likely initiate and drive larval drift within year as well as influence family composition in the river and drift between years (see Adams 2004). The propensity for fluvial dependents to drift into the backwater during the flood year was likely related to the coupling of temperature and flooding, while the relative absence of these species in the drift and the larval assemblage was related to their decoupling during the non-flood year. A lack of a spring flood pulse in 2005 may have resulted in lower reproduction or recruitment of fluvial dependents. These patterns stress the strong reliance of fishes' early life history on annual flood pulses.

Aside from the influence of abiotic cues on floodplain habitat use, the ecological role of the restored LS, as it contributes to fish reproduction in the river-floodplain, needs to be identified and evaluated. In other systems, slackwater areas contribute larvae and juveniles to the river such that densities downstream of the backwater-river confluence become higher (Muth and Schmulbach 1984; Sheaffer and Nickum 1986). However, we did not see any difference in river densities upstream or downstream of the backwater. Substantial movement of larvae between the backwater and river occurred, even though Swan Lake seemed to neither function as a major sink nor source of larvae. The LS backwater appears to continue to be a vital component of the lower ILR, with potential benefits to the Mississippi River due to its close proximity (USACE 1991). In particular, LS produced a large proportion of clupeids, which are important prey for piscivorous fishes and waterbirds.

Drift rates of macrohabitat generalists suggest a behavioral component to lateral drift, thereby discounting suspicions of LS acting solely as a sink. The dominant taxa in the larval assemblage are commonly found in the drift (Jurajda 1995; Reichard et al. 2001); yet in our study clupeids were less abundant in the 2004 drift than their ambient densities would have warranted. Similarly, during 2005, macrohabitat generalists drifted in lower rates despite their higher densities within the river and LS. This under-representation of macrohabitat generalists in the drift may reflect drift avoidance (Brown and Armstrong 1985; Reichard et al. 2001). Ultimately, these findings imply a purposeful lateral drift pattern, where some families may avoid or are less prone to drift.

Diel and size structure patterns of larval drift occurred, reflecting a behavioral component to drift. Larger larvae drifted at night during both years, while higher catch rates at night only occurred during 2005. Larvae, particularly larger fish, may innately drift at the bottom during the day and move to the surface at night to feed or evade predation (Gale and Mohr 1978; Muth and Schmulbach 1984; Johnston et al. 1995). Carter et al. (1986) captured larvae drifting in densities almost four times higher at night than during the day. Other abiotic factors, like water clarity, likely drove interannual diel variations, where catch rates showed no diel patterns during the flood year and were more apparent during low flow when water transparency was likely higher (Reichard et al. 2001; Araujo-Lima et al. 2001). Therefore, it is possible that diel patterns quantified in LS may result from phototaxic responses and/or changes in water clarity between years.

The tendency for larger fish to drift out of LS during 2004 supports conventional larval drift hypotheses, that backwaters function as nursery habitat, provide ideal conditions for growth, and later become a source for age-0 fish (Sheaffer and Nickum 1986). Lateral movement during 2005 may not have been necessary as a life history strategy given that portions of the lower ILR contained amenable habitat similar to backwaters, with slower flows, warmer habitat, and high plankton densities (Csoboth 2006). Furthermore, abundant juvenile clupeids emigrated from LS into the ILR during summer 2004, with juvenile sciaenid and moronid fish followed this same trend during fall 2004 (Schultz 2006). Thus, it is our thought that larvae entering the backwater or spawned in the backwater were able to feed and grow throughout the season, eventually making an ontogenetic habitat shift by exiting the backwater sometime that fall and recruiting to the river fishery (King 2004).

In restored backwaters, continuous connectivity should be maintained where possible because species use the floodplain throughout the spring and summer. Limiting backwater access or reducing river access could impair the recruitment potential of certain fishes, eventually leading to a less diverse riverine fish community (Turner et al. 1994). The LS connection to the ILR was about 99% wider than its post-restoration state, and likely permitted gradual changes in water level, direction, and velocity. After the restoration, water levels now lag behind the river, flow directions typically change multiple times per day, and velocities are dynamically variable and most likely higher through the restricted channel, especially during flood pulses. Although our data suggest that functionality has been maintained to some extent, it is possible that the narrowed connection (via the levee) and the constructed stop-log structure have negatively altered water flow patterns and lateral exchange of fishes through the structure.

This study was unique, as no other study to our knowledge has quantified lateral larval drift in a river-floodplain system. Lateral drift is an important component of fishes' life history in lotic systems, but these strategies and ontogenetic habitat changes may not withstand the anthropogenic disturbances in streams and rivers. Altered hydrology may affect larval drift ecology and influence recruitment of fishes. Lateral drift patterns in the restored LS-ILR system were strongly influenced by the flood pulse, but in the absence of a flood, the ambient biotic assemblage appeared to drive drift timing, magnitude, and composition. Generally, the Swan Lake HREP appears to have successfully altered the backwater to reduce sediment input while still maintaining some river connectivity and lateral drift functionality. Ultimately, future conservation efforts aimed at restoring hydrology should not focus on one particular habitat, but should equally consider main channel, floodplain, and tributary habitats (Galat and Zweimuller 2001).

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Table 1. Drift of fish during 2004 and 2005 in Swan Lake, Illinois River, with net sets averaged by time period (standard error represents ±1 of mean catch rate).

		N		Catch (#/minute)
Year	Time of day	Net sets	Fish	Mean ± SE
2004	Dawn (0450 – 0711)	24	1,956	1.80 ± 1.76
	Day (1215 – 1500)	24	2,551	2.38 ± 2.00
	Dusk (1815 – 2050)	22	1,137	0.96 ± 0.94
	Night (0000 – 0110)	20	112	0.12 ± 0.08
2005	Dawn (0450 – 0640)	24	490	0.45 ± 0.42
	Day (1145 – 1435)	23	77	0.07 ± 0.05
	Dusk (1815 – 2110)	26	70	0.06 ± 0.04
	Night (2340 – 0110)	20	869	0.93 ± 0.79

Note: Night net sets were not conducted during rising flood waters during 2004, which reduced net set count and likely mean night catch rate.

Table 2. Fish families grouped into one of three habitat use classes (Galat and Zweimuller 2001) with percent of total catch during 2004 and 2005 by gear. Drift net total catch for 2004: *N* = 5,756; for 2005: *N* = 1,506. Tow net total catch for 2004: *N* = 56,476; for 2005: *N* = 60,509. No fluvial specialists were caught in either drift or tow nets.

	2004		2005	
Family	Drift Nets	Tow Nets	Drift Nets	Tow Nets
Fluvial Specialist	0.00%	0.00%	0.00%	0.00%
Fluvial Dependent				
Catostomidae	8.37%	0.33%	1.59%	0.24%
Cyprinidae	28.82%	2.70%	1.00%	4.85%
Moronidae	0.16%	0.18%	0.00%	0.04%
Percidae	0.03%	< 0.01%	0.07%	0.00%
Sciaenidae	31.01%	5.04%	2.19%	0.44%
Macrohabitat Generalist				
Atherinidae	0.00%	0.01%	0.00%	0.19%
Centrarchidae	0.30%	1.82%	0.73%	1.08%
Clupeidae	30.59%	89.84%	92.56%	92.52%
Gasterosteidae	0.00%	< 0.01%	0.07%	0.00%
Ictaluridae	0.00%	0.00%	0.53%	< 0.01%
Lepisosteidae	0.07%	0.01%	0.00%	0.00%
Poeciliidae	0.03%	0.01%	1.20%	0.59%

Table 3. Kolmogorov-Smirnov tests for 2004 and 2005 pooled diel drift net data and Lower Swan Lake (LS) and the Illinois River (ILR) larval tow data, where test results are listed for treatments which collected significantly larger-sized fish.

	2004			2005		
Treatment	KSa	*P*	Result	KSa	*P*	Result
IN vs. OUT	10.91	< 0.01*	*Out*	4.41	0.01*	*In*
LS vs. ILR	8.83	< 0.01*	*LS*	5.37	< 0.01*	*LS*
NIGHT vs. DAWN	9.48	< 0.001*	*Night*	7.78	< 0.001*	*Night*
NIGHT vs. DAY	9.57	< 0.001*	*Night*	11.99	< 0.001*	*Night*
NIGHT vs. DUSK	10.93	< 0.001*	*Night*	2.55	< 0.001*	*Night*
DAWN vs. DAY	0.56	0.914	--	9.70	< 0.001*	*Dawn*
DAWN vs. DUSK	2.03	< 0.001*	*Dusk*	6.06	< 0.001*	*Dusk*
DAY vs. DUSK	1.48	0.026	--	4.74	< 0.001*	*Dusk*

* Indicates significant differences between treatments, where $P \leq$ *or* ”.

Figure Captions

Figure 1. Fixed drift net site during 2004 and 2005. Three tandem nets were floated on the Lower Swan (LS) side of the stop-log structure. Schematic depicts nets sampling ichthyoplankton drifting out of LS. Inlay portrays the drift net frame positioned to sample ichthyoplankton drifting into LS.

Figure 2. Mean daily temperature of the Illinois River and Lower Swan for 2004 and 2005. Mean daily depth data are depicted as solid black lines. Channel depth of Lower Swan was recorded at the stop-log structure between the backwater and the river. River stage and temperature data were collected at Hardin, Illinois, USA. Shaded regions represent sampling periods.

Figure 3. Mean daily depth (top), average velocity (middle), and differential velocity (bottom) in the channel between Swan Lake and the Illinois River. Average velocities are averaged across daily observations (N=48 per day) and direction was not incorporated. Differential velocity was calculated as the sum of channel velocities per day (i.e., the sum of 48 values). Positive velocity values represent net inflow of water into Swan Lake and negative values correspond to net outflow to the river. Continuous data were recorded using a submerged device anchored at the stop-log structure. No depth or velocity data were collected from September through November 2004, and no directional velocity data were collected from October 2004 through April 2005. Shaded regions represent the 2004 and 2005 sampling periods.

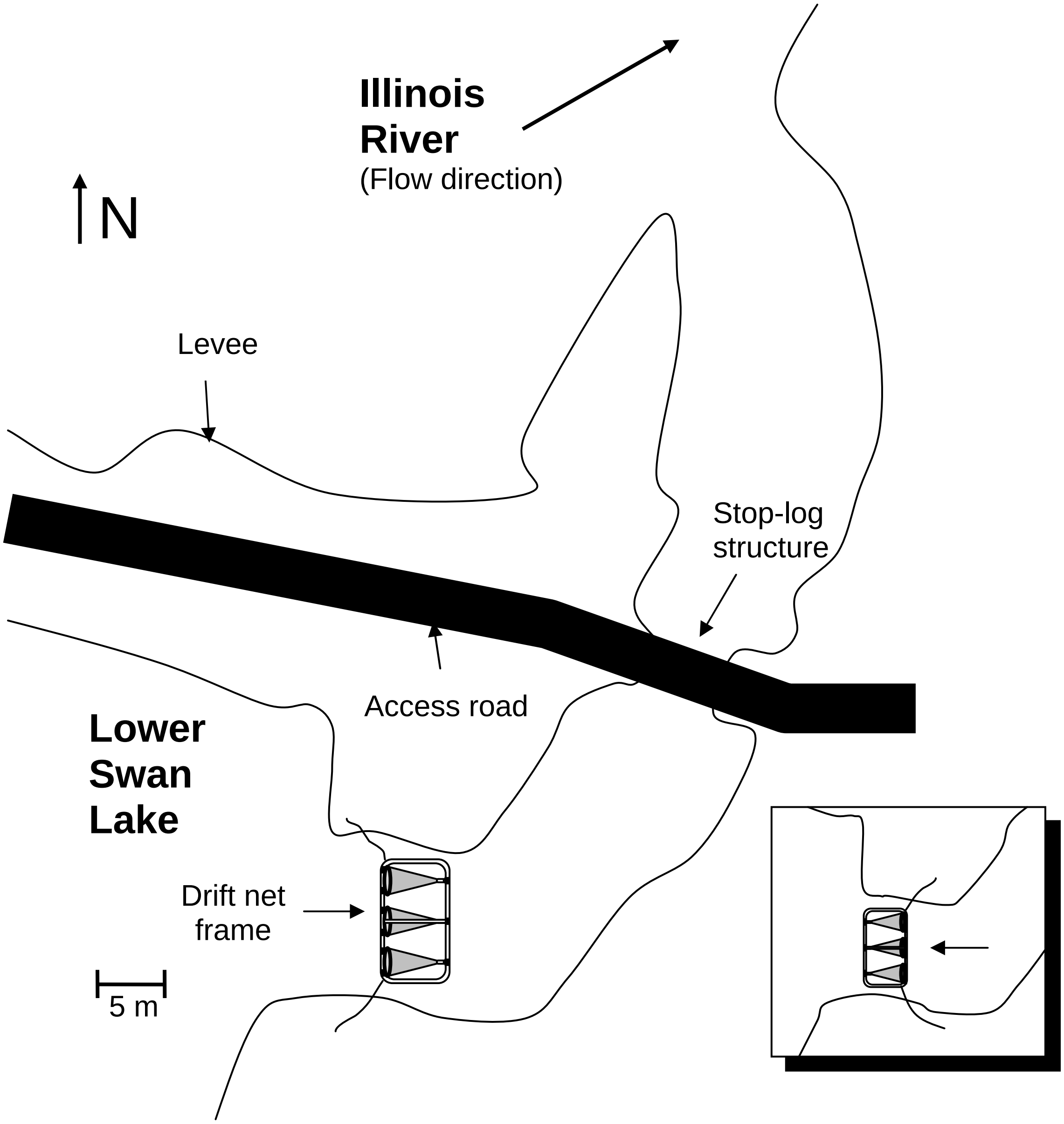
Figure 4. Drift of ichthyoplankton into (●) and out (○) of Swan Lake in 2004 and 2005. Symbols represent mean catch (#/minute) per net set and line data represent continuous depth data recorded at the stop-log structure. Error bars represent ±1 standard error for the mean of each net set.

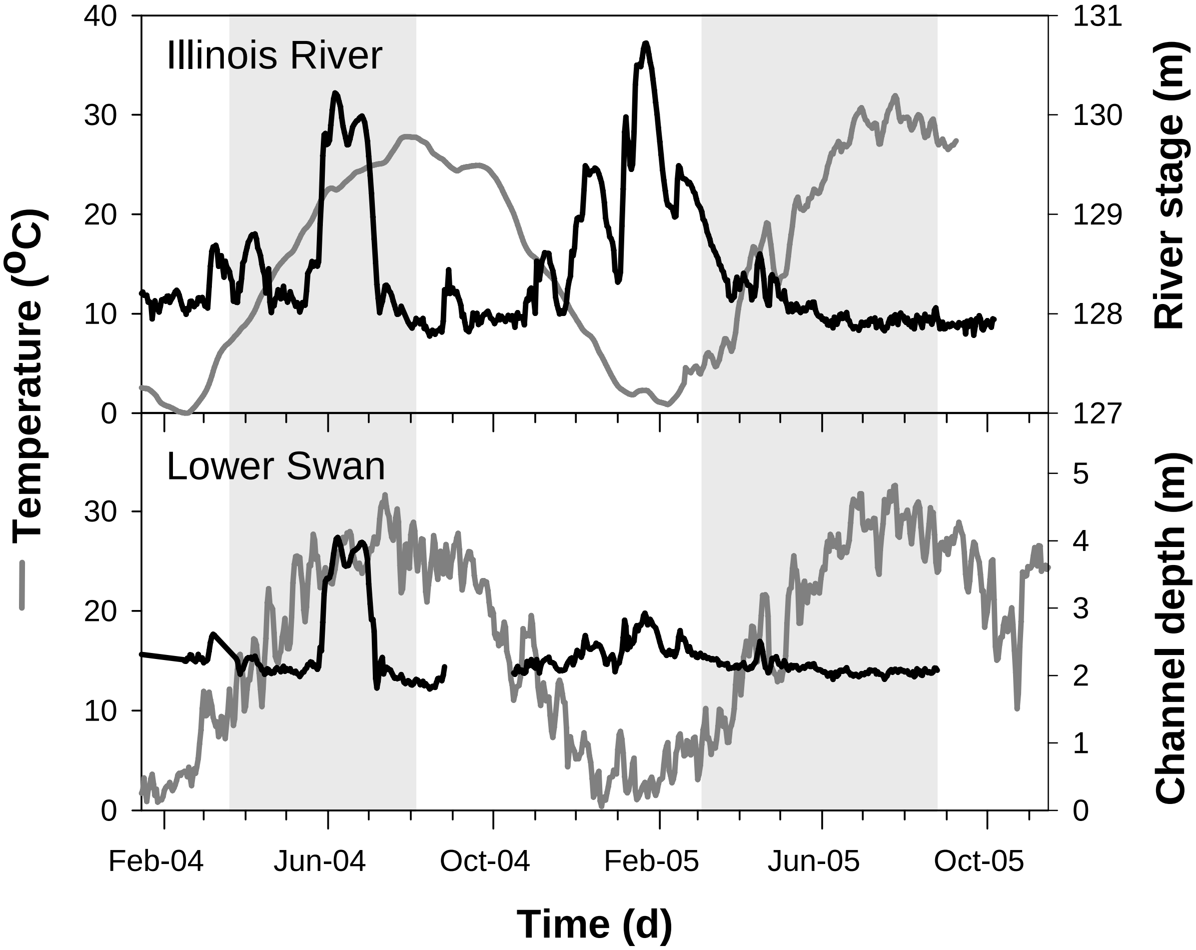
Figure 5. Mean density (#/m^3) per day of fish caught in the Illinois River and Swan Lake during 2004 and 2005. Error bars represent ±1 standard error of the mean transect density. Note change in scale for Lower Swan.

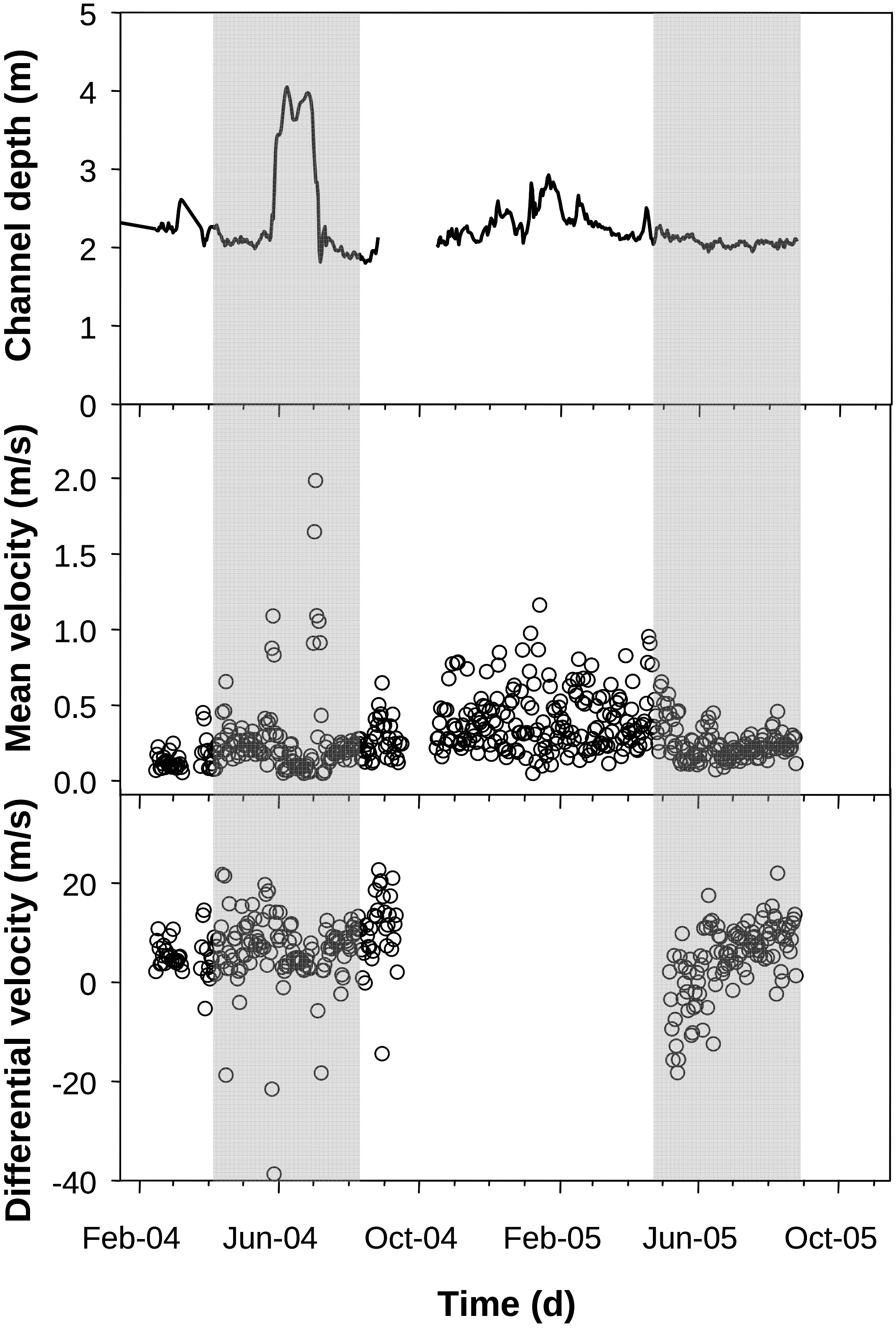
Figure 6. Mean catch rate (#/minute) of families drifting into and out of Swan Lake and mean density (#/m^3) of families in the Illinois River and Swan Lake grouped by habitat use classes during 2004 and 2005. All families were included and grouped into one of three habitat use classes as defined by Galat and Zweimuller (2001): fluvial specialists (FS), fluvial dependents (FD), and macrohabitat generalists (MG). Error bars represent ±1 standard error of the mean.

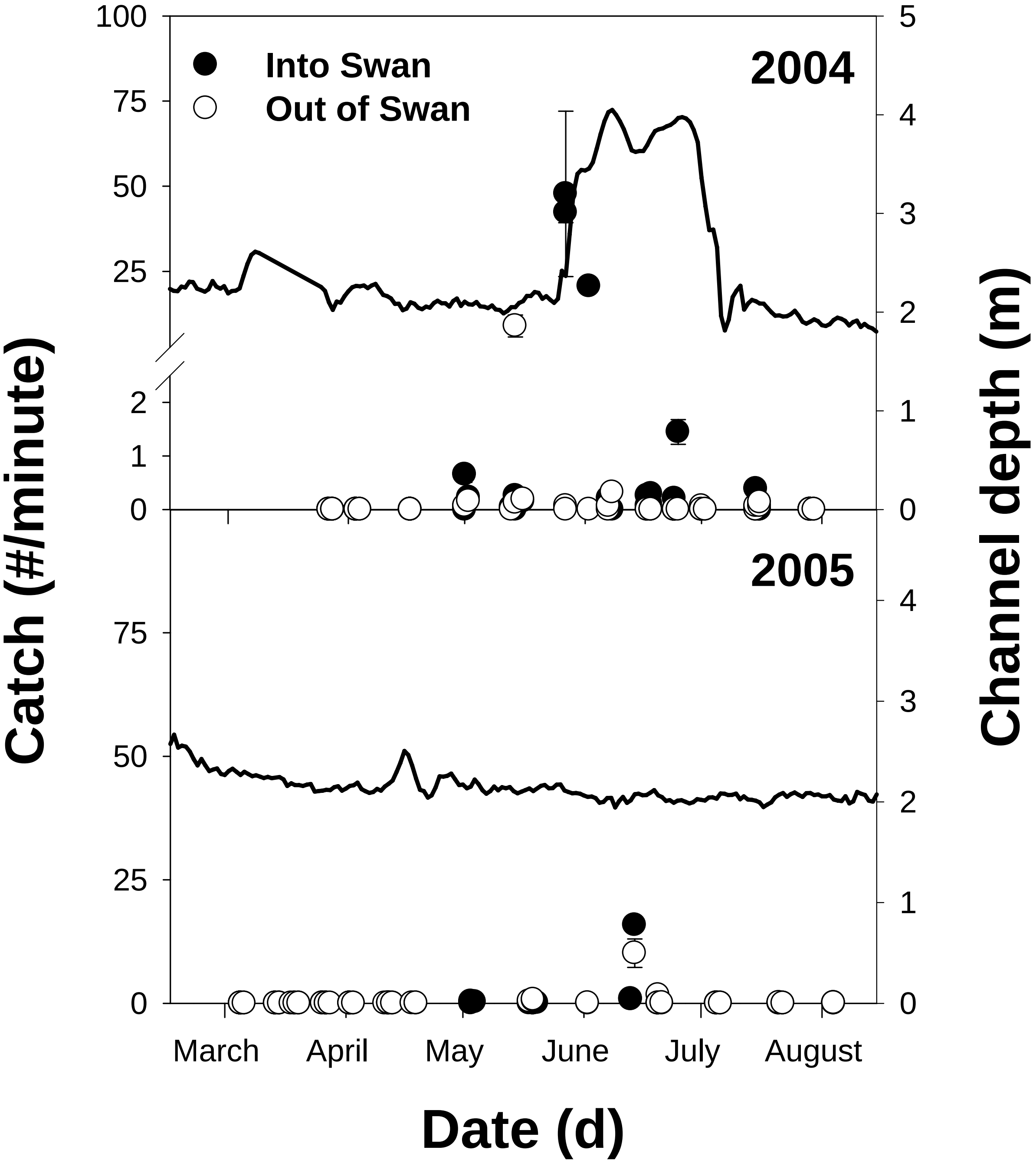
Figure 7. Length frequency distributions of all larval and juvenile fish caught in the Illinois River, Lower Swan, and in the drift during 2004 and 2005. Length frequencies are expressed as a percentage of the average caught. Descriptive statistics were calculated on raw length data.

Figure 8. Length frequency distributions of all fish caught drifting during each time of day for 2004 and 2005. Data of fish drifting into and out of Lower Swan were pooled by time of day (i.e., dawn, day, dusk, night). Length frequencies are expressed as a percentage of the mean number per net set (i.e., mean of three nets) summed over the sampling season. Descriptive statistics were calculated on raw length data.

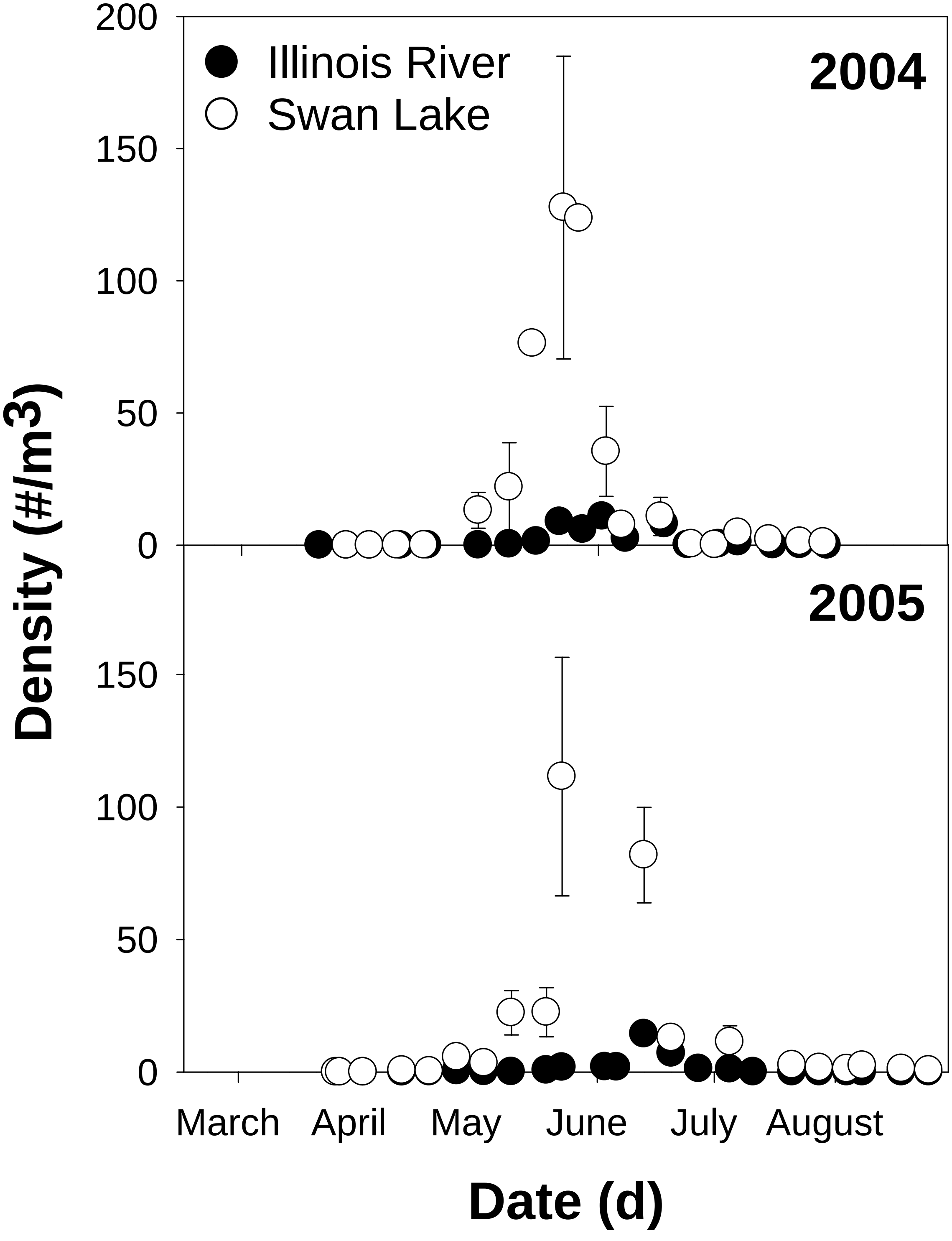

Illinois River
(Flow direction)
N
Levee
Stop-log structure
Access road
Lower Swan Lake
Drift net frame
5 m

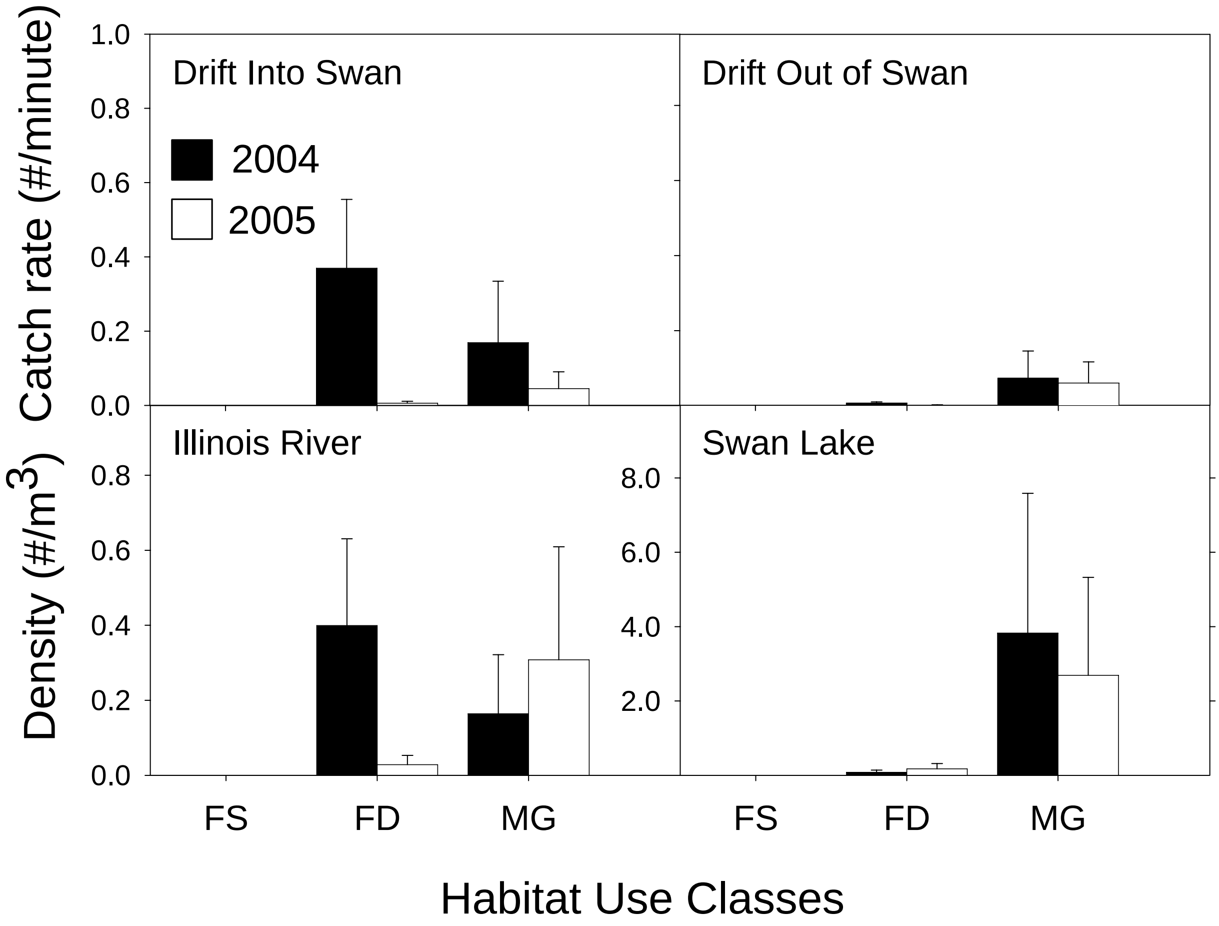

Illinois River
Lower Swan
— Temperature (oC)
River stage (m)
Channel depth (m)
Time (d)
40
30
20
10
0
131
130
129
128
127
5
4
3
2
1
Feb-04
Jun-04
Oct-04
Feb-05
Jun-05
Oct-05

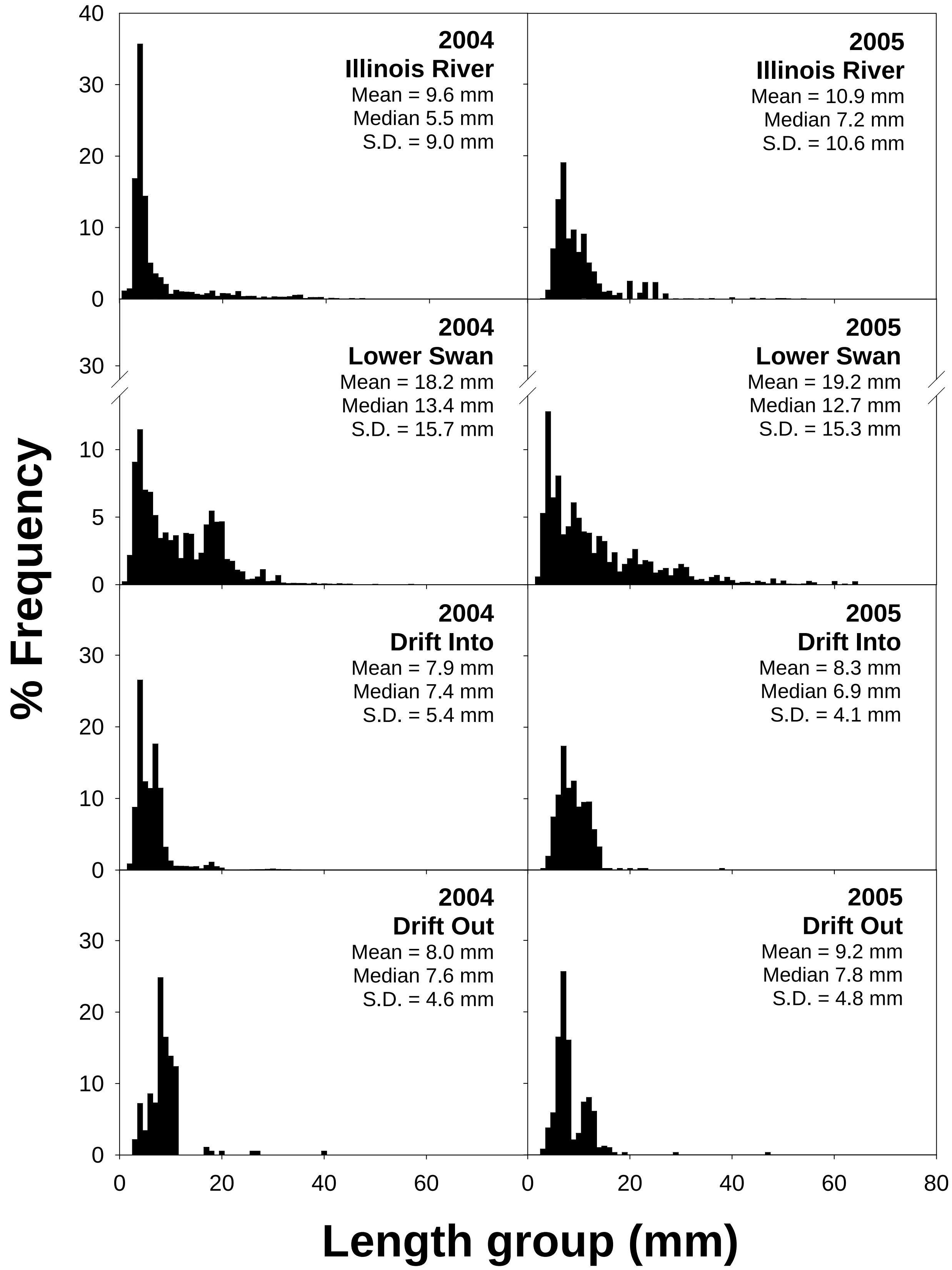

Channel depth (m)
5
4
3
2
1
0
Mean velocity (m/s)
2.0
1.5
1.0
0.5
0.0
Differential velocity (m/s)
20
0
-20
-40
Feb-04
Jun-04
Oct-04
Feb-05
Jun-05
Oct-05
Time (d)



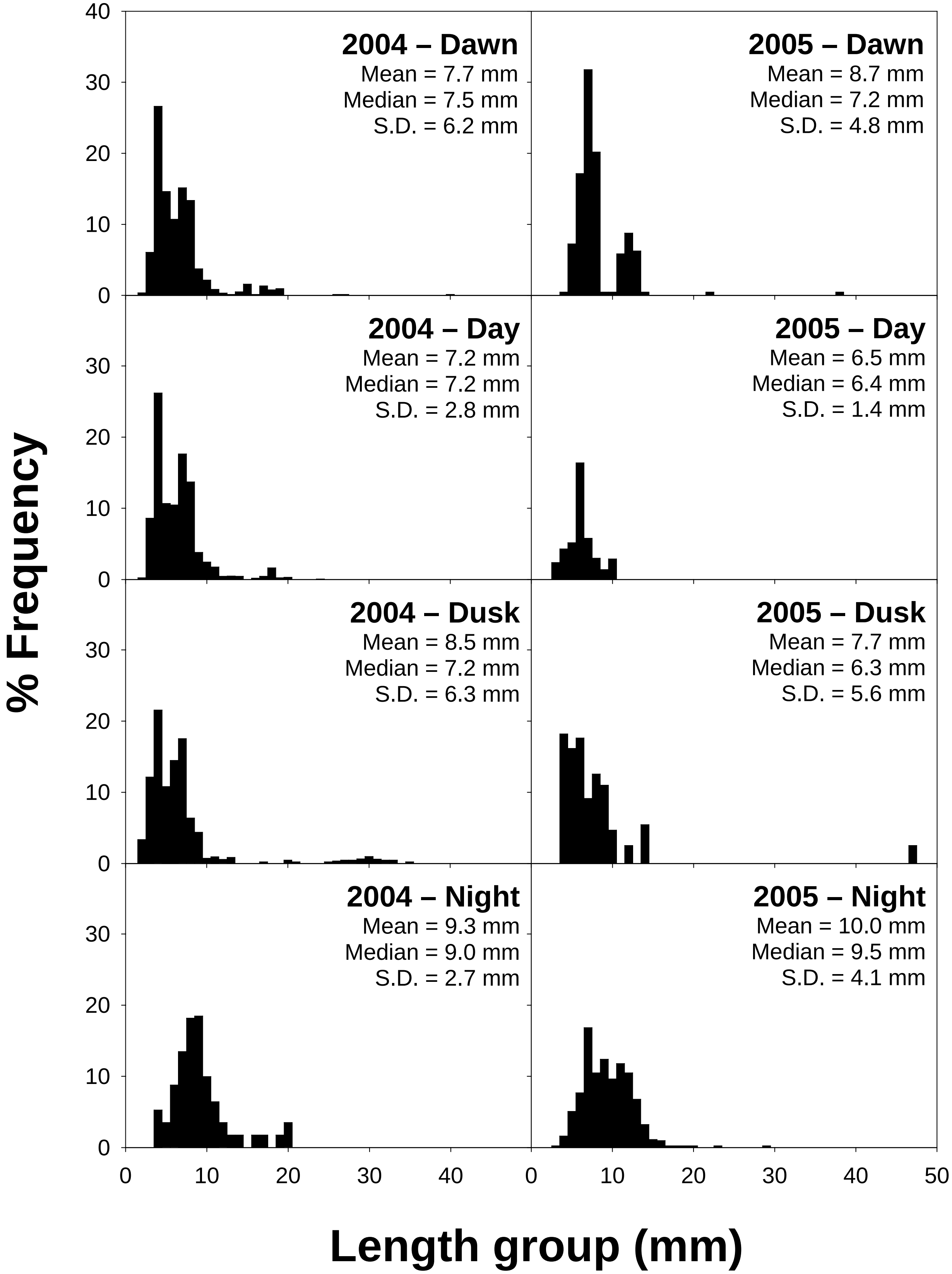
100
75
50
25
2
1
0
75
50
25
0
5
4
3
2
1
0
4
3
2
1
0
● Into Swan
○ Out of Swan
2004
2005
March
April
May
June
July
August
Catch (#/minute)
Channel depth (m)
Date (d)


200
150
100
50
0
150
100
50
0
● Illinois River
○ Swan Lake
2004
2005
Density (#/m3)
March
April
May
June
July
August
Date (d)


Drift Into Swan
Drift Out of Swan
Illinois River
Swan Lake
2004
2005
Catch rate (#/minute)
Density (#/m3)
1.0
0.8
0.6
0.4
0.2
0.0
8.0
6.0
4.0
2.0
FS
FD
MG
Habitat Use Classes



% Frequency
40
30
20
10
0
2004
Illinois River
Mean = 9.6 mm
Median 5.5 mm
S.D. = 9.0 mm
2005
Illinois River
Mean = 10.9 mm
Median 7.2 mm
S.D. = 10.6 mm
30
10
5
0
2004
Lower Swan
Mean = 18.2 mm
Median 13.4 mm
S.D. = 15.7 mm
2005
Lower Swan
Mean = 19.2 mm
Median 12.7 mm
S.D. = 15.3 mm
30
20
10
0
2004
Drift Into
Mean = 7.9 mm
Median 7.4 mm
S.D. = 5.4 mm
2005
Drift Into
Mean = 8.3 mm
Median 6.9 mm
S.D. = 4.1 mm
30
20
10
0
2004
Drift Out
Mean = 8.0 mm
Median 7.6 mm
S.D. = 4.6 mm
2005
Drift Out
Mean = 9.2 mm
Median 7.8 mm
S.D. = 4.8 mm
0
20
40
60
0
20
40
60
80
Length group (mm)



2004 – Dawn
Mean = 7.7 mm
Median = 7.5 mm
S.D. = 6.2 mm
2005 – Dawn
Mean = 8.7 mm
Median = 7.2 mm
S.D. = 4.8 mm
2004 – Day
Mean = 7.2 mm
Median = 7.2 mm
S.D. = 2.8 mm
2005 – Day
Mean = 6.5 mm
Median = 6.4 mm
S.D. = 1.4 mm
2004 – Dusk
Mean = 8.5 mm
Median = 7.2 mm
S.D. = 6.3 mm
2005 – Dusk
Mean = 7.7 mm
Median = 6.3 mm
S.D. = 5.6 mm
2004 – Night
Mean = 9.3 mm
Median = 9.0 mm
S.D. = 2.7 mm
2005 – Night
Mean = 10.0 mm
Median = 9.5 mm
S.D. = 4.1 mm
% Frequency
40
30
20
10
0
0
10
20
30
40
50
Length group (mm)

COMPARATIVE FISH REPRODUCTION

SUMMARY

Large river modifications have widely degraded backwaters, reducing critical habitat for larval fishes. During 2004 and 2005, we assessed how river backwaters function as fish nurseries by quantifying the response of larval fish communities to restoration of Swan Lake, a major (>1,000 ha) backwater complex on the lower Illinois River. The response was compared to a nearby unrestored backwater complex and to adjacent river segments. Zooplankton prey also were quantified. Families of fish changed between years, with fluvial-dependent taxa present during the flood pulse of 2004 and relatively absent during the drought of 2005. During both years, about tenfold greater larval densities were produced in backwaters than the river. Larvae were larger in backwaters and during the non-flood year, probably due to abundant zooplankton. All backwaters produced similar larval densities regardless of restoration. Growth was highest in the most isolated and regulated backwater portion of Swan Lake. Predictable flood pulses coupled with habitat heterogeneity in the backwaters may be important for larval production, assemblage diversity, and recruitment. Current site-specific restoration efforts constructed with the built-in flexibility to adapt to other management regimes will likely be most beneficial to all early life history strategies of fishes, while also providing widespread benefits and supporting the move towards system-wide management programs on large rivers.

Keywords: larval fish, backwater, restoration, early life history, flood pulse, Illinois River

Introduction

Natural river ecosystems support abundant and diverse species assemblages due to high habitat diversity and physical complexity. Off-channel aquatic habitats, like backwater lakes and sloughs, provide productive habitat and a lentic-lotic gradient whereby fishes find structural refuge, food resources, and spawning and nursery grounds (Junk et al. 1989). For many riverine fish species, floodplains are optimal spawning and nursery habitats and are actively sought out via lateral spring spawning migrations of adults (Molls 1999). However, river regulation and degradation have reduced backwater habitat quality and river connectivity, jeopardizing larval fish production and success (Havera and Bellrose 1985, Tockner and Standford 2002). Connectivity of river and backwater areas is likely related to larval fish diversity and recruitment, such that reduced access could alter riverine fish communities (Pezold 1998, Turner et al. 1994). Although this association is critical to backwater health (Gore and Shields 1995), few studies have related connectivity to early life history requirements and larval fish communities (Miranda 2005).

Despite the important links between backwaters and larval fish production, floodplain habitat has become severely degraded and is among the most endangered landscape in the world (Tockner and Stanford 2002). Dam and levee construction coupled with high sediment loads from agricultural lands extensively modified large river hydrology, altered the flood-pulse, reduced backwater habitat quality and river connectivity, and decreased system productivity (Havera and Bellrose 1985). In the 1980s, the loss of backwater habitat in the Illinois River spurred research and the development of projects aimed to restore essential fish and waterfowl habitat (Sheehan et al. 1990). A major (> $US 10 million) Habitat Rehabilitation and Enhancement Project (HREP), through the federally supported Environmental Management

Program (EMP), was initiated on a major backwater of the Illinois and Mississippi Rivers, Swan Lake. One goal of the Swan Lake HREP, to improve spawning and nursery habitat for fishes, was addressed by controlling river connectivity of the backwater and increasing habitat heterogeneity (USACE 1993).

HREP techniques must be evaluated because current paradigms, which influence river management and serve as the basis for these restorations, will continue to be implemented in future programs. However, the paucity of information about life histories and habitat needs in large river ecosystems, particularly for larval and juvenile stages, often hinders restoration efforts and effectual progress (Galat and Zweimuller 2001). All riverine fishes exhibit certain life history strategies based on their relative dependence on flowing water; some are specialized for riverine environments, while others require flowing water for only a portion of their life history. Lastly, generalist fishes reside mostly in lentic backwater areas, especially during their first year of life. Therefore, understanding larval dynamics and production within restored habitats will enable researchers and managers to determine the influence of habitat alterations on larval fish assemblages. We assessed the benefit of a restored HREP-supported backwater system relative to an unmanipulated off-channel complex by quantifying the response of larval fish communities. We also evaluated the how flood regimes and backwater habitat heterogeneity affected larval abundance and composition within managed and natural backwater habitats.

Methods

Study Area

Our study area was located on the lower Illinois River (ILR) and contained two backwater complexes, HREP-managed Swan Lake (river kilometer [RKM] 8.0, measuring from its confluence with the Mississippi River) and natural Calhoun Point (CP, RKM 1.6), and two adjacent river segments, RKM 1.6 and RKM 8.0 (measuring from its confluence with the Mississippi River; Fig. 1).

Historically, Swan Lake was only connected to the river at its downstream end where a 0.5 km-wide opening to the river existed. During restoration, a cross-levee was constructed to create a lower compartment (Lower Swan [LS], 567 ha) and middle compartment (Middle Swan [MS], 485 ha; Fig. 1), and the historic LS connection was restricted to the width of a stop-log water control structure, about 5 m wide. An additional river connection was created in MS via a stop-log structure. During normal pool stage, the stop-log water control structures at LS and MS were the only avenue through which larval fish movement between the compartments and river could occur. The cross-levee and stop-log connections allowed the compartments to be managed independently (Fig. 1). Lower Swan was managed to be continuously connected to the Illinois River, while the MS water control structure was opened in early winter, allowing spawning fish to access the lake before it was disconnected from the river in early spring and pumped several feet below pool level each June. Moderate flooding topped the MS stop-logs in the water control structure and the cross-levee, which served to connect MS to the river and LS during additional times of the year. Thus, LS was a restored, unmanipulated backwater, while MS was a restored and manipulated counterpart.

The CP backwater complex (466 ha), at the confluence of the Illinois and Mississippi Rivers, was unrestored and regularly connected to the rivers during floods via low-lying levees (Fig. 1). These levees provided limited flood control, and functioned more for isolating the backwater and maintaining water levels during non-flood periods. This backwater system was dendritic and contained many vegetated islands; however CP was also shallow and had an unconsolidated lakebed. This system represented an unrestored, unmanipulated backwater.

Larval Production

We quantified larval production using paired, bow-mounted ichthyoplankton nets (0.5 m diameter x 2 m long, 500-μm mesh). River and backwater habitats were sampled during late March through September 2004 and 2005 to understand drift patterns and compare the outcome of management practices on study sites. Four randomly stratified chosen transects within each Swan Lake compartment and two randomly chosen transects in the ILR (RKM 8.0) were sampled weekly. Every two weeks, we sampled CP and its adjacent segment of the ILR (RKM 1.6). We stratified backwater transects into inshore and offshore tows, with two inshore plus two offshore transects per Swan Lake compartment, and one inshore plus one offshore tow within CP ($N = 10$ backwater transects). River tows were conducted within one km upstream and one km downstream of each backwater-river confluence ($N =$ four river transects). In MS, sampling ended on 22 July 2004, and on 27 June 2005, due to the summer drawdown (i.e., water level management program). Calhoun Point was not sampled during 2005 due to low water levels.

At each transect, tows were conducted at the surface for five minutes at a constant speed, with a calibrated mechanical flow meter (Model 2030R, General Oceanics, Inc., Miami, Florida, USA) mounted in the mouth of one net to standardize volume sampled. Inshore backwater tows followed the shoreline, offshore backwater tows were straight transects, and river tows were straight transects conducted perpendicular to flow direction. River tows started at the main channel border, continuing across to the opposite main channel border. If five minutes had not passed, the direction was reversed with nets still in the water, and sampling continued until five minutes was reached.

Upon completion of each transect, contents were flushed into the cod end and preserved in 95% ethanol. Samples were split to approximately 200 fish using a Folsom plankton splitter (Aquatic Research Instruments, Hope, Idaho, USA). All age-0 fish were counted, identified to the lowest possible taxon, typically genus, and classified to a developmental stage (i.e., yolk-sac, larval, juvenile) using descriptions and keys in Soin and Sukhanova (1972), Auer (1982), Murty et al. (1986), Tweb et al. (1990) and voucher specimens from Southern Illinois University's Fluid Vertebrate Collection (Brooks Burr, Carbondale, Illinois, USA) and Colorado State University's Larval Fish Laboratory (Darrel Snyder, Fort Collins, Colorado, USA). A subsample of fish from each taxon and stage identified was measured (total length [TL]; mm) using Scion Image® software or metric calipers ($N = 10$ per net). During 2005, MS samples were counted, but not identified.

At the start of each transect, we sampled water chemistry that could affect the density of larval fish, including temperature (^{o}C) and dissolved oxygen (mg/L [YSI Model 52 Dissolved Oxygen Meter; Yellow Springs Instruments, Yellow Springs, Ohio, USA]), water depth, secchi depth, and average wind speed (km/h [Kestrel 1000, NFS – Radiation Protection Systems, Inc., Groton, Connecticut, USA]). Surface water velocity (cm/s) was measured in the main channel at river transects with an electronic (Flo-Mate Model 2000, Marsh McBirney, Inc., Frederick, Maryland, USA) or mechanical flow meter (Model 2030R Flowmeter, General Oceanics, Inc., Miami, Florida, USA). Continuous monitoring temperature loggers (8-bit Minilog-TR, Vemco Ltd., Nova Scotia, Canada) were located in each backwater and the river to supplement the water temperature data taken at each transect. River stage data were recorded at Hardin, Illinois by the

St. Louis District U.S. Army Corps of Engineers. Swan Lake channel depth data were recorded using a Doppler unit (Model 6526-51 Starflow Ultrasonic Doppler Flow Meter; Unidata America, Lake Oswega, Oregon, USA). This unit was anchored to the bottom of the water control structure and continuously recorded temperature (^{o}C) and depth (mm). Depth data were not available for MS and CP.

Habitat Use Classes

Fish collected were grouped by family into one of three generalized classes to better explain trends between years and treatment groups (Galat and Zweimuller 2001). The classes were: fluvial specialists, fluvial dependent, and macrohabitat generalists. Fluvial specialists inhabit streams and rivers throughout their entire life and rarely enter floodplain habitats (Galat and Zweimuller 2001). Fluvial dependent species regularly use lentic backwater or reservoir habitats, but certain life stages depend on lotic environments (Galat and Zweimuller 2001). These species are typically broadcast spawners, where developing eggs and larvae are semi-buoyant and passively drift in wind-induced or downstream currents (Holland 1986). Adult fluvial dependent fishes also may make lateral migrations into slow-flowing lentic areas to spawn—activities likely corresponding with increasing temperatures and rising water levels (Junk et al. 1989). Macrohabitat generalists include species commonly found in reservoirs and off-channel habitats that do not depend on lotic systems (Galat and Zweimuller 2001). When these fishes use the river, it is as a corridor to move among backwaters (Junk et al. 1989, Dettmers et al. 2001). Spawning usually occurs in off-channel habitats and offspring generally do not leave this habitat until the juvenile stage (Holland 1986). We grouped families based on Galat and Zweimuller (2001; taxonomy from Nelson [1994]), and the only deviation from their groupings was Sciaenidae, which we classified as fluvial dependent (Dettmers et al. 2001, Koel and Sparks 2002).

Zooplankton Density

Because zooplankton abundance and composition affect growth and survival of exogenous feeding larvae, zooplankton were sampled in each backwater and river segment. Similar to larval tows, we randomly stratified samples between inshore/offshore and upstream/downstream habitats. Backwaters were sampled before a corresponding inshore and offshore larval tow (*N* = two samples/backwater), and river sites were sampled in the thalweg at each upstream and downstream transect (*N* = two samples/river segment). Samples consisted of four, 1-m vertical hauls from the boat using a conical net (0.5 m x 2 m, 64 μm mesh). If depth was less than 1 m, the entire water column was sampled four times, with the depth noted to adjust volume sampled. During 2005, neither MS nor CP were sampled.

After each haul, net contents were flushed into a removable collecting bucket and rinsed into a 64 μm sieve. All four hauls from one site were preserved with 10% buffered sugar-formalin in a single container (Haney and Hall 1973), and returned to the laboratory for processing. Copepods including cyclopoids, calanoids, and nauplii, cladocerans including *Bosmina* spp., *Chydorus* spp., *Daphnia* spp., *Diaphanosoma* spp., and *Moina* spp., and other zooplankters including ostracods were identified and counted. Rotifers were dyed with a biological stain and counted, but not identified. Using a Henson-Stempel Pipette, each sample was processed until two taxa reached counts of 200 or until 10% of the sampled had been processed. Due to the high density of rotifers, they were counted until approximately 100 individuals had been processed. Zooplankton were grouped as rotifers and macro-crustaceans (i.e., copepods, cladocerans, nauplii, etc.) for data analysis due to large differences in density.

Data Analysis

Weekly means of abiotic variables were analyzed using principle component analysis (PCA) to reveal environmental relationships among sites and between years. The abiotic matrix, containing temperature (^{o}C), dissolved oxygen (mg/L), depth (m), secchi (cm), and wind (km/h) data, was run in PC-ORD using the correlation option to center and standardize parameters (McCune and Medford 1999).

Densities for larval tow and zooplankton data were calculated as fish/m^3 and zooplankton/L. Total larval production at each site was conservatively calculated as the sum of weekly densities, and when a site was not sampled or processed during a week, the larval density from the previous weeks was substituted for the missing value. All data were log-transformed to meet assumptions of normality. Two-way repeated measures ANOVA (proc MIXED, SAS Institute 1999) was used to test for differences among treatments sampled over time (Hurlbert 1984). For tow and zooplankton data, mean densities among sites were compared over time.

Length data were analyzed to determine whether sizes of fish differed among tow and drift treatments. Proportions of fish per 1-mm length group were calculated and used to make pairwise comparisons. Kolmogorov-Smirnov tests compared size structure of fish collected from tows and between year differences for LS and ILR tow data. Size structure among tow sites was compared using Bonferroni adjusted □-values. Because MS was drawn down before the end of the 2004 sampling season, pairwise comparisons within this site only included samples collected on or before 23 July.

We determined overall trends within and between years for larval and zooplankton densities. A one-way ANOVA design tested density differences between years for larvae and zooplankton. We investigated larval and zooplankton density differences within each year using a two-way ANOVA randomized block design. The randomized block design test assessed hypotheses about overall lake effects: (1) stratified habitats differed in density (i.e., inshore v. offshore), (2) sites differed in density (i.e., LS v. CP), and (3) fish families and zooplankton groups varied among sites. To control for experimentwise error rates, Tukey-Kramer post-hoc (Sokal and Rohlf 1995). Pearson correlation coefficients were calculated for untransformed larval and zooplankton densities to reveal any association between larvae and their food source.

Results

Environmental Factors

A moderate flood pulse occurred in 2004, where water levels in the lower Illinois River were above flood stage for approximately five weeks during June (Fig. 2). In contrast, water levels remained at or below normal pool level of 128 m during the 2005 sampling season.

Water temperatures in the river gradually rose and fell during 2004, peaking in late July at 28 ^{o}C, and varied in 2005, exceeding 30 ^{o}C twice (Fig. 2). Lower Swan conditions mimicked the river, although water temperatures were higher and more variable and depths fluctuated less (Fig. 2). The 2004 flood pulse topped the water control structure at MS and the low-lying levee at CP, causing water levels in these backwaters to rise rapidly and connecting all the backwaters to the river. Middle Swan and CP remained isolated during 2005.

From the PCA output, two axes, which had broken-stick eigenvalues less than the actual eigenvalues, were used to graphically represent the data (Jackson 1993). River and backwater sites were spatially separated by water and secchi depths, while water temperatures and other abiotic parameters were similar among all backwaters (Fig. 2 & 3, Appendix 1 & 2).

Larval Production

During both years, fish larvae in the backwaters first appeared in low densities during late March and early April, whereas larvae were not collected in river sites until late April (Fig. 4). Backwaters consistently produced higher densities of larvae than the ILR ($P < 0.01$, Table 1), with 2004 and 2005 ILR densities peaking at means of 11 and 14 fish/m^3 (Fig. 4), and production estimated at 40.5 and 31.0 fish/m^3, respectively. A peak of larvae occurred in all backwaters and the river during June 2004, the period of floodwater inundation (Fig. 4). During that time, Lower Swan and CP experienced a larval pulse of similar size (Fig. 4, Table 1), and had similar total production estimates of 434.8 and 408.7 fish/m^3, respectively. Middle Swan densities peaked the highest, at 400 larvae/m^3, though they were not statistically higher than LS (Table 1), and total production was estimated at 1,276.4 fish/m^3. These among site differences (two-way repeated measures ANOVA: $F_{3,10} = 55.37$, $P < 0.01$) differed across time (2004: $F_{21,133} = 55.06$, $P < 0.01$) and also interacted (2004: $F_{43,133} = 8.07$, $P < 0.01$). During 2005, sites also differed (two-way repeated measures ANOVA: $F_{2,9} = 53.37$, $P < 0.01$) across time (2005: $F_{22,115} = 15.05$, $P < 0.01$) and interacted (2005: $F_{31,115} = 5.00$, $P < 0.01$). Larval pulses did not occur in synchrony among sites, but occurred in MS during May, LS during late May, and the ILR during June 2005 (Fig. 4). Although MS produced less total larvae during 2005, at 515.7 fish/m^3, total larval production was higher than LS during the same time interval. Production in LS increased during 2005, to 530.5 fish/m^3.

Each site had homogeneous larval distributions, with no differences between inshore and offshore or upstream and downstream stratified transects ($P > 0.05$, Table 2). However during 2004, offshore transects in Swan Lake had higher densities than inshore transects (Table 2).

The larval assemblage in the backwaters was comprised predominantly of macrohabitat generalists during both years, but changed between years in the river (Fig. 5). In the ILR, fluvial dependent taxa, consisting mostly of sciaenidae, catostomidae, and cyprinidae, occurred at higher densities during 2004, but were relatively absent during the low water year of 2005 (Fig. 5). Macrohabitat generalists dominated the ILR during 2005 (Fig. 5). MS contained higher densities of fluvial dependents (i.e., catostomids, cyprinids) as compared to other backwaters during 2004 (Table 3, Fig. 5). Clupeids, the most abundant family, drove macrohabitat generalist patterns in all sites and heavily influenced system-wide trends in total density (Table 3). Despite being continuously connected to the ILR, LS had substantially higher densities of every family sampled except sciaenidae and moronidae during 2004. During 2005, higher densities of these families were collected in LS, though densities were lower than during 2004.

Larval sizes were larger in backwater systems than the ILR, regardless of year (Fig. 6). During 2004, Calhoun Point had larger fish than MS and LS, possibly indicating this backwater provided better fish nursery habitat ($P < 0.001$, Table 4, Fig. 6). Progressively smaller sized fish occurred in MS, LS, and the smallest were collected in the ILR (Table 4). Lower Swan and ILR size structures were significantly larger during 2005, though larger fish again occurred in the backwater (all $P < 0.001$, Table 4).

Zooplankton Density

Similar to larval fish abundance, patterns in zooplankton and rotifer density differed by site and appeared influenced by the 2004 flood pulse. During 2004, macro-crustaceans, consisting mostly of cladocerans, copepods, and nauplii, peaked during late May and June in the ILR, LS, and CP, while MS densities steadily declined during April through June (Fig. 7). The opposite occurred for rotifer densities, with the ILR, LS, and CP showing declining densities during late May and a pulse of rotifers occurring in MS during 2004 (Fig. 8).

Although rotifers dominated the zooplankton assemblage during both years, 2005 rotifer densities appeared greater while macro-crustaceans occurred in lower densities than 2004 (Table 5). In the ILR, while there was no between-year difference in rotifer density ($F_{1,34} = 1.28$, $P = 0.27$; Fig. 8), higher densities of macro-crustaceans occurred during 2004 ($F_{1,34} = 12.43$, $P < 0.01$; Fig. 7). In LS, the opposite occurred, with higher rotifer densities during 2005 ($F_{1,34} = 8.46$, $P < 0.01$; Fig. 8) and no difference of macro-crustacean densities between years ($F_{1,34} = 1.43$, $P = 0.24$; Fig. 7). The repeated measures analyses for 2004 revealed temporal variation in density, but no difference among sites for rotifers (site: $F_{3,63} = 0.80$, $P = 0.50$; week: $F_{15,63} = 2.61$, $P < 0.01$; lake*week: $F_{32,63} = 1.20$, $P = 0.27$) or macro-crustaceans (site: $F_{3,64} = 1.63$, $P = 0.19$; week: $F_{15,64} = 11.92$, $P < 0.01$; lake*week: $F_{3,64} = 5.37$, $P = 0.01$). During 2005, all effects were significant for both rotifer densities (site: $F_{1,2} = 91.56$, $P = 0.01$; week: $F_{21,33} = 2.74$, $P < 0.01$; lake*week: $F_{16,33} = 3.70$, $P < 0.01$) and macro-crustaceans (site: $F_{1,2} = 93.56$, $P = 0.01$; week: $F_{21,33} = 3.69$, $P < 0.01$; lake*week: $F_{16,33} = 2.89$, $P < 0.01$).

Macro-crustaceans in LS and the ILR were positively correlated to larval fish density during 2004 (LS: $r = 0.71$, $P < 0.01$; ILR: $r = 0.63$, $P = 0.01$). Only rotifer densities in MS correlated with larval densities ($r = 0.80$, $P < 0.01$). No other patterns occurred during 2004, and no correlations occurred during 2005 for either rotifers or macro-crustacean densities and fish larvae.

Discussion

Connectivity is a critical feature of floodplain habitats, and likely influenced larval production patterns in the lower ILR and its backwaters. Larval production during the flood year was synchronized among all sites; well-timed to the period of inundation where all sites were connected. More families in greater evenness also occurred during the flood year. Although typically isolated from the river, a large variety of families occurred in MS during 2004, mostly notably a large proportion of catostomids and cyprinids that likely were entrained in the backwater when floodwaters topped its levee. This restored and manipulated site also produced more larvae during both years than any other site in the same time frame, which hints towards the benefits garnered from regular periods of connectivity and increased aquatic vegetation.

During the non-flood year we saw a lack of fluvial dependent taxa in all sites, which was more pronounced in the riverine habitat than the backwaters. Moreover, macrohabitat generalists tended to have higher densities during the non-flood year, apparently capitalizing on the stable water levels and low flows. Similarly, Brown and Coon (1994) showed a decreased number of taxa during a non-flood year, with lower densities of fluvial dependent species (i.e., goldeye *Hiodon alosoides*, buffalo *Ictiobus* spp., and carpsuckers *Carpiodes* spp.) and an increased abundance of macrohabitat generalists (i.e., centrarchids) in most tributaries. Changes in family composition between years emphasize the importance of a predictable flood pulse in larval fish ecology where rising waters cue spawning activity and permit access to floodplain habitat (Junk et al. 1989).

In accordance with other studies that have found lower or no reproduction during non-flood years (Brown and Coon 1994, Agostinho et al. 2004), we expected significantly lower densities and larval production estimates during 2005. However, production of macrohabitat generalists in LS was higher than the year before, and only MS production and peak densities were markedly lower. Moreover, the ILR, which functioned nearly exclusively as spawning habitat during 2004, as indicated by the larval lengths, had a larger size structure during 2005, suggesting these segments of the river provided relatively better nursery habitat for larvae during the non-flood year. Lower velocities during summer 2005, often below 0.1 m/s, transformed the ILR into habitat amenable for rearing larvae. This compensatory pattern in larval production occurred due to certain species, mainly gizzard shad *Dorosoma cepedianum*, *Gambusia affinis*, brook silversides *Labidesthes sicculus*, and emerald shiners *Notropis atherinoides*, exploiting the low flow conditions, a concept dubbed the 'low flow recruitment hypothesis' (Humphries et al. 1999, King 2004). The hypothesis postulates that during low flow periods in the river, appropriately sized prey is concentrated, and under these conditions, some species spawn and can easily make the transition from endogenous to exogenous feeding, thereby having high recruitment (Humphries et al. 1999). We do not know whether recruitment differences occurred between years, but the significantly higher rotifer densities collected in LS during the low-flow period likely supported the increased rates of larval production (Aoyagui and Bonecker 2004).

Limited information exists on the interaction of larval fishes and zooplankton densities in large rivers. The positive correlations to between plankton and larvae may be due to mutually favorable abiotic conditions, such as warming temperatures (Wetzel 2001). However, it has been speculated that biotic factors, such as top-down effects of larvae, may play an important role in riverine foodwebs (Jack and Thorp 2002). During the flood year, plankton densities were positively correlated to fish densities; ILR and LS zooplankton to fish densities in those sites, and MS larvae to rotifer densities. The positive correlation may be due to autochthonous inputs that increased production in backwaters during inundation, allowing for large pulses of larvae and plankton (Junk et al. 1989). The drastic summer decline may be due to increased foraging from larger larvae or simply summer declines often observed in large rivers, whether temperature related or not, though the pattern remains unexplained (Gosselain et al. 1998). Although significant patterns were not found in either CP during 2004, or in LS and the ILR during 2005, total plankton densities increased with larval production and decreased mid-summer.

Promoting vegetation growth through annual drawdowns may have positive effects for restored backwaters. Vegetation could have provided food and cover for larvae, possibly promoting invertebrate populations in densely vegetated areas and affording protection from predation (Dewey et al. 1997, Flinn et al. 2005). However, fish survival in MS was likely low due to the drawdowns. High mortality often occurs through bird and fish predation (Crowder et al. 1997) and anoxic conditions in shallower waters. The true potential of MS for larval production may only be realized if the compartment is managed in a rotating fashion, whereby drawdowns would be conducted every few years to maintain vegetation growth and during other years, it would be continuously connected to the river.

Floodwater stability (gradual rise and fall) coupled with habitat heterogeneity in the backwaters may be important. The unrestored backwater, CP, had the largest larval sizes during 2004, which may have been due to more stable water levels where larvae were not stranded on lake banks and fish nests were not desiccated due to rapidly receding waters (Adams et al. 1999, Brown and Coon 1994). In many species, larval size has been shown to directly relate to survival and eventually recruitment to the adult population (Miller et al. 1988). Progressively smaller larvae were collected in MS and LS, a pattern which corresponds to the degree of structure available in each site, with CP having the most emergent vegetation and LS having none.

To enhance the complex biotic and abiotic interactions so valued in riverine-floodplain habitats, restoration projects must have a built-in flexibility to either create or sustain different habitat types. This approach will likely be most beneficial to all life history strategies by offering a wide variety of habitat characteristics to meet specific early life history environmental and habitat requirements (Grift et al. 2003). Restoration projects similar to the Swan Lake HREP offer the added benefit of flexibility. The compartmentalization allows managers the option of rotating management regimes between the two lower compartments to promote habitat diversity while maintaining river connectivity. One or both compartments may be left open to the river at any one time, ensuring backwater access to riverine fishes. Given the widespread benefits of these management practices and the move towards system-wide management programs on large rivers (Theiling 1995, Flinn et al. 2005), current site-specific restoration projects should be constructed with the flexibility to adapt to other management regimes.

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Table 1. Site comparisons tested for overall differences in larval production during 2004 and 2005 among Calhoun Point (CP), the Illinois River (ILR), Lower Swan (LS), and Middle Swan (MS), with adjusted *P*-values.

Site Comparison			*t*	df	*P*
2004					
CP	vs.	ILR	1.61	48.1	0.38
CP	vs.	LS	-1.50	48.3	0.45
CP	vs.	MS	-2.51	48.0	0.07
ILR	vs.	LS	-4.04	46.6	< 0.01
ILR	vs.	MS	-4.85	47.7	< 0.01
LS	vs.	MS	-1.32	47.7	0.55
2005					
ILR	vs.	LS	-4.43	35.3	< 0.01
ILR	vs.	MS	-7.11	36.6	< 0.01
LS	vs.	MS	-3.07	37.5	0.01

Table 2. Comparisons of stratified transects determined spatial distribution of larvae within each site for 2004 and 2005 in the lower Illinois River system, where backwater systems had inshore and offshore transects (Calhoun Point [CP], Lower Swan [LS], and Middle Swan [MS]), and the Illinois River (ILR) had upstream and downstream transects at CP and LS.

Within-Site Comparison	*t*	df	*P*
Inshore vs. offshore			
CP 2004	0.72	11.0	0.49
LS 2004	-2.08	18.9	0.05
LS 2005	-0.66	18.0	0.52
MS 2004	-1.96	13.8	0.07
MS 2005	-1.49	12.2	0.16
Upstream vs. downstream			
ILR at CP 2004	-0.04	10.0	0.97
ILR at LS 2004	0.94	22.0	0.36
ILR at LS 2005	-0.27	22.0	0.79

Table 3. Fish families grouped into one of three habitat use classes (Galat and Zweimuller 2001) with percent of total catch during 2004 and 2005 for the Illinois River (ILR), Lower Swan (LS), Middle Swan (MS), and Calhoun Point (CP).

	2004 % Catch				2005 % Catch	
Family	ILR	LS	MS	CP	ILR	LS
Fluvial Specialist	0.00	0.00	0.00	0.00	0.00	0.00
Fluvial Dependent						
Catostomidae	1.14	0.19	24.67	0.00	0.29	0.20
Cyprinidae	5.73	2.20	5.22	4.15	7.82	3.09
Hiodontidae	0.00	0.00	< 0.01	0.00	0.00	0.00
Moronidae	1.09	0.03	0.00	0.04	0.09	0.03
Percidae	0.00	< 0.01	0.00	0.00	0.00	0.00
Sciaenidae	34.03	0.19	< 0.01	0.00	0.58	0.65
Macrohabitat Generalist						
Atherinidae	0.02	0.01	0.00	0.45	0.20	0.15
Centrarchidae	1.45	1.89	2.30	8.46	1.31	0.56
Clupeidae	56.38	95.43	67.51	86.75	89.61	94.88
Gasterosteidae	0.01	0.00	0.00	0.00	0.00	0.00
Ictaluridae	0.00	0.00	0.00	0.00	0.00	< 0.01
Lepisosteidae	0.04	0.01	0.25	0.01	0.00	0.00
Poeciliidae	0.02	< 0.01	< 0.01	0.06	0.04	0.40
Total *N*	**8,086**	**48,390**	**105,906**	**11,500**	**4,476**	**56,033**

Table 4. Length analysis of larval fish caught in Lower Swan (LS), Middle Swan (MS), Calhoun Point (CP), and the Illinois River (ILR) using Kolmogorov-Smirnov pairwise comparisons, with results listing sites or years which collected larger-sized fish.

Site Comparison	KSa	*P*	Results
p-value = 0.008, Bonferroni adjusted			
2004			
LS vs. MS	4.67	< 0.001*	*MS*
LS vs. CP	3.49	< 0.001*	*CP*
LS vs. ILR	8.83	< 0.001*	*LS*
MS vs. CP	1.74	0.005*	*CP*
MS vs. ILR	12.79	< 0.001*	*MS*
CP vs. ILR	11.54	< 0.001*	*CP*
p-value = 0.05, not adjusted			
2005			
LS vs. ILR	5.37	< 0.01*	*LS*
2004 vs. 2005			
LS	2.53	< 0.01*	*2005*
ILR	10.55	< 0.01*	*2005*

* Indicates significant differences between treatments, where *P* ≤ *or* ”.

Note: MS was drawn down before the end of the 2004 sampling season. Therefore, pairwise comparisons with this site were conducted on truncated data sets that included only samples taken before 24 July.

Table 5. Mean zooplankton density (#/L) ±1 SE by taxa described as a percent of the total annual density for each site in 2004 and 2005.

	Cladoceran		Copepod		Nauplii		Rotifer	
Site	Density (SE)	%	Density (SE)	%	Density (SE)	%	Density (SE)	%
2004								
Illinois River	0.20 (0.08)	< 0.01	1.53 (0.45)	0.03	28.79 (7.38)	0.59	4868 (2753)	99.4
Lower Swan	0.15 (0.04)	0.01	1.14 (0.35)	0.06	35.00 (7.87)	1.81	1897 (365)	98.1
Middle Swan	1.10 (0.54)	0.05	8.47 (3.90)	0.38	105.54 (36.22)	4.76	2101 (542)	94.8
Calhoun Point	0.32 (0.09)	0.01	0.98 (0.38)	0.04	40.50 (13.00)	1.63	2443 (745)	98.3
2005								
Illinois River	0.87 (0.31)	0.05	0.69 (0.23)	0.04	11.04 (3.29)	0.58	1888 (241)	99.3
Lower Swan	0.31 (0.08)	0.01	1.74 (0.56)	0.05	61.97 (23.65)	1.61	3774 (618)	98.3

Fig. Captions

Fig. 1. Study site of the lower Illinois River and its major backwaters.

Fig. 2. Mean daily temperature of Illinois River and backwater sites during 2004 through 2005. Mean daily depth data are depicted as solid black lines for the Illinois River and Lower Swan. Channel depth of Lower Swan was recorded using a submerged device at the stop-log structure. River data were collected at Hardin, Illinois, USA. Depth was not recorded for Middle Swan and Calhoun Point. Shaded regions represent sampling periods.

Fig. 3. Principle component analysis (PCA) of abiotic variables for the lower Illinois River (☐) and backwater sites, Lower Swan (▲), Middle Swan (▼), and Calhoun Point (●). Weekly means of abiotic variables were included in the analysis: temperature (^{o}C), dissolved oxygen (DO, mg/L), depth (m), secchi depth (cm), and wind (km/h).

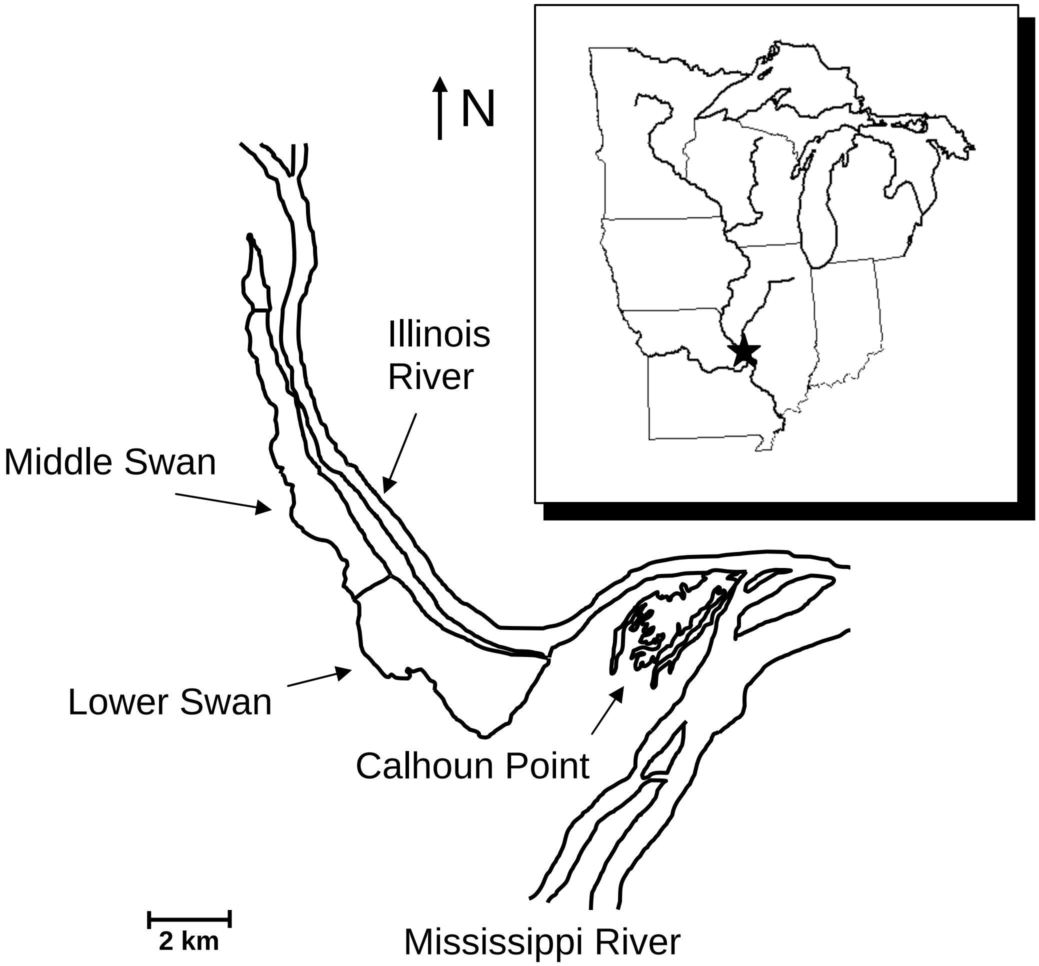
Fig. 4. Mean density (#/m^3) per day of fish caught in the lower Illinois River during 2004 and 2005. Error bars represent ±1 standard error of the mean transect density. Calhoun Point was not sampled during 2005 because of low water levels.

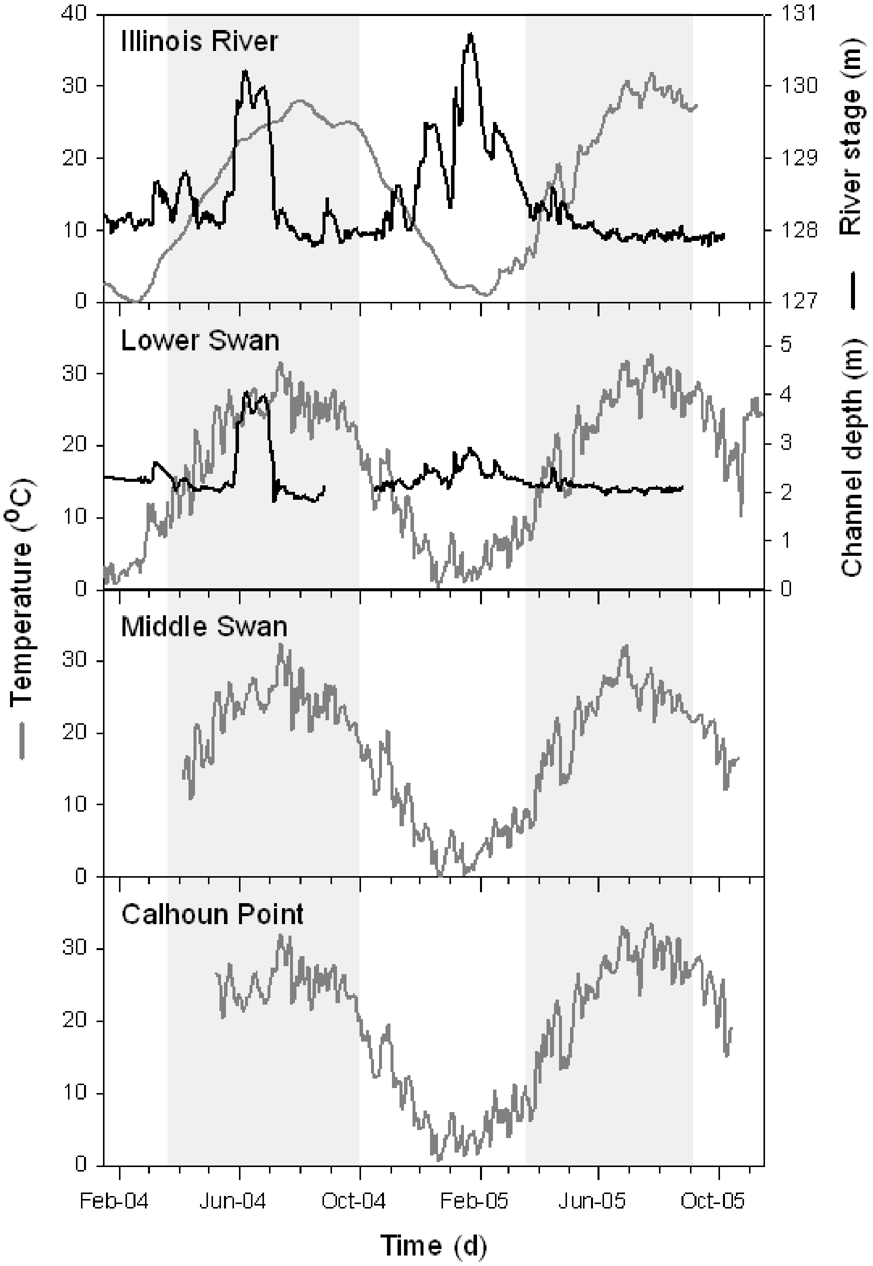
Fig. 5. Length frequency distributions of all larval and juvenile fish caught in each site in of the lower Illinois River system during 2004 and 2005. Sample size is an average of transects within that site summed over the sampling season. Length frequencies are expressed as a percentage of average catch. Middle Swan samples were not collected after 23 July 2004. During 2005, lengths were collected in Middle Swan samples, and Calhoun Point was too shallow to be sampled. Descriptive statistics were calculated on raw length data.

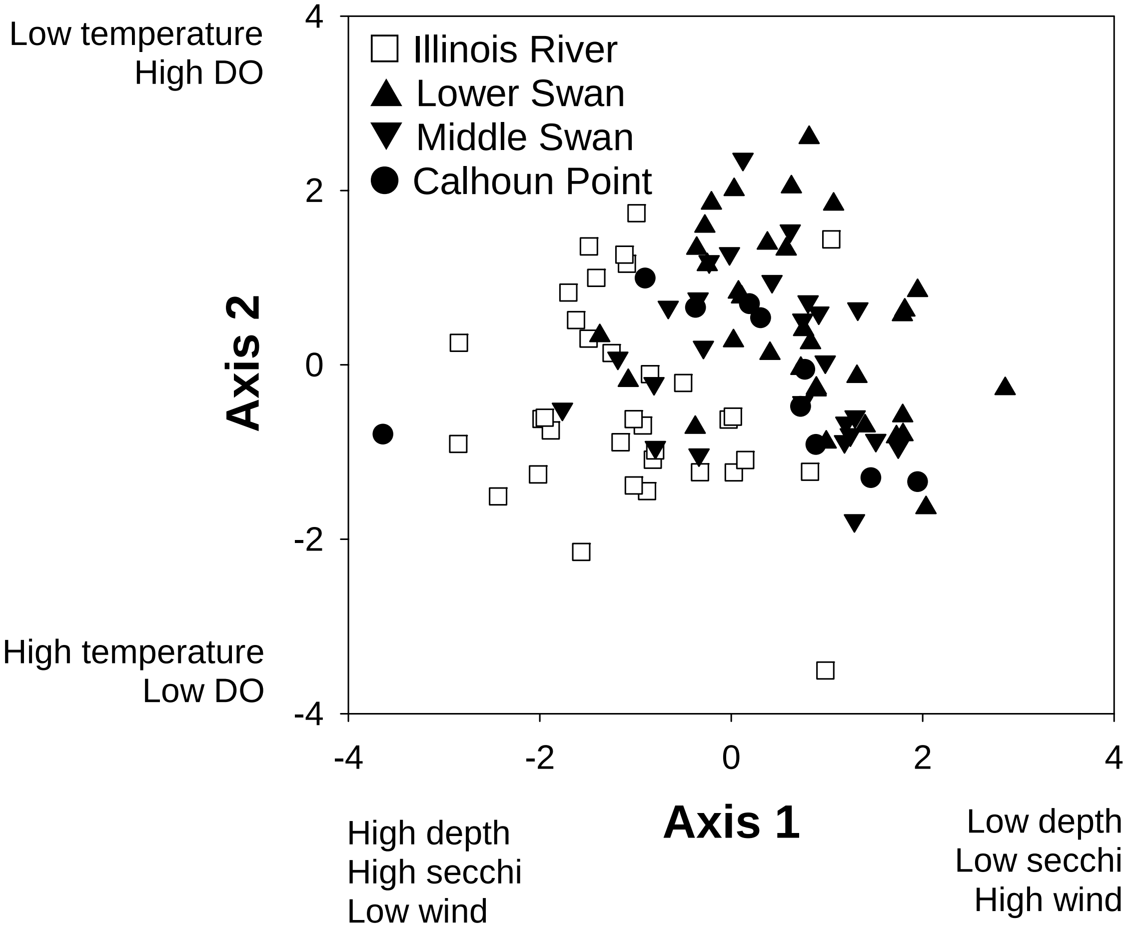
Fig. 6. Fluvial specialist (FS), fluvial dependent (FD), and macrohabitat generalist (MG) mean densities by site and year in the lower Illinois River system. Error bars represent ±1 standard error of the mean of dates sampled during that year.

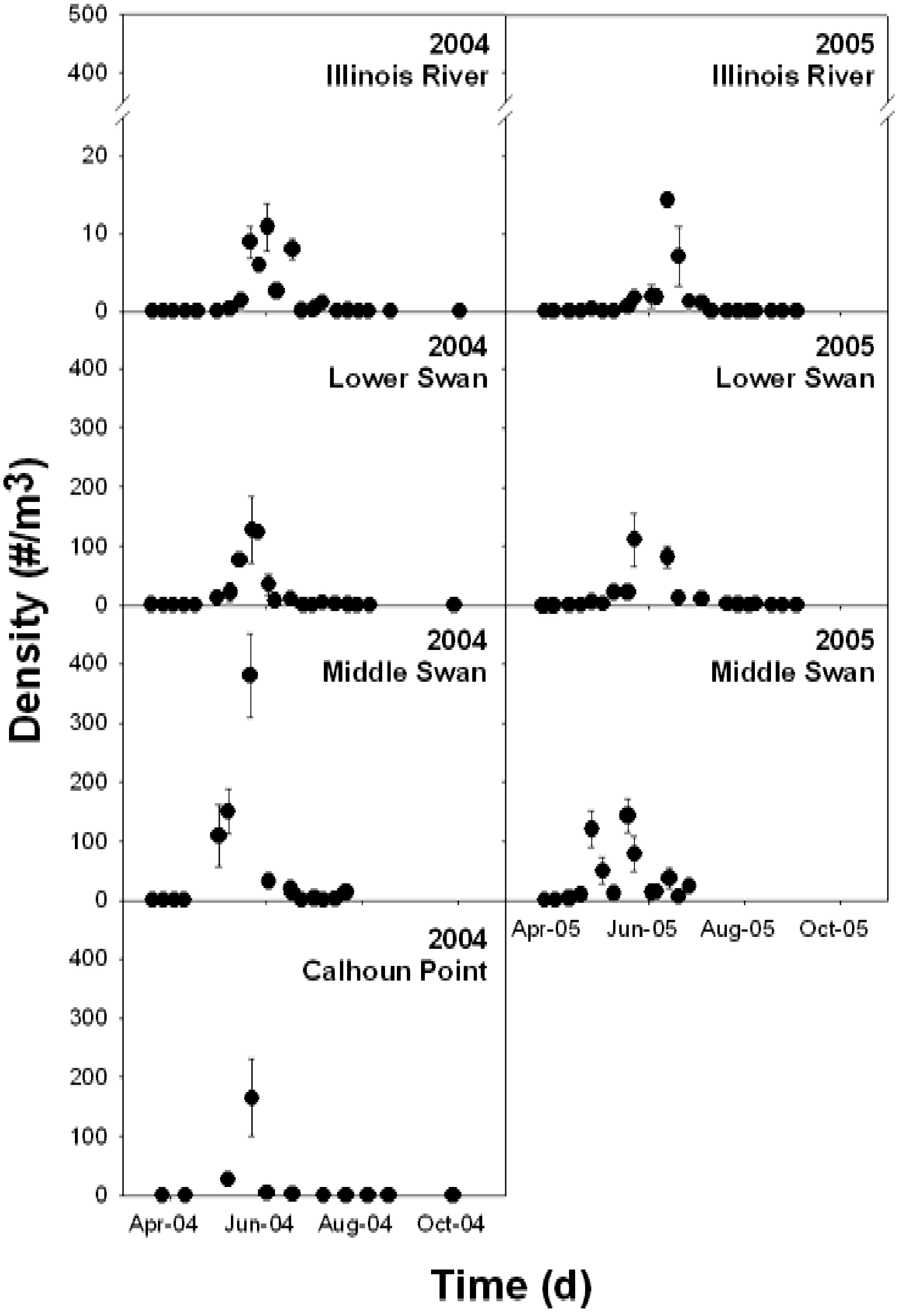
Fig. 7. Mean density (#/L) of cladocerans, copepods, nauplii, and other zooplankters per day in each site of the lower Illinois River system through 2004 and 2005. Error bars represent ±1 standard error of the mean of transects sampled that day. Middle Swan samples were not collected after 23 July 2004. During 2005, zooplankton data were not identified for Middle Swan and no zooplankton samples were taken in Calhoun Point.

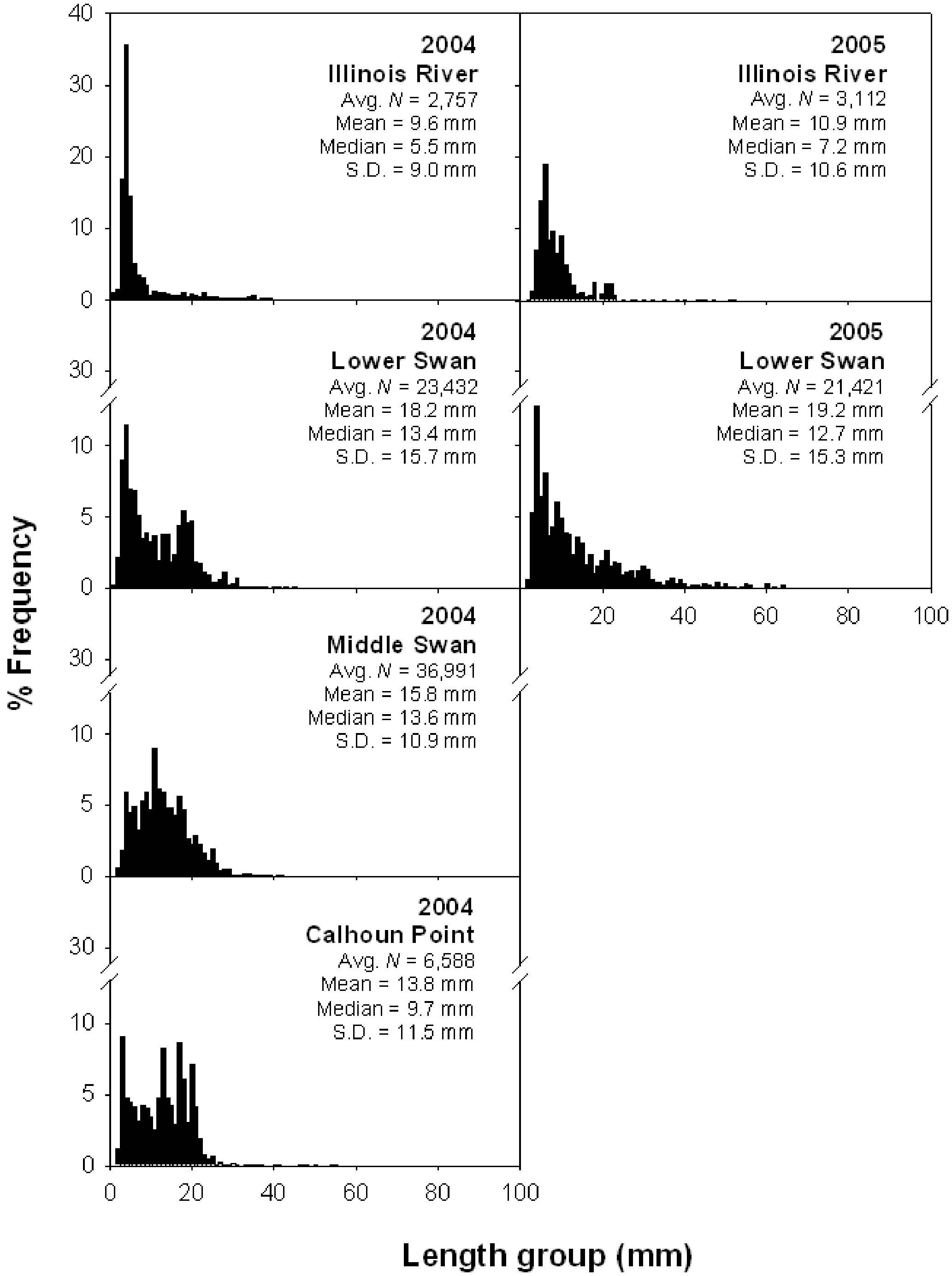
Fig. 8. Mean density (#/L) of rotifers per day in each site of the lower Illinois River through 2004 and 2005. Error bars represent ±1 standard error of the mean of transects sampled that day. Middle Swan samples were not collected after 23 July 2004. In 2005, zooplankton data were not identified for Middle Swan and no zooplankton samples were taken in Calhoun Point.

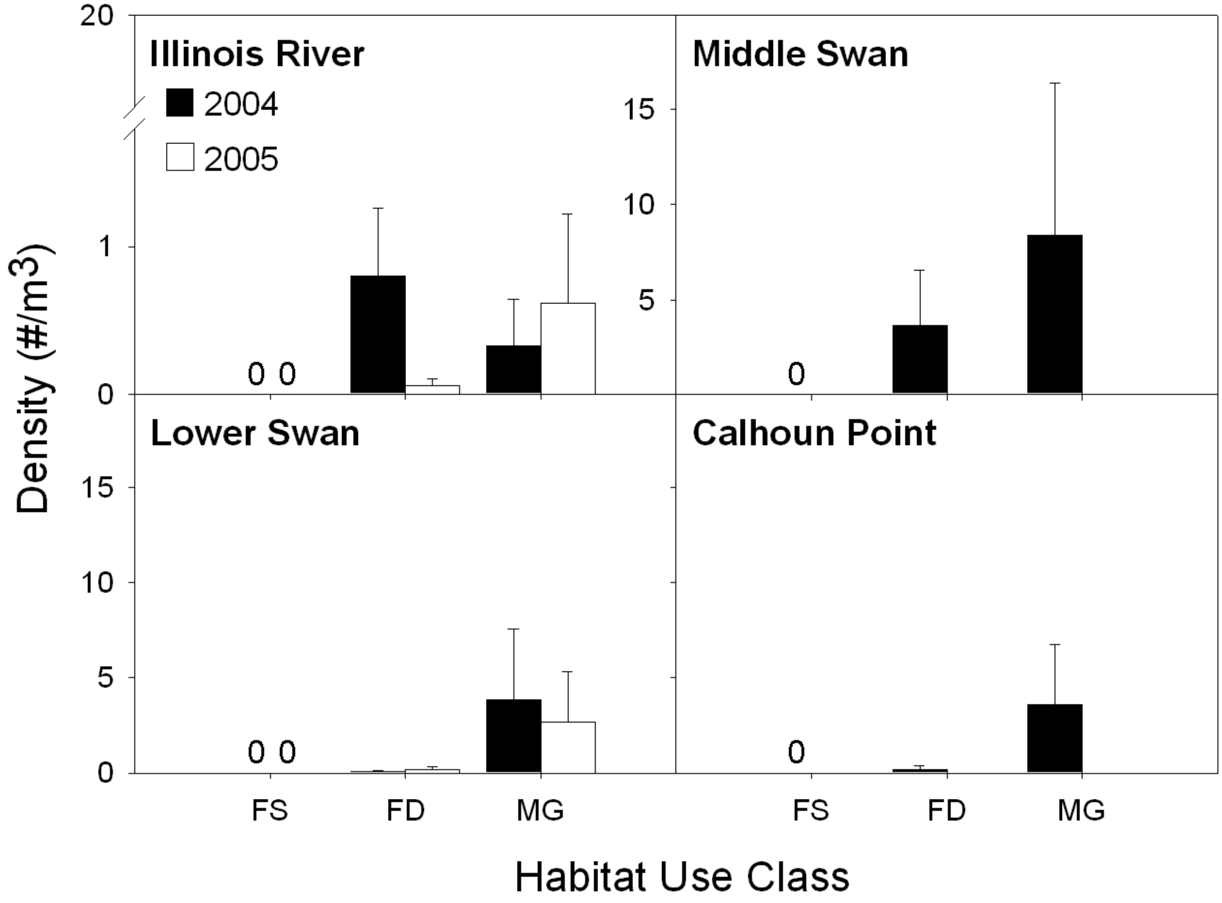

N
Illinois River
Middle Swan
Lower Swan
Calhoun Point
2 km
Mississippi River

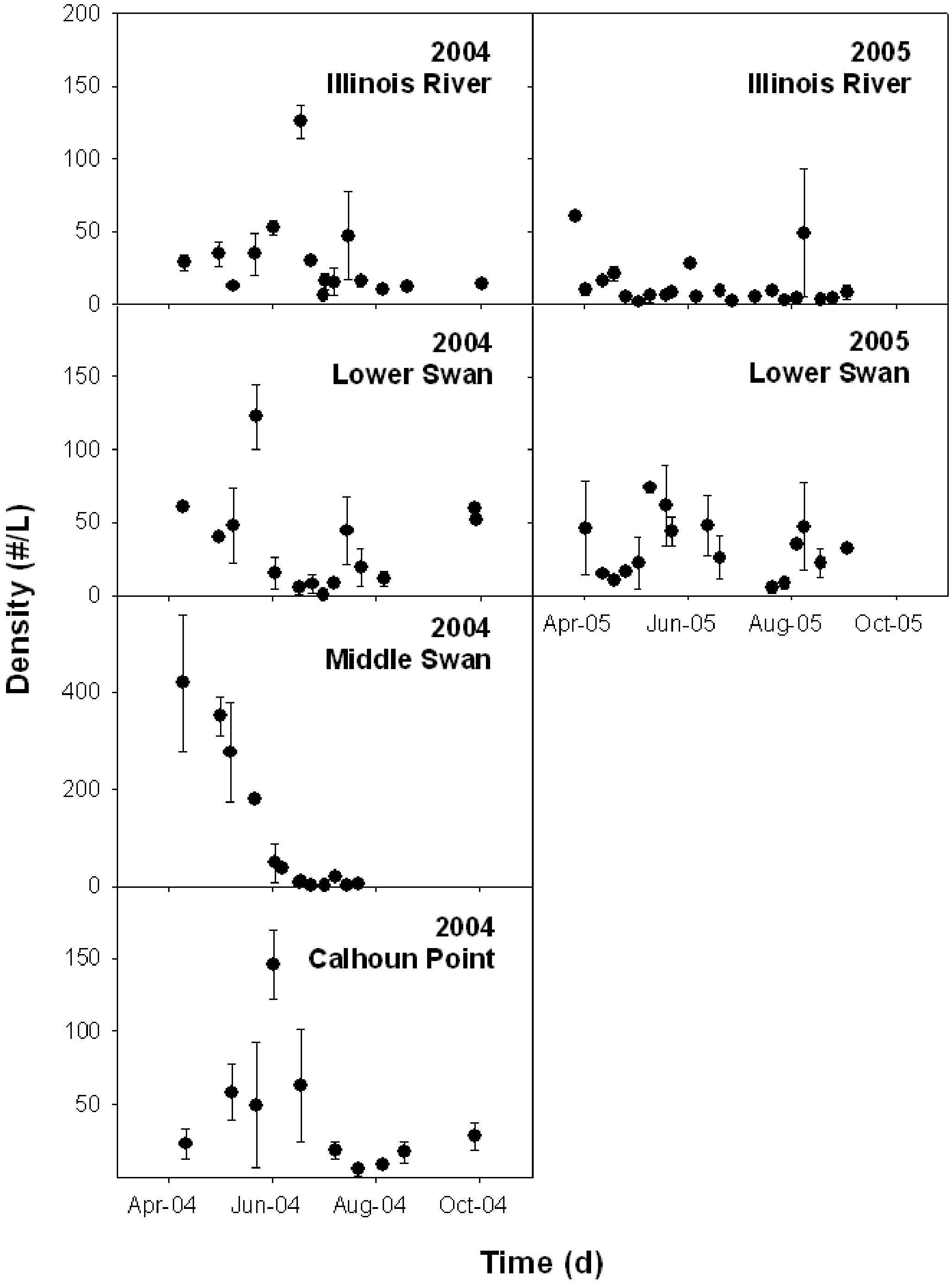

Illinois River
Lower Swan
Middle Swan
Calhoun Point
Temperature (°C)
River stage (m)
Channel depth (m)
Time (d)
Feb-04
Jun-04
Oct-04
Feb-05
Jun-05
Oct-05
0
10
20
30
40
127
128
129
130
131
0
1
2
3
4
5

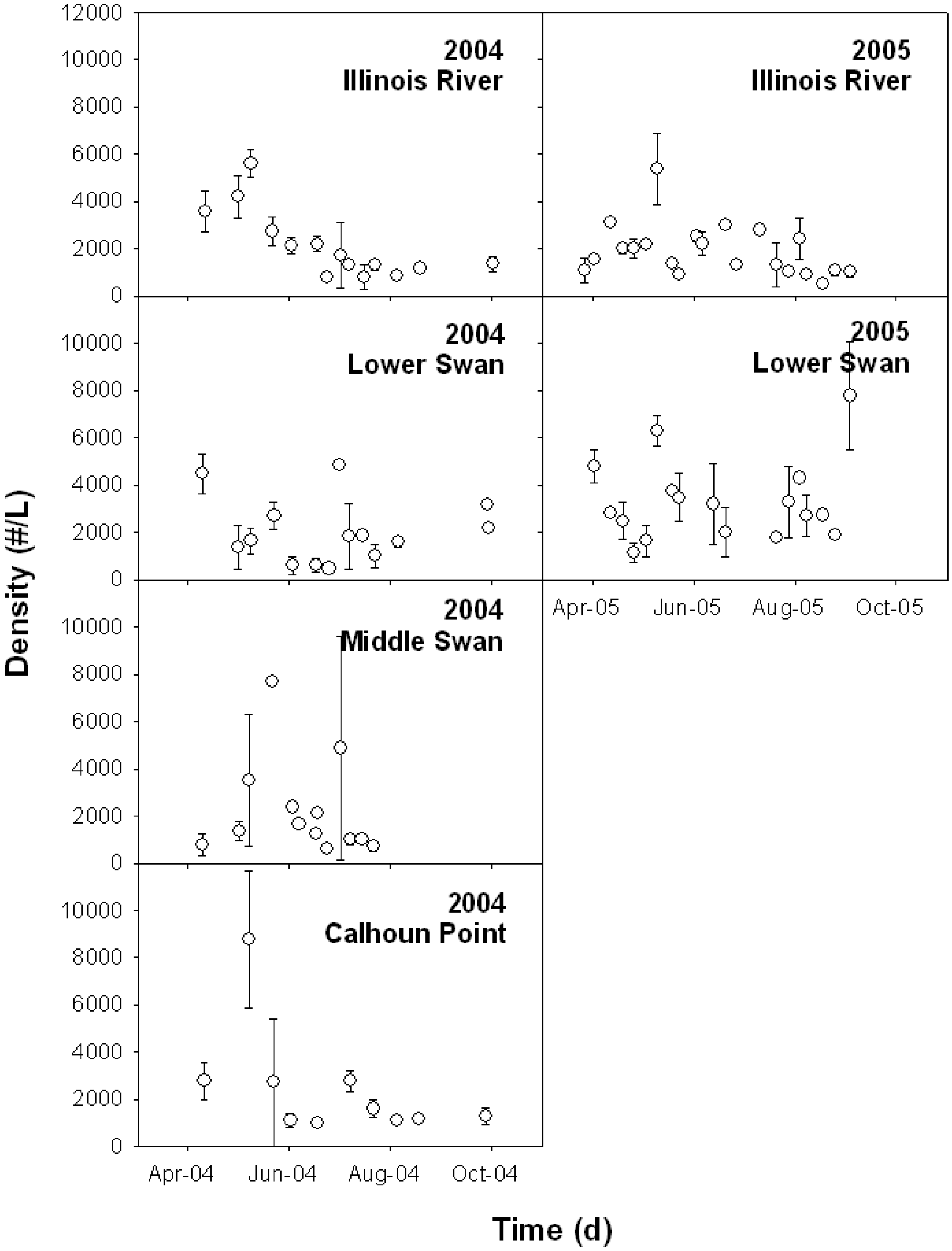

Low temperature
High DO
High temperature
Low DO
Axis 2
4
2
0
-2
-4
-4
-2
0
2
4
Axis 1
High depth
High secchi
Low wind
Low depth
Low secchi
High wind
□ Illinois River
▲ Lower Swan
▼ Middle Swan
● Calhoun Point


500
400
20
10
0
2004
Illinois River
2005
Illinois River
400
300
200
100
0
2004
Lower Swan
2005
Lower Swan
2004
Middle Swan
2005
Middle Swan
2004
Calhoun Point
Density (#/m3)
Apr-04
Jun-04
Aug-04
Oct-04
Apr-05
Jun-05
Aug-05
Oct-05
Time (d)


2004
Illinois River
Avg. N = 2,757
Mean = 9.6 mm
Median = 5.5 mm
S.D. = 9.0 mm
2005
Illinois River
Avg. N = 3,112
Mean = 10.9 mm
Median = 7.2 mm
S.D. = 10.6 mm
2004
Lower Swan
Avg. N = 23,432
Mean = 18.2 mm
Median = 13.4 mm
S.D. = 15.7 mm
2005
Lower Swan
Avg. N = 21,421
Mean = 19.2 mm
Median = 12.7 mm
S.D. = 15.3 mm
2004
Middle Swan
Avg. N = 36,991
Mean = 15.8 mm
Median = 13.6 mm
S.D. = 10.9 mm
2004
Calhoun Point
Avg. N = 6,588
Mean = 13.8 mm
Median = 9.7 mm
S.D. = 11.5 mm
% Frequency
40
30
20
10
5
0
20
40
60
80
100
Length group (mm)


Illinois River
■ 2004
☐ 2005
Middle Swan
Lower Swan
Calhoun Point
Density (#/m3)
20
1
0
15
10
5
0
0 0
0
0 0
0
FS
FD
MG
FS
FD
MG
Habitat Use Class


2004
Illinois River
2005
Illinois River
2004
Lower Swan
2005
Lower Swan
2004
Middle Swan
2004
Calhoun Point
Density (#/L)
Time (d)
200
150
100
50
0
400
200
Apr-04
Jun-04
Aug-04
Oct-04
Apr-05
Jun-05
Aug-05
Oct-05



2004
Illinois River
2005
Illinois River
2004
Lower Swan
2005
Lower Swan
2004
Middle Swan
2004
Calhoun Point
Density (#/L)
12000
10000
8000
6000
4000
2000
0
Apr-05
Jun-05
Aug-05
Oct-05
Apr-04
Jun-04
Aug-04
Oct-04
Time (d)

Appendix 1. Monthly means (± 1 SE) for abiotic variables (temperature [^{o}C], dissolved oxygen [DO, mg/L], water depth [m], secchi depth [cm], average wind speed [km/h], channel velocity [m/s]) collected at tow sites during 2004 in the Illinois River system.

Site	Temp		DO		Depth		Secchi		Wind		Velocity	
Illinois River												
March-04	7.81	(0.56)	7.58	(0.98)	.	.	22.6	(0.6)	7.0	(2.2)	.	.
April-04	11.22	(1.10)	7.86	(0.24)	6.25	(1.75)	27.5	(5.5)	9.2	(1.3)	0.46	(0.07)
May-04	18.62	(1.07)	5.03	(0.70)	6.07	(0.54)	25.2	(4.2)	6.4	(1.3)	0.54	(0.15)
June-04	23.28	(0.44)	4.49	(0.80)	7.89	(0.14)	23.6	(3.7)	8.3	(2.2)	0.63	(0.11)
July-04	25.91	(0.54)	3.61	(0.15)	6.62	(0.34)	30.3	(3.1)	7.0	(0.8)	0.55	(0.13)
August-04	26.87	(0.87)	3.85	(0.13)	6.44	(0.69)	29.1	(0.7)	8.4	(3.3)	0.19	(0.07)
September-04	23.77	.	8.18	(0.35)	6.81	(0.57)	25.6	(2.4)	7.7	(1.0)	0.08	.
Lower Swan												
March-04	13.90	(1.75)	7.26	(1.00)	0.87	(0.25)	15.3	(1.9)	13.6	(2.7)	.	.
April-04	16.42	(2.30)	8.02	(0.55)	0.77	(0.03)	23.2	(0.5)	9.0	(1.6)	.	.
May-04	22.03	(1.31)	6.31	(0.55)	1.01	(0.07)	19.5	(1.2)	8.2	(1.8)	.	.
June-04	25.05	(1.08)	8.61	(1.53)	2.09	(0.10)	32.7	(1.5)	6.8	(1.3)	.	.
July-04	27.58	(0.77)	6.30	(0.65)	0.96	(0.20)	22.7	(3.5)	4.7	(0.6)	.	.
August-04	24.02	(0.64)	3.78	(0.27)	0.70	(0.14)	14.3	(0.8)	6.3	(3.0)	.	.
September-04	20.24	(0.64)	5.08	(1.03)	0.77	(0.03)	14.1	(1.4)	6.4	(0.6)	.	.
Middle Swan												
March-04	.	.	7.64	(0.95)	1.33	(0.05)	25.6	(7.6)	4.0	(0.4)	.	.
April-04	14.40	(0.76)	9.53	(0.65)	1.26	(0.06)	27.8	(1.0)	10.0	(2.1)	.	.
May-04	22.29	(2.01)	6.50	(0.54)	1.13	(0.10)	24.1	(3.5)	7.8	(2.4)	.	.
June-04	25.18	(0.96)	8.08	(1.73)	2.03	(0.17)	34.5	(2.2)	9.6	(1.4)	.	.
July-04	28.41	(1.35)	7.46	(0.86)	1.25	(0.26)	33.6	(6.2)	7.6	(1.1)	.	.
Calhoun Point												
March-04	.	.	7.12	(0.42)	0.68	(0.33)	28.0	(1.0)	9.3	(5.4)	.	.
April-04	.	.	6.13	(0.05)	0.80	(0.00)	19.0	(1.0)	12.9	(5.7)	.	.
May-04	25.27	.	4.97	(1.27)	0.82	(0.08)	19.6	(0.6)	12.6	(0.3)	.	.
June-04	24.85	(1.80)	6.00	(3.31)	1.45	(0.20)	44.1	(18.4)	4.7	(3.5)	.	.
July-04	29.34	(2.44)	7.10	(0.96)	0.79	(0.11)	29.1	(3.6)	6.0	(1.9)	.	.
August-04	26.63	(0.82)	5.15	(0.70)	0.77	(0.03)	18.3	(0.8)	8.9	(2.3)	.	.
September-04	21.34	.	4.45	(0.05)	0.83	(0.02)	19.0	(1.0)	14.5	(0.6)	.	.

Appendix 2. Monthly means (± 1 SE) for abiotic variables (temperature [^{o}C], dissolved oxygen [DO, mg/L], water depth [m], secchi depth [cm], average wind speed [km/h], channel velocity [m/s]) collected at tow sites during 2005 in the Illinois River system.

Site	Temp		DO		Depth		Secchi		Wind		Velocity	
Illinois River												
March-05	6.57	.	.	.	5.25	(0.75)	27.0	(3.0)	14.5	(0.4)	0.44	.
April-05	15.16	(1.37)	6.66	(1.77)	6.91	(0.41)	28.9	(6.6)	9.9	(1.9)	0.50	(0.06)
May-05	18.68	(1.72)	9.66	(2.19)	6.51	(0.22)	18.1	(4.3)	10.1	(1.1)	0.37	(0.03)
June-05	26.64	(1.10)	9.69	(1.15)	6.36	(0.22)	25.6	(1.3)	5.8	(1.3)	0.16	(0.03)
July-05	29.60	(0.36)	7.04	(0.51)	6.00	(0.14)	24.9	(1.9)	4.6	(0.7)	0.15	(0.04)
August-05	28.44	(0.53)	6.78	(1.24)	6.50	(0.23)	25.9	(3.2)	6.1	(2.1)	0.12	(0.04)
September-05	26.65	.	8.94	(0.31)	6.13	(1.38)	25.1	(0.6)	3.4	(1.1)	0.15	.
Lower Swan												
March-05	8.86	(0.33)	3.93	(0.47)	0.78	(0.03)	12.5	(0.2)	7.7	(2.7)	.	.
April-05	16.15	(2.18)	8.31	(1.42)	2.11	(1.37)	15.3	(1.8)	13.2	(0.7)	.	.
May-05	20.94	(2.35)	9.25	(1.32)	0.75	(0.03)	14.5	(1.8)	8.9	(1.7)	.	.
June-05	27.09	(0.62)	10.02	(1.61)	0.73	(0.04)	14.4	(0.6)	5.0	(3.2)	.	.
July-05	29.47	(1.34)	9.54	(1.27)	0.68	(0.06)	14.5	(1.2)	7.1	(1.2)	.	.
August-05	27.53	(1.31)	5.76	(1.18)	0.73	(0.04)	13.5	(1.0)	5.6	(1.3)	.	.
September-05	26.62	.	12.72	(1.65)	0.70	(0.09)	13.3	(0.5)	4.5	(1.4)	.	.
Middle Swan												
March-05	7.74	.	4.21	(0.32)	1.08	(0.21)	23.3	(1.2)	9.1	(2.5)	.	.
April-05	16.81	(1.89)	8.31	(1.40)	0.95	(0.07)	20.9	(1.6)	12.3	(2.4)	.	.
May-05	21.45	(2.34)	9.25	(1.13)	0.89	(0.07)	21.5	(0.5)	8.6	(2.0)	.	.
June-05	27.54	(0.91)	7.45	(1.26)	0.97	(0.03)	19.1	(2.4)	7.6	(2.0)	.	.

ASIAN CARP REPRODUCTION

SUMMARY

Identifying how temporal variation in the environment affects reproductive success of invasive alien species will aid in predicting future establishment and tracking dynamics of established populations. Asian carps of the genus *Hypophthalmichthys* have become a nuisance in recent years in the Mississippi River basin. Their populations are apparently expanding, indicating favorable reproductive conditions. During 2004 and 2005, we quantified mean density of larval Asian carps, mean monthly gonadosomatic index (GSI) of adult males and females, and number of eggs within mature females in the lower Illinois River, a major tributary of the Mississippi River. A flood (rising water to peak nearing 0.7 m/s) and drought (declining water to < 0.2 m/s) occurred during apparent spawning in 2004 and 2005, respectively. During 2004, larval Asian carps occurred during 32% of sampling weeks; mean GSI and fecundity were relatively low for adult carps, likely reflecting partially spawned individuals and perhaps low reproductive investment. During the drought of 2005, larval stages were present during only one (5%) of the sampling weeks, while mean GSI and fecundity of adults were high through summer. Females resorbed eggs rather than spawned during this year. Spawning conditions during low water appear to be unsuitable for Asian carps, inhibiting adult spawning and yielding few larvae. Spawning conditions during 2004 appeared to be comparably better but still yielded low densities of larvae relative to native fishes. Reproduction in the lower Illinois River appears to be linked to flow and its impact on adult spawning decisions, but conditions for strong year-class production (i.e., high larval densities) may be rarer than previously expected.

INTRODUCTION

Many alien species successfully invade because they possess life history traits of r-selected species, generally exhibiting rapid growth rates, short generation times, exceptional dispersal capabilities, high reproductive output early in life, high abundance in their original range, and broad environmental tolerance (Ehrlich 1984; Lodge 1993). These opportunistic characteristics allow them to reach massive population numbers soon after establishing (Lodge 1993; Williamson 1996; McMahon 2002). Two non-native, river-dwelling cyprinids with many of these characteristics are the bighead carp *Hypophthalmichthys nobilis* and silver carp *H. molitrix*. They became established in the Mississippi River drainage in the early 1980s (Freeze and Henderson 1982; Costa-Pierce 1992) and appear to be able invaders (Kolar et al. 2005).

Bighead carp and silver carp are present in at least 23 and 16 US states, respectively (Fuller et al. 1999; Kolar et al. 2005). Commercial harvest of these fishes in the Illinois River increased by 124% during 2002 (Conover et al. 2006). Recently high population growth may lead to exploitative competition with native species (Koel et al. 2000; Schrank et al. 2003). For proper and effective management of these invasive species, temporal patterns of reproduction and spawning must be determined. If reproductive success varies annually with the environment or adult condition, then this information may be useful to forecast population growth and to potentially design control measures.

No data currently link adult Asian carp reproductive status to larval production in US waters; the successful production of early life stages by adults is likely attributed to physiological and environmental factors affecting both life stages. Reproductive needs of adult bighead and silver carps are similar and have been documented as requiring water temperatures of at least 17º C, with an optimum range of 21 – 26ºC (Verigin et al. 1978; Krykhtin and Gorbach 1981;

Abdusamadov 1987; Jennings 1988; Schrank et al. 2001), and a rise in water velocity approaching or exceeding 0.7 m/s (Krykhtin and Gorbach 1981; Abdusamadov 1987). Length of river required for successful spawning by silver carp may be ≥100 km (Gorbach and Krykhtin 1980). Conventional wisdom holds that uninterrupted river and swift current prevent the semibuoyant bathypelagic eggs from sinking and being covered with silt (Soin and Sukhanova 1972; Rothbard 1981). Adults are highly fecund in their native waters (Jennings 1988; Singh 1989; Kamilov 1990; Verigin 1991) and in the invaded waters of the Mississippi River basin (Schrank and Guy 2002; Kolar et al. 2005; Williamson and Garvey 2005). Thus, ideal spawning and rearing conditions should generate large cohorts. Adults forage in the river and backwaters (see DeGrandchamp 2006), feeding on zooplankton, phytoplankton, and detritus. Presumably, energy reserves and reproductive investment of these species will be coupled with food availability, likely enhanced by inputs from the floodplain.

In many species, rapid early development of embryos and larvae improves survival and eventually recruitment to the adult population (Miller et al. 1988). Asian carp offspring follow this pattern by rapidly developing into free-swimming larvae one day post-hatch, and exogenously feeding after 72 h (Soin and Sukhanova 1972; Murty et al. 1986). River regulation and flooding patterns also should influence larval success. High recruitment likely occurs in riverine environments when rising temperatures and river stage are coupled (Junk et al. 1989). However, the impact of a flood pulse on recruitment also may be a function of flood pulse predictability and the duration and area of inundation, where rapidly developing species with general spawning requirements are most successful (King et al. 2003). Larval Asian carps seem well adapted for recruiting in river-floodplain habitats.

In this manuscript, we quantify a field pattern comparing adult reproductive status to larval fish production during a high-water (2004) and low-water year (2005) in the lower Illinois River, a large tributary of the Mississippi where populations of both species expanded explosively in the early 2000s (Chick and Pegg 2001). Although purely correlative and limited temporally, these patterns provide insight into the relative roles of (1) flow rates (i.e., is 0.7 m/s required for spawning?) and (2) adult reproductive status on the production of larval densities of these congeners in North American rivers, thereby generating hypotheses that can be tested in the future with experiments, long-term data, and other novel approaches.

METHODS

Study Site

The lower Illinois River extends between the La Grange Lock and Dam (River kilometer, Rkm 130) at Beardstown, Illinois to the confluence of the Mississippi River (Rkm 0) at Grafton, Illinois. During spring through summer 2004 and 2005, we sampled Asian carps between Rkm 0 through Rkm 19 and within Swan Lake, a large, adjacent, and continuously connected backwater (1,100 ha) at Rkm 8. Adults occasionally move into Swan Lake (DeGrandchamp 2006) and larvae from the river become entrained within it (Csoboth 2006). The lower Illinois River is undammed, but has been channelized for navigation and is influenced by the Mel Price Lock and Dam in the Mississippi River downstream of the confluence. Despite a century of alterations due to dredging, water diversion from Lake Michigan, channelization, and levee construction, the river still retains an annual flood pulse (Karr et al. 1985; Sparks 1995). The Illinois River flooded its banks during late spring through summer 2004, while, in contrast, water levels declined during the same period in 2005 (DeGrandchamp 2006; Csoboth 2006; Figure 1).

Adult Fish Sampling

We collected adult *Hypophthalmichthys* spp. during April through October 2004 and March through August 2005 typically with experimental trammel nets (5.08 cm, 7.62 cm, 10.16 cm and 8.89 cm bar mesh panels; 3.66 m, 3.96 m, 4.27 m and 3.96 m outer wall respectively; 91.44 m in length); hoop nets (3.81 cm bar mesh, 0.91 m diameter fiberglass hoops), trap nets, and fish jumping into the boat also were sources. For each fish, species, sex, total length (TL, mm), and wet weight (Wt, g) were quantified. Gonads were removed from the fish to determine gender and weighed (wet mass, g). Ovaries were preserved in 10% buffered formalin. Oocytes within three, one-mL samples of each ovary sample were counted and the mean was multiplied by the total volume of the ovary sample to estimate number of eggs per female (hereby called ‘egg quantity’; Crim and Glebe 1990).

For each species, sex-dependent adult lengths were compared between years using two-way ANOVA (factors: sex and year; (proc GLM, SAS Institute 1999). Spawning periodicity was quantified by tracking the average weekly gonadosomatic index [GSI =100* wet gonad weight (g)/wet body weight (g)] (Crim and Glebe 1990) through time. A t-test was used to test differences in GSI between years for males and females of both species. Pearson correlation was used to test the relationship between female bighead and silver carp GSI for 2004 and 2005. Linear regression was used to determine the relationship between total length and egg quantity, and weight and egg quantity for each species for 2005. Body-size versus egg quantity data for 2004 were excluded from this analysis because many fish appeared to have completely or partially spawned, thus rendering size-dependent relationships inaccurate. ANOVA with

Tukey's HSD (proc GLM, SAS Institute 1999) was used to test differences in egg quantity among years for bighead and silver carps.

Larval Fish Sampling

During late March through September 2004 and 2005 (2004: 21 weeks; 2005: 22 weeks), surface ichthyoplankton tows were conducted weekly on the lower Illinois River and Swan Lake to quantify spawning of Asian carps. Weekly tows were conducted along four randomly chosen transects in the Illinois River (Rkm 8.0 and 1.6) and also along four transects in Swan Lake with a pair of bow-mounted ichthyoplankton nets (0.5 m-diameter x 2-m long, 500-µm mesh). A calibrated mechanical flow meter (General Oceanics [GO] Inc, Model 2030R) was mounted in the mouth of one net to estimate volume sampled (about 100 m^3 water sampled per transect). In the river, tows were conducted perpendicular to flow by beginning at the main channel border and sampling across the main channel. In Swan Lake, two tows were conducted near the shore and two near the center of the lake. Flow rate was quantified in the river channel during each sampling event using either a Marsh-McBirney electronic flow meter or the mechanical GO unit near the surface. Velocities (m/s) were averaged across sampling weeks within each month.

Net contents were preserved in 95% ethanol in individually labeled jars. Samples were subsampled using a Folsom plankton splitter (Aquatic Research Instruments, Hope, Idaho, USA) so approximately 200 fish per sample were processed and counted. Asian carp larvae were identified using voucher specimens (Colorado State University, Larval Fish Laboratory, Darrel Snyder, Fort Collins, Colorado, USA). Larvae were difficult to key to species and hybrids do occur (Kevin Irons, Illinois Natural History Survey, Havana, Illinois, personal communication), so we restricted analyses to genus. Data were standardized by volume of water sampled.

Densities from each pair of nets per transect were averaged. On each date, a single average was then calculated across all river and Swan Lake transects (i.e., 8 sample sites per date). Finally, monthly means were calculated across those daily averages. A one-way repeated measures ANOVA (proc GLM, SAS Institute 1999), was used to test for differences in mean monthly densities between years on $\log_{10}(x+1)$ transformed data. To quantify patterns among weeks, the presence or absence of Asian carp larvae in tows were compared using a two-tailed, binomial test where the expected distribution across weeks was generated from 2004 and compared to the observed distribution during 2005.

RESULTS

Female bighead carp were larger than males during both years (two-way ANOVA: sex, $F = 23.6$, df = 1, 349, $P < 0.0001$; sex X year, NS; Figure 2). Bighead carp were longer during 2005 than 2004 (two-way ANOVA: year, F = 23.6, df = 1, 349, $P < 0.0001$; Figure 2). Female silver carp were longer than males during 2005 (two-way ANOVA: sex, $F = 12.6$, df = 1, 134, $P = 0.0005$), but a sex* year interaction (two-way ANOVA: sex, $F = 7.5$, df = 1, 134, $P = 0.007$) indicated that average lengths were similar between sexes in 2004 (Figure 2). Mean lengths of silver carp did not differ between years (two-way ANOVA: year, NS, Figure 2).

Mean GSI showed no seasonal pattern, with individuals having high values as late as September (Figure 3). Bighead and silver carp GSI (averaged across all individuals caught during the sampling season) differed between 2004 and 2005 for females ($F = 58.36$, df = 1, 235, $P < 0.0001$; Figure 4) and males ($F = 15.57$, df = 1, 254, $P = 0.0001$; Figure 4). Monthly mean GSIs were positively correlated between female bighead carp versus female silver carp ($r = 0.87$, $P = 0.02$), while not being correlated between male bighead carp and male silver carp. Bighead and silver carp egg quantities differed between years (bighead $F = 35.16$, df = 1, 89, $P < 0.0001$; silver $F = 22.84$, df = 1,28, $P < 0.0001$). Mean (± SE) number of eggs per female in 2004 were $1.8 \pm 0.3 \times 10^5$ and $2.8 \pm 0.5 \times 10^5$ for bighead and silver carp and, in 2005, increased to $7.5 \pm 0.6 \times 10^5$ and $16 \pm 2.0 \times 10^5$. In 2005, bighead carp egg quantity increased with total length (# eggs = 2226.67 * TL – 1261512, $r^2 = 0.28$, $P < 0.0001$), and with weight (# eggs = 128.47 * Wt – 315083, $r^2 = 0.38$, $P < 0.0001$). Silver carp egg quantity in 2005 also was positively related to weight (# eggs = 325.14 * Wt – 581908, $r^2 = 0.24$, $P = 0.04$), but not with total length ($r^2 = 0.07$, $P = 0.30$). Of 137 bighead and silver carp females sampled in 2004, 19% had ovaries with mature oocytes and 5% appeared to have completely spawned. During 2005, 51% of 99 females sampled contained mature oocytes, but by fall, no female had appeared to spawn. About 27% of females in 2005 appeared to be resorbing their oocytes.

Larval and early juvenile Asian carps were present in ichthyoplankton nets in both the river (mean ± SD number/m^3 across dates where larvae were present: 0.03 ± 0.03) and Swan Lake (0.13 ± 0.25) during May through July 2004, peaking at an average near 0.06/m^3 across river and lake sites combined in May (Figure 3). Conversely, larvae appeared during only 1 week of sampling and only in Swan Lake during 2005 (Figure 3). On average, Asian carp larval densities were typically low during both 2004 (across months of May through July: mean number/m^3 ± SE = 0.03 ± 0.02) and 2005 (0.0006 ± 0.0006); Asian carp larvae appeared in larval tows on 32% of sampling weeks during 2004 and only 5% of weeks during 2005 (binomial test: $P = 0.008$; Figure 3).

DISCUSSION

During the two very different years, river flow appeared to influence spawning of the adult invaders. These results are consistent with reports from native waters in Asia, where an increase in activity and movement was associated with river stage (Krykhtin and Gorbach 1981; Abdusamadov 1987). Yi et al. (1988) (also see translations in Chapman 2006) discovered Asian carp eggs during the rise in river stage of the Yangtze River, China. In the Illinois River, river stage was the primary cue for movement of adults during the spawning season in the Illinois River (DeGrandchamp 2006). Adult reproductive condition (i.e., GSI and egg quantity) differed between years as well. We explore how river stage and adult reproduction were apparently related and thereby influenced the production of larvae in the Illinois River.

Adult Response to Flow

Adult reproductive condition differed dramatically between years for both species and likely affected larval output. Adults may use specific environmental conditions (such as river stage or temperature) to cue maturation and then initiate spawning. A lack of certain criteria would in turn adversely affect reproductive output of the adults, resulting in a lack of input to the system for that year. Spawning criteria documented to be associated with flow (rising water approaching 0.7 m/s; Krykhtin and Gorbach 1981; Abdusamadov 1987) for both species were only met in 2004. Reproductive characteristics in 2004 included relatively low GSI values for males and females, overall relatively low fecundity, and evidence of spawned-out fish. Previous spawning activity likely contributed to the lower fecundity and GSIs this year, although poor condition of non-spawning adults also may have been a mechanism. No spawned-out fish were observed in 2005, but many fish of both species had apparently started to resorb eggs, and had relatively higher fecundity and GSI values. The declining flow in 2005 may have curbed spawning in the river, yielding higher observed fecundity and GSI values. Retention of eggs often results in follicular atresia, which is a common phenomenon in teleosts that can be induced by stress, fasting, or environmental conditions, such as those not conducive to spawning (Nagahama 1983; Linares-Casenave et al. 2002). During 2005, shovelnose sturgeon *Scaphirhynchus platorynchus* in the Mississippi River also contained eggs that were apparently being resorbed rather than spawned (Sara J. Tripp and J.E. Garvey, Southern Illinois University, unpublished data). Thus, it is plausible that spawning and larval production were curbed in 2005 relative to 2004 across many species that require adequate flow to spawn.

Larval Response to Flow

Following spawning, it is possible that an annual flood pulse is necessary to keep eggs suspended in the water column, therefore increasing the chance of larval survival. With the exception of 2005, the Illinois River flooded during the apparent spawning season every year since the late 1990s (Figure 5), perhaps facilitating the recent increase of Asian carp populations in this system. In addition to keeping eggs and larvae suspended, it is likely that the regular inundation of the floodplain environment with each seasonal flood created suitable habitat for larval and juvenile fish, enhancing survival and recruitment (see Csoboth 2006).

Although we found nearly no larval production in the Illinois River during the low water year, there is still some speculation about adequate flow (i.e., 0.7 m/s) being the driving force behind successful reproduction. Kolar et al. (2005) cited an instance where bighead carp eggs were inadvertently sampled in a sediment study, and the eggs, although covered in mud, hatched and survived for four days. Our group also found recently hatched Asian carp larvae in an isolated, unconnected backwater of the Illinois River (Garvey et al. 2005). Further, the lower Illinois River, characteristic of a low-gradient stream, appears to rarely meet 0.7m/s throughout the spring and summer; only during times of high flooding does it appear to exceed this velocity. Even during the relatively high water of 2004 (c.f., 2005), the water velocities only approached 0.7 m/s during one week in June, although larvae occurred across several months (May through August). Thus, high river stage may augment egg and larval survival but not be critical for reproductive success.

Adult Condition

Although it is most likely that the interaction among adult physiology, environmental conditions, and adult behavioral decisions drove spawning, maternal condition of the adults also may affect reproductive output (Madenjian et al. 1996). A threshold maternal condition is presumably met when the enrichment of the floodplain environment supplies more food for adults, leaving them in high over-wintering condition. This condition also would be reflected during the following year by enhancing fecundity and reproductive potential. We saw this result in both species, with fecundity positively related to weight in 2005 following the 2004 flood year. If environmental conditions were favorable for adult spawning in 2005, then we might have quantified high production of larvae that year, given the high reproductive potential in the adults. However, poor larval production through time and apparent resorption of eggs suggest that good maternal condition is necessary but not sufficient for successful reproduction.

Implications

During this effort, both the bighead carp and silver carp populations in the Illinois River were dominated by cohorts produced during 2000 (Garvey et al. 2007). Apparently year-class strength in these species varies tremendously and probably depends on congruence among environmental conditions and adult spawning. In reality, neither 2004 nor 2005 likely were strong reproduction and recruitment years. Peak densities of all larval fish taxa combined in the Illinois River exceeded 10/m^3 during spring of both years (Csoboth 2006); Asian carp larvae were about 100 times less abundant relative to total larval abundance at its peak in 2004. Water levels in the Illinois River during the spring through summer of 2000 was intermediate between that in 2004 and 2005 (Figure 5), suggesting that flow was not exclusively responsible for the marked population expansions during that year. Conditions for successful adult spawning and larval production might be rarer than expected in the Illinois River, requiring congruence among adult maternal condition, a stable spring river rise, and perhaps sufficient flow for larval development.

Population growth in the Illinois River is likely driven by the high reproductive potential of both species and their relatively long life spans (see Williamson and Garvey 2005), which is necessary for persistence in variable environments (i.e., the storage effect, Warner and Chesson 1985). The regular spring flood pulse in the Illinois River probably serves to provide a predictable cue for spawning during most years. Even if spawning is not successful in the Illinois River, this system is near unimpounded reaches of the lower Mississippi and Missouri Rivers. Recent telemetry work in these systems has demonstrated that both species move widely during spring, spanning river reaches and crossing lock and dams (DeGrandchamp 2006). Thus, even if populations are poorly sustained in the Illinois River, then individuals from other reaches will continue to invade, potentially stabilizing population dynamics.

In our view, Asian carp populations within the Illinois River have high reproductive potential, but realization of this potential varies widely with climate and connectivity to other populations. At the regional and perhaps range-wide scale, this suggests that variable local responses to reproductive conditions are ameliorated by environmental heterogeneity among reaches. Key to predicting population dynamics and perhaps controlling these species is a refined understanding of the relative roles of connectivity, movement, and reach-specific reproduction in the Mississippi River basin.

Acknowledgments

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Figure captions:

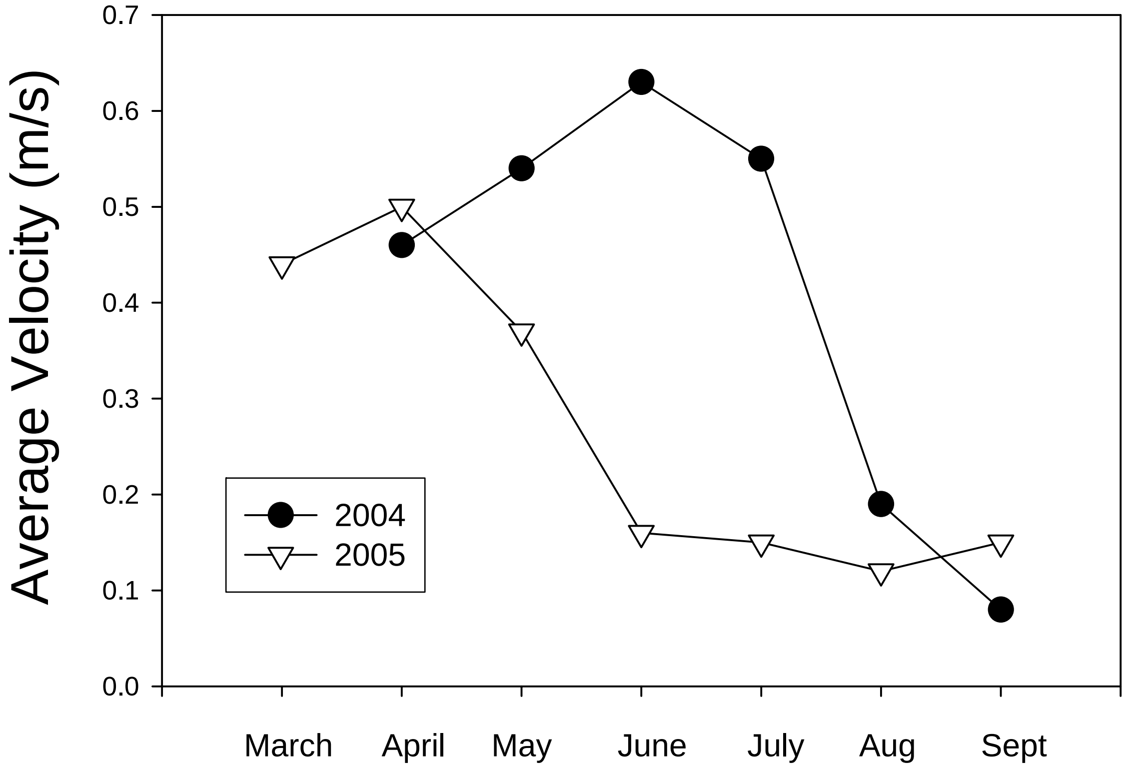
Figure 1. Mean monthly flow velocities (m/s) at larval fish tow sites (four per week then averaged across weeks) in the Illinois River during 2004 and 2005. Flow rates were averaged across sites and then among sampling dates. Standard error bars are smaller than each symbol.

Figure 2. Mean (+ 1 standard deviation) lengths of adult bighead and silver carp collected in the lower Illinois River during spring through summer 2004 and 2005.

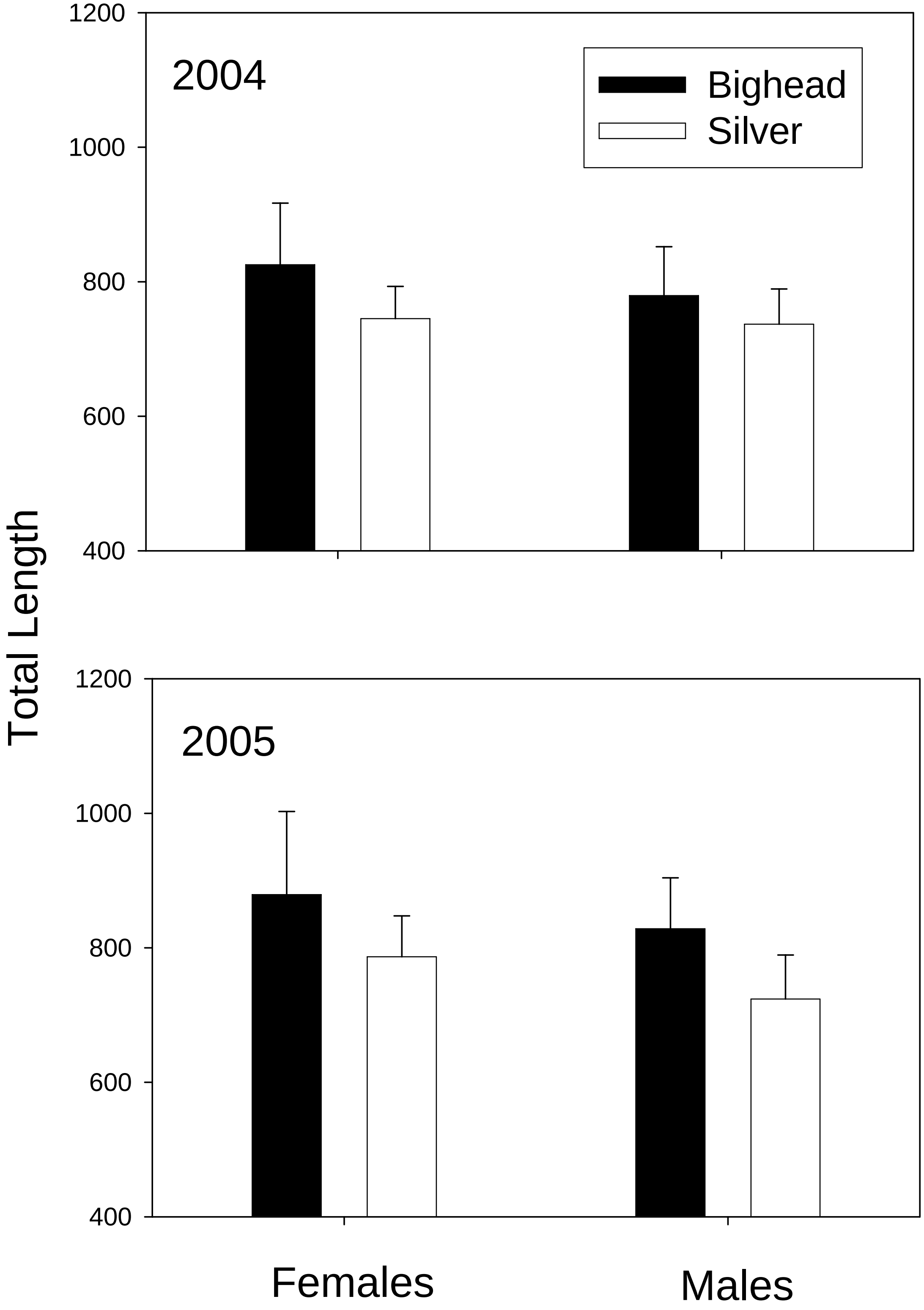
Figure 3. Seasonal patterns of mean monthly female gonadosomatic index (GSI) values for bighead and silver carp during 2004 and 2005 in the lower Illinois River. Male GSIs (not shown) also varied among months with no discernable peak. Mean larval densities of *Hypophthalmichthys* spp. (number per m^3) in the main channel Illinois River also are plotted.

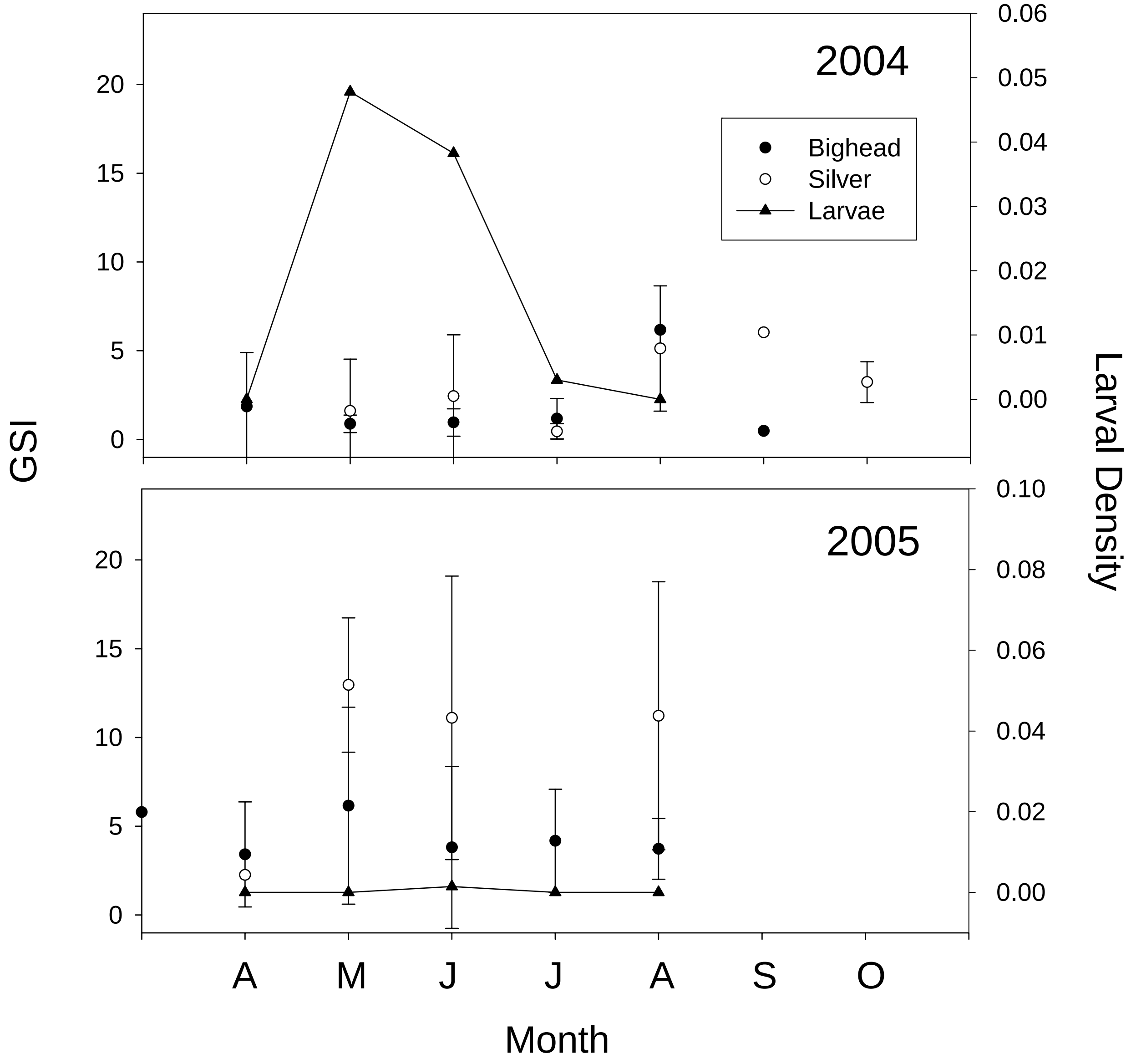
Figure 4. Mean GSI (+ 1 SE) of female (top) and male (bottom) bighead and silver carp from the lower Illinois River and Swan Lake during 2004 and 2005. Data were combined because species showed similar patterns between years. Females: 2004: *N*=137; 2005: *N*=99; Males:2004: *N*=153; 2005: *N*=102. GSI = (gonad weight/body weight) * 100.

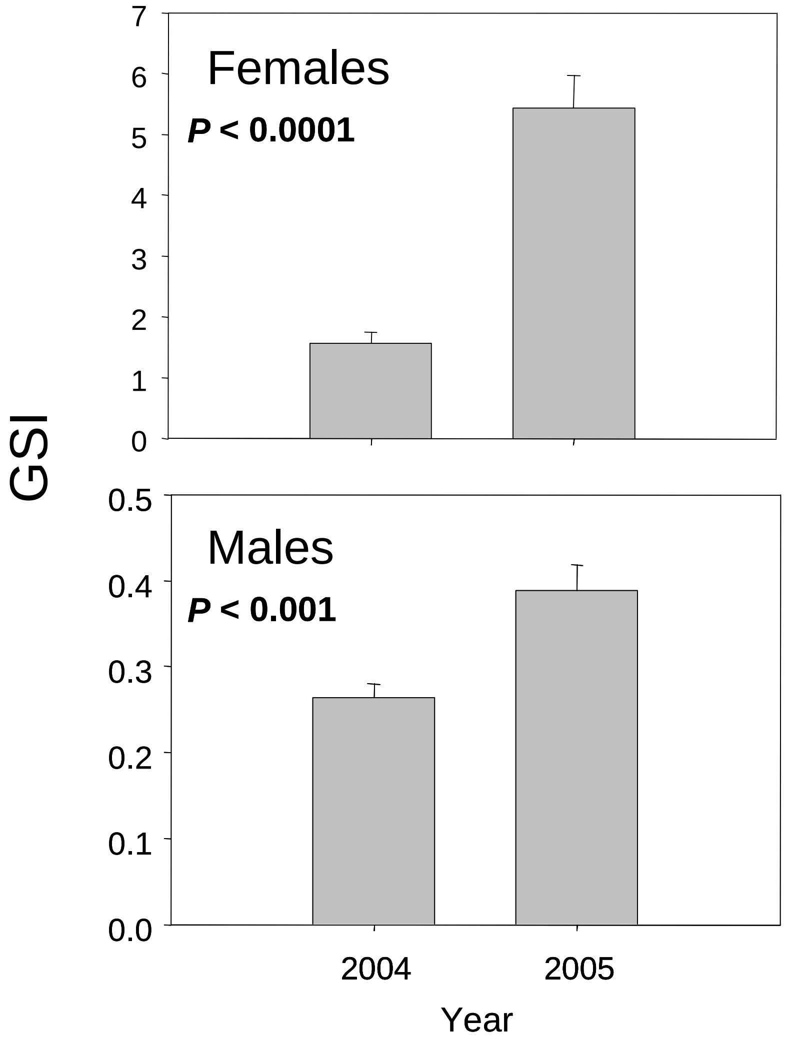
Figure 5. Maximum daily gage height (m) at the USGS gaging station at Hardin, Illinois River during January 1998 through January 2006. The dashed line is the gage height at which the river reaches flood stage. Shaded bars bracket the months during each year (May through October) when Asian carps may be spawning.

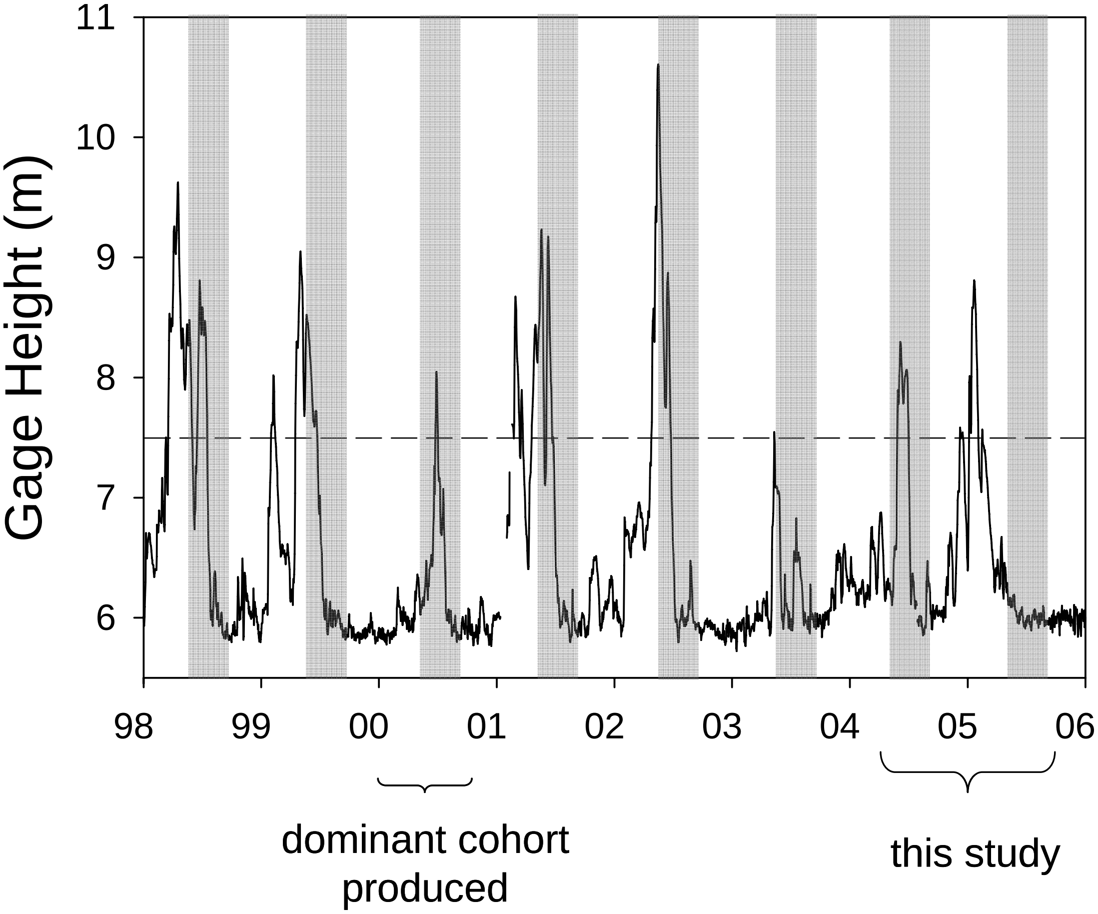


Average Velocity (m/s)
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
March
April
May
June
July
Aug
Sept
2004
2005


1200
1000
800
600
400
2004
Bighead
Silver
1200
1000
800
600
400
2005
Total Length
Females
Males


2004
2005
Bighead
Silver
Larvae
GSI
Larval Density
Month
A
M
J
J
A
S
O
0
5
10
15
20
0.00
0.01
0.02
0.03
0.04
0.05
0.06
0.08
0.10


GSI
Females
P < 0.0001
0
1
2
3
4
5
6
7
Males
P < 0.001
0.0
0.1
0.2
0.3
0.4
0.5
2004
2005
Year


11
10
9
8
7
6
Gage Height (m)
98
99
00
01
02
03
04
05
06
dominant cohort produced
this study

Section 5:

DEMOGRAPHIC RESPONSES OF FISHES

Colby Wrasse, Greg Conover, Nate Caswell, and Rob Simmonds

U.S. Fish and Wildlife Service
Carterville Fishery Resources Office
Marion, Illinois

Introduction

As part of the Swan Lake Habitat Rehabilitation and Enhancement Project (HREP), the US Fish and Wildlife Service, Carterville Fishery Resources Office (FRO) monitored a portion of the fisheries response to the project. Specifically we examined population age and size structure of selected species to assess fish reproduction and recruitment with and without the project, and under three water management regimes (Objective 2 in Scope of Work).

Swan Lake is a 1,200 ha Illinois River backwater lake located at Two Rivers National Wildlife Refuge in Calhoun County, Illinois. The Swan Lake HREP divided the lake into three separate compartments (Upper, Middle, and Lower). The water regimes of the individual compartments are independently managed through the use of stop-log gates and pumps.   Periodic drawdowns of the compartments are intended to improve fish and wildlife habitat by compacting sediments and promoting the growth of aquatic vegetation. Examining the fisheries population and growth characteristics of Middle and Lower Swan lakes, selected reference lakes, and historical data (Swan Lake 1994) provided insight into the effectiveness of the Swan Lake HREP and current water management practices.

Black crappie (*Pomoxis nigromaculatus*), bluegill (*Lepomis macrochirus*), white crappie (*Pomoxis annularis*), and freshwater drum (*Aplodinotus grunniens*) were selected as target species for age and growth analyses. These species were selected because they were relatively abundant during pre-project conditions and represent different reproductive guilds (Balon 1975). These species were expected to benefit from improved habitat conditions in Swan Lake. Black crappie are guarding, nest-spawning phytophils. These fish construct nests on the bottom, around algae and plant roots. White crappie are guarder substrate choosers, which deposit their eggs in algal growth, and alternatively will utilize a variety of aquatic plants. Bluegill are guarder nest spawning lithophils which deposit eggs on hard surfaces that have been previously cleaned. Freshwater drum are open substrate pelagophils that scatter non adhesive floating eggs in open water. White crappie reproduction, and to a lesser extent black crappie, could be expected to benefit from improved aquatic plant communities. Both species of crappie and bluegill would benefit from hardened and otherwise improved substrate conditions. Freshwater drum may be expected to respond to stable open water conditions.

Other studies have demonstrated that periodic drawdowns and dewatering of lakes can improve the condition of the littoral substrate, aquatic vegetation, macroinvertebrate production, and overall fish community (Wegner and Williams 1974; Moyer et al 1995). In many lakes and reservoirs, water level management and habitat manipulations are essential tools for improving desirable fish communities (Beam 1983; Moyer et al. 1995; National Research Council 1992).

In addition to research on target species, the scope of work called for evaluation of the population structures of bighead carp (*Hypophthalmichthys nobilis*) and silver carp

(*Hypophthalmichthys molitrix*). Summary of these data is attached as an addendum to this report (Appendix A).

Methods

We coordinated with the Illinois Natural History Survey (INHS) for the collection of fish samples for age analysis of target species. The INHS collected fish from Middle and Lower Swan lakes using standard fyke nets in single and tandem sets, from July 2004 through September 2006. The INHS also collected fish from the reference backwater lakes during the fall of 2004 and summer 2006 using daytime electrofishing, standard fyke nets, and minnow fyke nets. The reference lakes sample was a composite of several pool 26 backwater lakes including: Ellis Bay, Airport, Portage Island, Piasa, Brickhouse, Luesse, and Long lakes. Pre-project fish were collected by Carterville FRO from Swan Lake in April 1994 using standard fyke nets in single and tandem sets.

All fish were measured to the nearest millimeter and weighed to the nearest gram, when possible. Relative weights (W_r) were calculated for each fish. Relative weight is an index of condition based on the relationship between the actual weight of a fish of a given length and a “standard” weight developed for a given species (Murphy and Willis 1996).

Sagittal otoliths were removed from each fish and halved through the center of the nucleus. The otoliths were submerged in glycerin, and annuli were counted under a dissecting microscope with an independent light source. Two readers aged fish independently without knowledge of the other reader's age estimate. When readers disagreed, a third party aged the fish, and a consensus was reached.

Because samples from Middle and Lower Swan lakes were collected throughout the year, and reference lake fish were collected from early summer through fall, back-calculating length at age was necessary to make meaningful comparisons. Back-calculation is a technique that provides information on past growth of a fish based on the relationship between the radius of a hard part (e.g. otolith or fin ray cross-section) and fish length (Murphy and Willis 1996). Back-calculation measurements were performed using Scion Image software. Measurements were taken along a straight-line axis from the center of the nucleus to the dorsal edge of the otolith. Measurements to each annulus were taken at the center of the opaque band. Back-calculated lengths (BCLs) at age were found using the Fraser-Lee model with an intercept of zero, also known as Dahl-Lea model or direct proportion method (Murphy and Willis 1996).

Several researchers have demonstrated the validity of this method for back-calculating length at age for centrarchid species. Klumb et. al. (2001) found that when aging bluegill x green sunfish hybrids, the Frasier-Lee model (with zero intercept) produced more accurate BCLs than the Weisberg model. The Frasier-Lee model with a zero intercept was found to be more reliable than intercept-corrected direct proportion or regression methods when back-calculating the lengths of largemouth bass from otoliths (Schramm et al. 1992). Boxrucker (1986) found no significant difference in BCLs of white crappie when comparing otolith to scale BCLs. Back-calculated lengths for white crappie, ages 1 to 3, were found to be similar to length-frequency data collected from rotenone samples (Maceina and Betsil 1987). Heidinger and Clodfelter (1987) compared otolith BCLs of three species (smallmouth bass, striped bass, and walleye) using three different methods: log-log intercept, linearly derived intercept, and assumed zero intercept. They found that none of the various methods consistently affected whether or not the back-calculated mean total length and the corresponding empirical mean total length was significantly different. Back-calculated lengths for smallmouth bass using the Fraiser-Lee method (zero-intercept) were similar to empirical lengths (Heidinger and Clodfleter 1987).

Using BCLs calculated by the Fraiser Lee (zero intercept) model allowed us to compare samples collected throughout the growing season which was necessary to achieve adequate sample sizes and make meaningful comparisons. One potential drawback to using back-calculated lengths is Lee’s phenomenon (Murphy and Willis 1996). Lee’s phenomenon can cause BCLs of older fish to be smaller or larger than BCLs of younger fish in the sample. Since the great majority of the fish in our samples were less than 3 years of age, the effects of Lee’s phenomenon should be minimal.

Because Lower Swan Lake did not completely refill until 2003, we could be assured that any fish from year classes before 2003 were not hatched and reared in Lower Swan Lake. Therefore, statistical comparisons of back-calculated lengths excluded fish from year classes before 2003 and focused only on year classes 2003-2005. Data from year classes previous to 2003 were only used in an illustrative manner.

All data analyses were conducted using JMP Statistical Software (SAS Institute 2000). Mean back-calculated length at age and relative weight were compared across sites using one-way analysis of variance (ANOVA), Tukey-Kramer HSD. Statistical tests were performed using a significance value of $\alpha = 0.05$. To make for better readability and organization, we included results and discussion sections for each target species, while the summary section is a big picture overview and includes all species and sites.

Due to the nature of the samples collected, many of the analyses specified in the scope of work (SOW) were not possible or valid. The original SOW called for monitoring of all three Swan Lake compartments, but no sample was collected from Upper Swan Lake due to lack of water in this compartment. We could not construct von Bertalanffy growth curves since at least four age classes are needed to construct these curves and we had only three age classes to work with. Instead we used mean back-calculated length at age to assess growth. Annual mortality estimates were not valid since the populations we examined contained only young fish (ages 0-3), and any mortality estimates would be largely skewed due to our significant removal of fish. The SOW indicated that we would assess recruitment rates by indexing the abundance of age-0 and age-1 fish, but due to gear and seasonal biases age-0 (young-of- year) fish were greatly underrepresented in all samples. Because of this, young-of-year (YOY) fish were excluded from all analyses. However, we did get a rough idea of recruitment success by looking at the relative abundance of age-1 fish. For the purposes of this report, we considered fish to be recruited if they had survived one winter period.

Black Crappie

Results

Age Structure

Prior to the HREP, the Pre-Project Swan Lake (1994) black crappie catch contained 50% age-3 and older fish (Table 1). Conversely, post HREP the black crappie catches in both Lower and Middle Swan lakes were dominated by age-1 and age-2 fish, with lower percentages (< 8%) of age-3 or older black crappie. The reference lakes black crappie sample was composed of 17% age-3 and older fish. Lower Swan Lake (51%), reference lakes (39%), and Pre-Project Swan Lake (1994) (30%) contained relatively high percentages of age-2 black crappie, while the Middle Swan Lake sample contained only 8% age-2 black crappie. Maximum age for black crappie collected from Pre-Project Swan Lake (1994) was 7-years while no black crappie older than age-5 were collected at any site from 2004-2006.

The Pre-Project Swan Lake (1994) black crappie catch had nearly equal numbers of young fish (age 1-2) and adult fish (age 3-7) (Figure 1). In 2004, the black crappie samples from both Lower Swan Lake and Middle Swan Lake were dominated by age-1 fish (Figure 2-3). However, in 2005 age-2 was the most common age class of black crappie in Lower Swan Lake while age-1 was still the most common in Middle Swan Lake. By 2006, age-3 was the most common age class of black crappie in Lower Swan Lake, and there were relatively few age-1 black crappie. Conversely, in 2006 age-1 remained the dominant age class in Middle Swan Lake. Black crappie sample size for the reference lakes was too small to make comparisons between years (Figure 4).

Table 1. Summary of the black crappie sample for Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006). TL is fish total length. Percent (%) is an age group's relative contribution to the total catch for that species.

	Age (yr)	N	%	Mean back-calculated TL	SD	Range
Pre-Project Swan Lake (1994)	YOY					
	1	43	19.2	99	11	75-123
	2	68	30.4	189	13	152-220
	3	21	9.4	214	25	154-269
	4	84	37.5	259	28	203-328
	5	4	1.8	286	15	267-302
	6	3	1.3	267	87	168-331
	7	1	0.4			314
Lower Swan Lake (2004-2006)	YOY	17	8.5			
	1	65	32.7	106	15	58-155
	2	102	51.3	183	24	101-235
	3	12	6.0	233	40	165-284
	4	2	1.0			245-271
	5	1	0.5			301
	6					
	7					
Middle Swan Lake (2004-2006)	YOY	24	15.8			
	1	104	68.4	116	22	75-165
	2	12	7.9	206	27	144-249
	3	11	7.2	241	32	199-300
	4	1	0.7			240
	5					
	6					
	7					
reference lakes (2004-2006)	YOY	5	12.2			
	1	13	31.7	134	31	66-197
	2	16	39.0	229	37	135-278
	3	6	14.6	241	22	220-281
	4					
	5	1	2.4			316
	6					
	7					

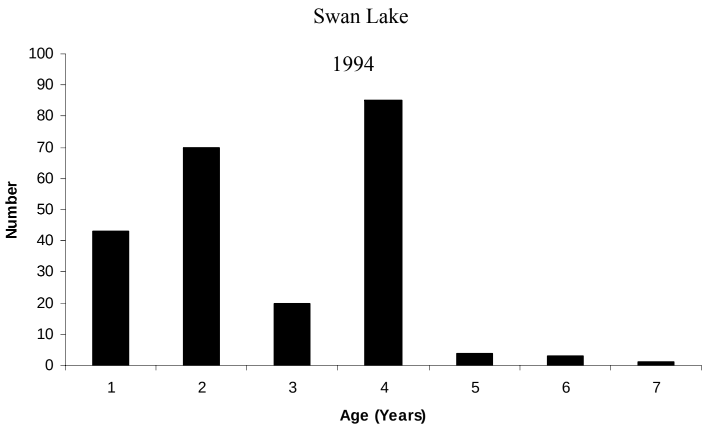


Figure 1. Age distribution of black crappie collected from Pre-Project Swan Lake (1994).

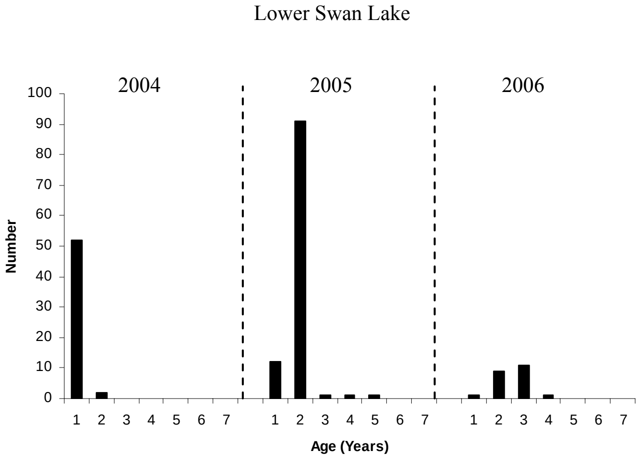


Figure 2. Age distribution of black crappie collected from Lower Swan Lake (2004-2006).

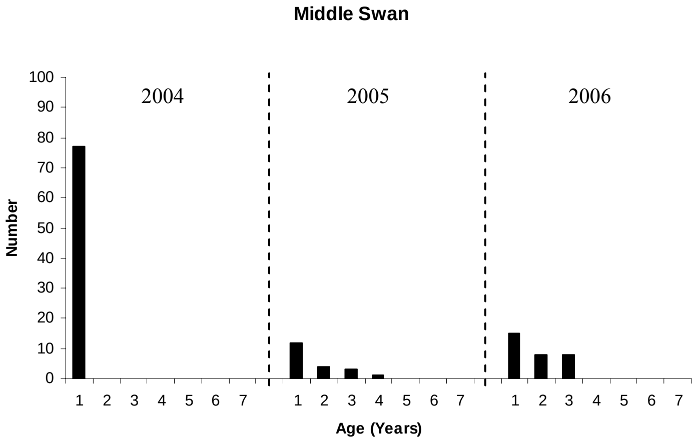


Figure 3. Age distribution of black crappie collected from Middle Swan Lake (2004-2006).

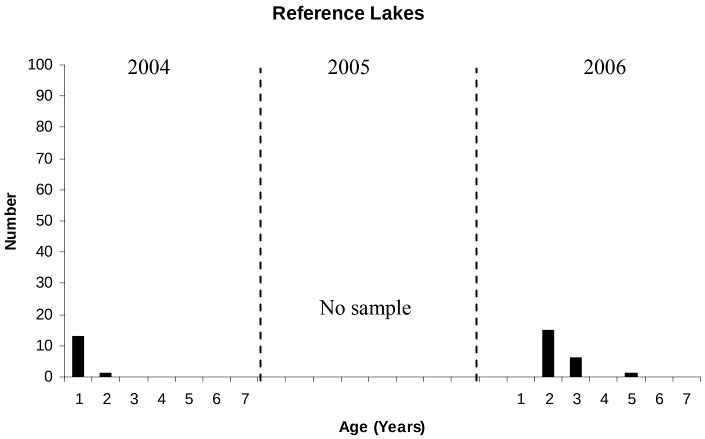


Figure 4. Age distribution of black crappie collected from reference lakes (2004 and 2006).

Length at Age

Pre-Project Swan Lake (1994) and Lower Swan Lake had similar age-1 mean lengths (Table 2). Age-1 mean length for Middle Swan Lake was significantly different than all other sites. Age-1 mean length for reference lakes was significantly different than all other sites ($P<0.01$).

Reference lakes had the highest age-2 mean length. Age-2 mean length for the reference lakes was significantly different when compared to all other sites. Middle Swan Lake had the second highest age-2 mean length. Age-2 mean length for Middle Swan Lake was significantly different when compared to all other sites ($P<0.01$).

Reference lakes and Middle Swan Lake had the highest age-3 mean lengths while Pre-Project Swan Lake (1994) had the lowest age-3 mean length. However, these differences were not significant ($P=0.11$).

Graphical representation of black crappie length at age illustrated the similarity between Swan Lake 1994 and Lower Swan Lake for all age classes (Figure 5). Reference lakes and Middle Swan Lake had greater lengths at age-1 and age-2, but lengths were similar for age-3.

Table 2. Mean back-calculated length at age for black crappie (ages 1-3) collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

Site	age-1 mean back-calculated length (mm)	age-2 mean back-calculated length (mm)	age-3 mean back-calculated length (mm)
Pre-Project Swan Lake (1994)	99	189	215
Lower Swan Lake (2004-2006)	106	183	233
Middle Swan Lake (2004-2006)	116[a]	205 [c]	241
reference lakes (2004-2006)	134[b]	229[d]	241

[a] Significant difference detected between Middle Swan Lake and all other sites.

[b] Significant difference detected between reference lakes and all other sites.

[c] Significant difference detected between Middle Swan Lake and all other sites.

[d] Significant difference detected between reference lakes and all other sites.

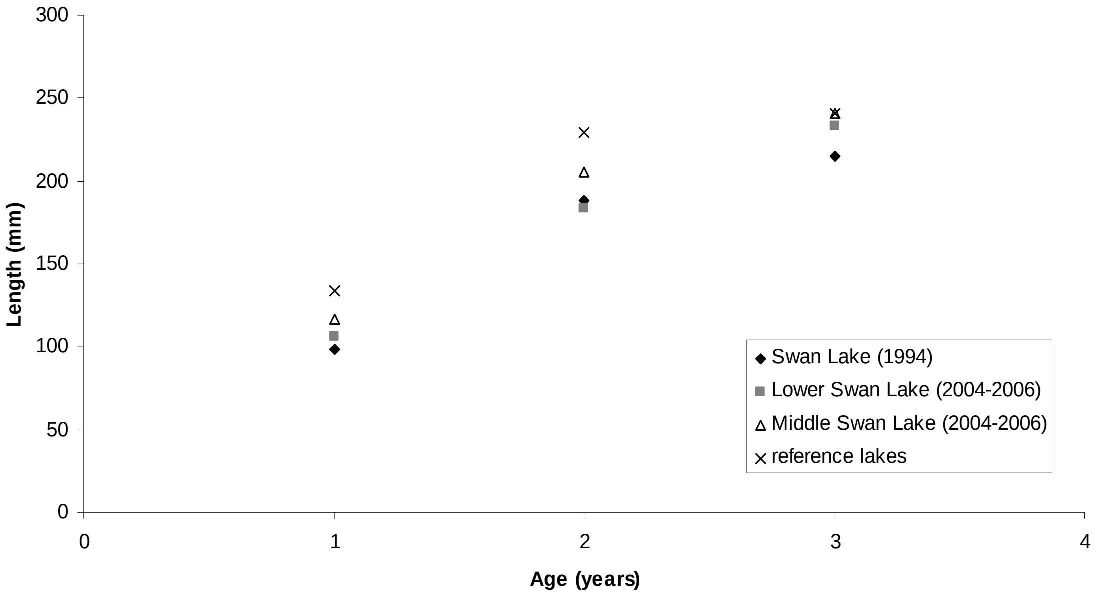


Figure 5. Mean back-calculated length (mm) at age (years) for black crappie (ages 1-3) collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

Relative Weights

Mean relative weights (W_r) for all sites were above 100 (Table 3). Lower Swan Lake and Middle Swan Lake mean W_r values were similar. Mean W_r values for Pre-Project Swan Lake (1994) and reference lakes were significantly lower ($P<0.01$) than the mean W_r values of Middle Swan Lake or Lower Swan Lake.

Table 3. Mean relative weights (W_r) for black crappie collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

black crappie (*Pomoxis nigromaculatus*)					
Site	Number	Mean W_r	Std. Error	Lower 95%	Upper 95%
Pre-Project Swan Lake (1994)	202	103.2[b]	0.9	101.4	105.1
Lower Swan Lake (2004-2006)	63	123.5	1.7	120.3	126.8
Middle Swan Lake (2004-2006)	68	126.8	1.6	123.6	129.9
reference lakes (2004-2006)	40	110.2[a]	2.1	106.2	114.3

[a] Significant difference detected between reference lakes and all other sites.

[b] Significant difference detected between Pre-Project Swan Lake (1994) and all other sites.

Discussion

Prior to the HREP project, the Swan Lake black crappie catch was fairly balanced with nearly equal numbers of young (ages 1-2) and old fish (ages 3-7). Immediately after completion of the HREP and 2002 drawdown, the Lower Swan Lake black crappie shifted towards a catch dominated by young fish. In 2004, nearly all black crappie collected were age-1. Paller (1997) noted that the size structures for some species shifted towards more small (young) fish after a reservoir drawdown. By 2005, nearly all black crappie collected were age-2. The preponderance of age-1 fish in 2004 and age-2 fish in 2005 indicated a strong 2003 year class of black crappie. By 2006, the 2003 year class (age-3 fish) was still the most common year class of black crappie with few age-1 fish in the Lower Swan Lake sample.

The relative strength of the 2003 year class, in comparison to the 2004 or 2005 year classes, indicated that black crappie may have enjoyed strong recruitment immediately after Lower Swan Lake had refilled. Reduced interspecific and intraspecific competition, relatively few predators, and improved spawning habitat through sediment compaction may have factored into the relatively high recruitment of the 2003 year class of black crappie. It was possible that there was strong recruitment of the 2003 year class throughout the river system and the relatively higher recruitment we noticed in Lower Swan Lake was due to factors beyond the HREP. Unfortunately, small sample sizes from the reference lakes did not allow us to further investigate this.

Any positive impacts the HREP may have had on black crappie recruitment in Lower Swan Lake appeared to have been short lived as the 2004 and 2005 year classes were not nearly as strong. It was likely that as competition and predation increased and sediments became more flocculent, the spawning success and recruitment rates for black crappie decreased in response.

The age structure of black crappie in Lower Swan Lake (post HREP), suggested that the black crappie population was rebuilding from within, with little immigration. This idea was further supported by the results of a fish movement study which found that black crappie did not frequently move through the water control structure separating Lower Swan Lake form the Illinois River (Schultz 2006). We speculated that a small number of adult black crappie entered Lower Swan Lake shortly after it had refilled (late 2002 or early 2003) and had one successful spawn in 2003. The fish from this year class made up the majority of the Lower Swan Lake black crappie catch from 2004-2006. Overall the black crappie catch in Lower Swan Lake shifted towards older fish during the course of the study.

Age-1 was the dominant age class of black crappie in Middle Swan Lake during each year of the study (2004-2006). Based on the relatively high numbers of age-1 fish in 2004 there appeared to be good recruitment of the 2003 year class. But unlike the Lower Swan catch, we did not see large numbers of age-2 fish in Middle Swan Lake during 2005. Although sample sizes from 2005 and 2006 were small, the black crappie age structure suggested that habitat and water management practices in Middle Swan Lake may have favored juvenile fish over adults. Middle Swan Lake may have value as nursery habitat for black crappie.

Since data indicated that the Lower Swan Lake black crappie population was a mostly captive population with little immigration, we could safely attribute most black crappie growth to the conditions found within Swan Lake, making comparisons between sites valid.

Length at age data indicated that black crappie, ages 1-2, in Middle Swan Lake had higher growth rates when compared to fish from Lower Swan Lake and Pre-Project Swan Lake (1994). Mean lengths at age were similar for black crappie from Lower Swan Lake and Pre-Project Swan Lake (1994). Overall data suggested that current habitat and water management regime of Middle Swan Lake was more favorable for growth of young black crappie when compared to Lower Swan Lake.

Although black crappie growth rates in Middle Swan Lake appeared greater than those of Swan Lake 1994, we could not attribute these differences to the HREP. Reference lakes also had significantly greater mean lengths than Swan Lake 1994, indicating that differences seen between Middle Swan Lake and Swan Lake 1994 may have been indicative of systemic changes over time and not related to the HREP. In fact, reference lakes had significantly greater mean lengths for age 1-2 black crappie, when compared to all other sites. These data may have indicated that growth rates for young black crappie were greater in backwater lakes that had a more consistent and natural connection to the mainstem river, when compared to lakes such as Lower and Middle Swan which have been largely separated from the river by levees. Researchers have noted that crappie in some diked backwaters and wetlands grew significantly slower than crappie in adjacent undiked areas (Markham et al 1997, Johnson et al 1997).

In general, the growth rates of black crappie in both Middle and Lower Swan lakes were above Illinois State averages (IDNR 2001) but were similar to historic growth seen in other Mississippi River backwater lakes (Carlander 1977). Black crappie in Middle and Lower Swan lakes approached typical angler harvest size at age-2, with fish averaging 205 mm (8.1") in Middle Swan Lake and 183 mm (7.2") in Lower Swan Lake. We should note that fast growth rates do not necessarily indicate healthy populations since high growth rates can be attributed to low density populations, which may not be desirable (Allen et al, 1998).

Although significant differences were noted in relative weights between sites, these comparisons probably had limited meaning since fish were collected from different seasons among sites. Fish condition has been noted to vary seasonally (Pope and Willis, 1996). Neumann and Murphy (1991) found mean relative weights for black crappie peaked during pre-spawn, declined after the spawn, and then increased throughout summer and fall. Hansen (1951) reported that crappie <165 mm reached maximum relative weights in June and July and declined afterwards. Additionally, differences in the size structure between populations can also affect relative weight comparisons (Pope and Willis, 1996). The most important insight we could glean from the relative weight data was that mean values from all sites were above 100 indicating that black crappie were in excellent condition, with ample forage for the current populations.

Bluegill

Results

Age Structure

Prior to the HREP, Pre-Project Swan Lake (1994) contained a high percentage (53%) of bluegill age-3 or older (Table 4). Conversely, post HREP the bluegill catch in Lower Swan Lake was dominated by age-1 and age-2 fish with relatively few (12%) age-3 bluegill. Similarly, the Middle Swan Lake bluegill catch was composed of mainly age-1 fish (86%) with few (5%) age-3 fish. The reference lakes bluegill catch contained 73% age-1 and age-2 fish and 26% age-3 and older fish. No bluegill greater than age-5 were collected in Pre-Project Swan Lake (1994), and no bluegill greater than age-4 were collected at any site from 2004-2006.

The age distribution for Pre-Project Swan Lake (1994) demonstrates a bluegill catch that had nearly equal numbers of fish ages 1-2 and fish age-3 and greater (Figure 6). In 2004, the bluegill populations from Lower Swan Lake, Middle Swan Lake, and reference lakes were dominated by age-1 fish (Figures 7-9). However, in 2005 age-2 was the most common bluegill age class in Lower Swan Lake while age-1 was still the most common bluegill age class in Middle Swan Lake. By 2006, the Lower Swan Lake bluegill population had strong representation from ages 1-3. Conversely, Middle Swan Lake was still dominated by age-1 bluegill.

Table 4. Summary of the bluegill sample for Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006). TL is fish total length. Percent (%) is an age group's relative contribution to the total catch for that species.

	Age (yr)	*N*	%	Mean back-calculated TL (mm)	Standard deviation	Length Range (mm)
Swan 1994	YOY					
	1	10	13.2	77	14	61-99
	2	26	34.2	140	21	98-201
	3	23	30.3	180	12	148-200
	4	15	19.7	192	9	182-211
	5	2	2.6	188	5	184-191
Lower Swan Lake (2004-2006)	YOY					
	1	93	55.4	71	15	40-121
	2	55	32.7	127	17	88-172
	3	20	11.9	163	15	120-184
	4					
	5					
Middle Swan Lake (2004-2006)	YOY					
	1	87	86.1	77	15	29-110
	2	9	8.9	128	30	77-170
	3	5	5.0	159	14	144-176
	4					
	5					
Reference Lakes (2004,2006)	YOY	1	1.2			
	1	57	67.1	79	41	55-109
	2	5	5.9	123	11	95-149
	3	19	22.4	164	19	133-214
	4	3	3.5	162	8	157-171
	5					

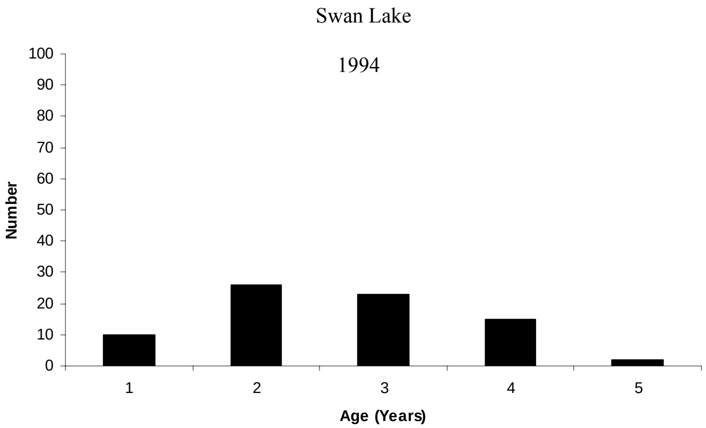


Figure 6. Age distribution of bluegill collected from Pre-Project Swan Lake (1994).

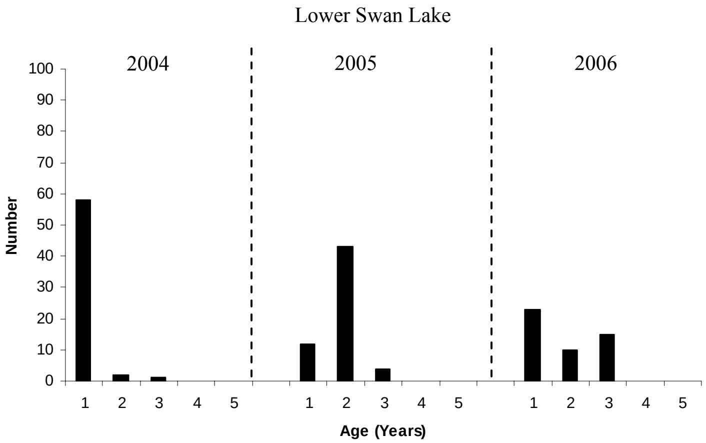


Figure 7. Age distribution of bluegill collected from Lower Swan Lake (2004-2006).

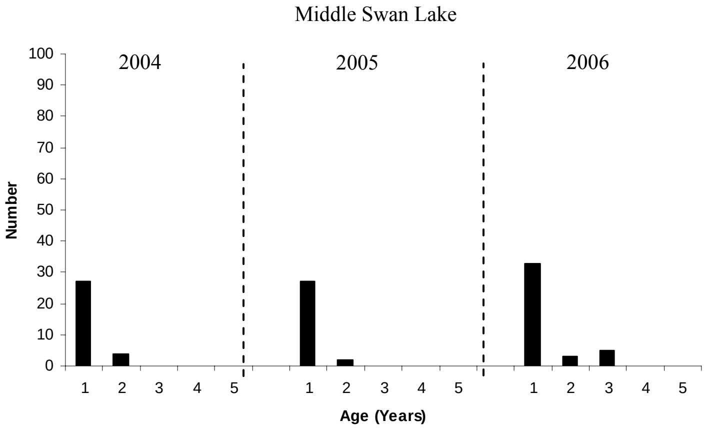


Figure 8. Age distribution of bluegill collected from Middle Swan Lake (2004-2006).

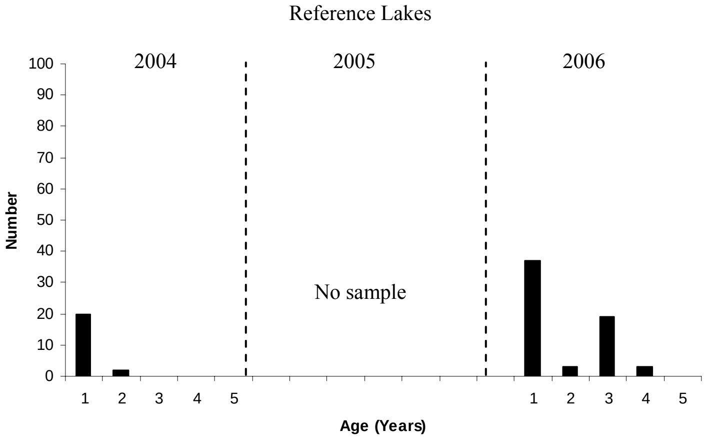


Figure 9. Age distribution of bluegill collected from reference lakes (2004 and 2006).

Length at Age

Bluegill age-1 mean length for Lower Swan Lake was significantly less (P<0.01) than mean lengths for Middle Swan Lake and reference lakes (Table 5). Bluegill age-2 mean length for Pre-Project Swan Lake (1994) was significantly greater (P<0.01) than mean lengths for Lower Swan Lake and reference lakes. Bluegill age-3 mean length for Pre-Project Swan Lake (1994) was significantly greater (P<0.01) than the mean lengths for all other sites. Graphical representation of bluegill length at age illustrated the similarity of lengths for age-1 fish from all sites (Figure 10). Pre-Project Swan Lake (1994) bluegill lengths at age-2 and age-3 were noticeably greater than other sites.

Table 5. Mean back-calculated length (mm) at age (years) for bluegill (ages 1-3) collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

Site	age-1 mean back-calculated length (mm)	age-2 mean back-calculated length (mm)	age-3 mean back-calculated length (mm)
Pre-Project Swan Lake (1994)	77	140[b]	180[c]
Lower Swan Lake (2004-2006)	71[a]	127	163
Middle Swan Lake (2004-2006)	77	128	159
reference lakes (2004-2006)	79	123	164

[a] Significant difference detected between Lower Swan Lake and Middle Swan Lake and between Lower Swan Lake and reference lakes.

[b] Significant difference detected between Pre-Project Swan Lake (1994) and Lower Swan Lake and between Pre-Project Swan Lake (1994) and reference lakes.

[c] Significant difference detected between Pre-Project Swan Lake (1994) and all other sites.

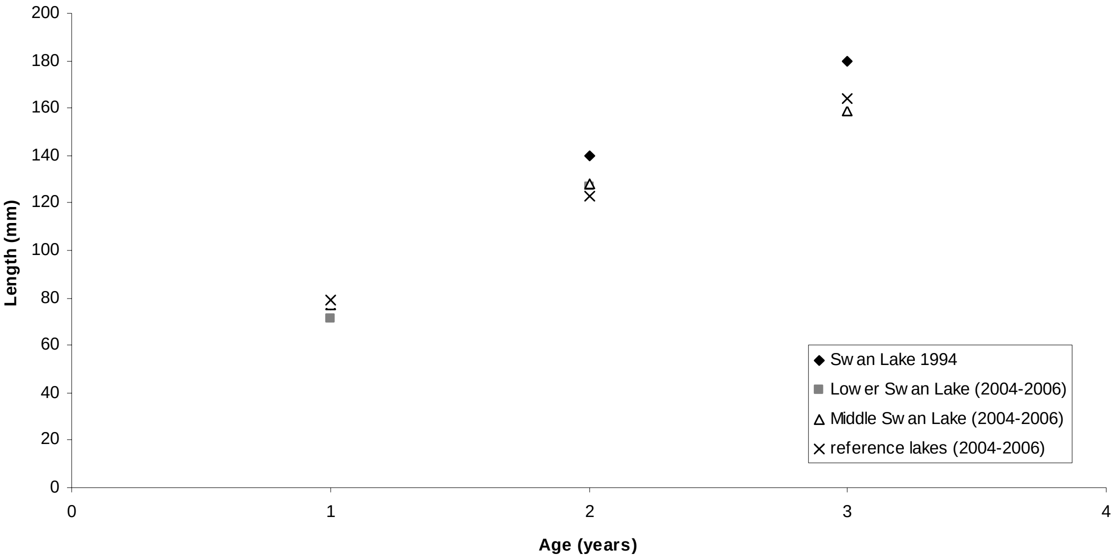


Figure 10. Mean back-calculated length (mm) at age (years) for bluegill (ages 1-3) collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006) and reference lakes (2004-2006).

Relative Weights

Bluegill W_r values for all sites were above 100 (Table 6). Lower Swan Lake and Middle Swan Lake had similar mean W_r values for bluegill. Mean W_r value for the reference lakes was significantly less than Lower Swan and Middle Swan lakes. Mean W_r values for Pre-Project Swan Lake (1994) were significantly lower ($P<0.01$) than all other sites.

Table 6. Mean relative weights (W_r) for bluegill collected from Pre-Project Swan Lake (1994), Lower Swan (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

bluegill (*Lepomis macrochirus*)					
Site	Number	Mean W_r	Std. Error	Lower 95%	Upper 95%
Pre-Project Swan Lake (1994)	71	105.9[b]	2.1	101.7	110.0
Lower Swan Lake (2004-2006)	94	136.1	1.8	132.5	139.7
Middle Swan Lake (2004-2006)	63	138.8	2.2	134.4	143.2
reference lakes (2004-2006)	83	119.6[a]	1.9	115.8	123.4

[a] Significant difference detected between reference lakes and all other sites.

[b] Significant difference detected between Pre-Project Swan Lake (1994) and all other sites.

Discussion

The bluegill age structures in Lower and Middle Swan lakes responded to the HREP in manner similar to that of black crappie. Prior to the HREP, the Pre-Project Swan Lake (1994) bluegill catch was composed of mostly age 2-4 fish, with only 10% age-1 fish. However, after the HREP and 2002 drawdown, the Lower Swan Lake bluegill catch was over 95% age-1 fish in 2004. In 2005, age-2 was the dominant age class of bluegill in Lower Swan Lake. By 2006, the bluegill catch was more balanced with similar number of age 1-3 bluegill. The Lower Swan Lake age structure for bluegill suggested that there was relatively strong recruitment of the 2003 year class and relatively weak recruitment of the 2004 age class. Wegener and Williams (1974) also found a strong year class of bluegill the year after their study lake refilled. As with black crappie, bluegill recruitment may have been aided by fewer predators, decreased competition for resources, and compacted sediments following the 2002 drawdown. Any positive impacts appeared to be short lived, as the 2004 bluegill year class in Lower Swan Lake was relatively week. Although the reference lakes sample was relatively small, there was some evidence that the 2003 bluegill year class may have experienced strong recruitment in the reference lakes as well. This suggested that bluegill recruitment for the 2003 year class may have been relatively high throughout the river system, and the changes we saw in the bluegill age structure of Lower Swan Lake may have been due in part to system wide conditions. Therefore, the effects of the HREP may not have been solely responsible for these changes.

Similar to black crappie, the age structure of bluegill in Lower Swan Lake suggested that the population was rebuilding from within with little influence from immigrant fish. This was reasonable to assume since bluegill did not frequently move through the water control structure separating Lower Swan Lake from the river (Schultz 2006). Since bluegill did not frequently immigrate into Lower Swan Lake, we could reasonably attribute most fish growth to the conditions of the lake.

The bluegill catch in Middle Swan Lake was dominated by age-1 fish in each year of the study. This age structure mirrored that of the black crappie catch in Middle Swan Lake. The habitat and water management practices in Middle Swan Lake appeared to favor juvenile bluegill over adults.

A significant difference in age-1 mean length was detected between Lower Swan Lake and Middle Swan Lake and Lower Swan Lake and the reference lakes. However, it was unlikely that a difference of 6 mm was biologically significant. With this exception, mean lengths at age for Lower Swan Lake, Middle Swan Lake, and reference lakes were similar. However, Pre-Project Swan Lake (1994) had significantly greater mean lengths for age-2 and age-3 bluegill.

It was interesting that bluegill growth rates in Pre-Project Swan Lake (1994) appeared to be higher than the other sites while black crappie growth rates in Pre-Project Swan Lake (1994) appeared lower than the other sites. A thorough investigation of the forage base in each lake would probably be needed to fully understand this phenomenon, and since this trend was mirrored in the reference lakes, we could be seeing river-wide changes in growth not related to the HREP.

In general, growth rates of bluegill in both Lower and Middle Swan lakes were slightly above Illinois state averages, with bluegill in both compartments reaching 127 mm (5 in) by age-2 (IDNR 2001). Relative weights for all sites were above 100 indicating that the bluegill populations were in excellent condition.

White Crappie

Results

Age Structure

Prior to the HREP, Pre-Project Swan Lake (1994) contained a high percentage (80%) of white crappie age-3 and older (Table 7). After the HREP, Lower Swan Lake contained 21% age-3 white crappie while Middle Swan Lake had only 4% age-3 white crappie. The reference lake white crappie sample was composed of 67% age-1 fish and 26% age-3 fish. In Pre-Project Swan Lake (1994), white crappie reached a maximum age of 7-years. From 2004-2006, no white crappie greater than age-3 were collected from Lower Swan Lake or Middle Swan Lake. The reference lakes sample contained crappie up to age-6.

The age distribution for Pre-Project Swan Lake (1994) illustrated the proportionally high numbers of adult white crappie in the catch (Figure 11). Middle Swan Lake was dominated by age-1 fish every year from 2004 to 2006 (Figure 12). The Lower Swan

Lake white crappie sample consisted of age-1 and age-2 fish in 2004 and nearly equal numbers of fish age 1-3 in 2005 and 2006 (Figure 13). The reference lake sample had nearly equal representation in each age class (Figure 14).

Table 7. Summary of the white crappie sample for Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006). TL is fish total length. Percent (%) is an age group's relative contribution to the total catch for that species.

	white crappie (*Pomoxis annularis*)					
	Age (yr)	N	%	Mean back-calculated TL	SD	Range
Pre-Project Swan Lake (1994)	YOY					
	1	7	14.0	82	16	67-112
	2	3	6.1	184	29	164-217
	3	12	24.0	245	21	213-273
	4	23	47.0	288	27	218-340
	5	2	4.1	293	1	
	6					
	7	2	4.1		85	220-340
Lower Swan (2004-2006)	YOY	7	18.4			
	1	12	31.6	120	24	85-151
	2	11	28.9	199	39	134-244
	3	8	21.1	284	7	277-293
	4					
	5					
	6					
	7					
Middle Swan (2004-2006)	YOY	9	19.6			
	1	35	76.1	121	21	74-172
	2					
	3	2	4.3			258-266
	4					
	5					
	6					
	7					
reference lakes (2004-2006)	YOY	1	1.2			
	1	57	67.1	130	27	81-177
	2	5	5.9	218	18	193-243
	3	19	22.4	277	16	258-301
	4	3	3.5			
	5					
	6					
	7					

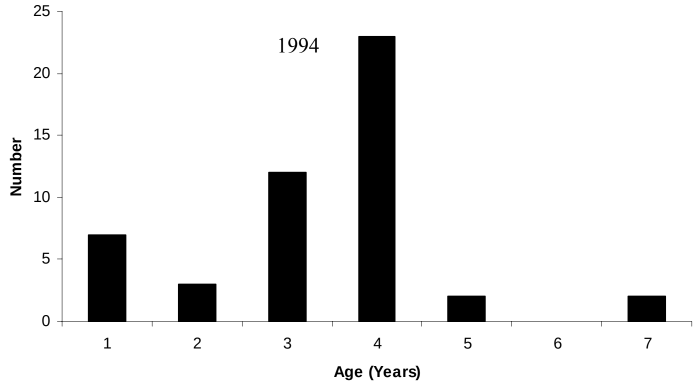


Figure 11. Age distribution of white crappie collected from Pre-Project Swan Lake (1994).

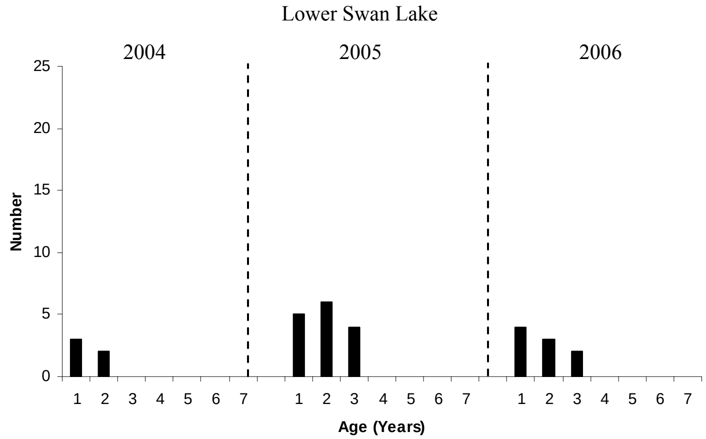


Figure 12. Age distribution of white crappie collected from Lower Swan Lake (2004-2006).

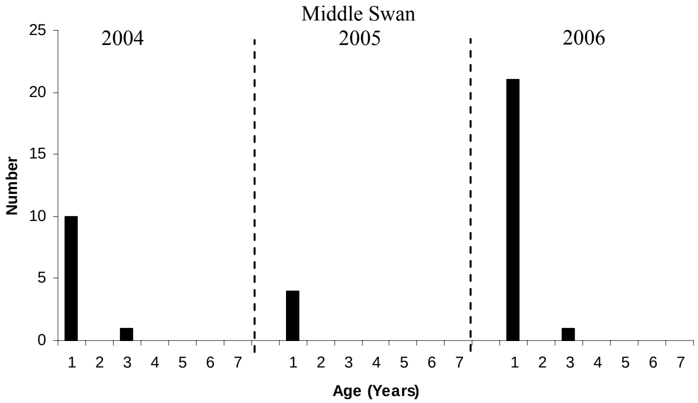


Figure 13. Age distribution of white crappie collected from Middle Swan Lake (2004-2006).

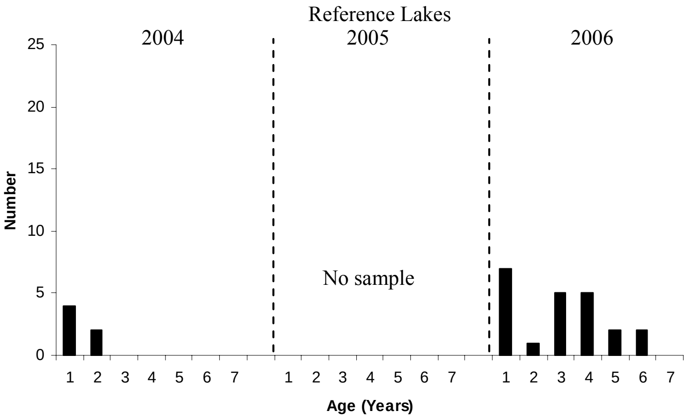


Figure 14. Age distribution of white crappie collected from reference lakes (2004 and 2006).

Length at Age

Mean length of age-1 white crappie in Pre-Project Swan Lake (1994) was significantly less (P<.01) than all other sites (Table 8). Lower Swan Lake, Middle Swan Lake and reference lakes had similar age-1 mean lengths.

Although based on small sample sizes, the graphical representation of mean length at age suggested lower mean lengths for Pre-Project Swan Lake (1994) at all ages (Figure 15).

Table 8. Mean back-calculated length at age for white crappie (age-1) collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

Site	age-1 mean back-calculated length (mm)
Pre-Project Swan Lake (1994)	82 [a]
Lower Swan Lake (2004-2006)	117
Middle Swan Lake (2004-2006)	121
reference lakes (2004-2006)	130

[a] Significant difference detected between Pre-Project Swan Lake (1994) and all other sites.

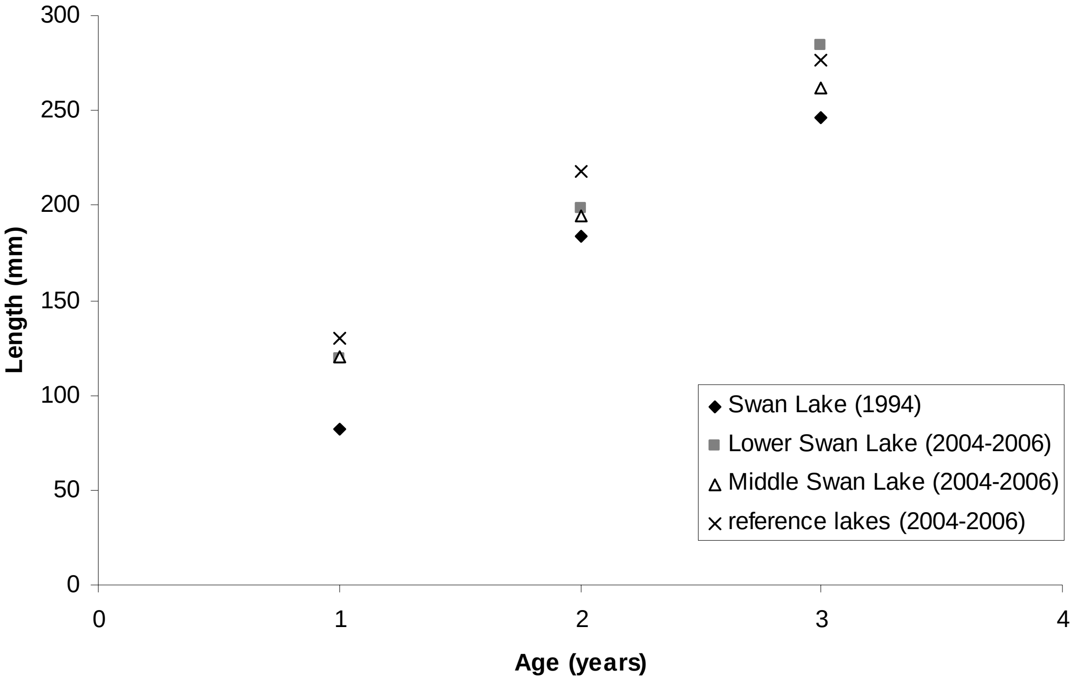


Figure 15. Mean back-calculated length (mm) at age (years) for white crappie (ages 1-3) collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

Relative Weights

White crappie mean W_r values for all sites were above 100 (Table 9). Mean W_r for Middle Swan Lake were significantly different ($P<0.01$) when compared to all other sites.

Table 9. Mean relative weights (W_r) for white crappie collected from Pre-Project Swan Lake (1994), Lower Swan (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

white crappie (*Pomoxis annularis*)					
Site	Number	Mean W_r	Std Error	Lower 95%	Upper 95%
Pre-Project Swan Lake (1994)	42	109.2	1.9	105.4	112.9
Lower Swan Lake (2004-2006)	14	115.8	3.3	109.4	123.3
Middle Swan Lake (2004-2006)	10	126.8[a]	3.9	119.1	134.4
reference lakes (2004-2006)	26	109.5	2.4	104.8	114.3

[a] Significant difference detected between Middle Swan Lake and all other sites.

Discussion

Small sample sizes for white crappie made data analyses difficult and in many cases not possible. While we were not able to draw definite conclusions from the white crappie data, we did make general observations which added to the overall discussion. Similar to black crappie and bluegill, white crappie did not frequently move through the water control structure at Lower Swan Lake (Schultz 2006), so this population could probably be viewed as isolated and distinct from populations at other sites.

Prior to the HREP, the Pre-Project Swan Lake (1994) white crappie catch had high percentages of adult (age-3-7) fish. Post HREP, catches in Lower Swan Lake and Middle Swan Lake were composed only of fish ages 0-3. Similar to the age structure of black crappie and bluegill, the vast majority of white crappie in Middle Swan Lake were age-1. However, we did not see a strong 2003 year class of white crappie in Lower Swan Lake. Whatever combination of factors that led to relatively high recruitment for bluegill and black crappie in 2003, did not affect white crappie in the same way. One possible explanation was the lack of vegetation in Lower Swan Lake. White crappie spawn in and around aquatic vegetation, and recruitment rates for white crappie have been shown to benefit from increased aquatic vegetation (Beam 1983) The age structure for Pre-Project Swan Lake (1994) suggested that recruitment rates of white crappie were probably not high even before the HREP project.

Sufficient length at age data for making comparisons existed only for age-1 white crappie. Lower Swan Lake, Middle Swan Lake and reference lakes had similar age-1 mean lengths, but Pre-Project Swan Lake (1994) had a significantly lower age-1 mean length. The apparent increase in white crappie growth rates, from 1994 to present, could be attributed to a number of factors including: decreased white crappie density, reduced interspecific competition, increase in available forage, or a combination of factors. However, no conclusions could be made.

White crappie mean relative weights at all sites were above 100, indicating that fish were in excellent condition with ample forage available.

Freshwater Drum

Results

Age Structure

Age-1 was the most common age class of freshwater drum in Pre-Project Swan Lake (1994), Lower Swan Lake and Middle Swan Lake (Table 10). Young-of-year freshwater drum was the most common age class collected at the reference lakes. Freshwater drum as old as 18-years were collected from Pre-Project Swan Lake (1994), Lower Swan Lake and Middle Swan Lake. The oldest freshwater drum collected in the reference lakes was age-9.

The age distribution for Pre-Project Swan Lake (1994) illustrated the relatively higher numbers of age-1 fish, while age classes 2-18 were composed of between 1 and 10 individuals (Figure 16). The relative numbers of age-1 fish in Lower Swan Lake and Middle Swan Lake increased from 2004-2006, while other age classes did not increase over the same time period (Figures 17 and 18). In 2006, the reference lakes contained a relatively high number of age-1 fish with few fish greater than age 1 (Figure 19). Data were insufficient to compare yearly differences for the reference lakes.

Table 10. Summary of the freshwater drum sample for Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006). TL is fish total length. Percent (%) is an age group's relative contribution to the total catch.

	Age (yr)	N	%	Mean back-	SD	Range
Pre-Project Swan	YOY					
	1	23	25.6	131	28	88-215
	2	2	2.2	180	39	116-
	3	4	4.4	227	49	160-
	4	10	11.1	268	43	192-
	5	1	1.1	272	40	215-
	6	4	4.4	300	40	231-
	7	6	6.7	324	44	248-
	8	6	6.7	343	44	264-
	9	7	7.8	370	47	276-
	10	6	6.7	386	46	293-
	11	9	10.0	411	52	305-
	12	2	2.2	423	55	315-
	13	2	2.2	433	50	327-
	14	1	1.1	429	46	337-
	15	4	4.4	433	49	344-
	16	1	1.1	451	27	426-
	17	1	1.1			436-
	18	1	1.1			445
Lower Swan Lake	YOY	34	16.6			
	1	81	39.5	100	28	68-164
	2	19	9.3	154	35	106-
	3	11	5.4	188	36	131-
	4	10	4.9	221	37	141-
	5	6	2.9	245	44	164-
	6	7	3.4	272	50	173-
	7	10	4.9	289	48	184-
	8	3	1.5	305	53	195-
	9	3	1.5	331	62	206-
	10	2	1.0	343	63	217-
	11	4	2.0	358	68	220-
	12	4	2.0	367	24	334-
	13	2	1.0	390	23	369-
	14	4	2.0	411	31	376-
	15	2	1.0	458	45	425-
	16	1	0.5			440-
	17	1	0.5			454-
	18	1	0.5			469
Middle Swan Lake	YOY	8	6.5			
	1	22	17.9	117	40	61-266
	2	14	11.4	186	38	90-257

	3	13	10.6	227	49	90-296
	4	12	9.8	255	49	180-
	5	15	12.2	263	53	201-
	6	5	4.1	283	49	219-
	7	3	2.4	297	49	237-
	8	3	2.4	319	50	249-
	9	5	4.1	335	54	264-
	10	3	2.4	346	53	276-
	11	2	1.6	359	59	288-
	12	3	2.4	378	65	301-
	13	3	2.4	406	78	320-
	14	5	4.1	420	76	336-
	15	2	1.6			326-
	16					
	17	2	1.6			350-
	18	3	2.4	506	74	465-
reference lakes	YOY	54	58.7			
	1	29	31.5	116	19	80-155
	2	2	2.2	198	22	169-
	3			253	26	213-
	4	2	2.2	303	36	245-
	5	3	3.3	351	39	307-
	6	1	1.1			387
	7					
	8					
	9	1	1.1			429
	10					
	11					
	12					
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	14					
	15					
	16					
	17					
	18					

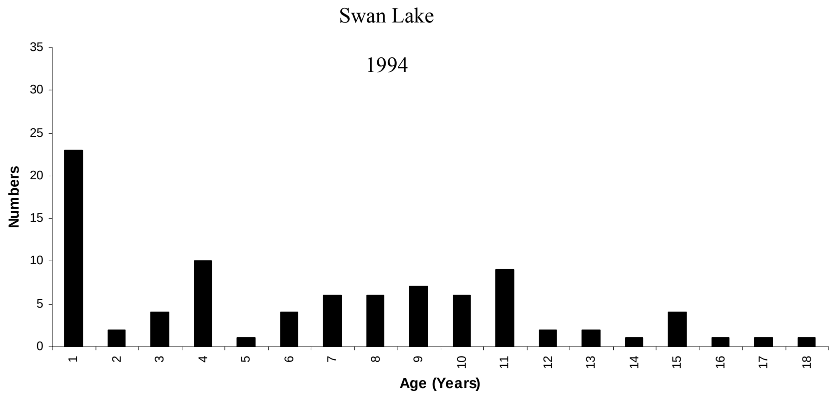

Figure 16. Age distribution of freshwater drum collected from Pre-Project Swan Lake (1994).

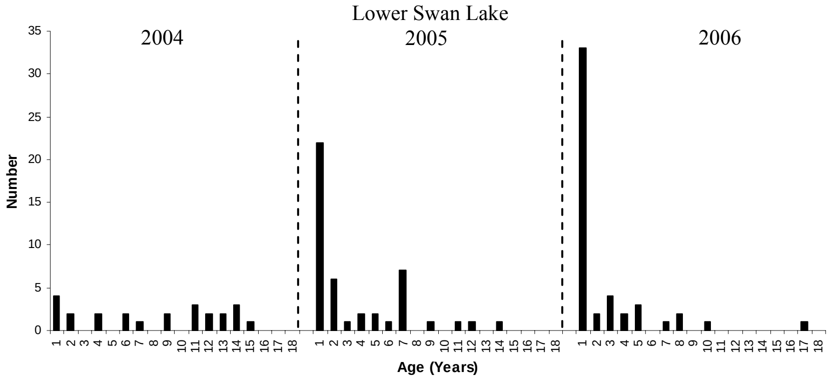

Figure 17. Age distribution of freshwater drum collected from Lower Swan Lake (2004-2006).

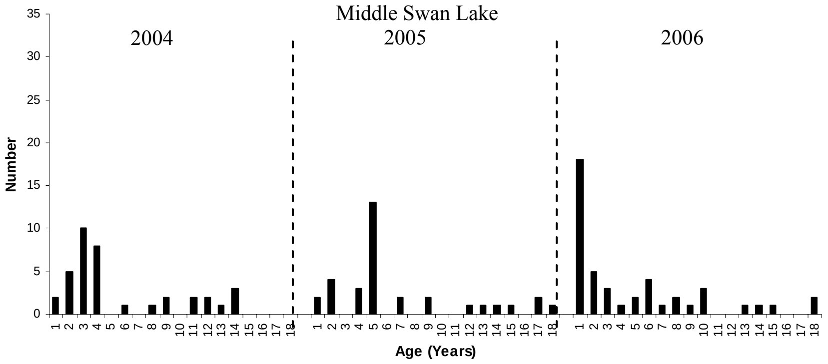

Figure 18. Age distribution of freshwater drum collected from Middle Swan Lake (2004-2006).

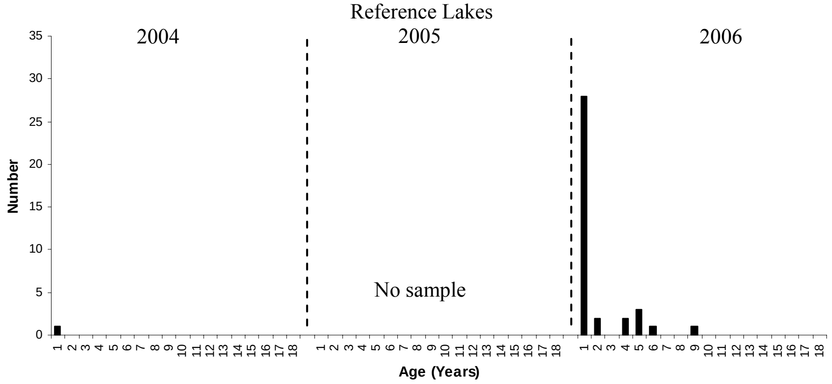

Figure 19. Age distribution of freshwater drum collected from reference lakes (2004-2006).

Length at age

Freshwater drum ages 6-13 in Pre-Project Swan Lake (1994) had higher mean back-calculated lengths than either Lower Swan Lake or Middle Swan Lake (Figure 20). Lengths for freshwater drum ages 6-13 were similar between Lower Swan Lake and Middle Swan Lake. However for ages 1-5, Middle Swan Lake had greater mean lengths than Lower Swan Lake and was similar to Pre-Project Swan Lake (1994). It should be noted that sample sizes were small for fish older than age-14. Also overall freshwater drum sample size from the reference lakes was small.

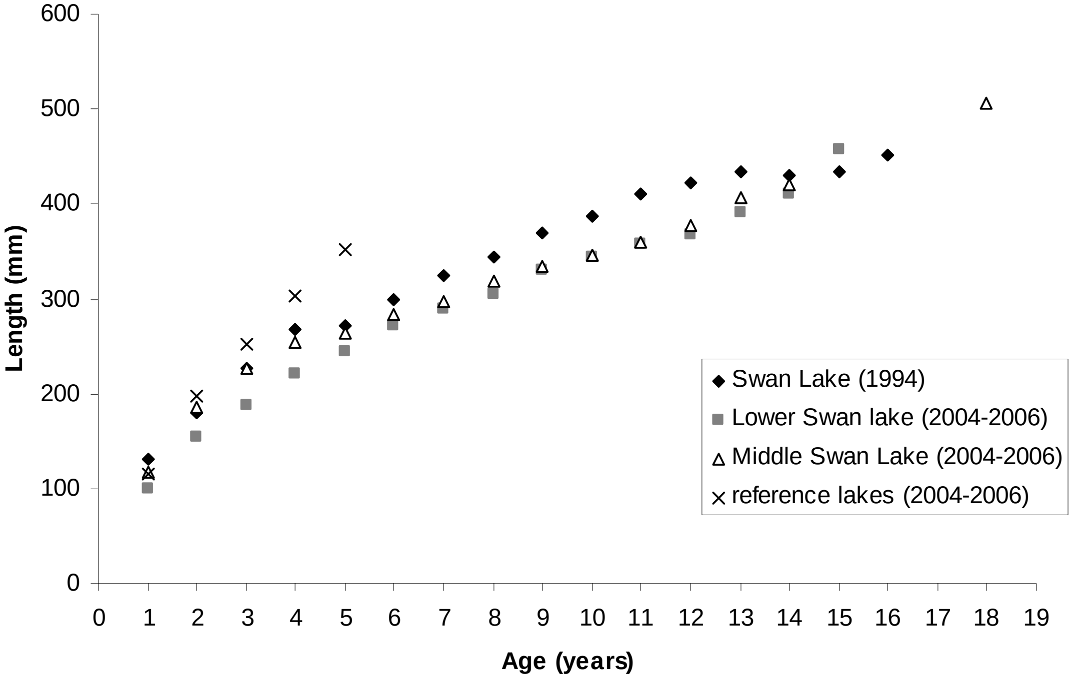

Figure 20. Mean back-calculated length (mm) at age (years) for freshwater drum collected from Pre-Project Swan Lake (1994), Lower Swan Lake (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

Relative Weights

Mean W_r values for all sites were near or greater than 100. The Mean W_r for Pre-Project Swan Lake (1994) was significantly different ($P=0.01$) when compared to all other sites.

Table 11. Mean relative weights (W_r) for freshwater drum collected from Pre-Project Swan Lake (1994), Lower Swan (2004-2006), Middle Swan Lake (2004-2006), and reference lakes (2004-2006).

freshwater drum (*Aplodinotus grunniens*)					
		Mean	Std	Lower	Upper
Site	Number	W_r	Error	95%	95%
Pre-Project Swan Lake (1994)	50	96.7[a]	3.5	89.9	114.7
Lower Swan Lake (2004-2006)	88	106.4	2.6	101.2	103.5
Middle Swan Lake (2004-2006)	91	110.4	2.6	105.3	111.5
reference lakes (2004-2006)	64	108.7	3.1	102.7	115.4

[a] Significant difference detected between Pre-Project Swan Lake (1994) and all other sites.

Discussion

Freshwater drum were shown to move frequently between Lower Swan Lake and the Illinois River (Schultz, 2006). Therefore we could not treat the Lower Swan Lake freshwater drum population as distinct and separate. Consequently we could not attribute any changes in freshwater drum population dynamics or age structure to the HREP project. However, we used comparisons of freshwater drum from Lower Swan Lake, Middle Swan Lake and reference lakes to illustrate the similarity in data between these sites, which further supported the idea that a single freshwater drum population encompassed these sites. Comparisons between Pre-Project Swan Lake (1994) and the present day sites (Lower and Middle Swan lakes, and reference lakes), may provide some insight into river wide changes in the freshwater drum population over the past decade.

Increases in percentage of age-1 fish were noted from 2004 to 2006 in Middle Swan Lake and Lower Swan Lake, possibly suggesting that river conditions were favorable for high recruitment of the 2005 year class of freshwater drum.

It appeared that growth rates for freshwater drum, ages 6-13, decreased from those noted in 1994 (Figure 20). This may have indicated a river wide shift in the freshwater drum population dynamics. We caution that data from older fish was based on a few individuals, and larger sample sizes would be needed to make any definitive conclusions.

Summary

The nature of the samples collected and a multitude of other factors presented challenges when analyzing the data. Having only three year classes of fish for study limited the type and depth of many analyses. Also the decade between the pre-project sample and the current study opened the door for possible system wide changes in fish populations, thereby making it difficult to detect changes due to the HREP. The seasonality of the samples (i.e., April for the 1994 sample and throughout the year for current sample) and slight variations in sampling technique likely affected results. In addition, we had no pre-project data for the reference lakes. The possibility of fish immigration into Lower and Middle Swan lakes was another factor which could have skewed data. Small sample sizes from the reference lakes, fish age-3 or older in general, and for white crappie at all sites, created holes in the data. It should be noted that these small samples sizes were *not* due to lack of effort, but rather lack of fish in the lakes.

The preponderance of confounding variables did not allow us to definitively answer the question, “What impact did the HREP have on the population structure and dynamics of targeted fish species.” However there were several observations that could be made from the data.

The Lower Swan lake black crappie and bluegill populations appeared to have reestablished mainly from within, with little immigration following the 2002 drawdown. Recruitment of black crappie and bluegill was relatively high the year after the lake had refilled. While we do not know the exact reasons for this, possible factors include: compacted spawning substrates, reduced competition, and reduced predation. Recruitment of black crappie and bluegill was relatively low in 2004 and 2005 possibly due to a change in these factors. White crappie in Lower Swan Lake did not experience strong recruitment following the drawdown. The lack of submerged vegetation may have explained why white crappie, which often spawn in and around vegetation, did not recruit well, while black crappie recruited fairly well.

The black crappie and bluegill populations in Lower Swan Lake were probably best characterized as having fast growth rates. However, growth rates of black crappie in Lower Swan Lake were slower than growth rates in Middle Swan Lake or the reference lakes. Bluegill growth rates were similar between the two compartments and the reference lakes.

Middle Swan Lake appeared to function as a nursery area for black crappie and bluegill, with young fish, but few adults, in the lake. Growth of both species was relatively fast (above state averages), but black crappie growth in Middle Swan Lake lagged behind growth seen in the reference lakes.

It was interesting to note that in Middle Swan Lake black crappie and white crappie grew faster in the years following the HREP, while bluegill actually grew slower. Since this phenomenon was also noted in the reference lakes, it may have represented a system wide change, so we could not with any certainty attribute this to the HREP.

Thus far, the water management strategies used in Lower Swan Lake and Middle Swan Lake have not produced desirable populations of sportfish (crappie and bluegill). The annual drawdowns in Middle Swan Lake led to fast fish growth and at least some yearly recruitment, likely due to compacted sediments and vegetation growth. Unfortunately this strategy does not allow for a stable, robust population of adult sportfish.

The complete drawdown of Lower Swan Lake in 2002, led to only one strong year class. This drawdown may have compacted sediments for a brief time allowing for one successful spawn. Unfortunately this drawdown did not fully compact lake sediments, and also removed most existing fish from the lake. Since sportfish do not frequently immigrate into Lower Swan Lake, the fish populations must rebuild from within, but rebuilding is hampered by low recruitment in the subsequent years.

In order to rebuild strong sportfish populations we need to improve recruitment (which occurred following drawdowns), but we also need to provide an opportunity for adult fish to persist. This may not be possible given current conditions, but this is necessary if a goal is to maintain quality sportfishing opportunities in Swan Lake. One strategy we considered to accomplish both involves dredging deepwater refuge areas and using partial annual drawdowns. Partial drawdowns would allow for compaction of bottom sediments in near shore areas, while still maintaining water in the deepwater refuge areas, thereby retaining adult fish in the lake that would be lost during a complete drawdown.

Unfortunately, it is likely that any deepwater areas would be short lived due to sedimentation and the benefits of this approach would never be realized.

Our near-term recommendation is for major drawdowns every 1-2 years (as funding permits) to address sediment compaction. Until the sediment can be stabilized to a larger degree, sustaining a quality sportfishery will be a major challenge. Because these drawdowns have resulted in good recruitment for black crappie and bluegill, we recommend taking measures to allow young fish to escape into the river during drawdowns. At least for the short-term, this would allow Swan Lake to function as a source of recruitment for the Illinois River and surrounding areas.

Our mid-term recommendation is for either partial drawdowns every 1-2 years or major drawdowns about every 5 years to maintain sediment compaction and subsequent vegetation growth once annual drawdowns have sufficiently compacted sediments. If regular partial drawdowns are successful in maintaining the habitat, then this should provide fairly ideal conditions for annual recruitment while providing habitat for adults to persist during drawdowns resulting in strong, steady sportfish populations. Major drawdowns every 5 years would most likely ‘reset’ fish populations which would result in more of a boom or bust type fishery, but would allow for a fishery assuming that growth remains good and fish begin to reach harvestable size at age 2.

Long-term it is difficult to determine what will work best. Until sediment is compacted, we won’t know how vegetation will respond. Until we know what habitat conditions will be, it is difficult to predict how fish will respond. Also, management actions necessary to maintain habitat will affect fish populations. For the long-term, we recommend using an adaptive management approach to move the project forward with the immediate need to compact sediment so that quality habitat is possible and to do this with the ever-present goal of maintaining quality, useable habitat for both resident and migratory fish.

Acknowledgments

We would like to thank Chad Dolan, Eric Ratcliff and anyone else from INHS that collected fish samples for us. Also a big thank you to Kelly DeGrandchamp and Doug Schultz of SIUC Fisheries for all the times they transported fish for us, and to John Mabery, Ken Dalrymple and the rest of the Two Rivers Refuge staff for their assistance and access to the refuge.

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Appendix A. Asian carp age and populations structures

Introduction

As part of Objective 2 in the scope of work for Biological Response Monitoring Swan Lake Habitat Rehabilitation and Enhancement Project, we examined the population structure of bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*Hypophthalmichthys molitrix*) utilizing Swan Lake. The study was viewed as an opportunity to collect additional data on Asian carps. This portion of the study was not intended to assess the effectiveness of the HREP, but it does allow us to gain valuable information on the life histories and population dynamics of these invasive fish which are spreading thoughout the entire Mississippi River system.

Methods

All Asian carp were collected by Southern Illinois University Carbondale (SIUC) between April and August in 2004 and between April and May in 2005 using various sampling techniques. Fish were collected from Lower Swan Lake, other nearby backwater lakes and the mainstem Illinois River. For the purposes of this report we analyzed only fish collected from Swan Lake, per the scope of work. Length, weight, and sex data were supplied by SIUC. The left pectoral spine from each fish was removed and sent to Carterville FRO. Pectoral fin spines were cross sectioned (0.7 mm thick) with an Isomet low-speed precision saw. Cross sections were placed directly on the microscope stage using bottom transmitted light, and the annuli were counted. Digital images of the spines were captured using Scion Image software. Carterville FRO performed the first age reading, and then spines were sent to SIUC for a second reading. When there was disagreement, readers met to discuss disagreement and reach consensus on a final age.

Results

For bighead carp collected in Lower Swan Lake we found 67% agreement between SIUC and CFRO age estimates, and 69% agreement for silver carp. The maximum age for bighead carp in our sample was 7-years. The maximum age for silver carp was 8-years. In 2004, age-4 was the most common age class in the sample for both species of Asian carp (Figure 1). In 2005, age-5 was the most common age class in the sample (Figure 2). In the combined 2004 and 2005 sample, female and male bighead carp had similar age structures (Figure 3) although there were more females (N=49) than males (N=37) in the sample. Both species of Asian carp appeared to grow rapidly up to age-3, and then growth slows (Figure 4). Bighead carp reached a maximum size of 1,181 mm, while silver carp reached a maximum size of 878 mm. Bighead carp had greater mean lengths for fish age-3 and older when compared to silver carp. For each age class, female bighead carp had greater mean lengths than males, and these differences in mean length appeared to increase as fish aged (Figure 5). Relationships between length and weight were similar for male and female bighead carp (Figure 6).

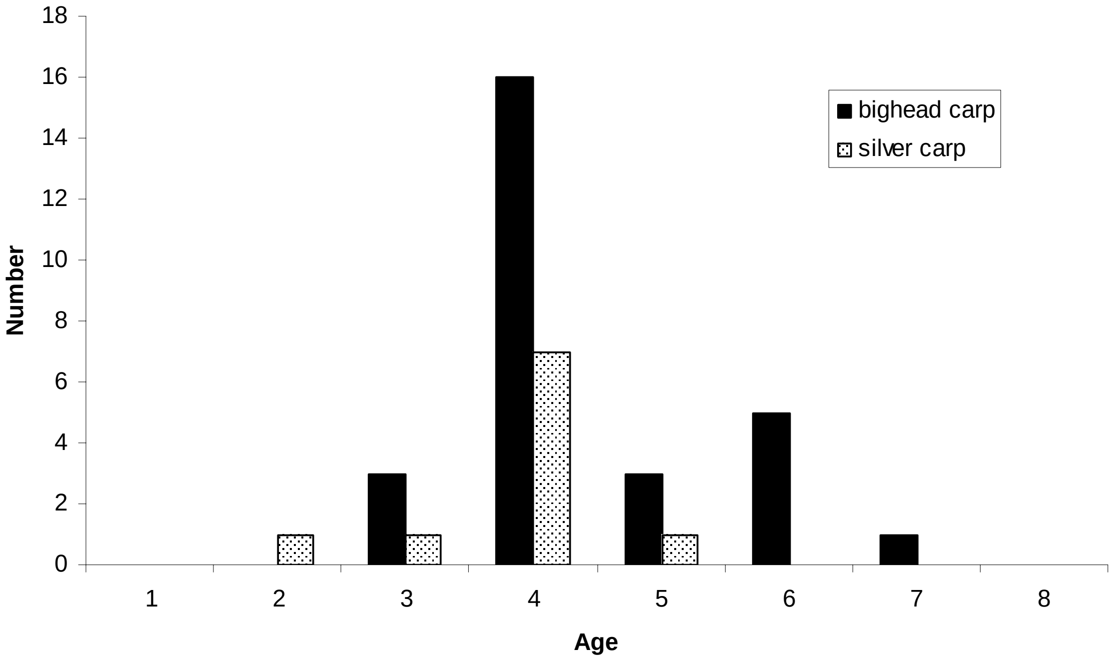

Figure 1. Age structure of bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*Hypophthalmichthys molitrix*) collected from Lower Swan Lake, Illinois during 2004.

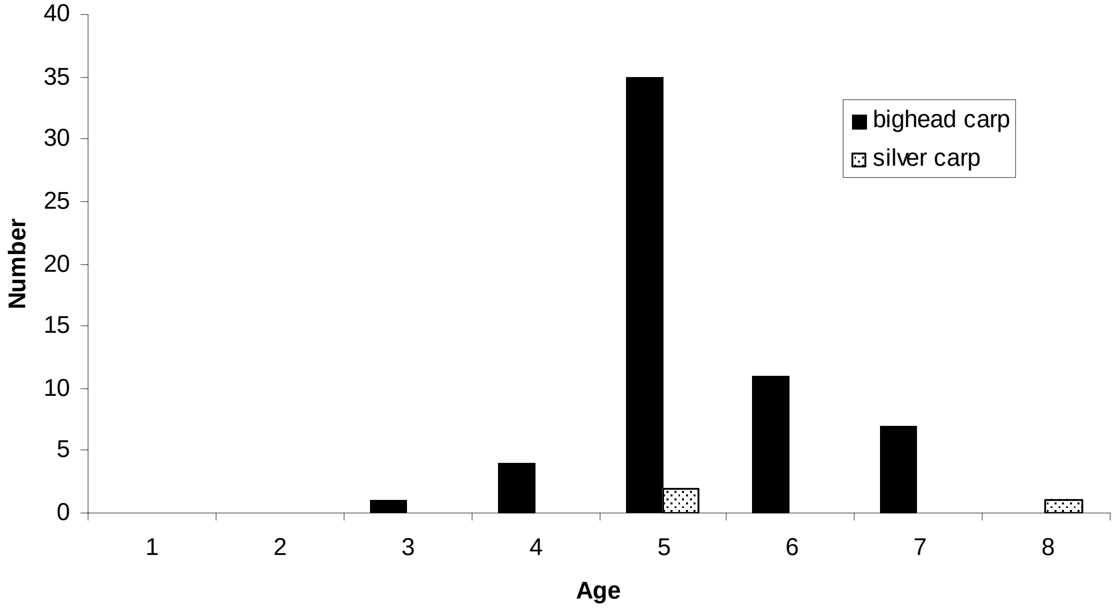

Figure 2. Age structure of bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*Hypophthalmichthys molitrix*) collected from Lower Swan Lake, Illinois during 2005.

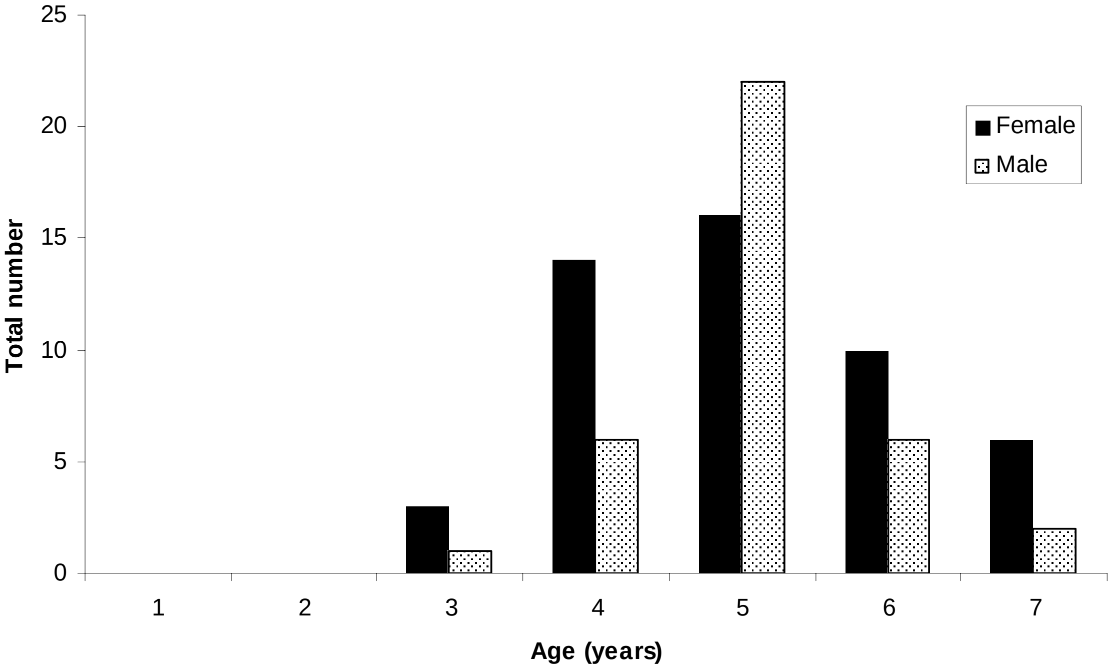

Figure 3. Age structure of female and male bighead carp (*Hypophthalmichthys molitrix*) collected from Lower Swan Lake, Illinois during 2004 and 2005.

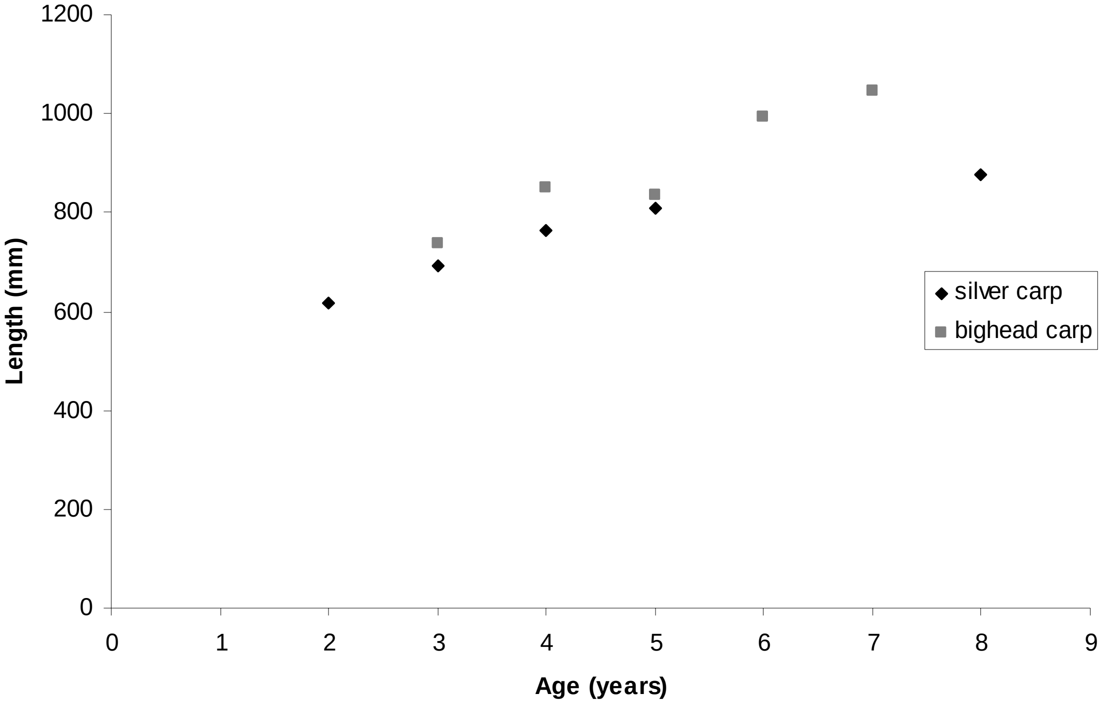

Figure 4. Mean length (mm) at age (years) for bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*Hypophthalmichthys molitrix*) collected from Lower Swan Lake, Illinois during 2004 and 2005.

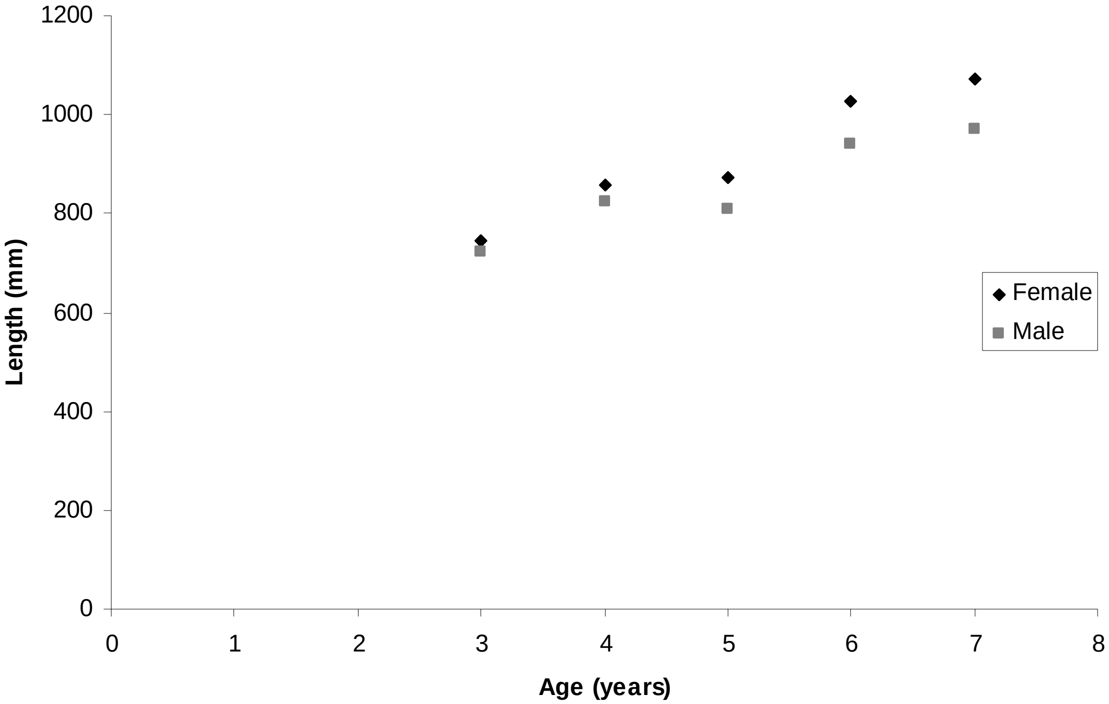

Figure 5. Mean length (mm) at age for female and male bighead carp (*Hypophthalmichthys nobilis*) in Lower Swan Lake, Illinois during 2004 and 2005.

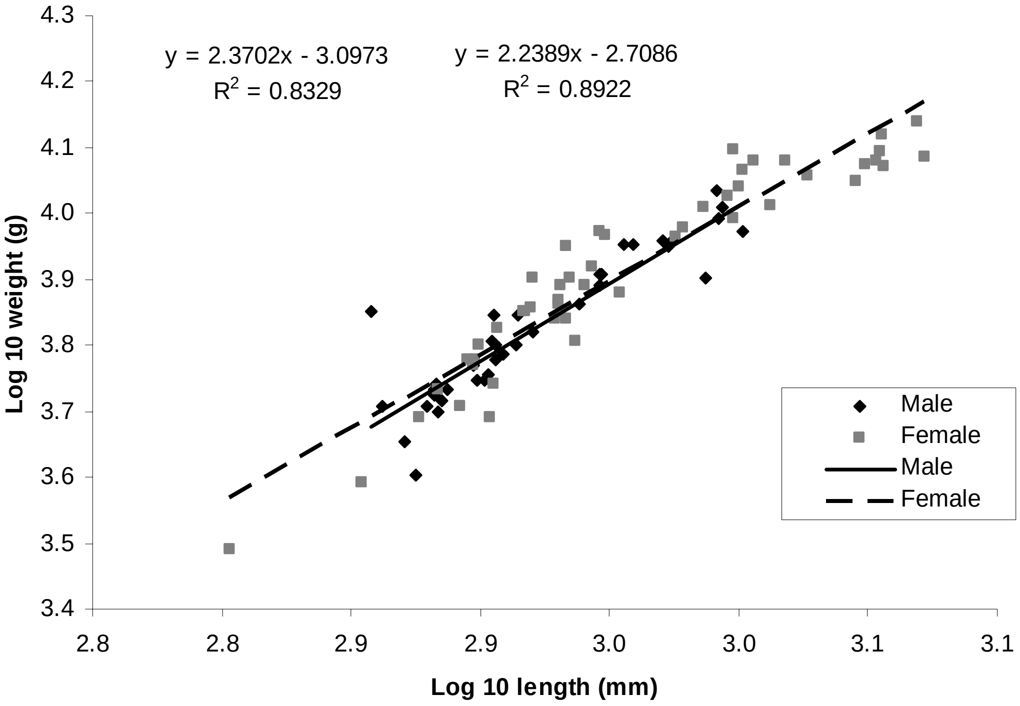

Figure 6. Length weight regression for male and female bighead carp collected from Lower Swan Lake, IL during 2004 and 2005.

Discussion

Asian carp are highly mobile fish and move frequently between Lower Swan Lake and the Illinois River (DeGrandchamp 2006; Schultz 2006). Therefore the Lower Swan Lake sample of Asian carp is probably representative of the Illinois River and Mississippi River population.

Gear bias likely caused young (age 0-3) Asian carp to be underestimated. Thus far abundances of Asian carp, especially young Asian carp, have been difficult to assess, and universally accepted sampling protocols have not yet been developed for these species. Because of this we could not draw any conclusions or make any generalizations about the dynamics of young Asian carp. Also, the small sample size for silver carp from Lower Swan Lake made analyses of these data difficult. Therefore, this discussion is focused mainly on bighead carp age-4 and greater.

The relatively large numbers of age-4 bighead carp in the 2004 sample, combined with the large number of age-5 fish in 2005 indicated that the 2000 year class had strong recruitment. Recruitment success is often affected by river level and floodplain inundation (King et. al 2003, Junk et al. 1989). River levels during 2000 were in general lower and more stable than other years during the time period of 1998-2002 (Figure 7). We found it somewhat surprising that the strongest recruitment year class was from a low water year. Asian carp young-of-year utilize off channel habitats as nursery areas (Nico and Fuller 2005; Fuller et al. 1999), so we suspected that greater amount of off channel habitats available during high water years would yield higher recruitment, but this did not appear to be the case. One possible explanation for high recruitment during the low water year of 2000 is that relatively few larval and juvenile Asian carp became trapped in off channel habitats, since many of these areas did not become inundated with water from the mainstem during 2000. We have found that juvenile Asian carp often become trapped in off channel areas when river levels recede, and this can lead to large scale die offs. During the low water year of 2000, young-of-year Asian carp may have been forced to utilize nursery areas that had more persistent connections to the mainstem rivers while in the high water years, young-of-year Asian carp utilized large areas of the floodplain that later became dewatered leading to fish kills. Interestingly, Shrank and Guy (2002) reported that their bighead carp sample from the lower Missouri River was dominated by the 1994 year class. In comparison to preceding or following years, 1994 was a relatively low and stable water year (Figure 8). Alacron (1996) also found variable recruitment, with the 1994 year class stronger than 1993,1995, and 1996 year classes. Alacron (1996) speculated that the 1993 year class may have been weaker because young-of-year bighead carp did not return from the floodplain to the Mississippi River. Additionally, Lubinski et al. (2004) postulated that common carp (*Cyprinus carpio*) in the Upper Mississippi River had poor recruitment during high or variable water years. While this is far from conclusive, we felt that it was worth noting, and it may warrant future study and could have management implications.

In general, the Asian carp population was characterized by fast growing, relatively short lived individuals. Age of bighead carp in our sample ranged from 3 to 7-years. Schrank and Guy (2002) also reported the same age range in the lower Missouri River. Alarcon (1996) reported maximum ages for bighead carp of 6-years in the Middle Mississippi River and 5-years in Pool 26 of the Mississippi River. The oldest silver carp in our sample was 8 years, while it is reported that they reach 20 years in their native Asian waters (Berg 1964, cited by Schofield et al. 2005). In our sample bighead carp growth was rapid in young fish and then slowed after the onset of sexual maturity (approximately age-3) (Huet 1970; Schrank and Guy 2006; Kolar et al. 2005). Schrank and Guy (2002) also reported that growth increments of bighead carp decreased at ages greater than 3-years while Alacron (1996) found fast growth up to age 4. Bighead carp had greater mean length at age and maximum lengths than silver carp. Female bighead carp had greater mean length at age than male bighead carp, and these differences appeared to increase as fish reached sexual maturity and continued to age. Length weight relationships for male and female bighead carp were similar. Schrank and Guy (2002) also found male and female length weight relationships to be similar for bighead carp.

There were more female than male bighead carp in the sample; however this could be due to the smaller males being underrepresented by the gear types used. Additionally, older male bighead carp could be proportionally overrepresented in the sample since these older (larger) males would be more likely to be caught than younger (smaller) males. Gear bias was likely partially responsible for the age structure of the Asian carp sample (Figures 1-3, Table 1).

The mean lengths at age for bighead carp in our sample were comparable to what Alacron (1996) found in the Mississippi River (Table 2). However we should note that Alacron used back-calculated lengths while we used empirical lengths at time of capture (April-June). It appears that bighead carp in the Mississippi River grow faster but are shorter lived than those reported in Asia (Galina 1991, cited by Alacron 1996).

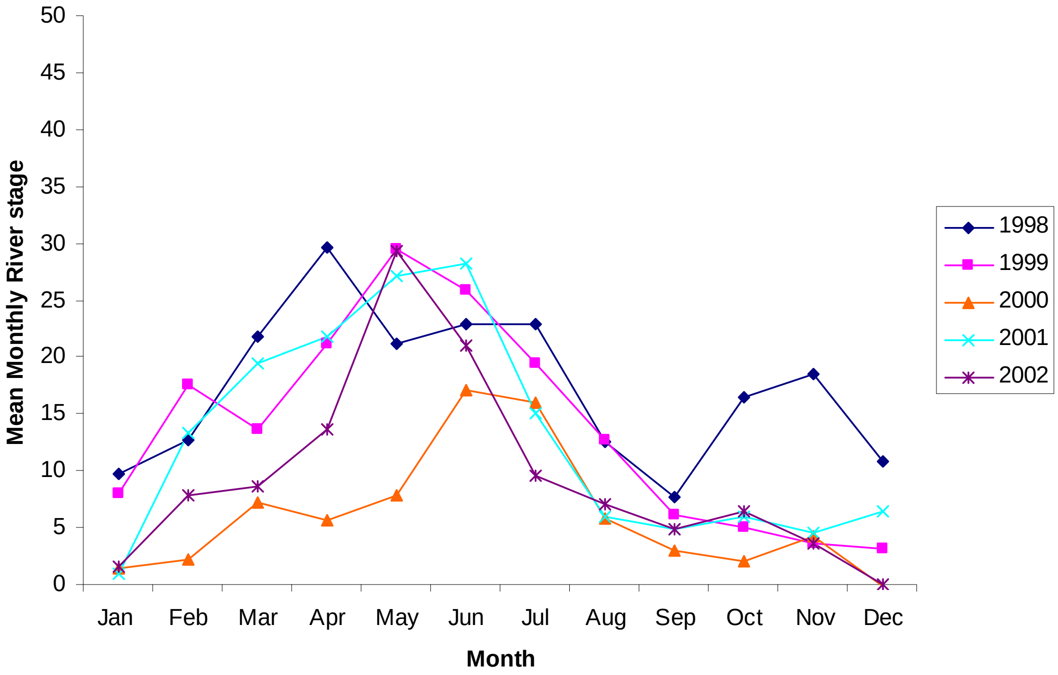

Figure 7. Mean monthly gauge reading at the St. Louis gauge on the Mississippi River from 1998-2002. Flood stage is 30 ft. Data courtesy of The U.S. Army Corps of Engineers, http://mvs-wc.mvs.usace.army.mil/archive/archindex.html

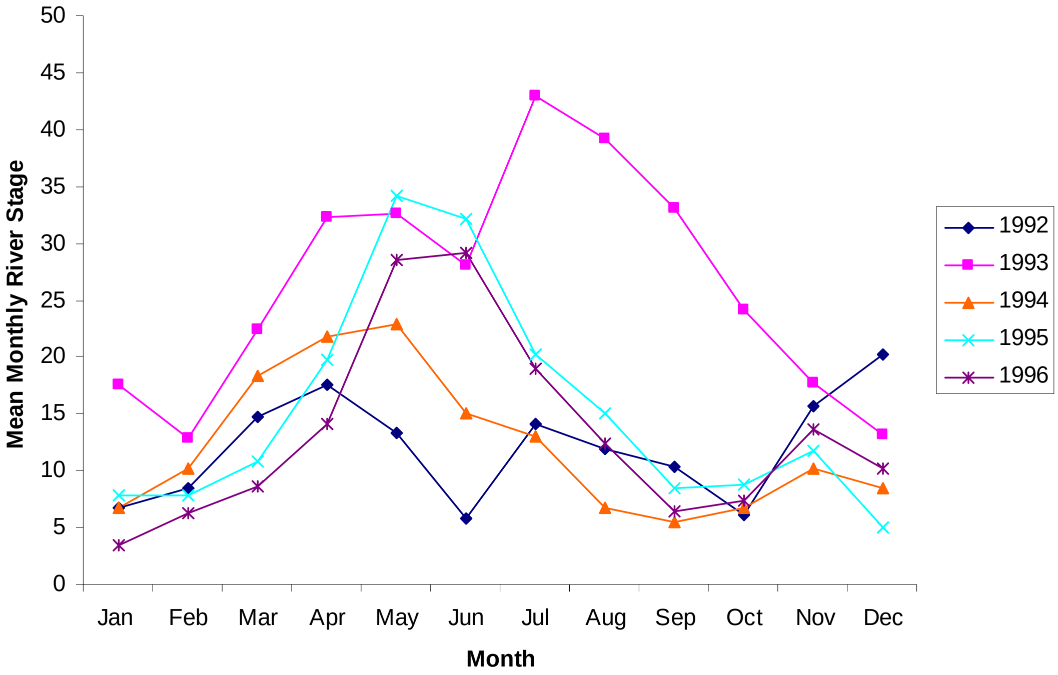

Figure 8. Mean monthly gauge reading at the St. Louis gauge on the Mississippi River from 1992-1996. Flood stage is 30 ft. Data courtesy of The U.S. Army Corps of Engineers, http://mvs-wc.mvs.usace.army.mil/archive/archindex.html

Table1. Summary of bighead carp (*Hypophthalmichthys nobilis*) catch from Lower Swan Lake, Illinois during 2004-2005.

	Age	Number	Proportion of catch (%)[a]	Mean total length (mm)	SD	Range
Male						
	3	1	(2.7)			721
	4	6	(16.2)	824	39.5	765-885
	5	22	(59.5)	809	87.2	728-983
	6	6	(16.2)	939	80.7	883-985
	7	2	(5.4)	969		935-1003
	totals	*37*	*43.0*	*839*		*721-1003*
Female						
	3	3	(6.1)	744	93.3	636-801
	4	14	(28.6)	859	87.1	753-1181
	5	16	(32.7)	873	89.7	715-1110
	6	10	(20.4)	1028	91.0	865-1172
	7	6	(12.2)	1071	79.0	970-1136
	totals	*49*	*57.0*	*917*		*636-1181*

[a] Numbers in parenthesis represent proportion of the total for that sex
italics indicate totals for the respective sexes

Table 2. Mean length (mm) at age (years) for bighead carp from selected waters. Lengths from Lower Swan Lake are empirical lengths at time of capture. All other lengths are back-calcualted.

	Length (mm) at age (years)									
Water Body	1	2	3	4	5	6	7	8	9	10
Lower Swan Lake (2004-2005)			738	849	836	995	1045			
Middle Mississippi River*	273	500	672	830	813	921				
Mississippi River, Pool 26*	233	447	611	702	807					
Lake Katlabukh (Ukraine)**	371	465	502	538	563	645	665	713	785	835
Kakhovka Reservoir (Ukraine)**		389	472	586	682	741	771	762		
Kremenchug Reservoir (Ukraine)**	239	374	470	565		701	746	800		

* Alacron (1996)

** Galina (1991) cited by Alacron (1996)

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SECTION 6:

RESPONSE OF WATERBIRDS AND FOOD AVAILABILITY

Michael W. Eichholz, Darin Grulkowski, Randy Smith

Cooperative Wildlife Research Lab
Southern Illinois University,
Carbondale, IL 62901-6504

MACROINVERTEBRATE RESPONSE DURING PERIODS OF USE BY MIGRATORY BIRDS TO THE SWAN LAKE HABITAT REHABILITATION AND ENHANCEMENT PROJECT (HREP) ON SWAN LAKE, IL.

INTRODUCTION

Many plants and animals have adapted to exploit natural hydrologic regimes of large river systems (Vannote et al. 1980, Sparks 1995, Junk et al. 1989, Bayley 1995, Poff and Allan 1997). A river's hydrologic pattern is essential for exchange of water, sediment, and nutrients between the main channel and its floodplain (Junk et al. 1989, Bayley 1995, Sparks 1995), and is critical for sustaining biological diversity and ecological integrity (Sparks 1995, Poff and Allan 1997). Current anthropogenic alterations (i.e. upland drainage, construction of dams and levees, channelization, and channel incision), however, have altered many natural processes that occur between a river and its floodplain (Fredrickson and Reid 1990, Bayley 1995, Poff and Allan 1997, Sparks et al. 1998, Nilsson and Berggren 2000). These modifications have had negative effects on associated floodplain biota (Poff and Allan 1997, Sparks et al. 1998, Nilsson and Berggren 2000). Therefore, efforts are needed to protect and restore these systems.

Currently it may not be possible to "restore" river floodplains to their original state, but "rehabilitation" can be a practical goal (Gore and Shields 1995). Habitat rehabilitation and/or management of large river riparian wetlands has become necessary due to wetland loss, and decreased productivity in remaining habitat (Fredrickson and Reid 1990, Kelley et al. 1993). An example of habitat rehabilitation is Swan Lake, a rehabilitated Illinois River floodplain lake located near the Mississippi and Illinois River confluence. Swan Lake has been recognized as an important site for migratory waterfowl and fish communities (U. S. Army Corps of Engineers 1991). Water quality within Swan Lake had been greatly reduced by high sediment loads from the Illinois River and local agricultural runoff. Other threats to this backwater site included wave action and water level fluctuations, all of which had decreased water quality and degraded aquatic plant communities. Decreased water quality and loss of vegetation negatively impacted food resources to waterfowl and fish that inhabited Swan Lake. To ameliorate this degradation, Swan Lake became a target for rehabilitation via the Swan Lake Habitat Rehabilitation and Enhancement Project (HREP).

A major goal of the Swan Lake HREP was to increase macroinvertebrate communities for the benefit of wildlife through improved water quality and restoring aquatic macrophytes (U. S. Army Corps of Engineers 1991). To achieve this particular goal, the U. S. Army Corps of Engineers identified 5 objectives: 1) reduce future sedimentation, 2) manage water levels during the vegetation growing season, 3) consolidate lake sediments, 4) decrease wind induced wave action, and 5) form small, independently manageable lake units. Riverside levees were constructed to reduce the sedimentation loads from the Illinois River. Lake islands were built to diminish wind generated wave action and, in turn, reduce sediment resuspension. Interior lake levees and water control structures divided Swan Lake into 3 lake units and permitted water level management within each lake compartment. Water level management should allow consolidation of bottom sediments, improve water clarity, and produce sediment characteristics that would promote the growth of aquatic macrophytes, possibly increasing abundance and biomass of associated aquatic macroinvertebrates (Wegener et al. 1974, Fredrickson and Taylor 1982).

Large scale habitat rehabilitation projects, like Swan Lake, are based on sound ecological principles. Habitat rehabilitation of large rivers, however, is less common, more expensive, more complex, and can be difficult to successfully implement in comparison to smaller streams and rivers (Gore and Shields 1995). Thus, due to the cost and biological impact of habitat rehabilitation, it is important such projects be evaluated to assure biological functions have been restored as intended. Habitat rehabilitation monitoring programs not only provide a measure of project success, they also provide information that can be used to modify aspects of the project (i.e. adaptive resource management) and provide insights to improve future restorations (Sparks et al. 1998). In many cases, macroinvertebrate recovery from disturbance has been shown to be a relatively rapid process (Niemi et al. 1990), so evaluation of current conditions may reflect project success. The U.S. Army Corps of Engineers and its collaborators recognized benefits of project evaluation, and specifically recognized importance of both pre- and post-rehabilitation monitoring.

Both sedimentation and vegetative habitat can have significant impacts on macroinvertebrate populations. Sedimentation is considered one of the main sources of degradation in streams, rivers, and wetlands (Richter et al. 1997, USEPA 1998). Studies have shown increased turbidity and/or sedimentation negatively impact some invertebrate populations (Rabeni et al. 2005, Wagener and LaPerriere 1985, Henley et al. 2000, Martin and Neely 2001, Donohue et al. 2003), and can also hinder growth of some wetland plants (Werner and Zedler 2002, Gleason et al. 2003). In particular to the Illinois River, reports have indicated that some macroinvertebrate communities have been negatively impacted by pollution and sedimentation rates (U. S. Army Corps of Engineers 1991, Theiling 1998). In addition to sedimentation, vegetation can have significant impacts on macroinvertebrate communities. Many studies have shown that vegetated sites contained more abundance and/or biomass of macroinvertebrate communities than compared to open water habitat (Krull 1970, Olson et al. 1995, Beckett et al. 1992, Thorp et al. 1997). More specifically, moist soil management has been shown to have positive impacts to macroinvertebrate abundance and biomass (Anderson and Smith 2000). The Swan Lake Habitat Rehabilitation and Enhancement Project was designed to reduce sedimentation/turbidity rates, and to manage levels to promote moist soil vegetation growth in order to increase macroinvertebrate communities for the benefit of wildlife.

STUDY OBJECTIVES/PREDICTIONS

Pre vs. Post-Habitat Rehabilitation and Enhancement Project (HREP)

The objective of this study is to compare pre-HREP macroinvertebrate abundance and biomass estimates (Smietanski 1994) to post-HREP estimates (data collected in 2004 and 2005) for evaluation of rehabilitation impacts on nutrient resources during waterfowl migration. Protein rich food sources, like macroinvertebrates, are important to waterfowl during various stages of their life cycle (Fredrickson and Reed 1988, Eldridge 1990, Fredrickson 1991). According to the goals proposed by the U. S. Army Corps of Engineers, significant improvements in macroinvertebrate abundance and biomass are expected post-HREP due to improved water quality (i.e. decreased sedimentation and turbidity) and increased macrophyte productivity. From these proposed expectations, we make the following predictions.

Prediction 1:

Macroinvertebrate abundance and biomass estimates will be significantly greater within the same habitat type post-rehabilitation (2004 and 2005) than compared to pre-rehabilitation estimates due to improved water quality.

Prediction 2:

Overall macroinvertebrate abundance and biomass will be greter under post-HREP conditions compared to pre-HREP conditions due to an increase in vegetative habitat and improved water quality.

METHODS

To accomplish our objectives and to assure the best comparison with pre-HREP data, our proposed post-HREP study will duplicate, as much as possible, equipment and methods used by Smietanski (1994). Fieldwork was conducted during early and late spring (March 18 and April 28), as well as early and late fall (October 13 and November1) for 2 years (2004 and 2005).

Aquatic Macroinvertebrate Sampling

Samples were collected during spring and fall in 2004 and 2005 at sites selected by Smietanski (1994) in his study of Swan Lake prior to habitat rehabilitation. When sites originally selected by Smietanski no longer contained the habitat type previously recorded, the sites were still sampled and the current habitat classification was recorded. Habitat site descriptions included emergent vegetation (river bulrush: *Scirpus fluviatilis*), flooded forest (mixed deciduous forest), moist soil (primarily *Echinochloa* and *Potamogeton* species), open water (areas completely devoid of vegetative or woody habitat), and submergent vegetation (sago pondweed: *Potamogeton pectinatus*). Two replicate 643cm^3 nektonic (D-frame sweep net) and 361cm^2 benthic (petite-Ponar dredge) samples were taken at each site. Benthic samples were placed in a 19 liter bucket and elutriated to remove organic material from the inorganic substrate. All samples were then rinsed through a 500μm mesh sieve. Materials retained within the sieve were placed in a plastic bag with a sample label. Formaldehyde (37% v/v) with Rose Bengal stain was then added to form 10% of the sample’s contents. Bags were sealed and transported back to Southern Illinois

University-Carbondale (SIUC) for lab processing. The following measurements were taken at each sample site: lake unit (lower, middle, and upper Swan), habitat type, water depth, date, time, weather conditions, collectors, substrate type/firmness, sampling device used, and comments were recorded.

Sample Processing

In the laboratory macroinvertebrates were sorted, identified, counted, and dried to estimate abundance and dry biomass. Samples were rinsed through the following series of sieve mesh sizes: (16mm), (4mm), (1mm), and (500µm). Macroinvertebrates were then separated from debris using forceps. Sorting was performed by eye or with a low-power magnification glass. Macroinvertebrate identification was performed under a dissecting microscope. Sieved material >1mm was completely sorted at all times; sieved material <1mm was subsampled when it contained a large number of macroinvertebrates (approximately 100 individuals or more) using a Folsom Wheel Splitter. A minimum of 50 macroinvertebrates were required in the subsample.

Aquatic Macroinvertebrate Identification and Quantification

Macroinvertebrates were identified using Merritt and Cummins (1996) and Smith (2001). Platyhelminthes and Nematoda were identified to phylum. Annelids were identified to class, Mollusca and Cnidaria to family. In the phylum Arthropoda, organisms in the subphylum Crustacean were identified to order and class, while Insecta were identified to family. Once identified and sorted, samples were dried at $65^{o}C$ for 48 hours. Dry mass of each sorted taxonomic group was weighed to nearest 10^{-4} grams. Biomass of minute macroinvertebrates was determined using length-dry mass regression equations according to Bottrell et al. (1976) and Benke et al. (1999). Abundance and dry biomass were recorded for analysis.

STATISTICAL ANALYSIS

Pre vs. Post-Habitat Rehabilitation and Enhancement Project (HREP)

To test the prediction that macroinvertebrate abundance and biomass will be greater within similar habitats under post-rehabilitation water quality conditions compared to pre-rehabilitation conditions, we only compared sites that contained the same habitat type post-HREP as sampled pre-HREP. Samples were analyzed separately by year, season, habitat type, and sample type. Two-tailed t-tests were used to determine differences in estimated means of macroinvertebrate abundance and biomass of pre-HREP (Smietanski 1994, published values) sites verses post-HREP (2004 and 2005, collected data) sites. In his analysis, Smietanski treated samples as replicates; however, we treated sites as replicates. Thus, when we analyzed differences in sites using two-tailed t-tests, we reduced Smietanski's reported *n* value by half. In all tests, a P-value of ≤ 0.05 was considered significant and P-values of ≤ 0.10 marginally significant. Nektonic and benthic samples were also combined for qualitative comparisons of differences between pre and post-HREP sites.

To test the prediction that overall Swan Lake will have significantly greater macroinvertebrate abundance and biomass under post-HREP conditions verses pre-HREP conditions due to increased vegetation production and improved water quality, we compared the overall estimated means of macroinvertebrate abundance and biomass of pre-HREP (Smietanski 1994, published values) estimates verses all post-HREP (2004 and 2005) site estimates. Estimated means for each year, season, and sample type under pre and post-HREP conditions were compared using nonparametric statistical tests (Kruskal-Wallis and Wilcoxon) with SAS version 9.1 software. In all tests, a P-value of ≤ 0.05 was considered significant with P-values of ≤ 0.10 marginally significant. Nektonic and benthic samples were also combined for descriptive comparisons of differences between pre and post-HREP sites.

RESULTS

Pre vs. Post-Habitat Rehabilitation and Enhancement Project (HREP)

For this study we collected a total of 394 nektonic and benthic samples during early and late spring (March 18 and April 28), as well as early and late fall (October 13 and November1) in 2004 and 2005. With the expected benefits of the Swan Lake Habitat Rehabilitation and Enhancement Project (reduced sedimentation/turbidity and increased macrophyte productivity) we predicted significantly greater macroinvertebrate abundance and biomass within the same habitat type, and among all sites combined, under post-rehabilitation conditions compared to pre- rehabilitation conditions.

Among habitats that had remained the same since rehabilitation, 80 pre/post-HREP comparisons of macroinvertebrate abundance and biomass were made (Table 1-4). Among the 80 comparisons, 31 revealed significant positive impacts while 13 comparisons revealed significant negative impacts. Also, pre/post-HREP comparisons of all sample sites combined revealed significant improvements in macroinvertebrate abundance and/or biomass within Swan Lake since rehabilitation in 4 of 8 spring sample comparisons, while significant decreases occurred in 3 out of 14 pre/post-HREP fall sample comparisons (Table 5-6). Thus, post-rehabilitation assessments seem to indicate the Swan Lake Habitat Rehabilitation and Enhancement Project was not completely successful in improving aquatic macroinvertebrate abundance and/or biomass.

Results appear to indicate that many pre/post-HREP comparison results were influenced by seasonal and annual variation. Some post-rehabilitation improvements in macroinvertebrate abundance and/or biomass occurred in one season, while not in others. For example, analysis of macroinvertebrate estimates within the same habitat type under post-HREP conditions compared to pre-HREP conditions revealed that significant increases among late spring nektonic abundance and biomass samples regardless of habitat type, while negative impacts were recorded for most early spring nektonic abundance and/or biomass estimates. Early spring nektonic macroinvertebrate biomass showed no improvement in emergent vegetation and flooded forest sites, while significant increases occurred in late spring. Also, open water sites showed significant improvements in benthic macroinvertebrate biomass for early spring while there were no significant differences between pre/post-HREP late spring benthic biomass comparisons. Analysis of all sample sites combined revealed no significant difference in nektonic biomass in early spring but significantly increased in late spring, with significant decreases in both early and late fall. Significant differences also varied between years. Late spring nektonic abundance among flooded forest and open water sites was significantly improved among pre/post-HREP 2004 comparisons; however, there was no difference among pre/post-HREP 2005 comparisons. Late fall nektonic pre/post-HREP 2004 comparisons revealed significant negative impacts in abundance (among open water sites) and biomass (among moist soil and open water sites), with no difference among pre/post-HREP 2005 comparisons. Finally estimated means of combined nektonic and benthic samples followed similar variation patterns (Table 7).

DISCUSSION

Pre vs. Post-Habitat Rehabilitation and Enhancement Project (HREP)

Water quality problems are still persistent in many regions of the Illinois River (Theiling 1998), and lack of improvement among many pre/post-HREP comparisons may be due to the fact that water quality has not shown dramatic improvements in Swan Lake since rehabilitation. In fact, estimated means of water turbidity have increased in 2 of the 3 lake units since project implementation (Chad Doland of the Illinois Natural History Survey, personal communication). Water quality data; however, was not continuously collected throughout the year, so it is difficult to conclude that water quality conditions are still hindering macroinvertebrate populations. In fact, qualitative site assessments during macroinvertebrate sampling periods indicated that water clarity was higher during fall sampling conditions compared to spring. All fall nektonic macroinvertebrate abundance and/or biomass habitat site comparisons, and all sample sites combined comparisons, however, showed no improvements post-HREP, while many spring nektonic abundance and/or biomass comparisons revealed significant improvements. Also, during both spring and fall sampling periods many habitat sites were observed containing very soft and unconsolidated sediments. Even with most habitat sites containing unconsolidated sediments, many significant comparisons between pre/post-HREP benthic samples have shown improvements in macroinvertebrate abundance and/or biomass since rehabilitation.

We believe the negative response in invertebrate abundance and biomass in fall and positive response in invertebrate biomass and abundance in spring is due primarily to the current water management practices. During the pre-HREP study, water levels were stable throughout the year relative to the water levels during the post-HREP study, allowing most habitats to maintain a relatively stable macroinvertebrate community. Under the current water management regime in the upper and middle units, however, water levels are reduced during summer to help consolidate bottoms and stimulate the growth of hydrophytes. Thus, during the fall sampling periods, a much larger proportion of the sampling locations are in newly colonized areas where macroinvertebrate communities are less developed, thus, during fall macroinvertebrate abundance and biomass of macroinvertebrate communities from the post-HREP period are low relative to the pre-HREP period. By spring, however, macroinvertebrate communities have had adequate time to completely recolonize the habitats that became available during the fall flooding period and responded to the improved conditions provided by the HREP activities, thus, abundance and biomass in most habitat types is greater in the post-HREP analysis relative to the pre-HREP study. These differences are most dramatic during the late spring period when macroinvertebrates have had the greatest amount of time to exploit the improved conditions.

The response in macroinvertebrate abundance in late spring coincides well with life history characteristics of the migratory waterfowl that exploit them. Mallards on swan lake appear to increase their dependence on invertebrates in late spring. Furthermore, lesser scaup, a species that relies more heavily on invertebrates, is a late spring migrant, thus, the relatively consistent increases in late spring macroinvertebrate abundance and biomass likely had a positive influence on both dabbling and diving ducks.

Impacts of Habitat Management

Habitat management has been a priority in Swan Lake, particularly the middle and upper lake units. Vegetation has been shown to have positive impacts on macroinvertebrate abundance and biomass of some macroinvertebrate communities (Voigts 1976). In particular, moist soil management has been reported to increase macroinvertebrate abundance and biomass (Anderson and Smith 2000), and epiphytic macroinvertebrates have been shown to provide a substantial proportion of resources for some wildlife species (Schramm and Jirka 1989). Management has increased moist soil vegetation area within the middle Swan Lake unit (Chad Doland of the Illinois Natural History Survey, personal communication). Even though some pre/post-HREP site comparisons revealed loss of vegetation, sites that were not sampled (sites that were not included among pre/post HREP comparisons) have increased in macrophyte productivity. So it is possible that gross macroinvertebrate abundance and biomass in late spring may have increased in middle Swan Lake not only due to increased productivity within specific habitat types, but, due to an increase in the amount of habitat (i.e. moist soil vegetation) that supports the greatest abundance and biomass of macroinvertebrates. Combined 2004 and 2005 estimates revealed greater average abundance (7,588 No./m^3) and biomass (0.74 g/m^3) in moist soil nektonic habitat verses open water nektonic habitat (abundance: 908 No./m^3; biomass: 0.04 g/m^3) in spring (early and late spring estimates combined). Also, combined 2004 and 2005 estimates revealed greater average abundance (7,128 No./m^3) and biomass (1.22 g/m^3) in moist soil nektonic habitat verses open water nektonic habitat (abundance: 1,320 No./m^3; biomass: 0.19 g/m^3) in fall (early and late fall estimates combined). Similar, but less pronounced differences in benthic abundance and biomass were also found between moist soil and open water habitat in the spring of 2004 and 2005. Thus, increasing moist soil habitat may increase the overall mean of nektonic (and possibly benthic) macroinvertebrate abundance and biomass. Furthermore, increasing vegetation may not only increase specific macroinvertebrate communities, it also has been shown to reduce sediment resuspension caused by wind and wave action (Dieter 1990). Thus, enhancement activities appear to have increase overall macroinvertebrate abundance biomass on the middle unit, the unit being managed to support moist soil vegetation. Furthermore, continued water level management may eventually provide conditions necessary to restore vegetative communities and water quality conditions that promote macroinvertebrate abundance and biomass.

CONCLUSIONS AND RECOMMENDATIONS

Although changes in invertebrate abundance between pre- and post-HREP were variable between habitat types and seasons, there appears to be an increase in abundance during late spring when invertebrates are most utilized by waterfowl. Thus, the HREP appears to have had a positive influence on macroinvertebrate abundance and biomass when waterfowl exploit them most.

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Table 1: Two tailed t-test results of comparisons between pre and post-HREP habitat sites for spring (early and late) nektonic (No./m^3 +/- SE) and benthic (No./m^2 +/- SE) abundance estimates at Swan Lake, Illinois River backwater.

Early Spring Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	6100	4100	3	3396	561	3	6340	2980	-0.65	10	ns	0.05	10	ns
Flooded Forest	10	7500	2700	7	3606	936	2	2224	1617	-1.36	15	ns	-1.68	10	ns
Moist Soil	4	2500	700	3	617	193	4	4028	1355	-2.59	5	P < 0.05	1.00	6	ns
Open Water	8	1900	900	8	39	39	8	464	139	-2.07	14	P < 0.10	-1.58	14	ns

Early Spring Benthic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	7056	2954	3	3536	743	3	8421	2067	-1.16	10	ns	0.38	10	ns
Moist Soil	4	10559	8789	4	5111	1454	4	30886	3935	-0.61	6	ns	2.11	6	P < 0.10
Open Water	8	4548	1116	8	11269	1476	8	14657	2814	3.63	14	P < 0.01	3.34	14	P < 0.01

Late Spring Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	100	<100	3	29990	8378	3	4126	1009	3.57	10	P < 0.01	3.97	10	P < 0.01
Flooded Forest	10	100	<100	2	11773	3530	3	2462	1093	3.31	10	P < 0.01	2.15	11	P < 0.10
Moist Soil	4	100	<100	4	10793	5367	4	14914	11844	1.99	6	P < 0.10	1.25	6	ns
Open Water	8	<100	<100	8	4503	1674	8	434	288	2.63	14	P < 0.05	1.09	14	ns

Late Spring Benthic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	2177	500	3	40023	12178	3	25360	13831	3.11	10	P < 0.05	1.68	10	ns
Flooded Forest	10	4064	800	2	40211	1901	3	24825	8959	17.52	10	P < 0.001	2.31	11	P < 0.05
Moist Soil	4	1238	280	4	19726	3949	4	49373	8786	4.67	6	P < 0.01	5.48	6	P < 0.01
Open Water	8	9278	2465	8	21141	4316	8	23487	6334	2.39	14	P < 0.05	2.09	14	P < 0.10

Table 2: Two tailed t-test results of comparisons between pre and post-HREP habitat sites for fall (early and late) nektonic (No./m^3 +/- SE) and benthic (No./m^2 +/- SE) abundance estimates at Swan Lake, Illinois River backwater.

Early Fall Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	15300	7100	-	-	-	-	-	-	-	-	-	-	-	-
Moist Soil	4	8000	2200	4	5531	1544	4	3044	1113	-0.92	6	ns	-2.01	6	P < 0.10
Open Water	8	800	100	8	719	483	7	2037	1281	-0.16	14	ns	0.96	13	ns
Submerged Vegetation	4	2000	200	-	-	-	-	-	-	-	-	-	-	-	-

Early Fall Benthic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Open Water	8	1619	432	8	16539	2520	7	10605	2503	5.84	14	P < 0.001	3.54	13	P < 0.01
Submerged Vegetation	4	2960	983	-	-	-	-	-	-	-	-	-	-	-	-

Late Fall Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	84900	20300	-	-	-	-	-	-	-	-	-	-	-	-
Moist Soil	4	32600	9400	4	10373	4228	3	9565	4493	-2.16	6	P < 0.10	-2.21	5	P < 0.10
Open Water	8	3700	600	7	220	137	7	2303	1088	-5.65	13	P < 0.001	-1.12	13	ns
Submerged Vegetation	4	37700	12700	-	-	-	-	-	-	-	-	-	-	-	-

Late Fall Benthic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
No Pre-HREP Data	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Table 3: Two tailed t-test results of comparisons between pre and post-HREP habitat sites for spring (early and late) nektonic (g/m^3 +/- SE) and benthic (g/m^2 +/- SE) dry biomass at Swan Lake, Illinois River backwater.

Early Spring Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	1.10	0.30	3	1.28	0.19	3	1.52	0.62	0.50	10	ns	0.61	10	ns
Flooded Forest	10	1.10	0.30	7	1.12	0.26	2	0.42	0.21	0.05	15	ns	-1.87	10	P < 0.10
Moist Soil	4	0.70	0.20	3	0.05	0.01	4	1.68	0.45	-3.24	5	P < 0.05	1.98	6	P < 0.10
Open Water	8	0.30	0.10	8	0.01	0.01	8	0.02	0.00	-2.95	14	P < 0.05	-2.82	14	P < 0.05

Early Spring Benthic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	1.20	0.30	3	1.36	0.46	3	5.64	3.25	0.29	10	ns	1.36	10	ns
Moist Soil	4	2.70	1.30	4	5.15	1.98	4	11.62	1.39	1.03	6	ns	4.69	6	P < 0.01
Open Water	8	1.90	0.40	8	8.04	1.73	8	4.95	0.78	3.47	14	P < 0.01	3.47	14	P < 0.01

Late Spring Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	<0.10	<0.10	3	6.08	1.65	3	2.20	0.94	3.61	10	P < 0.01	2.21	10	P < 0.10
Flooded Forest	10	<0.10	<0.10	2	2.60	0.16	3	1.16	0.98	13.12	10	P < 0.001	1.07	11	ns
Moist Soil	4	<0.10	<0.10	4	0.47	0.31	4	0.76	0.36	1.15	6	ns	1.73	6	ns
Open Water	8	<0.10	<0.10	8	0.11	0.04	8	0.02	0.01	0.07	14	ns	-0.78	14	ns

Late Spring Benthic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	1.90	0.50	3	8.67	2.51	3	8.63	4.86	2.65	10	P < 0.05	1.38	10	ns
Flooded Forest	10	1.50	0.60	2	3.74	0.01	3	9.12	3.03	3.73	10	P < 0.01	2.47	11	P < 0.05
Moist Soil	4	0.50	0.10	4	5.87	1.20	4	11.38	2.68	4.46	6	P < 0.01	4.06	6	P < 0.01
Open Water	8	20.10	13.00	8	6.70	0.72	8	4.02	0.48	-1.03	14	ns	-1.24	14	ns

Table 4: Two tailed t-test results of comparisons between pre and post-HREP habitat sites for fall (early and late) nektonic (g/m^3 +/- SE) and benthic (g/m^2 +/- SE) dry biomass at Swan Lake, Illinois River backwater.

Early Fall Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	5.00	1.60	-	-	-	-	-	-	-	-	-	-	-	-
Moist Soil	4	2.70	1.80	4	0.86	0.36	4	1.33	0.75	-1.00	6	ns	-0.70	6	ns
Open Water	8	0.60	0.20	8	0.27	0.18	7	0.32	0.13	-1.22	14	ns	-1.17	13	ns
Submerged Vegetation	4	45.30	44.00	-	-	-	-	-	-	-	-	-	-	-	-

Early Fall Benthic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Open Water	8	1.30	0.40	8	8.64	3.58	7	5.44	1.74	2.04	14	$P < 0.10$	2.32	13	$P < 0.05$
Submerged Vegetation	4	24.40	23.50	-	-	-	-	-	-	-	-	-	-	-	-

Late Fall Nektonic	Pre-HREP			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
Emergent Vegetation	9	17.40	3.50	-	-	-	-	-	-	-	-	-	-	-	-
Moist Soil	4	6.60	2.00	4	1.17	0.37	3	1.51	0.68	-2.67	6	$P < 0.05$	-2.41	5	$P < 0.10$
Open Water	8	0.30	<0.10	7	0.01	0.01	7	0.16	0.06	-2.84	13	$P < 0.05$	-1.15	13	ns
Submerged Vegetation	4	8.00	1.90	-	-	-	-	-	-	-	-	-	-	-	-

Late Fall Benthic	8.00			Post-HREP(2004)			Post-HREP(2005)			Pre vs. Post-HREP(2004)			Pre vs. Post-HREP(2005)		
Habitat	n	mean	SE	n	mean	SE	n	mean	SE	T-statistic	df	P-value	T-statistic	df	P-value
No Pre-HREP Data	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Table 5: Kruskal-Wallis test results of pre and post-HREP comparisons between all sites combined during spring and fall (early and late) for nektonic (No./m^3 +/- SE) and benthic (No./m^2 +/- SE) abundance estimates at Swan Lake, Illinois River backwater.

Season		Sample Type	Pre-HREP		Post-HREP(2004)		Post-HREP(2005)		Kruskal-Wallis Test Results			
			mean	SE	mean	SE	mean	SE	n	Chi-Squared	df	P-value
Spring												
	Early											
		Nektonic	4500	1364	1411	782	2299	730	4	3.50	2	P = 0.1738
		Benthic	7388	1743	6940	2173	18616	6262	3	3.29	2	P = 0.1931
		Total (No./Sample)	556		341		820					
	Late											
		Nektonic	100	0	10368	2098	4912	3361	4	8.29	2	P = 0.0158
		Benthic	4189	1795	26579	4692	36525	7149	4	8.00	2	P = 0.0183
		Total (No./Sample)	158		1626		1634					
Fall												
	Early											
		Nektonic	6525	3322	1587	1324	1629	612	4	3.11	2	P = 0.2106
		Benthic	2290	671	10248	6291	7945	2661	2	3.43	2	P = 0.1801
		Total (No./Sample)	502		472		392					
	Late											
		Nektonic	39725	16816	2854	2512	3950	2003	4	5.35	2	P = 0.0690
		Benthic	-	-	-	-	-	-	-	-	-	-
		Total (No./Sample)	-		-		-					

Table 6: Kruskal-Wallis test results of pre and post-HREP comparisons between all sites combined during spring and fall (early and late) for nektonic (g/m^3 +/- SE) and benthic (g/m^2 +/- SE) biomass estimates at Swan Lake, Illinois River backwater.

Season		Sample Type	Pre-HREP		Post-HREP(2004)		Post-HREP(2005)		Kruskal-Wallis Test Results			
Spring			mean	SE	mean	SE	mean	SE	n	Chi-Squared	df	P-value
	Early											
		Nektonic	0.80	0.19	0.44	0.26	0.78	0.36	4	0.81	2	P = 0.6668
		Benthic	1.93	0.43	5.12	1.69	7.51	2.08	3	4.62	2	P = 0.0992
		Total (g/sample)	0.12		0.21		0.32					
	Late											
		Nektonic	0.10	0.00	1.42	0.66	0.71	0.25	4	4.94	2	P = 0.0845
		Benthic	6.00	4.71	5.49	0.63	8.30	1.54	4	2.81	2	P = 0.2457
		Total (g/sample)	0.22		0.29		0.35					
Fall												
	Early											
		Nektonic	13.40	10.67	0.28	0.20	0.63	0.28	4	5.67	2	P = 0.0586
		Benthic	12.85	11.55	5.10	3.54	3.18	2.26	2	0.86	2	P = 0.6514
		Total (g/sample)	1.33		0.20		0.16					
	Late											
		Nektonic	8.08	3.53	0.36	0.28	1.19	0.69		5.65	2	P = 0.0592
		Benthic	-	-	-	-	-	-	-	-	-	-
		Total (g/sample)	-		-		-					

Table 7: Combined estimated means of nektonic and benthic abundance (No./sample) and biomass (g/sample) among habitat types during spring and fall (early and late) at Swan Lake, Illinois River backwater.

Early Spring Total	Pre-HREP		Post-HREP(2004)		Post-HREP(2005)	
Habitat	Abundance	Biomass	Abundance	Biomass	Abundance	Biomass
Emergent Vegetation	647	0.11	346	0.13	712	0.30
Moist Soil	542	0.14	224	0.19	1374	0.53
Open Water	286	0.09	409	0.29	559	0.18

Late Spring Total	Pre-HREP		Post-HREP(2004)		Post-HREP(2005)	
Habitat	Abundance	Biomass	Abundance	Biomass	Abundance	Biomass
Emergent Vegetation	85	0.08	3373	0.70	1181	0.45
Flooded Forest	153	0.06	2209	0.30	1055	0.40
Moist Soil	51	0.02	1406	0.24	2741	0.46
Open Water	341	0.73	1053	0.25	876	0.15

Early Fall Total	Pre-HREP		Post-HREP(2004)		Post-HREP(2005)	
Habitat	Abundance	Biomass	Abundance	Biomass	Abundance	Biomass
Open Water	110	0.09	643	0.33	514	0.22

Late Fall Total	Pre-HREP		Post-HREP(2004)		Post-HREP(2005)	
Habitat	Abundance	Biomass	Abundance	Biomass	Abundance	Biomass
NA	-	-	-	-	-	-

RESPONSE OF WATERFOWL TO THE SWAN LAKE HREP: SHOULD ABUNDANCE OR BEHAVIOR BE USED TO ASSESS HABITAT QUALITY FOR MIGRATING WATERBIRDS IN SPRING

SUMMARY

The predicted responses of ecosystem components following wetland restorations or enhancements are rarely evaluated. In the rare cases that waterbird responses to wetland restoration and enhancement activities are monitored, the response variable is typically the change in abundance, diversity, or richness of waterbirds. Abundance or density, however, may be a misleading measure of habitat quality for wildlife. An alternative for determining success of habitat improvements is to monitor feeding activity of waterbirds on the wetland. We used behavioral and abundance data from 2 guilds of ducks to determine if behavior provides a better indicator of habitat quality when quality is based on food availability. Behavioral data was a good predictor of habitat quality for dabbling ducks only, while indicators of abundance provided only marginal predictability of habitat quality for dabbling ducks and were negatively correlated to our estimate of habitat quality for diving ducks. We conclude that both behavioral data and abundance may only be appropriate for comparison among wetlands when ecological variation among waterbird guilds is considered, however, behavioral data may serve as a better predictor.

Introduction

Historically, the Illinois River Valley (IRV) and its flood plain wetlands supported a large number of migrating waterbirds during fall and spring (Bellrose et al. 1979). Factors associated with commerce and growing human population including: diversion of water from Lake Michigan in Chicago, creation of levee and drainage districts that decreased the number of lakes and flood storage capacity, more pronounced flooding events, lock and dam construction to aid commercial navigation, and extensive sedimentation have reduced and degraded this ecosystem throughout the last century (Mills et al. 1966; Bellrose et al. 1983). Despite being highly degraded, the IRV continues to support hundreds of thousands of waterbirds throughout spring and fall migration (Havera 1999). Because of its continued value to waterbirds, this region has been the focus of wetland restoration and enhancement activities by a large number of conservation agencies. One mechanism through which wetlands are restored and enhanced in this region is the U. S. Army Corps of Engineers Habitat Rehabilitation and Enhancement Projects (HREP). Swan Lake, a backwater wetland of the Illinois River, underwent a HREP because of its poor quality as fish and wildlife habitat (USACE 1991; Smietanski 1994). The HREP was intended to improve the quality of fish and waterbird habitat on Swan Lake by increasing food availability during the time waterbirds and fish are most abundant. A specified goal of the HREP was to monitor the restoration impact on fish and wildlife.

Walters (1986) first described adaptive resource management as implementing a specific management action and measuring the effectiveness of that action on the desired response, thus reducing the biological uncertainty in the response to specific management or restoration activities. Although restoration scientists view adaptive resource management as the most efficient way of increasing our understanding of the processes that underlie functioning ecosystems, this approach is rarely practiced (Walters 1986, Lancia et al. 1996). Wetland restoration and enhancement activities most often are designed to restore multiple trophic levels of the food web by promoting the propagation of native primary producers with the implicit assumption that lower and higher order consumers, such as aquatic macroinvertebrates and waterbirds, will respond favorably by exploiting the primary producers. In the rare case that the response of a secondary consumers (e.g. waterbirds) to wetland restoration or enhancement activities is monitored, the response variable is typically the change in abundance, diversity, or richness of waterbirds using the wetland (e.g**.** Murkin & Kadlec 1986; Delphy & Dinsmore 1993; Brown 1995; Hartman 1994; Brown & Smith 1998; Dugger et al. 2005). In addition to metrics of abundance, residence time at migratory stopover sites has been used as an index of wetland quality. Bellrose et al. (1979) found an index of residence time at stopover sites during spring migration to be positively correlated to food availability on those sites. Thus, residence time on stopover sites may also be used to evaluate the success of wetland enhancement or restoration activities because length of time individuals spend at stopover sites is related to current condition or quality of a wetland (Bellrose et al. 1979). Metrics of abundance of wildlife, however, may be a misleading measure of habitat quality (O'Conner 1981; Van Horne 1983). At low population densities, only the best habitat is used, but when population density is greater, even marginal habitat may be used extensively (Fretwell 1972; O'Connor 1981; Van Horne 1983). Additionally, a number of factors may influence diversity or abundance of waterbirds on a specific wetland that are unrelated to the wetland restoration and enhancement activities or the ability of the wetland to provide forage (Hochbaum 1944; Mills et al. 1966; Bellrose & Crompton 1970; Bellrose 1974; Brown & Dinsmore 1986; 1991; Hemesath & Dinsmore 1993; Hartman 1994; VanRees-Siewart & Dinsmore 1996; Havera 1999). For example, waterfowl often roost on large open wetlands during the day, and fly to smaller more productive wetlands during morning and evening feeding bouts. A more direct alternative to monitoring waterbird abundance, diversity, or residence time may be to monitor feeding activity of waterbirds on the wetland to determine the success of habitat improvements designed to increase the production of lower order producers for exploitation by waterbirds. Optimal foraging theory predicts animals should increase foraging intensity when feeding in areas of greater food availability (Stephens & Krebs 1986; Fritz et al. 2001). Monitoring feeding behavior should provide a more direct measure for determining if higher order consumers are responding to restoration and enhancement activities as assumed, thus, be impacted less than indicators of abundance and residence time by external influences (Smith unpublished data).

The first objective of our study was to test the implicit assumption that waterbirds will respond to the increase in food produced from the HREP by modifying their migratory behavior and daily activities. More specifically, we test the prediction that ducks using post-HREP Swan Lake will respond to an increase in food availability by increasing in peak abundance, duck-use-days (DUDs), residence time, and percent time spent feeding relative to the pre-HREP period as estimated by Smietanski (1994). Because the HREP activities designed to promote growth of primary and secondary producers, those most often exploited by ducks, were restricted to the middle unit of Swan Lake, we also predict ducks on this unit will account for most of the changes in abundance and behavior between the pre- and post-HREP periods.

Unfortunately, only a limited number of empirical studies have tested and supported this theoretical relationship between feeding behavior and food availability (Murkin & Kadlec 1986; DuBowy 1997). Thus, the second objective of our study is to test the theoretical relationship between food availability and feeding intensity for ducks, a higher order consumer that feeds on moist-soil plant seeds and aquatic macroinvertebrates. More specifically, test the prediction that the proportion of time spent feeding will be more closely correlated than duck abundance to food availability.

Methods

Study site

Swan Lake is an 1100 ha backwater wetland adjacent to the Illinois River near the confluence of the Mississippi River in Calhoun County, IL. Swan Lake is jointly owned by the United States Fish and Wildlife Service (USFWS) and Illinois Department of Natural Resources (IDNR). The USFWS portion is managed by the Calhoun Division of Two Rivers National Wildlife Refuge (NWR), which is part of the Mark Twain NWR complex. Established as a NWR because of its importance to migrating birds using the Mississippi and Illinois River basins, Swan Lake is a mosaic of open-water, moist-soil, and shoreline habitats (Figure 1). Historically, Swan Lake was oligotrophic with flocculent bottom sediments composed of organic matter (Bellrose et al. 1983). These water conditions supported a diverse plant community that was excellent waterbird foraging habitat (Bellrose et al. 1979). The Alton Dam (Lock and Dam #26) was installed on the Mississippi River at Alton, IL in 1938. This increased Swan Lake's surface area, temporarily increasing the value of Swan Lake as waterbird habitat (Bellrose et al. 1979). Within years, however, sedimentation in the lake increased from flood events on the Illinois River and runoff from adjacent agricultural lands. Increased sedimentation decreased water clarity, aquatic plant growth, and lake depth, thereby, reducing Swan Lake's value as fish and wildlife habitat (Bellrose et al. 1983). In an evaluation of waterfowl use, habitat, and food availability conducted in 1992 and 1993, Smietanski (1994) reported that Swan Lake provided poor waterfowl habitat based on low waterfowl abundance, waterfowl spending little time feeding, and low availability of waterfowl foods. To address the various environmental problems on Swan Lake, a HREP was initiated by the USACE in 1994. To accomplish the goals of improving food availability for ducks and fish, the USACE (1991) proposed several objectives including: reduce sedimentation rates into the lake and stabilize water levels during the summer growing season by isolating Swan Lake from the Illinois River with a levee, consolidate bottom sediments and control wave action to improve water quality, and allow for the independent management of waterfowl and fish with multiple lake units. The plan divided the USFWS owned portion of the lake into two units that could be individually managed for fish and waterfowl. Constructed features included a levee isolating Swan Lake from the Illinois River, equipped with water control structures and pumps, a levee dividing the lake into 2 units (Lower and Middle Swan Lake) with a water control structure, and chains of islands intended to reduce wind-fetch, wave action, and re-suspension of sediments. During our study, Middle Swan Lake was managed to produce moist-soil (Fredrickson & Taylor 1982) and submersed aquatic vegetation, providing foraging habitat for macroinvertebrates and ducks. Correspondingly, most of this unit was de-watered in summer to expose and consolidate bottom sediments, and promote growth of annual moist-soil plants.

The unit was slowly flooded during fall. Lower Swan Lake was managed primarily to provide fish habitat and remained connected to the Illinois River, allowing lake water levels to fluctuate with river water level and to allow unrestricted fish passage (Schultz 2006). Habitat in Lower Swan Lake differed little from pre-HREP conditions. A third unit, Upper Swan and Fuller Lakes (collectively known as Fuller Lake Game Management Area) is owned and managed by the IDNR and was unaffected by the HREP. The IDNR manages the area as a large moist-soil unit, primarily for waterfowl hunting (approximately 25 October – 25 December). In addition to Lower and Middle Swan Lake, Two Rivers NWR manages several smaller moist-soil impoundments adjacent to the south end of Swan Lake, which provide approximately 100 ha of shallow-water wetlands for migratory waterbirds and were unaltered by the HREP.

Field methods

Abundance

We conducted surveys of duck abundance twice weekly during spring migration on all waterfowl habitats within the Swan Lake complex. We began surveys shortly after ice-out in early March in 2004 and early February in 2005 and ceased mid to late April. We enumerated all ducks by species and recorded the associated lake unit. We varied survey start time and location to prevent counting the same location at the same time each survey. To avoid double counting, we counted flooded-forest habitats last, because ducks that flushed from forests settled in other habitats. We conducted surveys from points and transects previously established by Smietanski (1994). We also established a new transect along the newly built east levee of Middle and Lower Swan Lake to survey ducks in an area that had been surveyed using a boat by Smietanski (1994,

Figure 1). We observed birds using 10x binoculars and a 20-60x variable power spotting scope and identified, counted, and recorded all visible ducks at points or along transects.

Duck-use-days (DUDs) are the number of ducks using an area for a specified period of time (in days) and is commonly used as an indicator of duck use of a specific wetland, complex, or region over a specified time. For example if 10 ducks were observed on a wetland every day for 10 days, then the DUDs for that 10 day period would be 100. We estimated DUDs for each of the 3 units of Swan Lake as well as the entire complex by assigning the number of ducks counted during a survey to each day following, until we conducted another survey. Using DUDs as an indicator of abundance allows us to account for the variation in abundance over the entire migratory period.

Behavior

We used scan sampling to estimate the proportion of time dabbling ducks spent in various behaviors and focal sampling to estimate the proportion of time diving ducks spent in various behaviors (Altmann 1974). When scan sampling, we used a spotting scope to scan from one individual to the next, recording each individual's behavior and sex as they passed the center of the scope, sampling one species per scan. Methods for scan sampling closely followed Smietanski (1994), and used a flock size of 60 individuals as 1 sample. Flocks >120 individuals were divided into multiple samples. Flocks of <60 individuals in the same habitat were scanned sequentially to comprise 1 sample (Smietanski 1994). We were careful to ensure data collected were comparable to pre-HREP data and other studies to make inferences about the influence of change in quality of habitat on Swan Lake. Because it was impossible to distinguish between resting and the inter-dive loaf when scanning through a flock of diving ducks, we classified birds that appeared this way as resting.

Scan sampling underestimates the amount of time diving ducks spend feeding, because birds feeding underwater or in an inter-dive loaf are not observed or are misidentified (Hohman 1984, Baldassarre et al. 1988). Thus, we used focal sampling to obtain a more accurate depiction of diving duck behavior on Swan Lake to correlate with food availability. When focal sampling, we randomly selected an individual from a flock or small group and observed for 10 minutes (Altmann 1974). Small groups were used because unmarked individuals can not be followed in large feeding flocks of diving ducks. During this period, we recorded the amount of time an individual spent in each activity. Behavior categories for both sampling methods included feeding, resting, locomotion, social, and “other” following Smietanski (1994). Feeding included surface feeding, tipping up, diving, and the inter-dive loaf for diving ducks observed using focal sampling.

Statistical analysis

To determine if time spent feeding differed between the pre- and post-HREP periods, we used Kruskal-Wallis tests to evaluate differences in proportion of time dabbling and diving ducks spent feeding between lake units and years. Kruskal-Wallis tests compare median values; however, comparing means may be a more useful measure of central tendency for these data. Therefore, results are reported as mean percent time feeding ± 1 standard error. We considered differences significant when $P < 0.05$.

To determine if there was a relationship between duck use and habitat quality we correlated lake-unit specific estimates of DUDs per hectare of habitat with estimates (grams/m^2) of aquatic macroinvertebrates and moist-soil seeds for each lake unit. We estimated DUDs per hectare of habitat by dividing the total DUDs of a specific lake unit by the total hectares of the lake unit to control for size variation among lake units. Similarly, to determine if there was a relationship between feeding behavior and habitat quality, we correlated mean percent time feeding of ducks on each lake unit with estimates (grams/m^2) of aquatic macroinvertebrates and moist-soil seeds for each lake unit. Because various guilds of ducks (diving ducks vs. dabbling ducks) likely select habitats based on different criteria, we repeated these analyses for each guild separately. Because > 90% of the diet of dabbling ducks in the spring of 2004 was moist-soil seeds (Smith 2007), we correlated estimates of moist-soil seed availability only to percent time feeding and DUDs of dabbling ducks in 2004, whereas, moist-soil seeds and macroinvertebrate biomass were combined as estimates of food availability for all other correlations.

Results

Response to restoration

Because metrics of abundance based on a complete census have no associated estimates of variance, we were unable to conduct a statistical test to determine if differences observed in our abundance metrics were statistically significant. The large differences in these metrics between the pre and post-HREP periods, however, suggest that use of the Swan Lake complex by ducks increased after the completion of the HREP (Table 1).

Feeding was the primary dabbling duck behavior during 2004 and 2005 (Table 2); dabbling ducks spent 66.6 ± 2.6% (SE) of their time feeding. Dabbling duck feeding differed by lake unit ($\chi^2_3 = 20.5$, $P < 0.001$). Dabbling ducks fed most in moist-soil units (82.0 ± 4.4%), followed by the Upper (68.8 ± 3.1%), Middle (56.4 ± 4.2%), and Lower (< 1%) units of Swan Lake. Alternatively, diving ducks spent more time resting (58.0 ± 1.5%) than feeding (20.3 ± 1.6%). Time spent feeding by diving ducks differed by year ($\chi^2_1 = 59.8$, $P < 0.001$), but did not differ by lake unit ($\chi^2_2 = 0.145$, $P = 0.930$).

Percent time feeding was significantly greater during both post-HREP years for dabbling ducks($\chi^2_1 =48.7$, $P < 0.001$ [2004], $\chi^2_1 = 39.8$, $P < 0.001$ [2005]) and significantly greater during one year for diving ducks ($\chi^2_1 = 38.5$, $P < 0.001$ [2004], $\chi^2_1 = 3.3$, $P = 0.071$ [2005], Table 2). Additionally, DUDs, peak abundance, and residence time were all greater for both post-HREP years relative to the pre-HREP year. These differences were most pronounced on Middle Swan Lake, the unit modified specifically to provide habitat for ducks.

Abundance vs. Behavior

In 2004 neither DUDs nor feeding behavior were strongly correlated to food availability among wetland units (Fig. 2). Similarly, in 2005, DUDs of the 3 units was only marginally correlated to the density of food available in those units, however, feeding behavior of the ducks was strongly correlated to food availability among the 3 units (Fig. 2). When partitioning data among the two guilds, diving duck DUDs and percent time feeding were negatively correlated to our measure of lake unit specific food availability in both 2004 and 2005 (Fig. 3), whereas, dabbling duck DUDs were weakly positively correlated to food availability in both 2004 and 2005 and feeding behavior was strongly correlated to food availability in both 2004 and 2005 (Fig. 4).

Discussion

Response to restoration

As predicted, estimated peak abundance on Middle Swan Lake increased considerably (760% in spring 2004 and 805% in spring 2005) from pre- to post-HREP (Table 1). The observed increase in abundance could be due to an increase in the continental population level, a change in local use patterns, or an increase in individual residence time due to ambient temperature or increased food availability (Hochbaum 1944; Bellrose 1974).

Continental duck population could affect abundance at the local scale. Duck populations have generally increased since the pre-HREP evaluation of Swan Lake (Wilkins et al. 2005). Greater continental populations of ducks should result in more duck use of individual wetlands or wetland complexes; thus, a proportional increases may be observed at Swan Lake. A liberal estimate of continental duck population increase is around 20% (Wilkins et al. 2005). Peak abundance on Middle Swan Lake increased 740% in 2004 and 784% in 2005 and DUDs increased 823% in 2004 and 1382% in 2005 from the pre-HREP year. These results indicate that little of the observed increase in abundance of ducks on Swan Lake is due to an increase in the continent wide population.

The observed increase in peak abundance and DUDs could be due to increased residence time (Bellrose et al. 1979). Because groups of ducks continuously arrive at and leave stopover sites during migration, an increase in residence time would lead to an increase in peak abundance and DUDs. Residence time was most likely affected by either ambient temperature or food availability. Reproductive success declines as nest initiation date increases in waterfowl as well as other organisms (Drent & Daan 1980), thus, waterfowl likely have a time-selected spring migration pattern (Alerstam & Lindstrom 1990). Time minimizers should depart a stopover site when resources are at a level equal to or below the expected levels at future stopover sites (Stephens & Krebs 1986; Alerstam & Lindstrom 1990). Therefore, ducks should remain at spring stopover sites when they experience abnormally cold temperatures, despite depleted food sources, because wetlands farther north will likely be frozen, providing even fewer resources. In fact, ambient temperature is reported as the climatic variable that has the greatest impact on waterfowl migration patterns (Richardson 1977; Flickinger 1981; Baldasarre and Bolen 1984). Mean monthly temperatures were slightly warmer in both spring 2004 and spring 2005 than in spring 1993 (National Oceanic and Atmospheric Administration). Therefore, we predict there to be shorter residence times, lower peak abundance, and fewer DUDs in the post-HREP years if ambient temperature is the major cause of variation in duck abundance between the pre- and post-HREP periods. Our results are contradictory to this prediction, therefore, variation in weather is an unlikely explanation for our observed increase in peak abundance and duck use post-HREP.

We conclude that the most likely explanation for the increase in peak abundance and DUDs is a change in local feeding distribution and increased individual stopover duration in response to increased food availability. We propose the HREP at Swan Lake increased food resources sufficiently to alter feeding sites of some local migrants and allowed ducks to remain on the site for a longer period of time. As described above, our observed increase in the index of residence time could explain both the observed increased peak abundance and DUDs. An additional and indistinguishable explanation, however, is that local migrants have modified their site selection to spend more time feeding on the Swan Lake complex. We define local migrants as ducks that have a history of migrating through this area. Swan Lake lies near the confluence of several major migration corridors in the Mississippi Flyway, causing a large number of ducks to use, or pass nearby this wetland during migration periods. Waterfowl quickly and readily exploit newly available food sources (e.g. recently harvested agricultural fields, rising flood waters); therefore, establishment of abundant, high quality foods likely elicited a larger proportion of the ducks that historically passed through this migration corridor to use the Swan Lake complex as a stopover site. The majority of this increase occurred on Middle Swan Lake, indicating a direct reflection of greater food availability resulting from the HREP (Table 1). Additionally, the theoretical relationship between feeding behavior and food availability was strongly supported by our comparison between pre- and post-HREP behaviors. Time spent foraging by both diving and dabbling ducks was low when food resources were scarce (Smietanski 1994), but increased when food resources became abundant (this study, D. Grulkowski, Southern Illinois University, Carbondale, IL, personal communication).

Surprisingly, the increase in estimated peak abundance and DUDs on the overall complex was not entirely due to an increase in peak abundance and DUDs on Middle Swan Lake. Our estimate of peak abundance also increased on other units during both post-HREP years by an average of 157%, although peak abundance on the moist-soil units remained relatively stable between the pre- and post-HREP periods. Furthermore, DUDs increased on all other lake units (including moist-soil units) by an average of 399%. We propose that increased duck use of unmodified Swan Lake units is due to the way in which ducks identify and select feeding patches, and may have important implications for the juxtaposition of restored or enhanced wetlands. Ducks are crepuscular feeders, flying to feeding areas from roost sites during both morning and evening. Although we consider the Swan Lake complex to have numerous units (i.e. Upper/Fuller, Middle, and Lower Swan Lake, and moist-soil units), these units are only separated by narrow (<10 m) levees. Waterfowl commonly fly >30 km from roosting to feeding locations, thus, this complex is likely perceived as 1 feeding area by ducks (Ely & Raveling 1989; Austin & Humburg 1992; Hill & Fredrick 1997; Ackerman et al. 2006). Foods consumed by ducks (aquatic macroinvertebrates and moist-soil seeds) are distributed patchily throughout the environment. Ducks, like numerous other animals, likely have imperfect knowledge of their environment; thus, they sample probable feeding areas until a high quality patch of food is located (Stephens & Krebs 1986). Because ducks likely perceive the complex as one feeding area, they search both highly productive areas affected by the HREP, and less productive neighboring units that were unaffected by the HREP. Ducks were likely able to locate more food patches on lake units unmodified by the HREP, because a potentially greater number of ducks were searching the area for a longer time. We hypothesize, under the assumption that nutrient acquisition is the most limiting factor for waterbirds outside of the breeding season (Fretwell 1972; Loesch et al. 1994), directly increasing food availability on one unit of Swan Lake may have indirectly increased food availability on neighboring units. Ducks spent more time searching for and finding a greater proportion of the available food on the unmodified units, leading to an unexpected increase in food supplied by the overall Swan Lake complex. We would like to remind readers, however, that although our results suggest that directly impacting resources on one wetland may have indirect impacts on food resources available to waterbirds on neighboring wetlands, results of this project have only elicited this hypothesis; future research is needed for it to be adequately tested.

Abundance vs. Behavior

Wetland enhancement and restoration activities are most often designed to restore a hydrologic regime that supports hydrophytes with the assumption that individuals from higher trophic levels will respond by exploiting these hydrophytes as a nutritional source. Estimates of diversity and abundance of waterbirds are often used to assess wetland value or success of restoration efforts. We found neither feeding behavior nor indicators of abundance were strongly correlated to food availability when using data collected from 2 guilds of ducks (dabbling and diving ducks, Fig. 2). Thus, although indicators of abundance of waterbirds, such as ducks, may be useful for assessing temporal variation of habitat quality within a site (Smith 2007), it may not be a good predictor of habitat quality among sites. When analyzing the data for the 2 guilds separately, we found that abundance indicators and feeding behavior were negatively correlated to our estimate of food availability for diving ducks (Fig.3). Conversely, abundance and behavior were positively correlated to food availability for dabbling ducks; although, feeding behavior was more highly correlated to food availability than indices of abundance for dabbling ducks (Fig. 4).

We believe the negative correlations between diving duck feeding behavior, DUDs and food availability may be a result of inappropriately defined food availability etimates for diving ducks in this study, and the physical structure of the habitat that diving ducks prefer. Diving ducks have adapted to feeding in deep water (> 30 cm), effectively reducing competition with dabbling ducks (Nudds 1983). Primary productivity of deeper, more permanent wetlands is lower than shallower wetlands which have intermittently exposed substrates, thus, resources ducks exploit are less abundant in deeper permanent wetlands (Fredrickson & Taylor 1982). During our study, water levels on >80% of Upper Swan Lake and >50% of Middle Swan Lake were < 30 cm deep and managed to promote moist-soil plant growth, while deeper portions of both lake units remained flooded, even during maximum drawdown. Water levels in these areas remained fairly shallow throughout spring migration, except during a portion of spring of 2004 when water levels were higher than normal. Alternatively, during our study, > 95% of Lower Swan Lake was > 30 cm in depth and managed primarily as fish habitat. Thus, although diving duck behavior and DUDs were negatively correlated to our estimate of food availability, they were both positively correlated to the proportion of the lake unit with water depth preferred by diving ducks. Interestingly, in 2004, high water levels flooded normally shallow (< 30 cm), highly productive, moist-soil habitat to a depth more suited for diving ducks (> 30 cm). Diving ducks responded with feeding rates significantly higher (49.1 ± 2.9%, $\chi^2_1 = 59.8$, $P < 0.001$) than in 2005 (22.3 ± 1.7%, Smith 2007), a year when only the less productive, more permanently flooded habitats were at a depth >30 cm.

Alternatively, diving duck feeding behavior may not be as strongly related to food availability as dabbling duck feeding behavior because of the differences in feeding techniques. A tradeoff associated with increased foraging time is decreased vigilance, and increased predation risk (Guillemain et al. 2001, Fritz et al. 2002). This risk varies by feeding strategy based on whether the duck has some field of vision while feeding (i.e. dabbling with eyes above water, tipping up with head submerged, or diving below the surface, Guillemain et al. 2001).

Diving below the surface to feed may make diving ducks more susceptible to predation, thus predation pressure may play a stronger role than food availability in determining habitat selection and feeding intensity in diving ducks. Therefore, diving ducks may be selecting habitat based more on predator avoidance than food availability.

The correlation between feeding behavior and food availability was much stronger than the correlation between DUDs and food availability for dabbling ducks, supporting our prediction that feeding behavior is a better indicator of habitat quality. Unlike diving ducks, dabbling ducks tend to forage in shallow waters (< 30 cm deep) and roost in deeper open waters. Substrates of shallow waters are frequently exposed during the summer growing period, therefore, shallow wetlands tend to be more productive (Fredrickson & Taylor 1982). When wetland quality is based on productivity, estimates of bird abundance may return biased results; birds will be counted in productive feeding habitat, as well as less productive roosting habitat. We conclude that feeding behavior is likely a better predictor of wetland quality, as determined by food availability, than indicators of abundance if researchers control for ecological differences among wetland bird guilds.

Implications for practice

- Waterbirds responded to habitat enhancement through increased exploitation of an increased food supply.
- Waterbird abundance and feeding behavior appear to be good indicators of temporal change in productivity within a wetland.

● Increasing resource availability within one unit of a wetland complex may increase waterbird exploitation of resources in neighboring units.

● Neither feeding behavior nor indicators of abundance of waterbirds are good indicators of habitat quality when habitat quality is defined by the abundance of food availability and data are derived from birds from multiple guilds.

● Both feeding behavior and indicators of abundance are correlated to habitat quality (food availability) at the guild specific level but feeding behavior appears to be more closely correlated.

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Table 1. Duck use-days (DUDs), peak abundance, and an index of individual residence time (R/T) in days by lake unit at Swan Lake, Illinois during the pre-HREP (1993) and post-HREP (2004 and 2005) evaluation periods. Residence time is estimated as the ratio of DUDs to peak abundance.

	1993			2004			2005		
Lake Unit	DUDs	Peak	R/T	DUDs	Peak	R/T	DUDs	Peak	R/T
Lower	65,617	3,158		98,126	4,291		272,141	7,793	
Middle	19,818	953		183,052	8,005		293,798	8,412	
Moist-soil	24,427	1,175		31,777	1,390		59,165	1,994	
Upper/Fuller	21,156	1,017		45,030	1,969		261,727	7,495	
Total	131,018	6,303	20.8	357,985	15,655	22.9	891,710	25,694	34.3

Table 2. Percent of time spent feeding and standard errors (SE) of dabbling and diving ducks estimated using scan and focal sampling at pre-HREP (spring 1993) and post-HREP (spring 2004 and 2005) Swan Lake, Illinois.

Year	Dabblers	Divers	
		Scan	Focal
1993	17.5 2.0	4.5 0.4	
2004	66.2 6.2	41.4 2.5	63.4 5.7
2005	61.1 5.3	11.4 0.8	41.8 3.8

Figure legends

Figure 1. Location and depiction of Swan Lake, Calhoun County, Illinois. Points and lines denote survey points and transects.

Figure 2. Correlations of lake-unit specific duck use days (DUDs) per hectare of wetland, lake-unit specific percent of time spent feeding by ducks and estimated g/m2 of moist-soil seeds and macroinvertebrates commonly consumed by ducks for dabbling and diving ducks combined in 2004 and 2005 at Swan Lake, IL.

Figure 3. Correlations of lake-unit specific duck use days per hectare of wetland, lake-unit specific percent of time spent feeding by only ducks and estimated g/m2 of moist-soil seeds and macroinvertebrates commonly consumed by ducks for diving ducks only in 2004 and 2005 at Swan Lake, IL.

Figure 4. Correlations of lake-unit specific duck use days per hectare of wetland, lake-unit specific percent of time spent feeding by only ducks and estimated g/m2 of moist-soil seeds and macroinvertebrates commonly consumed by ducks for dabbling ducks only in 2004 and 2005 at Swan Lake, IL. We used moist-soil seeds only instead of moist-soil seeds and macroinvertebrates combined as an estimate of food availability in 2004 because < 10% of the biomass of dabbling ducks diet consisted of macroinvertebrates, whereas, in 2005 > 25% of the diet of dabbling ducks was macroinvertebrates.

Figure1

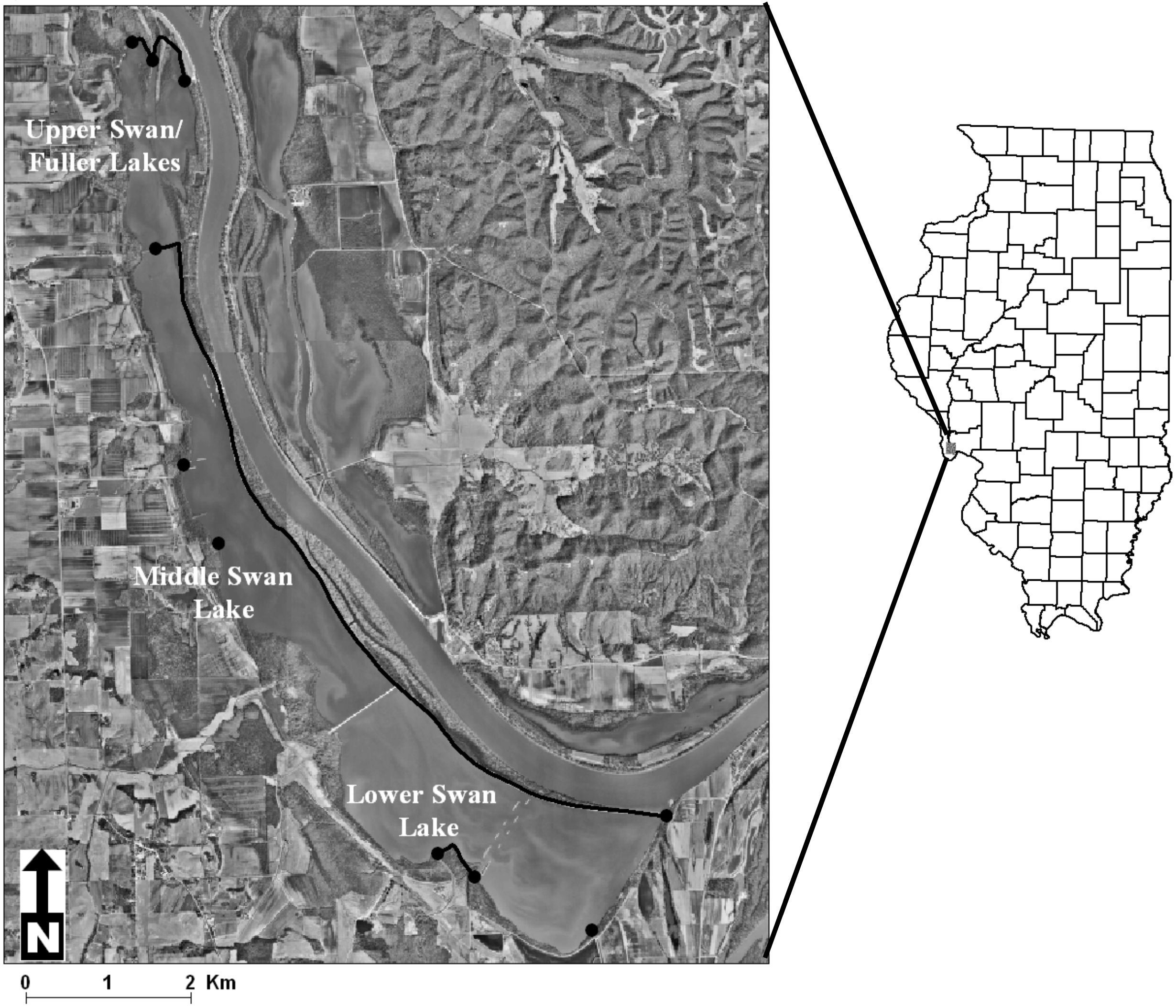
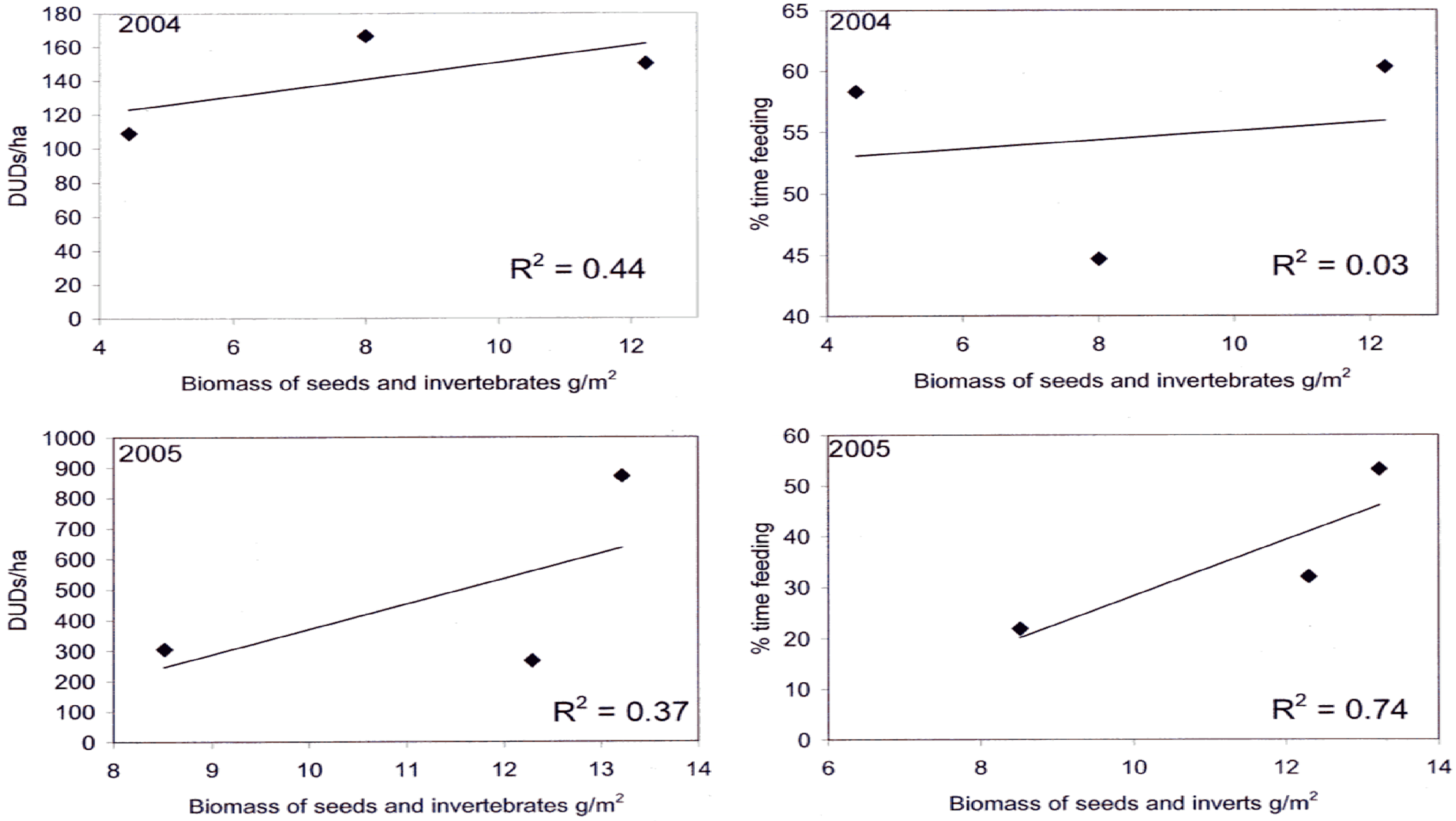

Upper Swan/
Fuller Lakes
Middle Swan
Lake
Lower Swan
Lake
N
0 1 2 Km

Figure 2



2004
180
160
140
120
100
80
60
40
20
0
DUDs/ha
4
6
8
10
12
Biomass of seeds and invertebrates g/m2
R2 = 0.44
2004
65
60
55
50
45
40
% time feeding
4
6
8
10
12
Biomass of seeds and invertebrates g/m2
R2 = 0.03
2005
1000
900
800
700
600
500
400
300
200
100
0
DUDs/ha
8
9
10
11
12
13
14
Biomass of seeds and invertebrates g/m2
R2 = 0.37
2005
60
50
40
30
20
10
0
% time feeding
6
8
10
12
14
Biomass of seeds and inverts g/m2
R2 = 0.74

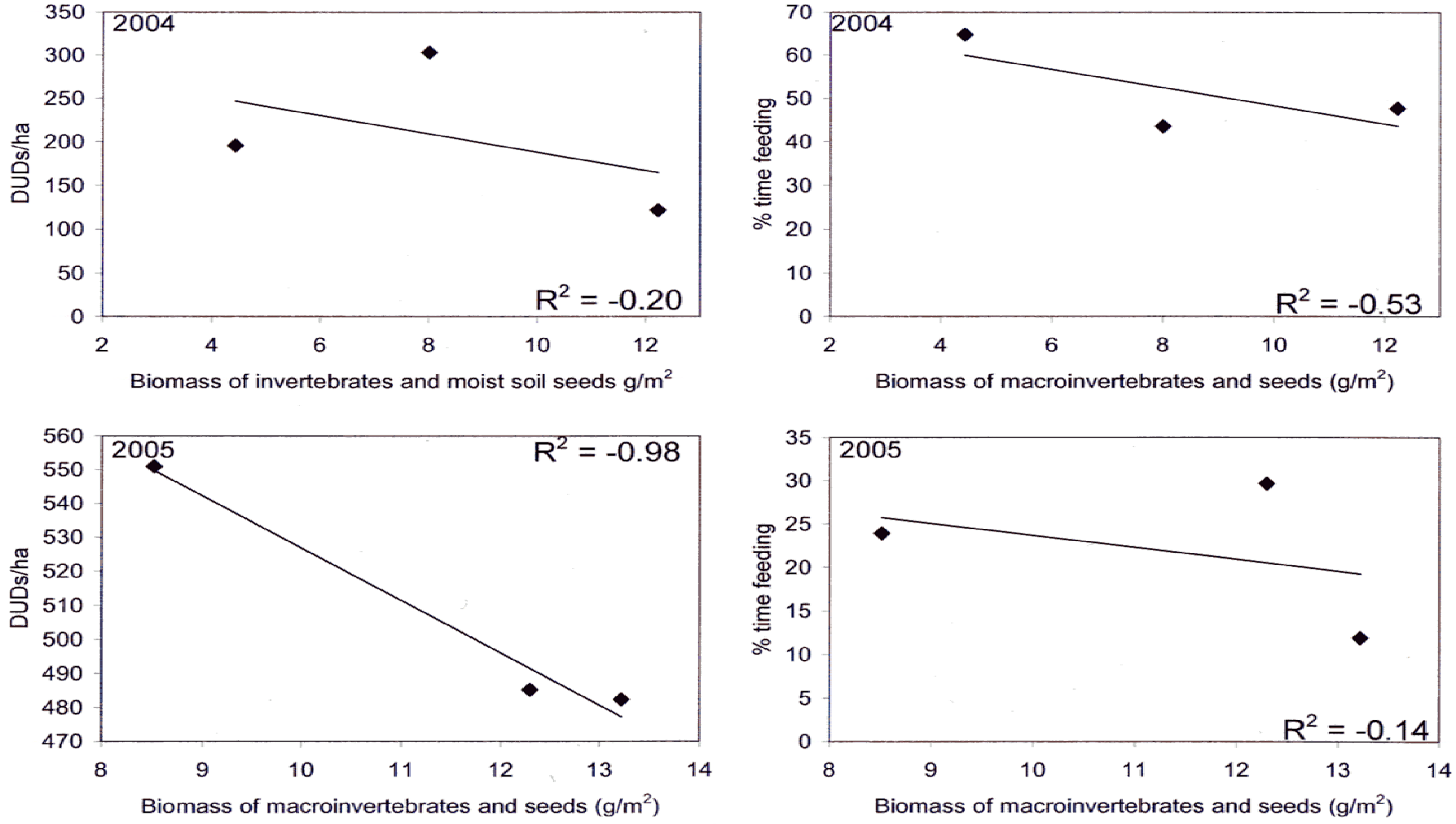

2004
DUDs/ha
350
300
250
200
150
100
50
0
2
4
6
8
10
12
R² = -0.20
Biomass of invertebrates and moist soil seeds g/m²
2004
% time feeding
70
60
50
40
30
20
10
0
2
4
6
8
10
12
R² = -0.53
Biomass of macroinvertebrates and seeds (g/m²)
2005
R² = -0.98
DUDs/ha
560
550
540
530
520
510
500
490
480
470
8
9
10
11
12
13
14
Biomass of macroinvertebrates and seeds (g/m²)
2005
% time feeding
35
30
25
20
15
10
5
0
8
9
10
11
12
13
14
R² = -0.14
Biomass of macroinvertebrates and seeds (g/m²)

Figure 3

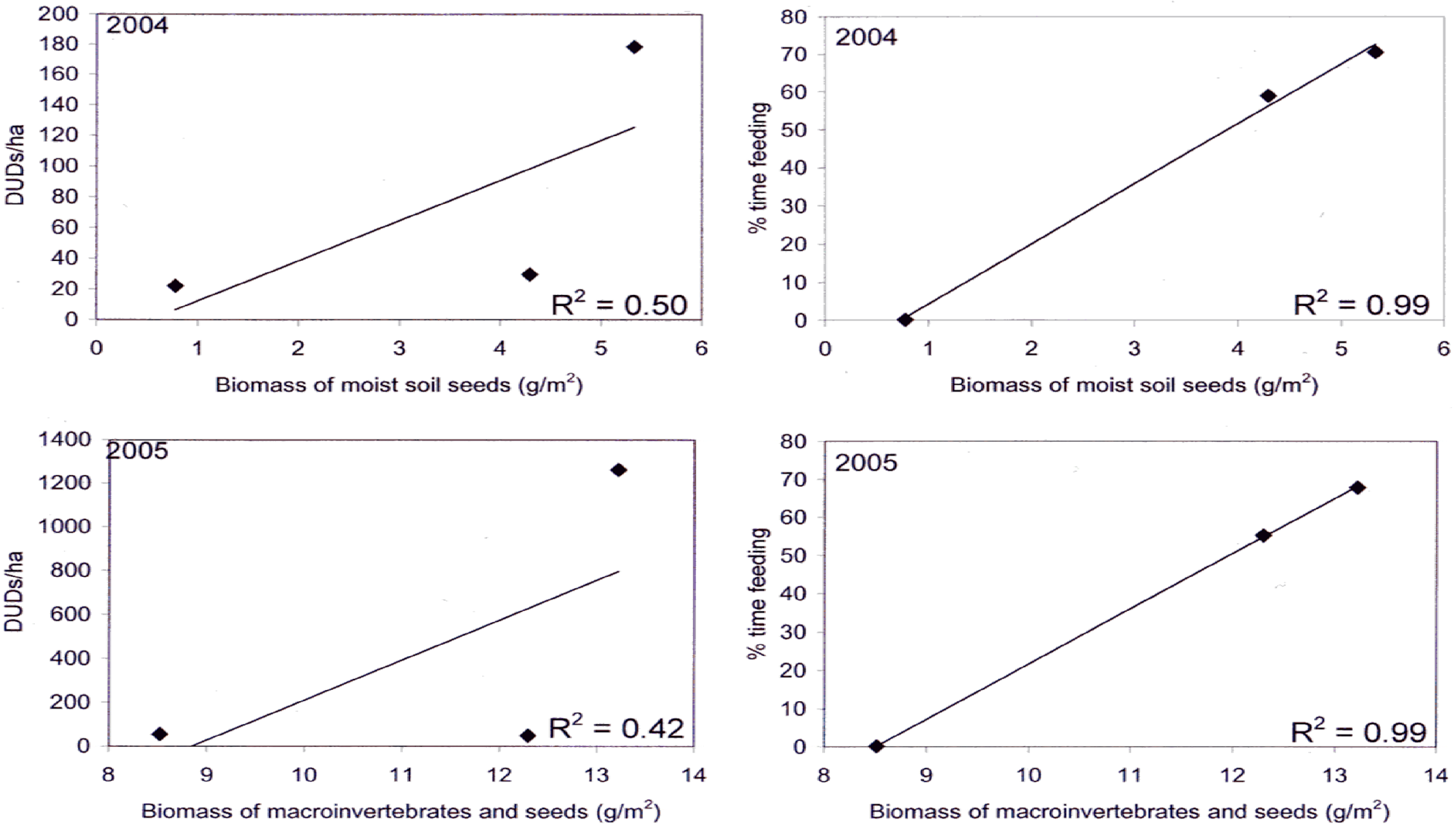

2004
DUDs/ha
200
180
160
140
120
100
80
60
40
20
0
R² = 0.50
0
1
2
3
4
5
6
Biomass of moist soil seeds (g/m²)
2004
% time feeding
80
70
60
50
40
30
20
10
0
R² = 0.99
0
1
2
3
4
5
6
Biomass of moist soil seeds (g/m²)
2005
DUDs/ha
1400
1200
1000
800
600
400
200
0
R² = 0.42
8
9
10
11
12
13
14
Biomass of macroinvertebrates and seeds (g/m²)
2005
% time feeding
80
70
60
50
40
30
20
10
0
R² = 0.99
8
9
10
11
12
13
14
Biomass of macroinvertebrates and seeds (g/m²)

Figure 4.

MALLARD AND LESSER SCAUP DIET AND FOOD SELECTION DURING SPRING MIGRATION ON SWAN LAKE, ILLINOIS

INTRODUCTION

Food Selection

Nutritional needs and energetic demands of waterfowl vary over the course of the 5 periods of the annual cycle (i.e. breeding, brood rearing, fall migration, over-winter, and spring migration). Waterfowl require sufficient protein and lipid reserves during the breeding season to support egg production and incubation (Krapu 1981, Krapu and Reinecke 1992). These reserves are acquired on wintering areas, migration areas or on the breeding grounds prior to reproduction, depending on the species of waterfowl (Ankney and MacInnes 1978, Krapu 1981, Hohman et al. 1988, Alisauskas and Ankney 1992). High protein requirements continue into brood rearing to rebuild reserves lost during incubation and provide protein for feather replacement during molt (Hohman et al. 1992). During fall migration, waterfowl require high carbohydrate diets to provide energy for migration and thermoregulation (Gruenhagen and Fredrickson 1990). High carbohydrate diets continue into the over-winter period, which allows for maintenance of body condition through extreme weather events (Wright 1959, Wills 1972, Allen 1980, Jorde et al. 1983, Baldassarre and Bolen 1984, Heitmeyer 1985, Delnicki and Reinecke 1986, Gruenhagen and Fredrickson 1990, Combs and Fredrickson 1996).

Near the end of winter or early in spring migration some waterfowl switch to diets higher in protein (Taylor 1978, Pederson and Pederson 1983, Heitmeyer 1985, Gruenhagen 1987, Lovvorn 1987, Miller 1987, Gammonley and Heitmeyer 1990, Afton et al. 1991, Manley et al. 1992, Thorn and Zwank 1993, McKnight and Hepp 1998). The reasons and exact timing of this transition remain unclear. One explanation is that by the end of winter foods high in carbohydrates become depleted, thus ducks must switch to a diet containing more invertebrates to maintain a level of energy acquisition adequate for maintenance and future reproduction (Lovvorn 1987). An alternative explanation is ducks switch to a diet high in protein because waterfowl require large amounts of protein for migration and breeding (Krapu 1981, Afton 1984). For example, reserves acquired by arctic nesting geese on wintering grounds or during spring migration are used extensively in clutch formation (Barry 1967, Raveling and Lumsden 1977, Ankney and MacInnes 1978, Raveling 1979, Bromley and Jarvis 1992). Geese arriving on breeding grounds with greater nutrient reserves lay larger clutches and have a higher probability of successfully hatching a nest (Ankney and MacInnes 1978). Nutrient reserves acquired during winter or spring migration can affect reproductive success of some duck species as well (Afton and Ankney 1991, Alisauskas and Ankney 1992, Dubovsky and Kaminski 1994). Females that do not acquire necessary reserves prior to arrival must spend longer periods building reserves before breeding in order to reach the nutrient reserve threshold necessary for clutch production (Reynolds 1972, Laurila and Hario 1988, Ankney and Alisauskas 1991, Esler et al. 2001). Further, Dubovsky and Kaminski (1994) found that winter diet restriction delayed nesting in mallards by 1 to 3 weeks. Early nesting and hatching dates have been linked to increased reproductive success (Dow and Fredga 1984, Barzen 1989, Gauthier 1989, Hepp et. al. 1989, Rohwer 1992). Therefore, females arriving on breeding grounds with sufficient nutrient reserves to begin reproduction immediately should be more successful.

Although understanding food type selection during a critical period of the annual cycle such as spring migration appears to be of fundamental importance for waterfowl management, surprisingly few studies have simultaneously examined diet and food availability to determine food type selection, thus nutritional requirements, during spring (Pederson and Pederson 1983, Miller 1987, Manley et. al. 1992). Furthermore, no studies have determined food selection in mid-migration areas of the Mississippi Flyway during spring. Studies that evaluated diet or food selection during spring have returned mixed results; some reporting the importance of high carbohydrate diets (LaGrange 1985, Heitmeyer 1985), while others found foods high in protein were more important (Miller 1987, Afton et al. 1991, Manley et al. 1992, Strand 2005, Anteau and Afton 2006, Badzinski and Petrie 2006).

One assumption explaining diet shift is that both dabbling and diving ducks are able to select between a diet high in protein versus a diet high in carbohydrates. It has been demonstrated with geese that individual foods can be selected (Sedinger and Raveling 1984). It is unclear, however, as to whether ducks can actually select specific foods while foraging underwater, or if most food items are consumed as they are encountered. Ball (1990) determined that canvasbacks (*Aythya valisineria*) foraging underwater in experimental conditions did indeed select individual food items, not just food patches. Furthermore, studies have documented ducks eating certain foods in greater or less amounts than they were available, indicating selection (Miller 1987, Manley et. al. 1992, Anderson et al. 2000). It is unclear, however, whether ducks are selecting food patches with abundant preferred foods then consuming items as they are encountered within that patch, selecting specific food types within a patch, or using both strategies.

STUDY OBJECTIVES

The goal of this study was to determine if common waterfowl habitat management practices provide adequate food resources for spring migrating mallards and lesser scaup on mid-migration habitat in the Mississippi Flyway. Therefore, my specific objectives were to determine if mallards and scaup can select specific foods from those available in the environment, at what scale the selection might take place (patches or within patches) and what types of diet are being selected, diets high in protein or carbohydrates at mid migration stopover sites during spring.

Study Species

We chose mallard ducks and lesser scaup as study organisms for my research. As the most abundant and widely distributed waterfowl species in North America (Bellrose 1980), mallards are well studied, and used extensively to make management decisions (Johnson et al. 1993, Nichols et al. 1995, Johnson et al. 2002). Because of their abundance and importance as a game animal, mallards are a species of concern among many interest groups. Despite being well studied, little is known about mallard food selection during spring migration. Three studies have examined diet of mallards during spring migration, but only 1 of those studies (Pederson and Pederson 1983), conducted at Klamath Lake National Wildlife Refuge on the border of California and Oregon, estimated availability, allowing for an assessment of food selection (Pederson and Pederson 1983, Heitmeyer 1985, LaGrange 1985). Clearly, further research is necessary to elucidate food selection by spring migrating mallards. Examining mallard food selection during spring migration can provide researchers and managers with important insights regarding habitat use and management, and help identify areas along migration routes where nutrient reserves are acquired.

Unlike mallards, lesser scaup (hereafter scaup) abundance has been steadily declining since the late 1970’s (Austin et al. 1998, Afton and Anderson 2001, Wilkins et al. 2005), despite an increase in nearly all other waterfowl species during the mid 1990's (Wilkins et al. 2005). As the most abundant diving duck species in North America (Bellrose 1980, Austin et al. 1998), scaup declines have raised concern among biologists and sportsmen alike. Several hypotheses have been developed to explain the cause of these declines (Austin et al. 2000). Some relate to habitat conditions on the breeding grounds (see Austin et al. 2000), possible over-harvest (Allen et al. 1999), and habitat changes along migration routes (Afton and Anderson 2001, Anteau and Afton 2004, Kenyon et al. 2005, Anteau 2006, Anteau and Afton 2006, Badzinski and Petrie 2006). Two distinct theories have emerged relating to habitat changes on migration areas. One concerns the acquisition of contaminants from food sources leading to decreased reproductive output (Custer et al. 2003, Fox et al. 2005, Anteau et al. 2007). The second, the spring condition hypothesis, suggests insufficient food resources along spring migration routes contribute to decreased female body condition upon reaching the breeding grounds (Afton and Anderson 2001, Anteau 2006). Reduced body condition may lead to decreased reproductive output and survival (Ankney and MacInnes 1978, Alisauskas and Ankney 1992, Afton and Anderson 2001, Barboza and Jorde 2002, Anteau and Afton 2004), since nutrient reserves acquired during migration may be used for breeding (Afton 1984, Afton and Ankney 1991, Esler et al. 2001). This is of particular concern in the Mississippi Flyway, since approximately 40% of the scaup population winters in and migrates through this region each year (Afton and Anderson 2001).

Several researchers have reported that scaup feed primarily on aquatic invertebrates (Gammonley and Heitmeyer 1990, Afton et al. 1991, Strand 2005); though the proportion of animal matter in the diet varied by study (Gammonley and Heitmeyer 1990, Afton et al. 1991, Strand 2005, Anteau and Afton 2006, Badzinski and Petrie 2006). Food availability data were not collected in these studies, therefore, it is unknown if scaup actively selected specific food types, or consumed foods in proportion to their availability during migration. One study evaluating scaup food selection on their breeding grounds reported that scaup consumed a higher proportion of plant seeds than were available (Afton and Hier 1991), suggesting scaup have the ability to select specific foods.

Swan Lake has recently changed management regimes to focus on moist-soil plant production (Chapt.1). Given that scaup feed primarily on animal material in spring (Gammonley and Heitmeyer 1990, Afton et al. 1991, Strand 2005), it is important to evaluate scaup diets as well as foods available at Swan Lake since it is unknown if moist-soil management provides preferred foods for scaup during spring. Further, Austin et al. (2000) recommend obtaining more information about lesser scaup during spring migration through studies of nutrient reserve dynamics, habitat use, and food habits, as well as evaluating food availability on migration staging areas.

METHODS

Waterfowl Collection

We collected feeding female mallards and lesser scaup with a shotgun to determine diet during spring migration at Swan Lake, Illinois (see Chapt. 1). Duck collections began shortly after ice-out (early February) and concluded when target species were no longer abundant on Swan Lake (mid-April). Individuals were collected in proportion to habitat use, determined during abundance surveys conducted prior to collection days (Chapt. 1). We categorized the different vegetation associations frequented by waterfowl in and around Swan Lake as habitats, including: moist-soil, flooded forest, open water, and near shore. We attempted to collect individuals that had been observed feeding for at least 15 minutes to ensure sufficient food for analysis, and that the duck had not been feeding in another location. Immediately after collection, we injected 10% buffered formalin into the upper digestive tract of collected waterfowl to prevent post-mortem digestion (Dillery 1965, Swanson and Bartonek 1970), and recorded the location where the bird had been feeding with a GPS unit so the site could be revisited for food availability sampling. Ducks were then numbered, tagged, placed in plastic bags, and refrigerated until they could be transported to the lab for processing.

We divided collected ducks into early, mid, and late migrant groups. Early migrants were those collected from the beginning of the sampling period (5 March 2004, and 11 February 2005) through approximately the first week of March (6 March 2004, 5 March 2005). Mid migrants were collected from approximately the first week of March (11 March 2004, 10 March 2005) through the end of March (21 March 2004, 31 March

2005), and late migrants were all birds collected in April (1 - 23 April 2004, 1 -8 April 2005).

Middle Swan Lake was primarily managed to produce moist-soil plants throughout the study period, with slow late-season drawdowns during summer and slow re-flooding in fall. Water levels varied throughout the study, and were not always characteristic of moist-soil management. We categorized the majority of ducks collected on this unit as being collected in moist-soil habitat, with the exception of scaup in 2004 collected from deep, unvegetated portions of the unit, which we classified as open water.

Food Availability Sampling

Food availability samples were taken at waterfowl collection sites within 1 to 4 days from the time of duck collection by collaborator and fellow SIU-C graduate student Darin Grulkowski. Two nektonic sweep net samples and 2 benthic samples from either a core (10 cm diameter and depth) or a petite ponar dredge (361 cm^2) when in deep water, were taken at each collection site. Samples were rinsed through a 500 µm sieve before being stored in plastic bags, and preserved with formalin for transportation to the laboratory for further analysis. Stratified random samples were also taken at points located in all habitats throughout Swan Lake to quantify macroinvertebrate and seed abundance. Sites were visited 4 times during spring to account for temporal variation as the season progressed.

Laboratory Analysis

We transported collected ducks and food availability samples to the Cooperative Wildlife Research Laboratory Annex at SIU-Carbondale. We removed, rinsed and sorted contents of the esophagus and proventriculus to determine diet. We examined all contents under a dissecting microscope to ensure no small food items were missed. We identified seeds, and when possible plant material, to genus, and invertebrates to family. Availability samples were processed similarly, and rinsed through and sorted from 1 inch, 10 mm, 1 mm, and 500 µm sieves and food items were dried to constant mass (± 0.001g) at 60° C for ≥ 48 hours.

Statistical Analysis

We summarized diet and food availability data by percent occurrence and aggregate percent mass (Swanson et al. 1974). Diet data were summarized separately for mallards and scaup, years, and by early, middle, and late migration periods to determine if diet or availability changed with period of migration. To better describe which component of the wetland provides resources for ducks, we compared food availability data collected from nektonic and benthic habitats to diet. We combined foods into general categories as moist-soil plant seeds (high carbohydrate foods) and invertebrates (high protein) to allow for an easy comparison of which food types were more important in waterfowl diets. We tested for differences between diet and food availability using Chi-square tests, and we considered differences statistically significant at $P < 0.05$.

We compared food availability from random locations sampled every 2 weeks to sites where feeding ducks were collected to determine if waterfowl selected specific feeding sites based on food availability. We summarized seed and invertebrate foods from all random locations from a given sampling period, and compared these to samples taken at duck collection sites during the same period. We used Chi-square tests in SAS (SAS Institute 1999) to determine differences between food availability at random and collected duck sites and to determine selection of feeding sites based on foods available.

RESULTS

Mallards and scaup collected during spring migration 2004 and 2005 contained, on average, 0.83 (± S.E. 0.25g, mallard) and 0.24 (± S.E. 0.09g, scaup) grams of dried food, respectively. Moist-soil seeds were primary foods for both species, comprising 78.9% (mallard) and 58.7% (scaup) aggregate mass for years combined.

Mallard Food Selection

Fifty-seven mallards were collected during spring migration 2004 (n = 17) and 2005 (n = 40). Of these, 52 contained sufficient food for analysis, 15 from 2004, and 37 from 2005. Fewer mallards were collected in 2004 because high water on Middle Swan Lake forced mallards to feed in flooded forest habitats where they were difficult to approach within shotgun range. In 2004, 11 mallards were collected from flooded forest habitats and 4 from moist-soil habitats. Mallards in 2005 were collected from moist-soil (n = 25), flooded forest (n = 8), and near shore habitats (n = 4).

Years. Mallard diets in springs 2004 and 2005 were significantly different (χ^2_1 = 4.68, P = 0.03), yet both were primarily composed of moist-soil seeds, 87.5% (±2.0%) and 73.5% (±6.1%) aggregate mass, respectively **(Figure 2)**. Invertebrates made up 12.5% (±2.0%, 2004) and 23.9% (±6.1%, 2005) aggregate mass of mallard diets. Important foods by aggregate percent mass included rice cut-grass (*Leersia oryzoides*), smartweed (*Polygonum spp*.), root parts, aquatic sow bugs (Isopoda), and millet (*Echinochloa spp*.) in 2004 (Table 2.1). Millet, smartweed, rice cut-grass, snails (Gastropoda), and tubers were the most important foods by aggregate percent mass in 2005 (Table 2.2). Samples taken at collection sites revealed that 46.6% (±6.9%) aggregate mass of available foods were high carbohydrate seeds and 53.4% (±6.9%) were

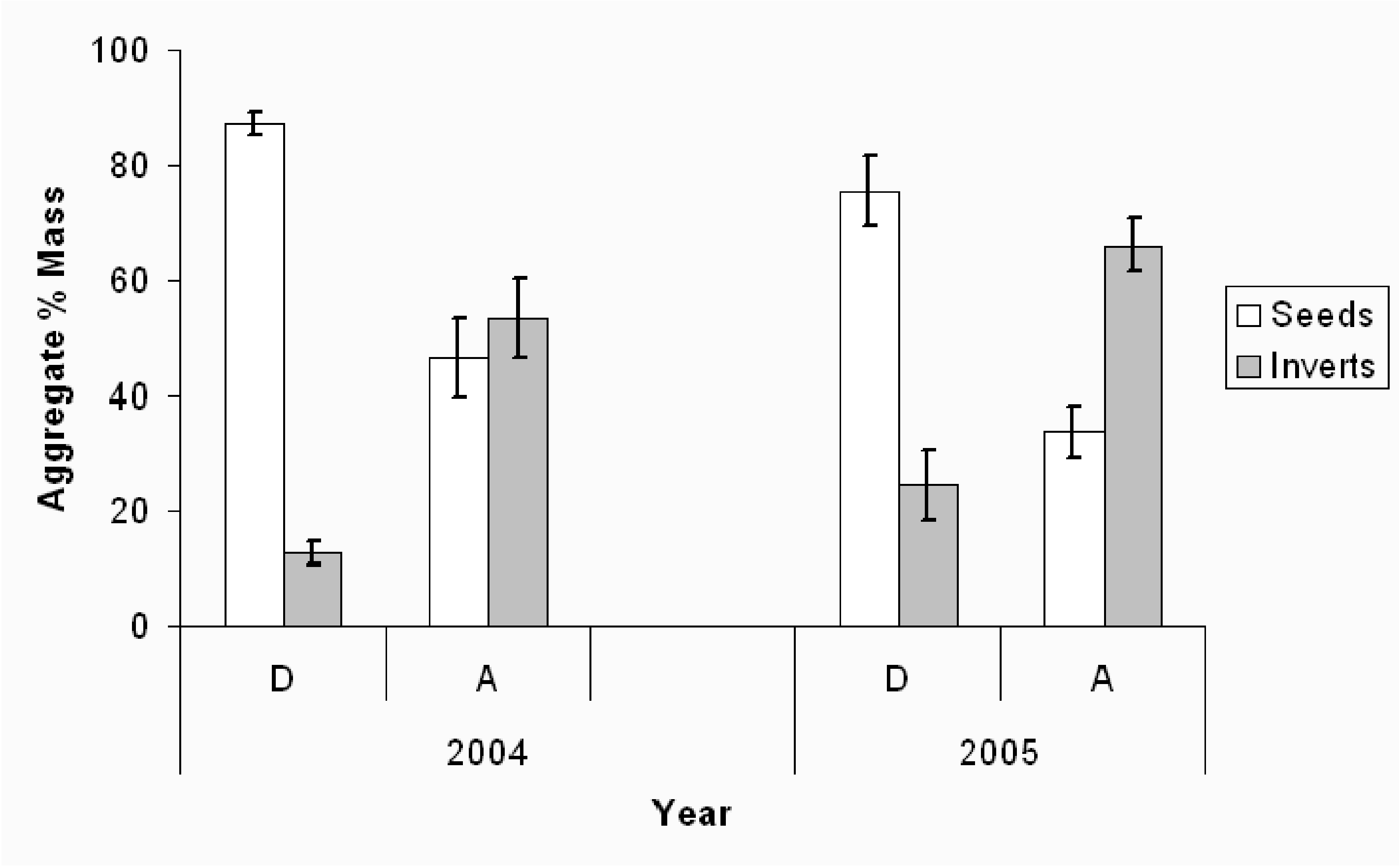


Figure 2.1. Diet (D) and food available (A) at mallard collection sites, with standard errors, during springs 2004 and 2005 at Swan Lake, IL.

Table 2.1. Foods consumed by mallards (n = 15) during spring migration 2004 on Swan Lake, Illinois. Foods making up less than 0.1% aggregate mass of diet are listed as trace (tr.).

Food Item	Aggregate %	% Occurrence
Animal Material (High Protein)	12.5	67
Isopoda (aquatic sow bugs)	7.3	40
Culicidae (mosquito)	3.6	13
Unknown Invertebrate Material	0.6	7
Other Diptera Larvae	0.3	7
Gastropoda (snails)	0.3	20
Amphipoda (scuds)	0.3	33
Hydrophilidae (beetle)	0.1	7
Curculionidae (beetle)	tr.	7
Oligochaeta (worms)	tr.	7
Sphaeridae (fingernail clams)	tr.	7
Plant Material (High Carbohydrate)	87.5	100
Leersia oryzoides (rice cut-grass)	31.3	67
Polygonum spp.(smartweed)	22.9	60
Root Parts	19.0	20
Echinochloa spp. (millet)	6.8	27
Unknown Seeds	3.0	47
Bidens spp. (beggars ticks)	2.9	67
Amaranthus spp.(pigweed)	1.1	27
Polygonum spp. (tearthumb)	0.5	7
Cuscuta spp. (dodder)	tr.	7

Table 2.2. Foods consumed by mallards (n = 37) during spring migration 2005 on Swan Lake, Illinois. Food items making up less than 0.1% aggregate mass of diet are listed as trace (tr.).

Food Item	Aggregate %	% Occurrence
Animal Material (High Protein)	24.5	62
Gastropoda (snails)	8.4	38
Isopoda (aquatic sow bugs)	6.1	35
Sphaeridae (fingernail clams)	5.1	11
Chironomidae (midges)	3.1	19
Hirudinea (leeches)	0.4	14
Odonata (Coenagionidae/Aeshnidae)	0.3	8
Trichoptera (caddisflies)	0.3	5
Amphipoda (scuds)	0.2	16
Corixidae (water boatmen)	0.2	19
Oligochaeta (worms)	0.1	11
Coleoptera (Dytiscidae/Hydrophilidae beetles)	0.1	14
Platyhelminthes (flatworms)	0.1	3
Belostomatidae (giant water bugs)	0.1	5
Unknown Invertebrate Material	tr.	5
Cladocera (water fleas)	tr.	5
Culicidae (mosquito)	tr.	5
Nematoda (roundworms)	tr.	5
Diptera - Tabanidae (house flies)	tr.	3
Ephemerpotera (mayflies)	tr.	3
Acariformes (aquatic mites)	tr.	5
Ostracoda (seed shrimp)	tr.	5
Hymenoptera - Formicidae (ants)	tr.	3
Copepoda	tr.	3
Collembola (springtails)	tr.	3
Unknown Invertebrates	tr.	8
Plant Material (High Carbohydrate)	75.5	100
Echinochloa spp. (millet)	22.6	57
Polygonum spp. (smartweed)	13.6	68
Leersia oryzoides (rice cut-grass)	11.4	43
Tubers	7.2	14

Table 2.2 continued.

Food Item	Aggregate %	% Occurrence
Unknown Seeds	6.1	41
Cuscuta spp. (dodder)	4.0	19
Cephalanthus occidentalis (buttonbush)	3.8	22
Cyperus spp. (nut sedges)	3.6	49
Bidens spp. (beggars ticks)	2.2	11
Polygonum spp. (tearthumb)	1.0	8
Potamogeton spp. (pondweed)	tr.	3
Amaranthus spp. (pigweed)	tr.	14
Sagittaria lattifolia (arrowhead)	tr.	3

high protein invertebrates in 2004. In 2005, seeds and invertebrates comprised 33.8% (±4.6%) and 66.2% (±4.6%) aggregate mass of foods available, respectively. Although there tended to be more seeds available in 2004 than 2005, the difference was not statistically significant ($\chi^2_1 = 3.5$, $P = 0.06$). In both years, mallards selected a diet higher in seeds relative to their availability [$\chi^2_1 = 36.18$, $P < 0.001$ (2004), and $\chi^2_1 = 33.9$, $P < 0.001$ (2005)].

Combined data from 2004 and 2005 revealed diet was composed of 78.9% (±5.0%) seeds and 21.1% (±5.0%) invertebrates, while food available at collection sites was composed of 37.3% (±3.9%) seeds and 62.7% (±3.9%) invertebrates. Mallard diet contained significantly more seeds than were available ($\chi^2_1 = 36.2$, $P < 0.001$).

Migration Periods. We divided collected mallards into early, mid, and late migrant groups (**Figure 2**, Tables 2.3 and 2.4). Although diet differed between 2004 and 2005 for all birds combined ($\chi^2_1 = 4.7$, $P = 0.03$), and the individual migration periods [$\chi^2_1 = 18.9$, $P < 0.001$ (early), $\chi^2_1 = 10.0$, $P = 0.002$ (mid), $\chi^2_1 = 13.7$, $P < 0.001$ (late)], the trend was similar enough to combine years. Diet [79.5% (seeds), 20.5% (invertebrates) (±8.1%) aggregate mass] and food availability [42.5 (seeds), 57.5% (invertebrates) (±8.2%) aggregate mass] were significantly different for early migrant mallards ($n = 17$) ($\chi^2_1 = 28.6$, $P < 0.001$). Mid migrant mallards ($n = 26$) ate more seeds (91.2%, ±5.0% aggregate mass) than invertebrates (8.8%, ±5.0% aggregate mass), and diet was significantly different ($\chi^2_1 = 65.3$, $P < 0.001$) than food available at collection sites [35.7 (seeds), 64.3%, ±4.9% (invertebrates) (±4.9%) aggregate mass]. Late migrants ($n = 9$) consumed more invertebrates (57.6%, ±14.6% aggregate mass) than seeds (42.4%, ±14.6% aggregate mass), but diet and availability [81.0% (invertebrates), 19.0% (seeds)

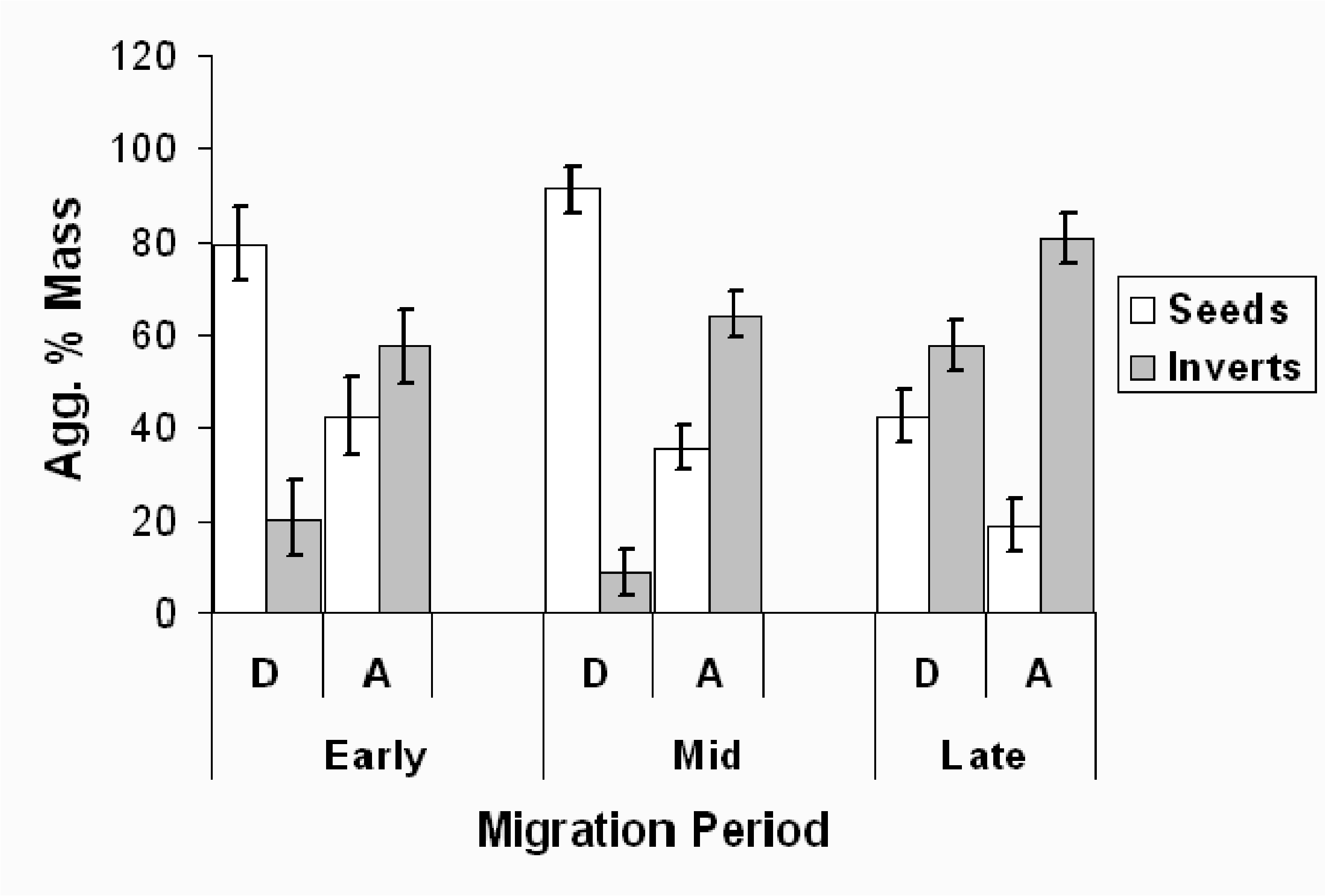


Figure 2.2 Diet (D) and food available (A) at mallard collection sites, with standard errors, during 3 migration periods at Swan Lake, IL during springs 2004 and 2005.

Table 2.3. Mallard diets by food category and migration period on Swan Lake, Illinois during spring migration 2004.

		Diet		Availability	
Migration Period	Food Category	Aggregate %	Std Err	Aggregate %	Std Err
Early Migration	High Carbohydrate Seeds	97.4	2.1	52.4	26.5
	High Protein Invertebrates	2.6	2.1	47.6	26.5
Mid Migration	High Carbohydrate Seeds	98.8	0.9	51.0	8.9
	High Protein Invertebrates	1.2	0.9	49.0	8.9
Late Migration	High Carbohydrate Seeds	57.3	24.5	33.6	6.5
	High Protein Invertebrates	42.7	24.5	66.4	6.5

Table 2.4. Mallard diets by food category and migration period on Swan Lake, Illinois during spring migration 2005.

		Diet		Availability	
Migration Period	Food Category	Aggregate %	Std Err	Aggregate %	Std Err
Early Migration	High Carbohydrate Seeds	75.6	9.5	47.5	8.2
	High Protein Invertebrates	24.4	9.5	52.5	8.2
Mid Migration	High Carbohydrate Seeds	87.9	7.2	28.9	5.2
	High Protein Invertebrates	12.1	7.2	71.1	5.2
Late Migration	High Carbohydrate Seeds	30.5	18.1	7.4	2.9
	High Protein Invertebrates	69.5	18.1	92.6	2.9

(±5.5%) aggregate mass] were still significantly different ($\chi^2_1 = 12.5$, $P < 0.001$), more seeds were eaten than were available.

Random Sites. In 2005, mallard collections began on 12 February; random sampling did not begin until 5 March. Over the course of the random sampling period, invertebrate biomass did not vary substantially (Figures 2.1, 2.2). Since random site availability did not differ substantially from 5 March to 30 April, it is unlikely that food available in February was substantially different, validating comparisons of diet, collection site availability and random site availability.

We compared food availability from mallard collection sites and random sites which were summarized by the week random samples were taken for 2004 and 2005 combined. Mallard collections coincided with 3 sampling weeks in both years (weeks 1, 3, 5, **Figure 2**). Food available at mallard collection sites [40.3% (seeds), 59.7% (invertebrates) (±5.8%) aggregate mass] contained more seeds and was significantly different from random sites [19.0% (seeds), 81.0% (invertebrates) (±2.6%) aggregate mass] during the first sampling week ($\chi^2_1 = 10.6$, $P = 0.001$). Food available at mallard collection sites from the third [33.6% (seeds), 66.4% (invertebrates) (±6.4%) aggregate mass] ($\chi^2_1 = 2.4$, $P = 0.119$) and fifth weeks [23.6% (seeds), 76.4% (invertebrates) (±6.0%) aggregate mass] ($\chi^2_1 = 0.1$, $P = 0.744$) were not significantly different than food available at random sites [24.3% (seeds), 75.6% (invertebrates) (±2.9%), 26.4% (seeds), 73.6% (invertebrates) (±2.8%)aggregate mass] for the third and fifth weeks, respectively.

Habitats. We compared diet and food availability from mallards collected in various habitats. Mallard diets primarily contained high carbohydrate foods in flooded forest (n = 19) [82.8% (seeds), 17.2% (invertebrates) (±7.4%) aggregate mass] and moist-soil (n =

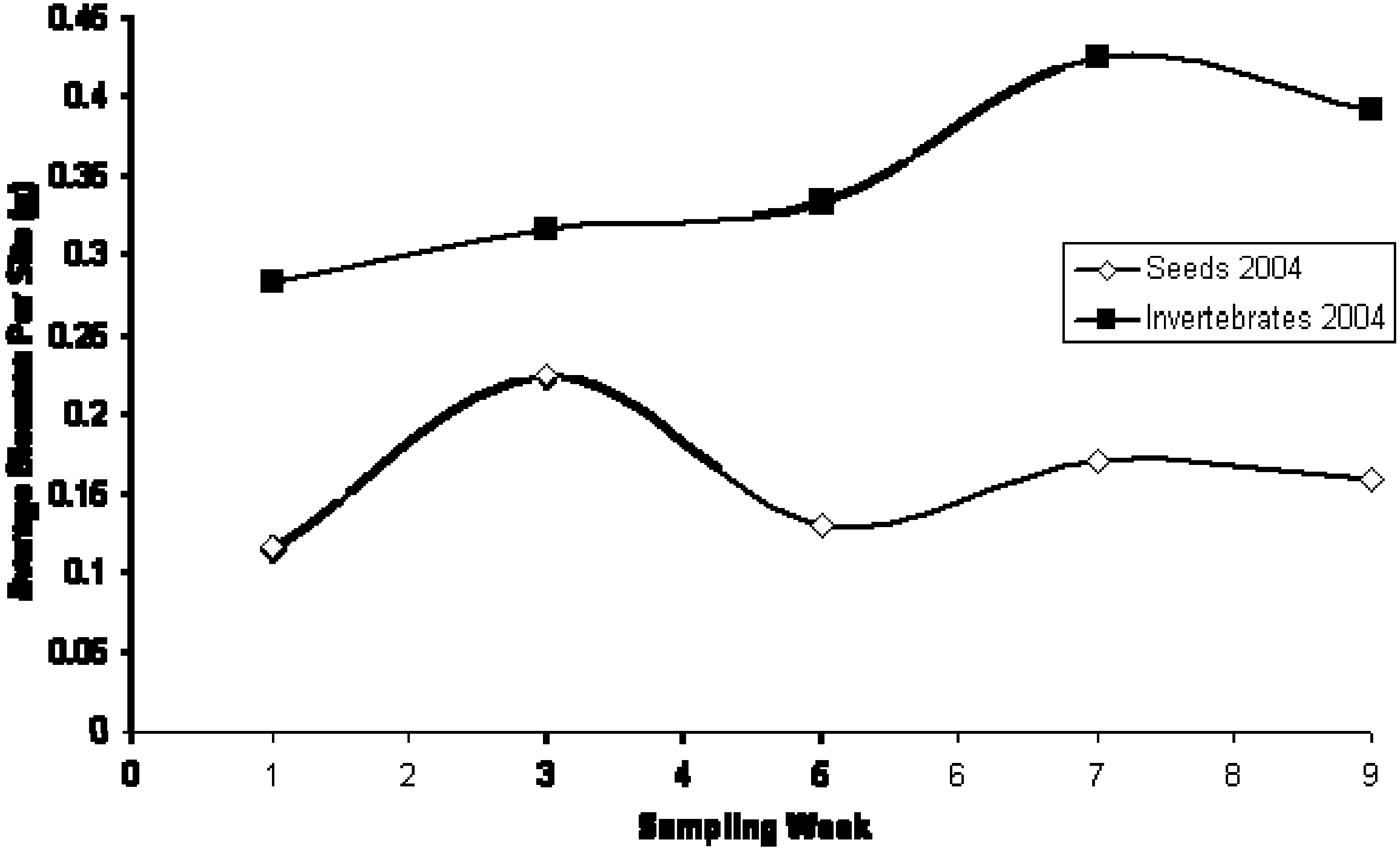


Figure 2.3. Average biomass of seeds and invertebrates available at random sampling locations on Swan Lake during spring 2004. Sampling began 2 March (week 1), and concluded 28 April (week 9).

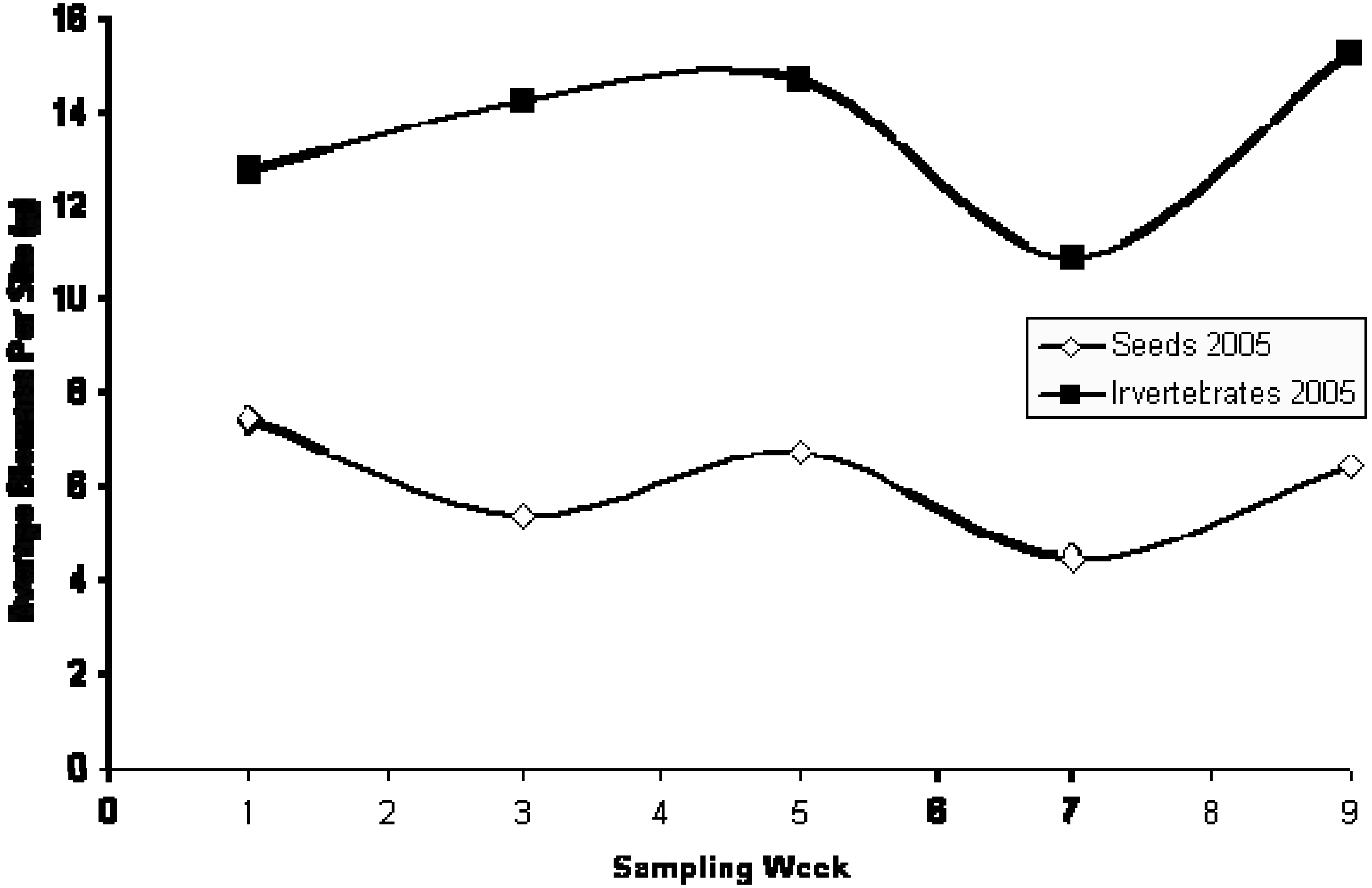


Figure 2.4. Average biomass of seeds and invertebrates available at random sampling locations on Swan Lake during spring 2005. Sampling began 5 March (week 1), and concluded 30 April (week 9).

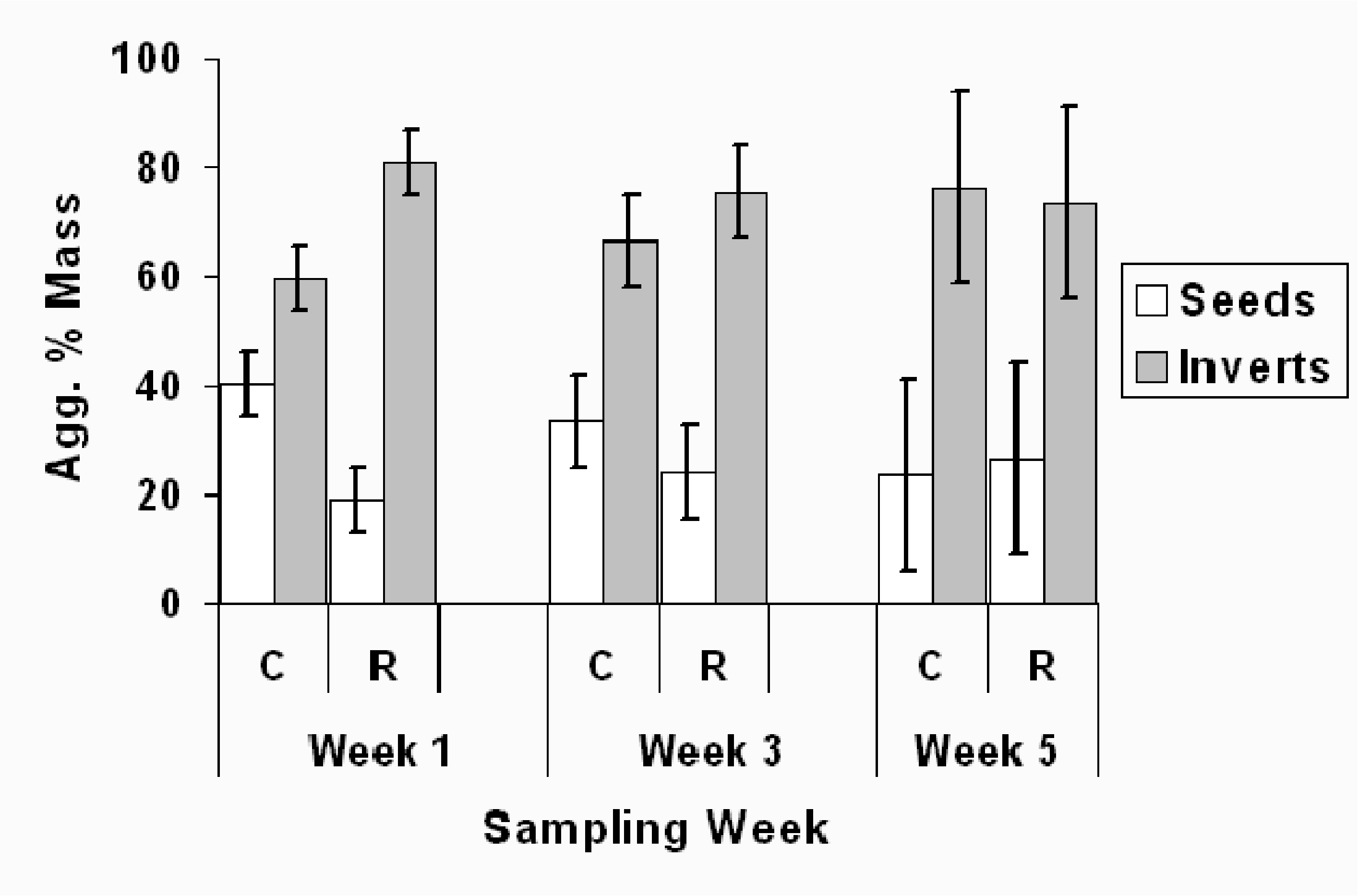


Figure 2.5. Food available at mallard collection sites (C) and random sites (R) with standard errors, during 3 sampling periods on Swan Lake, IL, springs 2004 and 2005.

29) [82.9% (seeds), 17.1% (invertebrates) (±6.1%) aggregate mass] habitats, and were not significantly different between habitat types (χ^2_1= 0.0, P = 1.0). Food availability samples collected in flooded forest [42.6% (seeds) and 57.4% (invertebrates) (±6.5%) aggregate mass] and moist-soil [37.0% (seeds) and 63.0% (invertebrates) (±5.0%) aggregate mass] were not significantly different (χ^2_1 = 0.75, P = 0.39). Mallards ate a higher percentage of high carbohydrate seeds in flooded forest (χ^2_1 = 34.32, $P < 0.001$) and moist-soil (χ^2_1 = 44.08, $P < 0.001$) habitats than were available in those habitats. Diet of mallards collected from near shore habitats (n = 4) contained a higher percentage of invertebrates [32.1% (seeds), 67.9% (invertebrates) (±23.1%) aggregate mass] and were significantly different than the diets of mallards from flooded forest (χ^2_1 = 53.22, $P < 0.001$) and moist-soil (χ^2_1 = 53.27, $P < 0.001$). Food available at near shore collection sites contained few seeds [3.2% (seeds), 96.8% (invertebrates) (±1.0%) aggregate mass]. Although mallard diets from near shore sites contained primarily invertebrates, diet and availability were still significantly different (χ^2_1 = 29.13, $P < 0.001$) because mallards ate more seeds than were available. Availability at near shore sites was also significantly different than availability at flooded forest (χ^2_1 = 45.17, $P < 0.001$) and moist-soil sites (χ^2_1 = 36.13, $P < 0.001$).

Lesser Scaup Food Selection

We collected 70 lesser scaup during spring migration 2004 (n = 30) and 2005 (n = 40), 61 contained sufficient food to include in analysis, 26 from 2004, and 35 from 2005. In 2004, 22 scaup used for analysis were collected on Middle Swan Lake, 3 on Upper Swan/Fuller Lakes, and 1 on Lower Swan Lake. In 2005, 30 scaup were collected on Middle Swan Lake, 4 on Upper Swan/Fuller Lakes, and 1 on Lower Swan Lake. Many scaup were observed on Lower Swan Lake (> 10,000 individual observations, spring 2005), but seldom fed there, precluding more collections.

Years. Lesser scaup diets from springs 2004 and 2005 differed significantly (χ^2_1 = 29.30, P < 0.001). Scaup diets were primarily composed of animal material [37.2% (seeds) and 62.8% (invertebrates) (±8.0%) aggregate mass] in 2004, whereas 2005 diets contained higher amounts of seeds [74.7% (seeds) and 25.3% (invertebrates) (±5.8%) aggregate mass]. The 5 most important foods in scaup diets in 2004 were snails (Gastropoda), smartweeds (*Polygonum spp*.), midge larvae (Chironomidae), aquatic sow bugs (Isopoda), and dodder (*Cuscuta spp*.) (Table 2.5). The most important foods in scaup diets in 2005 were rice cut-grass (*Leersia oryzoides*), smartweeds, snails, millets (*Echinochloa spp*.), and nut sedges (*Cyperus spp*.) (Table 2.6). Food availability samples taken at collection sites contained 23.8% (±5.4%, 2004) and 33.9% (±4.8%, 2005) aggregate mass seeds, 76.2% (±5.4%, 2004) and 66.1% (±4.8%, 2005) aggregate mass invertebrates, and were not significantly different between years (χ^2_1 = 2.4, P = 0.12). Chi-square tests comparing diet and availability of seeds and invertebrates for 2004 (χ^2_1 = 4.0, P = 0.046) and 2005 (χ^2_1 = 33.9, P < 0.001) indicated significant differences; more seeds were consumed than were available (**Figure 2**).

Migration Periods. Early (n = 3) and mid (n = 10) migrant scaup ate primarily high protein invertebrates [71.2% ±5.2% (early) and 74.3% ±11.4% (mid) aggregate mass] and diets were similar between the 2 earliest time periods (χ^2_1 = 0.23, P = 0.64) in 2004. Late migrant scaup (n = 13), however, ate a higher percentage of seed material [48.1% (seeds) and 51.9% (invertebrates) (±13.1%) aggregate mass] and differed significantly from the other migration periods (χ^2_1 = 12.6, P = 0.002). When separating

Table 2.5. Foods consumed by lesser scaup (n = 26) during spring migration 2004 on Swan Lake, Illinois. Food items making up less than 0.1% aggregate mass of diet are listed as trace (tr.).

Food Item	Aggregate %	% Occurrence
Animal Material (High Protein)	62.8	96
Gastropoda (snails)	23.4	42
Chironomidae (midges)	8.7	27
Isopoda (aquatic sow bugs)	8.5	38
Nematoda (roundworms)	6.6	65
Cladocera (water fleas)	4.9	42
Hirudinea (leeches)	3.3	15
Coleoptera (Dytiscidae/Hydrophilidae beetles)	2.0	8
Unknown Invertebrate Material	1.6	27
Copepoda	1.2	42
Oligochaeta (worms)	1.0	4
Ostracoda (seed shrimp)	0.7	27
Corixidae (water boatmen)	0.5	8
Acariformes (aquatic mites)	0.3	27
Coenagrionidae (damselflies)	0.1	4
Amphipoda (scuds)	0.1	12
Plant Material (High Carbohydrate)	37.2	85
Polygonum spp. (smartweed)	20.0	62
Cuscuta spp. (dodder)	7.2	12
Leersia oryzoides (rice cut-grass)	2.3	8
Echinochloa spp. (millet)	2.3	15
Cyperus spp. (nut sedge)	2.1	58
Tubers	1.4	4
Bidens spp. (beggars ticks)	1.0	8
Unknown Seeds	0.8	31
Sagittaria lattifolia (arrowhead)	0.1	8
Cephalanthus occidentalis (buttonbush)	tr.	4
Amaranthus spp. (pigweed)	tr.	8

Table 2.6. Foods consumed by lesser scaup (n = 35) during spring migration 2005 on Swan Lake, Illinois. Food items making up less than 0.1% aggregate mass of diet are listed as trace (tr.).

Food Item	Aggregate %	% Occurrence
Animal Material (High Protein)	25.3	80
Gastropoda (snails)	13.0	46
Nematoda (roundworms)	2.9	17
Sphaeridae (fingernail clams)	2.8	9
Isopoda (aquatic sow bugs)	2.5	23
Chironomidae (midges)	1.2	20
Oligochaeta (worms)	0.7	6
Unknown Invertebrate Material	0.6	6
Acariformes (aquatic mites)	0.5	20
Hirudinea (leeches)	0.4	6
Unknown Invertebrates	0.3	17
Corixidae (water boatmen)	0.2	14
Amphipoda (scuds)	0.1	14
Trichoptera (caddisflies)	tr.	6
Ostracoda (seed shrimp)	tr.	14
Coleoptera - Dytiscidae (beetles)	tr.	6
Cladocera (water fleas)	tr.	11
Copepoda	tr.	6
Culicidae (mosquito)	tr.	3
Plant Material (High Carbohydrate)	74.7	100
Leersia oryzoides (rice cut-grass)	29.6	51
Polygonum spp. (smartweed)	14.9	54
Echinochloa spp. (millet)	12.4	37
Cyperus spp. (nut sedges)	5.9	69
Cuscuta spp. (dodder)	4.6	14
Unknown Seeds	3.1	23
Cephalanthus occidentalis (buttonbush)	2.7	9
Potamogeton spp. seeds (pondweeds)	0.8	14
Bidens spp. (beggars ticks)	0.5	20
Amaranthus spp. (pigweed)	tr.	9
Sagittaria lattifolia (arrowhead)	tr.	20

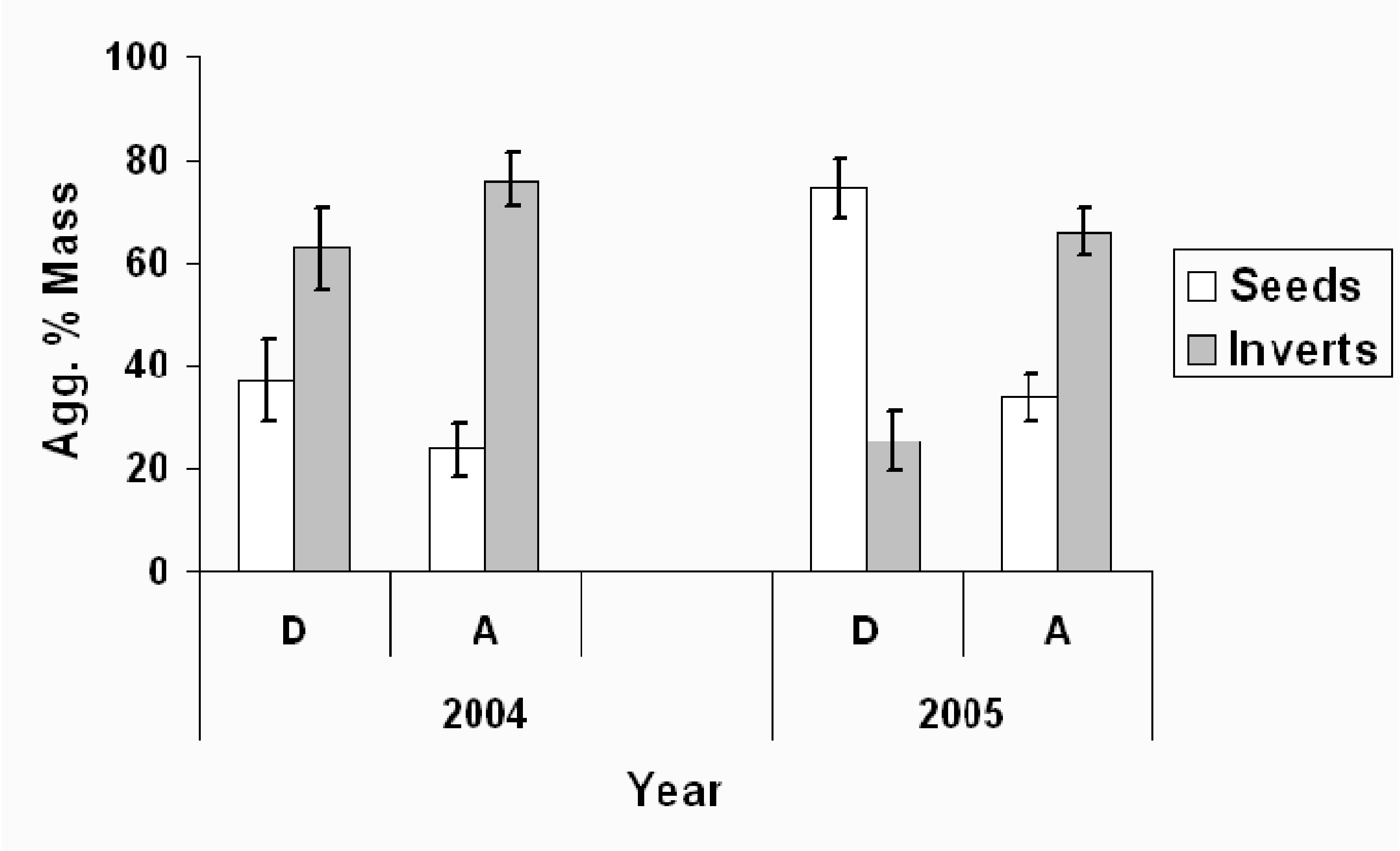


Figure 2.6. Diet (D) and food available (A) at lesser scaup collection sites, with standard errors, during springs 2004 and 2005 at Swan Lake, IL.

food availability at collection site data from 2004 into the same 3 migration periods, percentages of seed and invertebrate material available at scaup collection sites was nearly equivalent over the last 2 migration periods in 2004 (Table 2.7), but more moist-soil seeds were available at early migrant collection sites [38.4% (seeds) and 61.6% (invertebrates) (±25.0%) aggregate mass]. Chi-square tests revealed no differences between diet and availability for early (χ^2_1 = 1.82, P = 0.178) and mid (χ^2_1 = 0.44, P = 0.620) migration periods, but a significant difference did exist between diet and availability for the late migration period (χ^2_1 =14.9, P < 0.001), in which scaup ate more seeds than were available at collection sites in 2004 **(Figure 2)**.

Scaup diets were similar (χ^2_1 = 0.02, P = 0.88) between early (n = 11) [70.1% (seeds), 29.9% (invertebrates) (±11.1%)] and mid (n = 16) [70.5% (seeds), 29.5% (invertebrates) (±9.6%)] migrants (Table 2.8) in 2005. Late migrant scaup (n = 8) ate higher percentages of seeds [89.3% (seeds) and 10.7% (invertebrates) (±5.9%) aggregate mass] than early and mid migrant scaup (χ^2_1 =12.8, P = 0.002). Food available at collection sites consistently contained lower percentages of seeds [early: 22.9% (seeds), 77.1% (invertebrates) (±7.5%), mid: 38.6% (seeds), 61.4% (invertebrates) (±8.4%), late: 44.7% (seeds), 55.3% (invertebrates) (±5.6%) aggregate mass] than occurred in the diet, and differed significantly from diet during all migration periods [early: (χ^2_1 = 44.4, P < 0.001), mid: (χ^2_1 = 20.7, P < 0.001), late: (χ^2_1 = 43.8, P < 0.001)] **(Figure 2)**.

Random Sites. In 2005, scaup collections began on 18 February; random sampling did not begin until 5 March. Over the course of the random sampling period, invertebrate biomass did not vary substantially (Figure 2.2). Since random site availability did not differ substantially from 5 March to 30 April, it is unlikely that food

Table 2.7. Lesser scaup diets by food category and migration period on Swan Lake, Illinois during spring migration 2004.

Migration Period	Food Category	Diet		Availability	
		Aggregate %	Std Err	Aggregate %	Std Err
Early Migration	High Carbohydrate Seeds	28.8	5.2	38.4	24.0
	High Protein Invertebrates	71.2	5.2	61.6	24.0
Mid Migration	High Carbohydrate Seeds	25.7	11.4	22.3	7.9
	High Protein Invertebrates	74.3	11.4	77.7	7.9
Late Migration	High Carbohydrate Seeds	48.1	13.1	21.5	7.4
	High Protein Invertebrates	51.9	13.1	78.5	7.4

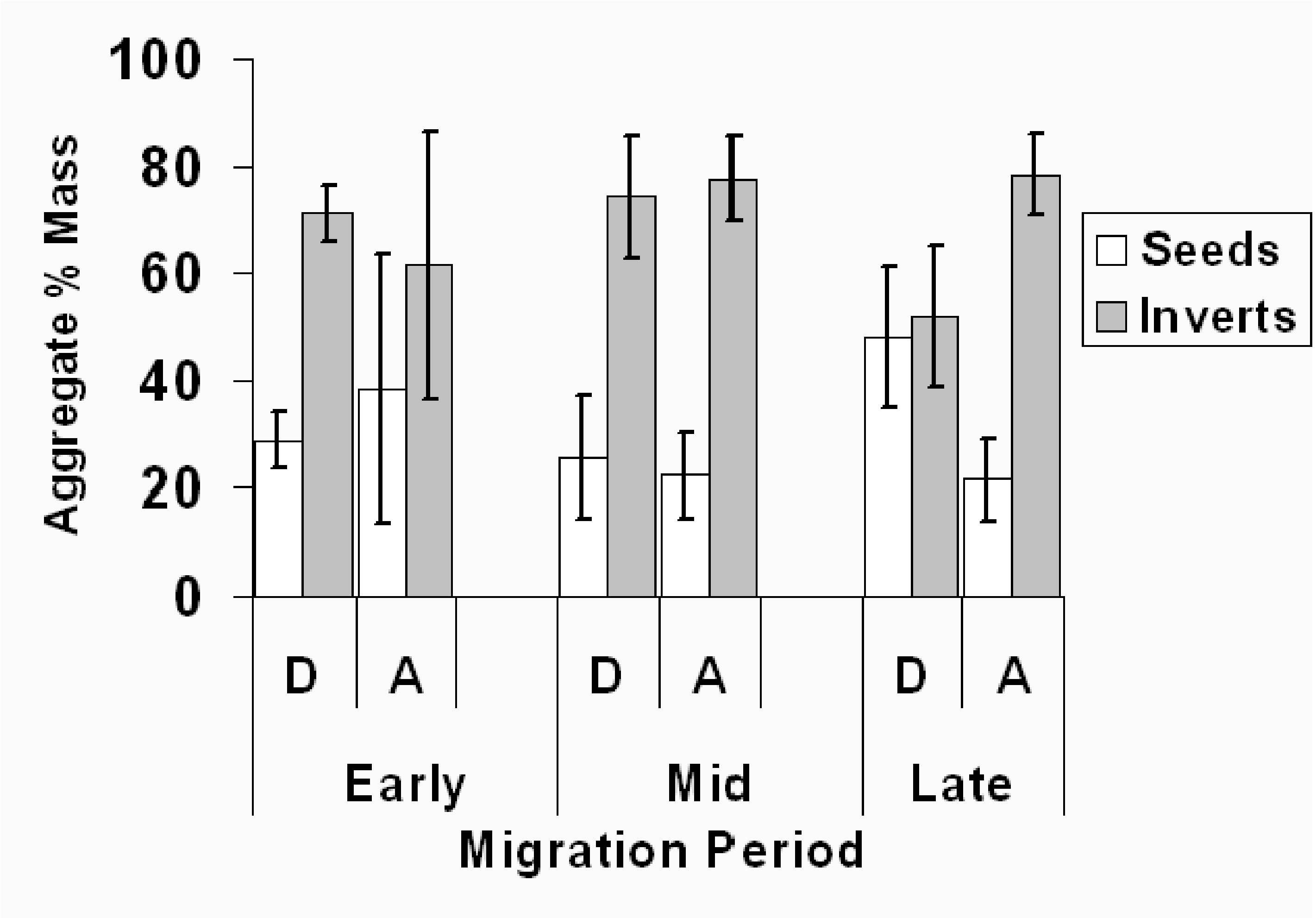


Figure 2.7. Diet (D) and food available (A) at lesser scaup collection sites, with standard errors, during 3 periods of spring migration 2004 at Swan Lake, IL.

Table 2.8. Lesser scaup diets by food category and migration period on Swan Lake, Illinois during spring migration 2005.

		Diet		Availability	
Migration Period	Food Category	Aggregate %	Std Err	Aggregate %	Std Err
Early Migration	High Carbohydrate Seeds	70.1	11.1	22.9	7.5
	High Protein Invertebrates	29.9	11.1	77.1	7.5
Mid Migration	High Carbohydrate Seeds	70.5	9.6	38.6	8.4
	High Protein Invertebrates	29.5	9.6	61.4	8.4
Late Migration	High Carbohydrate Seeds	89.3	5.9	44.7	5.6
	High Protein Invertebrates	10.7	5.9	55.3	5.6

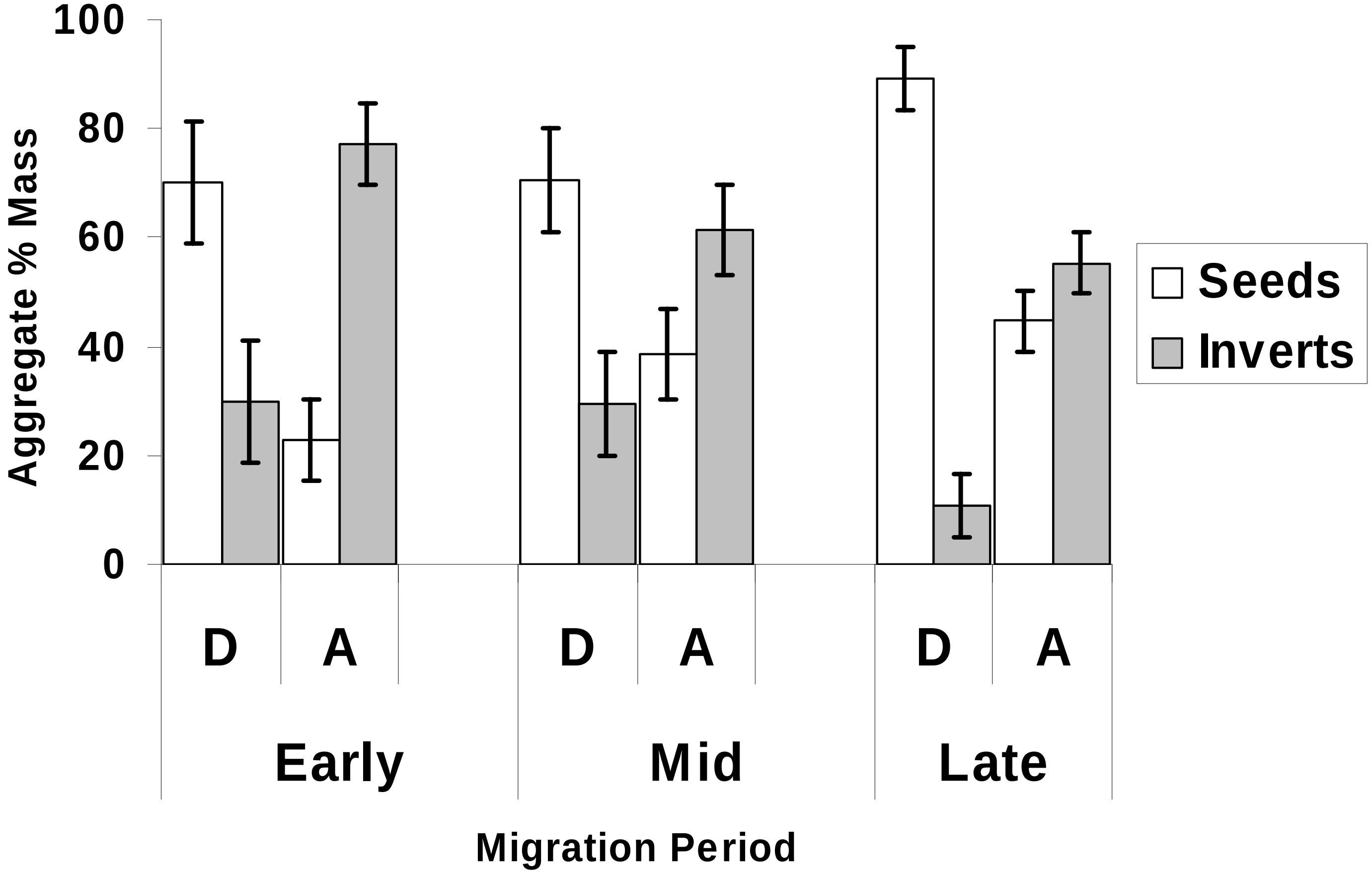


Figure 2.8. Diet (D) and food available (A) at lesser scaup collection sites, with standard errors, during 3 periods of spring migration 2005 at Swan Lake, IL.

available in February was substantially different, validating comparisons of diet, collection site availability and random site availability.

We compared food availability from scaup collection sites and random sites which were combined by the week random samples were taken (**Figure 2**). During spring 2004, scaup were collected during all 5 sampling weeks, however, only 1 scaup was collected during the first week, and was not included in the analysis. Food available at scaup collection sites did not differ significantly from random sites during week 3 ($\chi^2_1 = 0.03$, $P = 0.871$), week 5 ($\chi^2_1 = 0.10$, $P = 0.755$), or week 7 ($\chi^2_1 = 0.57$, $P = 0.450$). There was a significant difference ($\chi^2_1 = 22.0$, $P < 0.001$) between random and collection sites during week 9 in which percentage of seeds available at scaup collection sites was less than those available at random sites. Percentage of seeds at scaup collection sites declined dramatically from roughly 30% for weeks 3-7, to 6% in week 9. Seeds available at random sites remained nearly constant (26-35% aggregate mass seeds) throughout the sampling period, while invertebrates increased through spring (Figure 2.1).

In 2005, scaup were collected during 3 random sampling periods **(Figure 2.X)**. Food available at scaup collection sites [week 1: 34.2% (seeds), 65.8% (invertebrates) (±7.0%), week 3: 32.4% (seeds), 67.6% (invertebrates) (±6.7%) aggregate mass] was not different than food available at random sites [week 1: 24.9% (seeds), 75.1% (invertebrates) (±3.7%), week 3: 22.0% (seeds), 88.0% (invertebrates) (±4.2%) aggregate mass] during the first ($\chi^2_1 = 1.9$, $P = 0.163$) and third ($\chi^2_1 = 2.5$, $P = 0.111$) random sampling periods. During the fifth week, collection sites [37.6% (seeds), 62.4% (invertebrates) (±10.2%) aggregate mass] contained more moist-soil seeds and were significantly different ($\chi^2_1 = 6.1$, $P = 0.013$) than random sites [21.6% (seeds), 78.4%

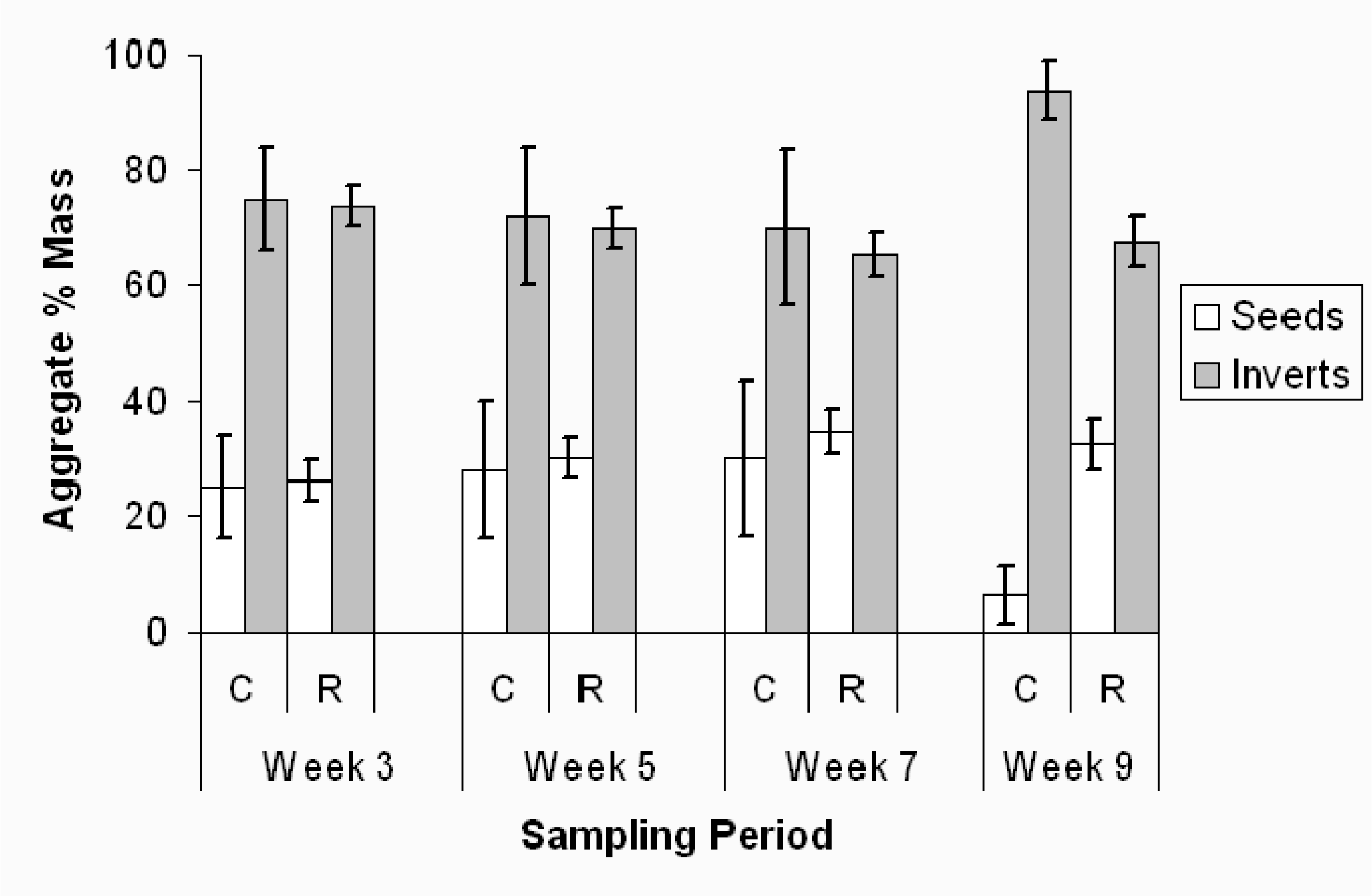


Figure 2.9. Food available at lesser scaup collection sites (C) and random sites (R) with standard errors, during 4 sampling periods on Swan Lake, IL, spring 2004.

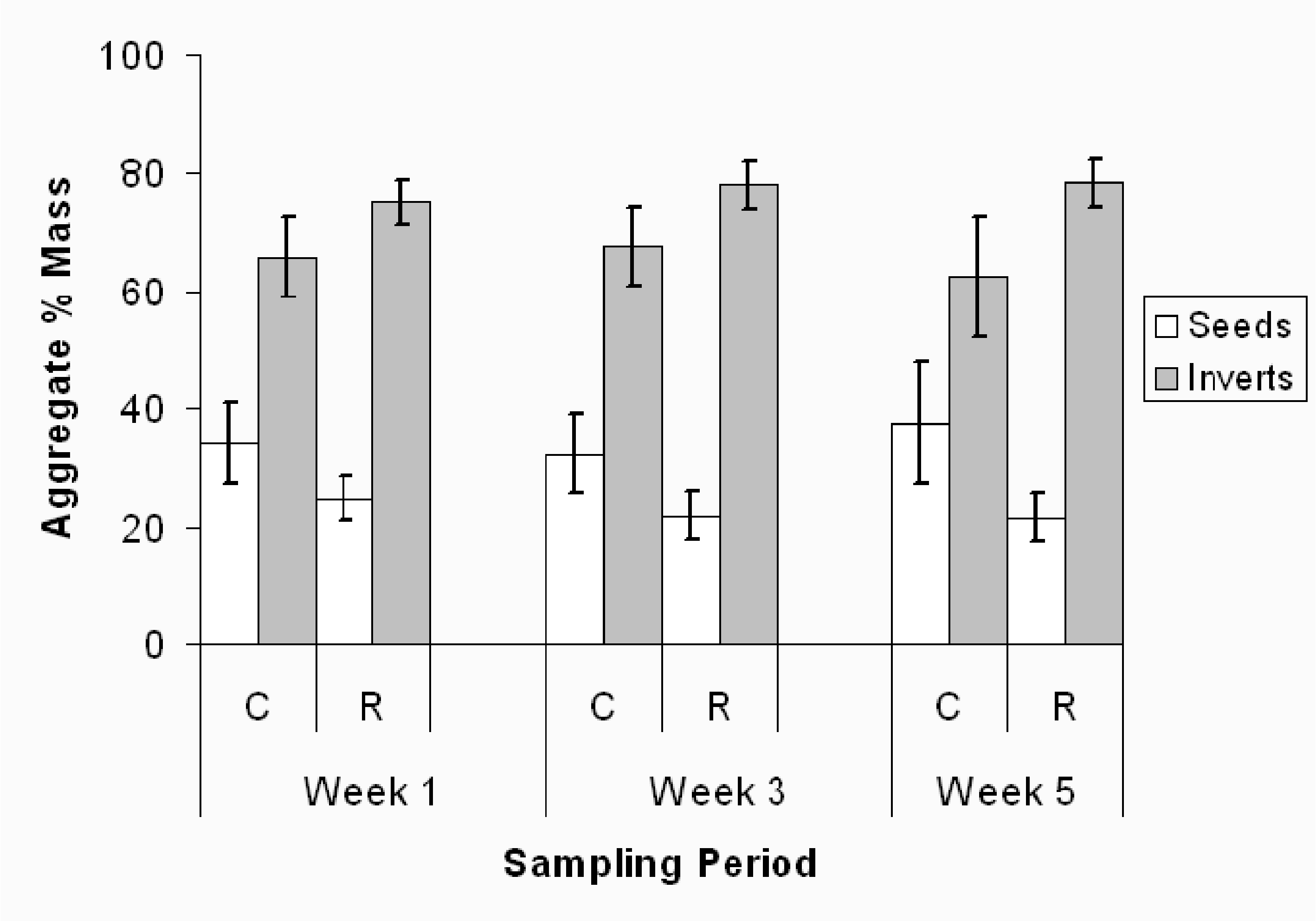


Figure 2.10. Food available at lesser scaup collection sites (C) and random sites (R) with standard errors, during 4 sampling periods on Swan Lake, IL, spring 2005.

(invertebrates) (±4.0%) aggregate mass]. Seed biomass remained nearly constant, or declined slightly, while invertebrate biomass increased slightly throughout spring at random sampling points (Figure 2.2).

Habitats. The majority of scaup were collected from Middle Swan Lake in 2004. This lake unit was managed to produce abundant moist-soil vegetation, however, during spring 2004, water levels varied considerably and may not have mimicked typical moist-soil management. Despite varying water levels, we grouped scaup collected from vegetated areas of Middle Swan Lake into the moist-soil category. Scaup were collected in open water (n = 15), and moist-soil (n = 11) habitats in 2004. Scaup collected in open water ate higher percentages of invertebrates (74.4% ±9.5% aggregate mass) than seeds (25.6 ±9.5% aggregate mass). Scaup collected in moist-soil habitats ate a higher percentage of seeds (53.0% ±12.8%) than invertebrates (46.9% ±12.8% aggregate mass). Open water and moist-soil habitats had higher percentages of invertebrates available [84.4% ±5.1% (open water) and 65.0% ±9.9% (moist-soil) aggregate mass] than seeds [15.6% ±5.1% (open water) and 35.0% ±9.9% (moist-soil) aggregate mass]. Diet and food availability were not significantly different for open water (χ^2_1 = 3.01, P = 0.08), but were significantly different for moist-soil (χ^2_1 =6.57, P = 0.01) where scaup ate more seeds than were available.

Water levels on Middle Swan Lake were more characteristic of moist-soil habitat in 2005 than in 2004; therefore, we classified all scaup collected on Middle Swan Lake as feeding in moist-soil habitat in 2005. We collected scaup from moist-soil (n = 18) and open water (n = 16) habitats in 2005. Diet from these habitats were not significantly different (χ^2_1 = 0.43, P = 0.51). Scaup collected from both habitats ate more seeds [76.8% ±8.9% (open water) and 73.4% ±8.2%

(moist-soil) aggregate mass] than invertebrates [23.2 % ±8.9% (open water) and 26.6% ±8.2% (moist-soil) aggregate mass]. Food available at collection sites contained a higher percentage of invertebrates [66.0% ±6.3% (moist-soil), 58.6% ±8.2% (open water) aggregate mass] than seeds [34.0% ±6.3% (moist-soil), 41.4% ±8.2% (open water) aggregate mass]. Scaup ate more seeds than were available and diet and food availability were significantly different for moist-soil ($\chi^2_1 = 30.6$, $P < 0.001$) and open water ($\chi^2_1 = 26.8$, $P < 0.001$) habitats. Diet differed significantly between 2004 and 2005 in open water ($\chi^2_1 = 52.07$, $P < 0.001$) and moist-soil ($\chi^2_1 = 8.58$, $P = 0.003$) habitats.

DISCUSSION

Of the 3 food selection studies of dabbling ducks that have been conducted during spring, all have found ducks eating high protein animal material at a higher proportion than was available at the collection site (Pederson and Pederson 1983, Miller 1987, Manley et al. 1992). Pederson and Pederson (1983) found that mallards and northern pintails (*Anas acuta*) feeding in the Klamath Basin of California and Oregon increased the amount of animal material in their diets from winter to spring, and ate midge larvae (Chironomidae) in higher proportions than they were available at collection sites (Pederson and Pederson 1983). A similar example is provided by Miller (1987) for pintails in the Sacramento Valley of California. Although this study focused primarily on fall and winter food selection, an increase in consumption of invertebrates, primarily midge and beetle (Coleoptera) larvae, was reported late in the February-March sample period (Miller 1987). The most recent example is provided by Manley et al. (1992), with male blue-winged teal (*Anas discors*) staging for migration in Louisiana. Teal selected a higher proportion of invertebrates than were available in their selected habitats. Animal material made up roughly 20% of available food, but comprised 69% to 80% of teal diets (Manley et al. 1992). These studies contrast with my results, which indicate that waterfowl primarily selected a high carbohydrate diet during spring migration at Swan Lake.

Mallard Food Selection

Food available at mallard collection sites was different than food available at random sites only during the first random sampling week, and was not different during the third and fifth sampling weeks. During all sampling periods, however, mallard diet differed substantially from food available at collection sites. These results indicate that mallards selected individual foods on a local scale, but other factors, such as predator risk, are important when selecting feeding sites.

Previous research evaluating mallard diet in spring has returned mixed results regarding the types of foods mallards eat during this period. LaGrange (1985) reported that mallards consumed high carbohydrate foods such as seeds and agricultural grains during spring; however, this may have reflected a bias in collection habitats. Heitmeyer (1985) noted that female mallards increased protein consumption during the pre-basic molt late in winter, then switched to a diet higher in carbohydrates to prepare for spring migration, however, no food availability data was collected in this study. Pederson and Pederson (1983) found mallards increased the percentage of high protein invertebrates in their diets from late winter into spring. This increase, however, was reported as proportional to availability (Pederson and Pederson 1983). My results indicate that mallards primarily selected natural plant foods during the migration period, and increased invertebrate consumption once reproduction began.

Evidence from mallard diets and food availability at collection sites indicate that mallards selected moist-soil plant seeds while feeding on Swan Lake during spring migration. Mallards responded positively to favorable water conditions in 2005, distributing more evenly throughout Swan Lake, as opposed to 2004 in which deep water forced mallards into a few select areas where favorable water depth occurred. Early and mid migrant mallards ate greater percentages of moist-soil plant seeds than were available at the sites they were collected from. Mallards selected, and ate, a higher percentage of seeds in all migration periods, and habitats except for late migrants, which ate a higher percentage of invertebrates than seeds, yet selection for seeds still occurred since seed availability was very low at late migrant collection sites.

Migration Periods. Late in migration, invertebrate biomass increased in food availability samples. This was caused by an increase in invertebrate biomass at mallard collection sites late in spring, and not a depletion of high carbohydrate foods, which stayed relatively constant throughout the migration period. Stratified random sampling covering all of Swan Lake revealed that invertebrate biomass increased through spring (Figures 2.1, 2.2). The increase in invertebrate biomass at mallard collection sites was much greater than the average increase in invertebrate biomass at random sites, suggesting that mallards selected sites with abundant high protein foods.

Mallards collected late in migration (April) made a shift in the amount of invertebrate material eaten from 20.5% and 8.8% aggregate mass for early and mid migrants, respectively, to 42.4% for late migrants. It is well documented that waterfowl eat primarily invertebrates that are high in protein on the breeding grounds (Krapu 1981, Krapu and Reinecke 1992). Inspection of internal anatomy of late migrant mallards revealed that at least one had initiated nesting.

Macroscopic examination of the ovary showed that the bird was well into rapid follicle growth, and contained a nearly fully formed egg in the ovi-duct. A study conducted by the Illinois Natural History survey from 1998 to 2003 found mean nest initiation date for mallards in central Illinois ranged from 22 April to 6 May, and first nest attempt was as early as 4 April (A. Yetter, INHS, pers. comm.). Based on these findings, the date of collections (1 to 7 April), physical evidence of laying, and anecdotal evidence of other local breeding mallards (isolated pairs, hens flushed from nests), we believe all mallards classified as late migrants, were resident breeding birds. These birds should be considered resident mallards and should not be considered spring migrants. Therefore, a shift to a high protein diet is expected in preparation for nesting (Swanson et. al. 1979, Krapu 1981, Krapu and Reinecke 1992), and does not reflect a switch in diet prior to arrival on the breeding grounds.

Positive selection for moist-soil seeds indicates that mallards are not using food resources on Swan Lake to build protein reserves. This also supports the theory that mallards rely heavily on exogenous sources of protein for nesting, even though endogenous reserves are used to meet lipid and energy requirements of reproduction (Krapu 1981).

Lesser Scaup Food Selection

Swan Lake is not known as a primary scaup staging area, compared to other areas in central IL (i.e. Pool 19 of the Mississippi River). Scaup abundance, however, reached 2,935 in spring 2004 and 7,175 in spring 2005 on Swan Lake. Scaup also accounted for 52,997 and 149,210 use days over the duration of spring migration in 2004 and 2005, respectively. Further, the Illinois River Valley historically hosted large numbers of migrating scaup (Anderson 1959, Mills et. al. 1966). Scaup numbers decreased dramatically following loss of a major food source, and the displaced birds presumably switched migration routes to the Mississippi River Valley (Mills et. al. 1966). Determining if preferred scaup foods are abundant in the Illinois River Valley could be important for managers and researchers concerned with issues affecting lesser scaup.

Similar to mallards, food available at scaup collection sites did not differ from food available at random sites (Figure 2.1) throughout most of spring migration 2004 or 2005. This suggests that scaup were not actively selecting foraging sites based exclusively on food resources available. Scaup diet was significantly different, however, than food available at collection sites, indicating that, similar to mallards, once scaup reached a foraging site they selected preferred foods on a local scale.

The majority of previous studies examining lesser scaup diet during spring migration stated that scaup fed primarily on invertebrates, especially in spring (Rogers and Korschgen 1966, Gammonley and Heitmeyer 1990, Afton et al. 1991, Strand 2005, Anteau 2006, Anteau and Afton 2006, Badzinski and Petrie 2006), although one study reported that scaup selected seeds on their breeding grounds (Afton and Hier 1991). Seeds, however, made up a relatively small portion of the diet by aggregate percent mass, but were virtually absent in food availability samples (Afton and Hier 1991). Gammonley and Heitmeyer (1990) reported that scaup migrating through the Klamath Basin of California and Oregon fed predominantly on animal material (77% aggregate mass) and that chironomidae larvae made up the highest percentage of all foods consumed by scaup. Afton et al. (1991) reported that diets of scaup collected in northern Minnesota contained primarily animal material (91.8% aggregate mass). Similarly, Strand (2005) reported that spring migrating scaup in South Dakota commonly consumed chironomids, amphipods, and gastropods, however, amounts of plant material and seeds were not reported. Badzinski and Petrie (2006) reported that mollusks [zebra mussels (*Dreissena polymorpha*) and snails] made up the majority (77.7% aggregate mass) of scaup diets on the lower Great Lakes during spring. None of these studies examined food availability. Anteau (2006) evaluated scaup diets and estimated the abundance of Amphipoda throughout the upper Midwest during spring. Amphipods are known to be a preferred scaup food (Rogers and Korschgen 1966, Bartonek and Hickey 1969, Swanson and Nelson 1970, Swanson and Duebbert 1989, Afton and Hier 1991, Afton et al. 1991, Lindeman and Clark 1999, Strand 2005, Anteau 2006); therefore, food selection was not specifically determined. Animal foods dominated scaup diets (91.9% aggregate mass); seeds and vegetation were common in diets (73% and 23% occurrence, respectively), but comprised a small component of the diet by mass (8.1% combined aggregate mass) (Anteau 2006).

My results show not only that scaup feed on moist-soil plant seeds during spring migration, but that seeds were selected over invertebrates at Swan Lake. Selection for moist-soil plant seeds at Swan Lake and indicates that seeds may be a more important food for migrating scaup than previously thought during spring in the Mississippi Flyway.

Migration Periods. Several researchers have suggested that waterfowl increase the amount of protein in their diets as they approach the breeding season (Taylor 1978, Pederson and Pederson 1983, Heitmeyer 1985, Gruenhagen 1987, Lovvorn 1987, Miller 1987, Gammonley and Heitmeyer 1990, Afton et al. 1991, Manley et al. 1992, Thorn and Zwank 1993, McKnight and Hepp 1998). In both 2004 and 2005, scaup on Swan Lake selected the largest percentage of seeds during the last migration period. We are uncertain why late migrant scaup on Swan Lake selected a diet high in carbohydrates over one based on high protein invertebrates even though invertebrates became more abundant later in migration.

Plant Material. Plant fragments were found in nearly all scaup diets (78.5% occurrence for all scaup collected in 2004 and 2005), but since large amounts of plant material were collected in availability samples and discarded, it was impossible to compare to food availability. Plant fragments constituted a significant percentage of the diet (26.5% aggregate mass for all scaup collected in 2004 and 2005) when included in calculations of aggregate % mass, and 4 scaup diets' contained only plant fragments. Other studies of scaup diet that included plant material in analysis found high occurrence in diets, but relatively low importance based on aggregate percent mass (Gammonley and Heitmeyer 1990, Afton et al. 1991). We are uncertain why plant material constituted such a large percentage of the diet of scaup collected on Swan Lake.

Anteau and Afton (2004) suggest investigating declines in scaup body condition at northern locations (i.e. Minnesota, USA and Manitoba, Canada) during spring migration by assessing the ability of scaup to obtain historically preferred foods on migration areas, and factors affecting the availability of those foods. My research indicates scaup are not exclusively consuming historically dominant foods, but are selecting moist-soil plant seeds during some periods of migration. It is unknown how a high carbohydrate diet during spring migration could affect body condition of birds collected on staging areas farther north, or upon reaching the breeding grounds. Parts of Swan Lake are intensively managed for moist-soil plant production. Results of studies examining invertebrate response to moist-soil management have varied, but indicate that different invertebrate communities respond differently to varying amounts of vegetation and open water (Flinn et. al. 2005, D. Grulkowski personal communication). Regardless, the habitat assemblages on Swan Lake provided foods historically consumed by scaup, however, scaup preferred moist-soil seeds, a food not considered common for scaup.

Another question arises about what foods are truly available to feeding mallards and scaup. Although benthic invertebrates were abundant in core samples taken at collection locations, it is debatable if these food items could be obtained by feeding ducks. Research conducted by the Illinois Natural History Survey on Swan Lake indicates that benthic sediments are flocculent and deep, and benthic invertebrates may be found as deep as 75 cm below the benthic surface (T. Timmerman personal communication). Although most invertebrates occurred in the top portion (0 to 6 cm) of the benthos, a high percentage also occurred deeper. If the average duck bill is approximately 4 - 6 cm long, this leaves many invertebrates out of reach of the ducks. Core samples taken during this study were 10 cm deep, potentially as much as half of the food 'available' could not be accessed by foraging ducks.

IMPLICATIONS AND FUTURE CONSIDERATIONS

Current wetland management regimes outside of breeding areas generally focus on maximizing the production of high carbohydrate food sources by managing for moist-soil plant species (Fredrickson and Taylor 1982). This management regime may provide an excellent food source during fall migration and the over winter period, as well as maximizing recreational value on public lands (Fredrickson and Taylor 1982). It was previously unknown, however, if these wetlands provided adequate nutrition through all periods of the annual cycle waterfowl use these wetlands. My research indicates that moist-soil management may provide an excellent source of food for spring migrating waterfowl.

Future research should attempt to determine if mallards are selecting high carbohydrate diets at other mid-migration locations. Management efforts to benefit spring migrant mallards should try to maximize availability of moist-soil seeds throughout spring by producing moist-soil vegetation and by manipulating water levels to benefit migrating mallards. This would include shallowly flooded (~10 – 15 cm) areas of moist-soil vegetation, as well as shallowly flooded forests.

No prior research has found lesser scaup actively selecting moist-soil seeds mid-migration, therefore, future research should examine scaup food selection during spring at other mid-migration locations, and determine the impact on body condition that this diet may have. Study sites should focus on, or include areas intensively managed for moist-soil plant production. It may also be important to determine why scaup are selecting high carbohydrate foods. Scaup may favor these foods because of a lack of energy reserves necessary to complete migration, or possibly physical factors within wetlands (i.e. unconsolidated bottom sediments, high turbidity) are preventing scaup from finding adequate invertebrate foods. For instance Tome and Wrubleski (1988) reported that lesser scaup often visually located food items in the substrate or associated with vegetation in the water column. Unconsolidated bottom sediments or poor water clarity could potentially reduce foraging efficiency by scaup by reducing their ability to visually locate foods. Scaup were also reported to use tactile location to find foods located in sediments (Tome and Wrubleski 1988), yet common scaup foods, such as amphipods, are often free-swimming, or found in vegetation. If scaup are forced to use tactile location because of reduced visibility they may be forced to only consume stationary food items located in the benthos (e.g. seeds). Management efforts focused on improving habitat for scaup should attempt to increase water clarity, consolidate bottom sediments, and provide vegetation that produces seeds as well as a substrate for invertebrates.

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Appendix A

Lateral Springtime Invertebrate Drift between a Restored Backwater and a Large Midwestern River

Laura A. Csoboth, Darrin L. Martin, James E. Garvey,

and Matthew R. Whiles

Abstract

Despite their oftentimes degraded state, backwater habitats are vital to many large river species, particularly when river connectivity remains intact. However, the importance of backwater-river connectivity to macroinvertebrates is poorly understood. During spring 2005, we investigated invertebrate drift into and out of a restored backwater, Swan Lake, at its confluence with the lower Illinois River. Weekly sampling at the backwater-river confluence revealed diel and taxonomic patterns of invertebrate drift, and relationships with abiotic conditions. Invertebrates collected represented 17 orders and 1 phylum, and were dominated by zooplankton. No differences between drift into or out of Swan Lake occurred, although aquatic invertebrates drifted more at dusk and night. Drift into, but not out of, Swan Lake was related to abiotic factors, surface velocity, water temperature, and water depth. These findings suggest that lateral drift may be of biological importance to invertebrates in large rivers.

Introduction

River floodplain areas are productive landscapes, oftentimes critical to biological processes in large rivers and supporting ontogenetic habitat changes in many species' life histories (Junk *et al.* 1989, Greenwood and Richardot-Coulet 1996, King 2004). Connectivity of the floodplain to the river allows for the exchange of organic matter and organisms, which may influence or maintain species assemblages at various trophic levels (Junk *et al.* 1989, Miranda 2005). For aquatic invertebrates, habitat heterogeneity created by various levels of river-backwater connectivity may affect species' distribution and abundance, where species actively or passively drift among habitats for survival, feeding, and reproduction (Greenwood and Rickardot-Coulet 1996). The drift of organisms between the river's main channel and its adjacent floodplain areas will, herein, be defined as lateral drift. Although the invertebrate movement between the main channel and floodplain areas may be as important as downstream drift, this lateral drift is not well understood (Arscott *et al.* 2005). These invertebrate drift events are not discrete, but interact temporally, spatially, and on a diel basis, with millions of invertebrates drifting downstream in large rivers (Greenwood and Richardot-Coulet 1996). The contribution of invertebrates from a backwater to the river has been investigated (Eckblad *et al.* 1984, references in Greenwood and Richardot-Coulet 1996, Arscott *et al.* 2005), but the extent of lateral movement in both directions between a river and backwater has, to our knowledge, never been explored. The lateral drift dynamics of invertebrates between a restored backwater system and an adjacent segment of the lower Illinois River were investigated to determine density and taxa patterns on a temporal and diel basis. Abiotic factors, such as temperature, river stage, and velocity were examined to determine their influence on drift patterns. Our study was focused during the spring when macroinvertebrate drift is highest (Koetsier & Bryan 1995).

Methods

Study Area

Restoration of Swan Lake (SL), a 1,100-ha Illinois River (ILR) backwater located between river kilometer 8 and 21, was completed through the federal Environmental Management Program (EMP) during the 1990s to improve its functionality and river connectivity. Historically, SL was connected to the ILR through a 0.5-km wide opening at its downstream end. Restoration of the backwater complex restricted this river connection to the width of a stop-log water control structure, about 5-m wide, which was nested within a rip-rap lined channel (c. 50 m long x 12 m wide). During normal pool stage, the stop-log structure is the only avenue through which exchange between the backwater and river occurs, making the stop-log structure the focal point of this study site.

Drift Sampling

To quantify bidirectional invertebrate drift between SL and the ILR, three conical drift nets (0.5 m x 2 m, 500-µm mesh) were attached to a floating, anchored PVC frame and fished during March through May 2005 (Figure 1). We sampled invertebrates for 15 minutes at the surface (approximately one-third channel depth) every week on the lake-side of the SL stop-log structure (Figure 1). Two directional net sets, one sampling invertebrates potentially moving into SL and one set sampling invertebrates leaving SL, were conducted at dawn, mid-day, dusk, and mid-night within 24-36 hours. Sampling was reduced to biweekly during May.

At each sampling time (e.g., dawn, mid-day, dusk, mid-night), surface water temperature (^{o}C) and dissolved oxygen (mg/L [YSI Model 52 Dissolved Oxygen Meter]), and surface water velocity (cm/s [Flo-Mate Model 2000]) were quantified. A Doppler meter (Model 6526-51 Starflow; Unidata America, Lake Oswega, Oregon, USA), anchored to the bottom of the water control structure, recorded continuous temperature (^{o}C) and depth (mm) twice an hour.

Upon completion of each drift net set, contents were flushed into the cod end and preserved in 95% ethanol. Each sample was split to approximately 100 individuals using a Folsom plankton splitter (Aquatic Research Instruments, Hope, Idaho, USA) and identified to Order under a dissecting microscope. Non-insects were identified using Smith (2001), and insects were identified using taxonomic keys in Merritt & Cummins (1996).

Data Analysis

We analyzed macroinvertebrates and zooplankton separately by investigating directional (i.e., into vs. out of backwater), diel, and seasonal trends in drift data, as well as correlations with abiotic factors. Lateral invertebrate drift represented exchange between the backwater and river and was calculated as number of organisms per minute. Some invertebrates were caught in nets set opposite to the direction of flowing water when velocities were 0.1 m/s or greater (e.g., frame positioned to catch organisms drifting out of SL while water flowed into SL at 0.11 m/s; Scheidegger and Bain 1995). This was not drift and thus was removed from data sets before analyzing. All data were $\log_{10}(x+1)$ transformed to meet assumptions of normality.

Two-way repeated measures ANOVA (proc MIXED, SAS Institute 1999) were used to test for differences among treatments sampled over time (Hurlbert 1984): (1) drift catch rates into and out of the backwater and (2) drift catch rates at dawn, day, dusk, and night. Non-zero catch rates of drift data for each direction were regressed (multiple regression, proc REG, SAS Institute 1999) against environmental parameters (e.g., depth, temperature, velocity). To control for experimentwise error rates, we used Tukey-Kramer post-hoc (Sokal and Rohlf 1995).

Results

There was no spring flood pulse during 2005. Water levels remained at or below normal pool level (i.e., 128 m), and temperatures rose to 25 ^{o}C during May (Figure 2). Movement of water through the SL water control structure was bi-directional, often changing direction multiple times a day, and velocities varied greatly. Mean channel velocities, which can reach velocities of greater than 2 m/s during rising flood waters, peaked at 0.32 m/s during this study.

Invertebrates were collected during every sampling time and date, comprising a total of 17 orders and 1 phylum (Table 1). Cladocerans and copepods dominated the catch into SL, while cladocerans, copepods, and dipterans dominated the drift out of the backwater (Table 1).

Temporal and diel drift patterns were observed, although no directional pattern was detected (macroinvert.: $F_{1,6} = 0.60$, $P = 0.41$; zoop.: $F_{1,6} = 0.03$, $P = 0.87$). Zooplankton drift rates were higher in March and April ($F_{8,27} = 2.55$, $P = 0.03$), though no temporal patterns were observed for macroinvertebrates ($F_{8,27} = 2.01$, $P = 0.08$; Figure 3). Macroinvertebrates, which were dominated by dipterans and hemipterans, displayed a diel drift pattern where lower catch rates were observed during dawn and daytime sampling (Time of day: $F_{3,4} = 7.28$, $P = 0.04$; Week: $F_{8,10} = 4.53$, $P = 0.01$; Time of day*Week: $F_{23,10} = 2.44$, $P = 0.07$; Figure 4). Zooplankton drift during dawn sampling was not significant (Time of day: $F_{3,4} = 1.51$, $P = 0.34$; Week: $F_{8,10} = 2.13$, $P = 0.13$; Time of day*Week: $F_{23,10} = 0.55$, $P = 0.87$; Figure 4).

Despite no difference between drift rates into and out of SL, invertebrates drifting into the backwater were influenced by abiotic factors (e.g., water temperature, surface velocity, and channel depth). Macroinvertebrate drift rates were significantly correlated to surface velocity (positively) and depth (negatively) in the multiple regression model (Model: Catch = 38.07 – 0.07*Temp + 0.16*vel – 16.72*depth; $F_{3,20} = 13.10$, $P < 0.01$). The regression model was not significant for zooplankton drifting from the ILR into SL (Model: Catch = 374.50 – 1.43*Temp + 0.17*vel – 158.25*depth; $F_{3,20} = 2.38$, $P = 0.10$). Drift rates of macroinvertebrates and zooplankton drifting out of SL were not significantly correlated to abiotic factors.

Discussion/Conclusion

During spring, lateral macroinvertebrate drift, predominantly out of SL, was dominated by dipterans (Chironomidae) and drift was highest at dusk and at night. Koetsier & Bryan (1995) also found that dipterans (*Chaoborus*, Chironomidae) dominated the drift in March, May, and June samples. There was a significant effect of time of day for invertebrate drift during this study, but it is most likely due to the presence of the hemipterans and not the chironomids. This is also suggested by Waters (1972) in which they found that while Chironomidae larvae may be abundant, they show little tendency to drift with diel periodicity. A significant positive correlation was found with macroinvertebrates and velocity ($P<0.0001$) and a significant negative correlation between macroinvertebrates and depth ($P=0.0452$). Both of these correlations are expected and explained by each other. Velocity is negatively correlated to depth meaning that as depth increases, velocity decreases. This is seen in the data as well. Macroinvertebrate drift increases with velocity and also increases with decreasing depth.

Zooplankton drift was highest during March and April. There was no significant drift into SL by zooplankton, but there seemed to be a trend of water depth and temperature on their drift rates. Both abiotic factors were inversely related to zooplankton drift. The relationship with depth is the same as with the macroinvertebrates. It is possible that with more data over a longer period, this relationship could be significant. The dominant taxa collected was Cladocera, primarily *Daphnia*. Cladocerans are lentic species more suited for backwater areas. Movement into backwater areas during periods of more connectivity allows these organisms to grow their populations in times of lower flow and connectivity (Eckblad *et al.* 1984). Once flow or connectivity increases, the increased population is then transferred back to the main river channel as part of drift. Eckblad *et al.* (1984) reported that these reduced flushing times of backwater lakes allows those species that are typically more lentic to increase their population.

Zooplankton made up 78.9% (by individual) of the total drift (in and out) of SL. Drift out of SL consisted of 65.3% of the total individuals caught. Lateral invertebrate drift may be substantial enough to influence adult fish congregations in these backwater-river confluences (Schultz 2006) or stimulate lateral larval fish drift by serving as a prey source (Eckblad *et al.* 1984, Sheaffer and Nickum 1986). This may be especially true for planktivorous fish species such as the invasive Asian carp (*Hypophthalmichthys* spp.).

This exchange between backwater areas and the main channel also serves as an exchange of biomass and productivity which may not have previously been accounted for or interpreted as only downstream movement. We acknowledge that once it is in the river, it becomes downstream drift, but it may not always remain as such. During the course of this study, the backwater habitat acted as a source of invertebrates, particularly zooplankton, contributed to the larger river system. Our study focused only on spring drift during a non-flood year. To accurately characterize lateral drift, a more comprehensive study should be conducted, involving all seasons and varying flood regimes.

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Table 1. Total numbers and mean catch rates (#/minute, ± SE) of laterally drifting invertebrates between Swan Lake and the lower Illinois River during spring 2005. Data are grouped by subclass or order.

	Into Swan			Out of Swan		
Taxon	*N*	Catch	(SE)	*N*	Catch	(SE)
Acariformes	225	0.197	(0.055)	420	0.384	(0.128)
Amphipoda	47	0.041	(0.017)	78	0.072	(0.029)
Anostraca	1	0.001	(0.001)	0	0.000	(0.000)
Arachnidae	2	0.002	(0.001)	0	0.000	(0.000)
Cladocera	11,134	9.700	(3.128)	17,136	15.755	(7.637)
Coleoptera	1	0.001	(0.001)	2	0.002	(0.001)
Collembola	19	0.017	(0.008)	16	0.015	(0.008)
Copepoda	7,423	6.486	(2.378)	11,444	10.499	(3.992)
Decapoda	0	0.000	(0.000)	2	0.002	(0.001)
Diptera	497	0.435	(0.116)	8,788	8.005	(5.212)
Ephemeroptera	96	0.084	(0.045)	37	0.034	(0.022)
Hemiptera	1,187	1.045	(0.411)	1,008	0.920	(0.354)
Hirudinea	4	0.003	(0.004)	12	0.011	(0.006)
Isopoda	1	0.009	(0.001)	7	0.001	(0.005)
Mollusca	4	0.004	(0.002)	0	0.000	(0.000)
Oligochaeta	4	0.004	(0.004)	2	0.002	(0.002)
Podocopa	3	0.003	(0.002)	103	0.096	(0.073)
Tricoptera	3	0.003	(0.003)	7	0.006	(0.114)

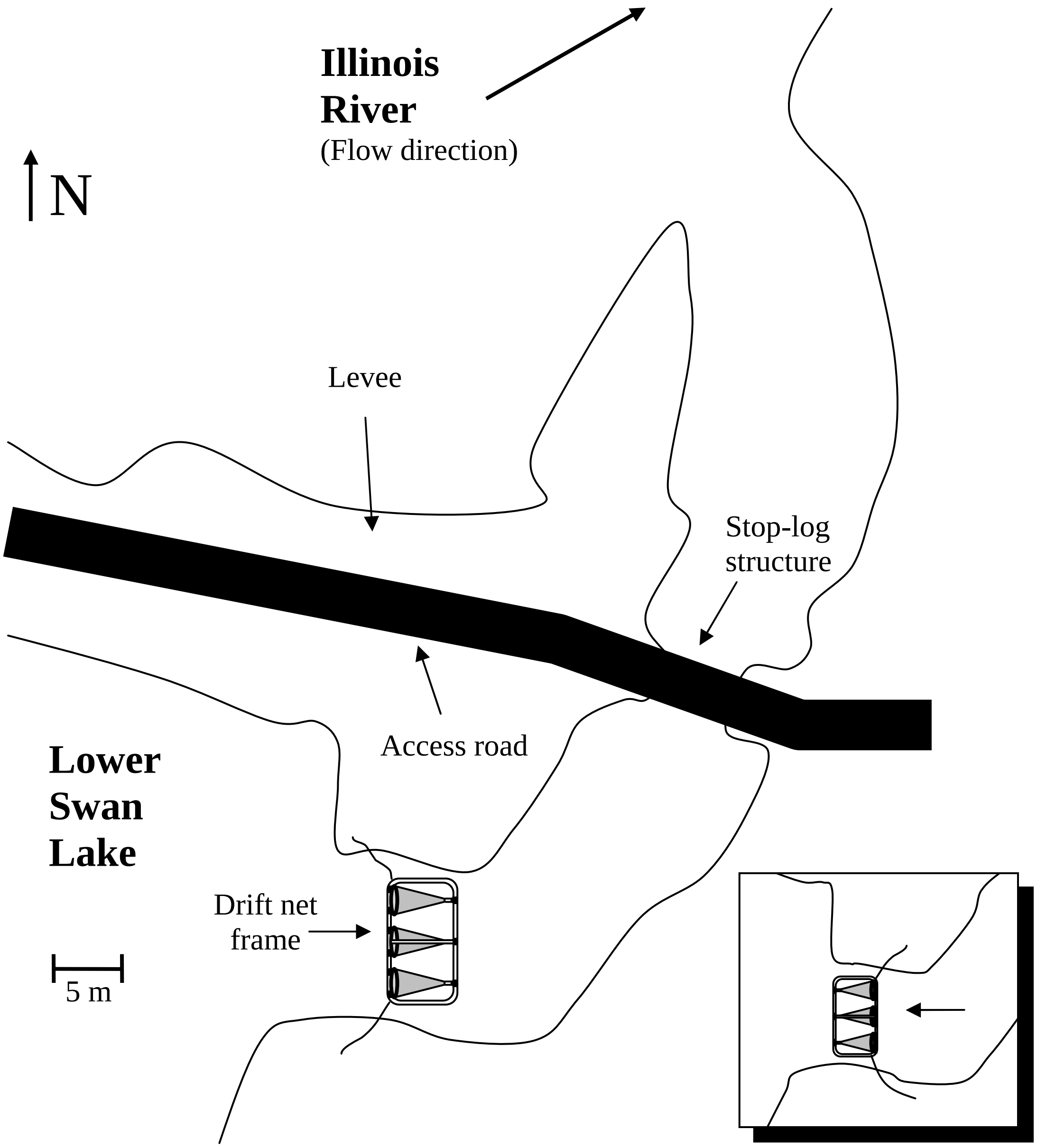

Figure 1. Fixed drift net site during 2004. Tandem nets were floated on the Swan Lake (SL) side of the stop-log structure. Schematic depicts nets sampling invertebrates drifting out of SL. Inlay portrays the drift net frame positioned to sample invertebrates drifting into SL.

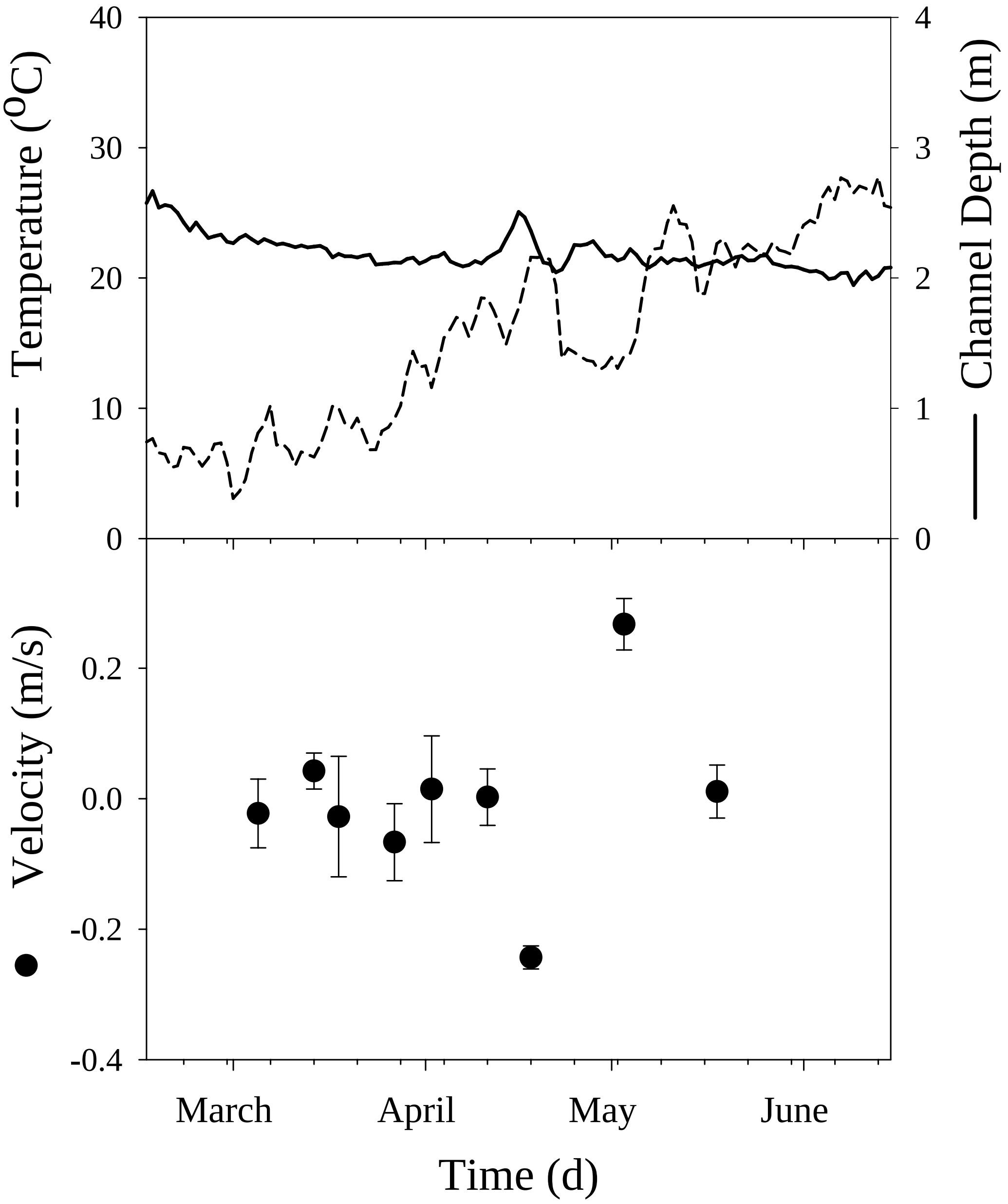

Figure 2. Mean daily water temperature (dashed line) and water depth (solid line) at the Swan Lake stop-log structure, collected with the Doppler unit during spring 2005. Mean water velocity data (●, ± SE) collected during sampling time points on the Swan Lake side of the stop-log structure.

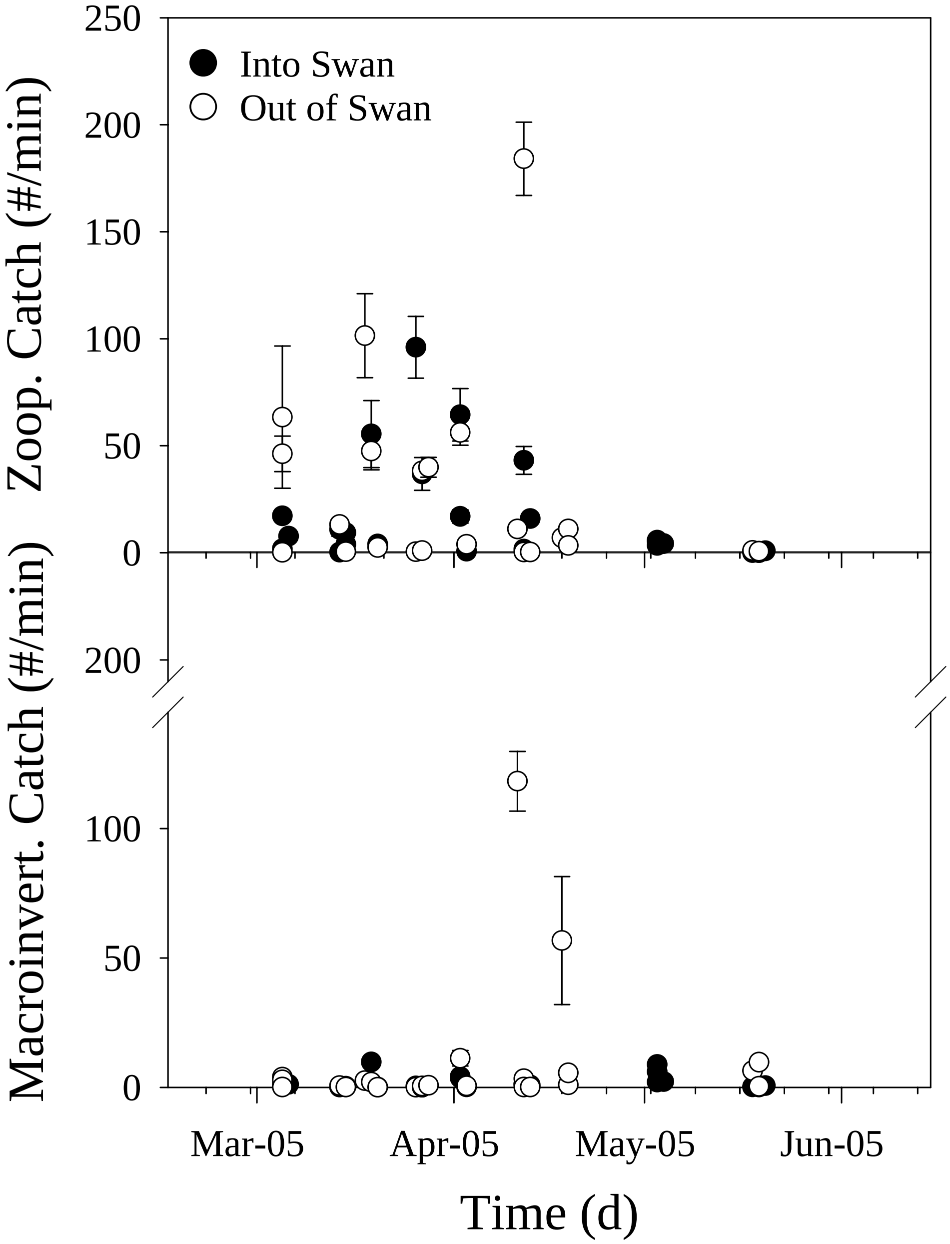

Figure 3. Mean catch rates (± SE) of zooplankton and macroinvertebrates into (●) and out (○) of Swan Lake during spring 2005.

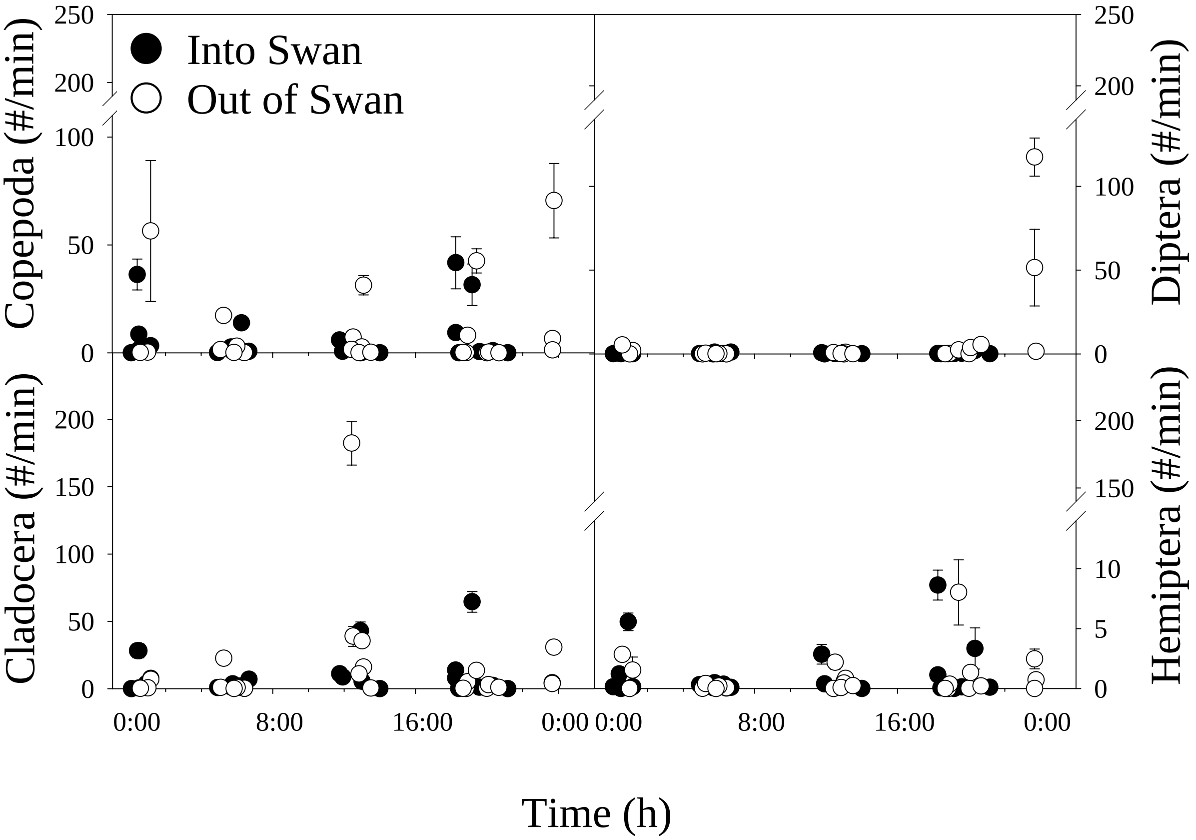

Figure 4. Mean diel drift rates (#/minute, ± SE) for the four main taxa sampled at the Swan Lake water control structure: copepods, cladocerans, dipterans, and hemipterans.

Appendix B

EFFECT OF BACKWATER LAKE MANAGEMENT ON THE FORAGING HABITAT OF FIVE COMMON RIVERINE FISHES IN SWAN LAKE, CALHOUN COUNTY ILLINOIS

BY

THOMAS TIMMERMANN

B.S., Western Illinois University, 2004

THESIS

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CHAPTER 1: INTRODUCTION

Ecosystem Restoration of Backwater Lakes

My study examines the effectiveness of improving habitat quality through management in a backwater lake. Backwater lakes, like most systems on earth, have been severely altered through anthropogenic effects. Pristine ecosystems are all but non-existent today. Restoration ecology attempts to restore ecosystems, such as backwater lakes, to a pristine or a more natural state, through manipulations and management of those ecosystems (Davis and Slobodkin 2004, Harris and vanDiggelen 2006). Often restoration ecology is coupled with the idea of Adaptive Management, in which ecological manipulations are set up as experiments with assessment and learning (Murray and Marmorek 2003). Restoration ecology often encompasses broad spatial and temporal scales. Some restoration efforts require modifications to entire systems, such as watersheds (Harris and vanDiggelen 2006). Temporally, these efforts may require years to assess any benefits and require constant adaptation of methods based on acquired knowledge (Murray and Marmorek 2003, Mant and Janes 2006). My study offers a unique opportunity to examine the effects that an adaptive management experiment has on an entire system.

Backwater lakes occur in the floodplain of rivers or on islands, and can be contiguous with the river for much of the year or remain isolated except during severe floods. They receive the majority of their water from the river, either through flooding events or through backflow from the river, and may have some flow at times. Backwater lakes are critical habitat in large river ecosystems. The microhabitats within backwater lakes are essential for many life history functions of fish and wildlife and also contribute to the general ecological integrity of the river (Junk et al. 1989).

Fluctuations in river stage provide a diversity of habitats within backwater lakes (Junk et al. 1989). Historically, temperate river systems in North America have experienced variations in water levels characterized by a high spring flood pulse and a moderate fall flood pulse, with a summer low water period between these flooding events (Sparks 1995). During this summer drying period, water flows out of backwater lakes and they begin to dry. This exposes the flats along the backwater lake’s edges, causing an increase in growth of emergent vegetation (Theiling 1998, Sparks 1995, Middleton 2002). As the summer proceeds, sediments in exposed flats compact and harden, allowing for decreased lake turbidity and providing a suitable rooting substrate for vegetation (Theiling 1998, Sparks 1995, Middleton 2002). New microhabitats are created in these backwaters as emergent vegetation grows and submerged vegetation becomes established. Changes in river stage also lead to the creation of two distinct types of backwater lakes: contiguous backwaters, which are connected year round to the river, and isolated backwaters, which are only connected during periods of high flow. These backwater lakes function differently ecologically within the system as riverine organisms have constant access to contiguous backwaters but only sporadic access to isolated backwaters.

Backwater lakes provide several different types of habitat critical to a variety of plants and animals found within the river system. There is evidence that backwaters serve as a refuge for native Unionid mussels from invasive zebra mussels (*Dreissena polymorpha*; Tucker and Atwood 1995) and that alterations made in backwater habitats may affect the diversity of these native mussels (Tucker et al. 1996). Fish such as sunfish (*Lepomis* spp.), cyprinids, gizzard shad (*Dorosoma cepedianum*) and freshwater drum (*Aplodinotus grunniens*) utilize backwaters for spawning grounds (Winemiller et al. 2000, Dewey and Jennings 1992, Sheaffer and Nickum 1986, Zigler and Jennings 1992). The importance of backwater lakes continues from spawning to larval and juvenile stages where the fish are provided refuge from current and a heightened supply of food. Juvenile fish are more prevalent in backwaters than in any other river habitat, consume prey found in the aquatic vegetation, and utilize the vegetation as cover from predators (Sheaffer and Nickum 1986). Johnson et al. (1998) also cited the benefits of backwater lakes as a winter refuge where fish seek to escape main channel currents and find warmer water.

Waterfowl also make heavy use of backwater lakes during fall migration and the spring return flight. Almost 36% of migrating waterfowl in North America use the Mississippi River and its backwaters (Havera et al. 1996). Backwaters provide an abundance of food for waterfowl including fingernail clams (Saphariidae) and other benthic macroinvertebrates (Schneider 2000). Vegetation and plant seeds in the backwaters are also food sources for dabbling ducks (Schneider 2000) and provide cover for ducklings in the spring (Havera 1999).

Backwater lakes are also critical for anthropogenic purposes. In addition to providing feeding and spawning habitat, the abundance of plant life removes toxins such as ammonia (Havera and Bellrose 1985). Backwater lakes also reduce the high sediment load of the main channel by allowing excess sediments to settle and compact through seasonal drying (Schneider 2000). Finally, backwaters provide a means of flood water control by holding, or allowing the flow through, of spring and fall flood pulses (Schneider 2000, Havera and Bellrose 1985).

Damming and channelization of rivers severely alters the link between rivers and backwater lakes. In the 1930's dams were installed on the Illinois River to allow commercial navigation. These dams maintain a specific water level and eliminate the summer drying period by reducing annual fluctuations in river stage (Koel and Sparks 2002, Nelson et al. 1994, Sparks et al. 1998). Additionally, channelization of the main channel and tributaries throughout the watershed allows for quick removal of water from upland areas. This, combined with draining of wetlands to provide more agricultural fields, increases the rate and amount of sediments deposited into the Illinois River and its backwaters (Koel and Sparks 2002, Poff et al. 1997, Havera and Bellrose 1985). A survey by Lee and Stall (1977) concluded that from 1903 to 1973 the Illinois River basin yearly lost an average of 15.4 million tons of sediment. As of 1993, many of the 53 backwaters along the Illinois River lost anywhere from 30-100% of their early 1900's volume because of increased sedimentation (Bhowmik 1993). Increased sedimentation not only causes backwaters to fill in but also results in great reductions in, or complete loss of, macrophyte populations, including *Potamogeton* spp. and *Nelumbo lutea* (Bellrose et al. 1979, Middleton 2002, Reese and Lubinski 1983, Delong 2005).

To combat sedimentation and the loss of vegetation in backwater lakes a series of management strategies have been implemented. Common along the Illinois River are the Habitat Rehabilitation and Enhancement Projects (HREP) that typically have the goals of reducing watershed sediment inputs and improving habitat in backwater lakes

(Muncy 1996). These projects usually include a series of physical improvements such as dredging of sediments, installing closing structures, and using islands to divert flow. Other management strategies include river level manipulation or conversion of backwater lakes into wetlands or marshes. River level manipulation entails the periodic lowering of the entire river water level allowing for exposure of mudflats which consolidates sediments and promotes vegetation growth (Sparks 1995). Conversion of backwater to wetlands or marshes involves isolating the lake from the river channel and then actively draining the lake (via pumping structures) which allows for vegetation growth. This is the most common form of management and is generally termed “moist soil management” (Havera et al. 1996, Havera and Bellrose 1985).

Management options used for reducing sediment deposition in Illinois River backwaters have both benefits and detriments. Lowering the entire water level of the river to allow for draining of backwaters restores the natural process of the summer drying period, consolidates sediments and promotes vegetation growth, but reduces the navigability of the river to both commercial barges and recreational boaters (Sparks 1995). Moist soil management also mimics the natural water regime and low water levels during the summer and exposes mudflats for a period of 70 days for plant germination and 90 days for plants to seed (Havera et al. 1996, Anderson and Smith 2000). Given the higher river levels present since the construction of locks and dams, this is done by installing levees separating the backwater lake from the river and then using pumping structures to remove the water and expose mudflats. Moist soil management increases growth of aquatic vegetation, lowers turbidity and compacts sediments; however, severing the lake from the river isolates backwater lakes to the point where the connection to the river is maintained only during severe flooding events. This greatly limits access to backwater lakes for fishes and other riverine based organisms (Sparks et al. 1998).

Swan Lake, a large backwater lake of the Illinois River (Figure 1.1), followed a similar trend to other Illinois River backwaters. Once highly productive habitat for fishes, aquatic vegetation and wildlife, but increases in sedimentation and the lack of summer dry periods have caused a recent decline in these biota (Theiling et al. 2000, USACE 1993). Swan Lake was targeted by the United States Army Corps of Engineers (USACE) as a lake that would benefit from a HREP because of these problems with sedimentation and loss of aquatic vegetation, as well as fluctuations in water level and wind generated waves (Muncy 1996). To address these issues the USACE made several modifications to the lake including: a large levee separating Swan Lake from the Illinois River, levees that divided the lake creating 3 management units, stop log structures, pumping structures and island chains (Figure 1.2; Muncy 1996). The HREP project began in 1995 and was finished in 2001.

With Swan Lake divided into separate units, there exists the ability to manage the units of Swan Lake independent from each other. The upper unit has consistently been managed by the Illinois Department of Natural Resources (IDNR) as a traditional moist soil unit, with levees separating the backwater from the river and aggressive pumping to expose mudflats and promote emergent vegetation growth. In the remaining two units of the lake, the United State Fish and Wildlife Service (USFWS) and the USACE have been attempting to utilize the benefits of moist soil management while providing for riverine fish access to these lakes by practicing adaptive management. In 2004 and 2005 this resulted in the middle unit being pumped down in the late summer to allow for compaction of sediments and vegetation growth, but a connection between the backwater and the river was maintained to allow access by fish for the remainder of the year. The lower unit was not pumped down and the connection between the backwater and the river was maintained throughout the entire year. These differences in management should alter the habitat within each unit. This study focuses on the effect that these management techniques have on fish habitat. The upper unit with its reduced connection to the river and aggressive drawdowns is not managed with riverine fish in mind; for that reason this study focuses on the effect that management had on the fish habitat within the middle and lower units only.

Theoretically, draw downs will increase the amount and diversity of aquatic vegetation, lower turbidity and harden sediments. Establishment of aquatic vegetation allows for colonization of a diversity of macroinvertebrate taxa (Anderson and Smith 2000, Anderson 1997). Studies have shown a positive correlation between vegetation and macroinvertebrate abundance (Gregg and Rose 1985). Declines in turbidity may change zooplankton, plant and macroinvertebrate communities (Lloyd et al. 1987, McCabe et al. 1983, Thorp et al. 1994) and hardening of the sediments could alter both the composition and the location of macroinvertebrates within the sediment (Persson and Svensson 2006, Newrkla and Wijegoonawardana 1987). Crowder and Cooper (1982), applied these ideas to fish diet showing that bluegill consumed the fewest number of prey taxa at low vegetation density, slightly more at the highest vegetation density and the most at an intermediate density.

For this study I present data used to test the hypothesis that the different management strategies used in the middle and lower units (facilitated by improvement made to the lake through the HREP) cause changes in the prey community and subsequently alter the diets of the fish inhabiting the lakes. I predict that middle unit drawdowns would harden sediments, increase the amount of emergent vegetation, and cause diversification in the prey base. The lower unit, with no active drawdown, should remain relatively homogeneous in terms of a lack of vegetation and sediment characteristics. With a homogeneous habitat, one would expect a less diverse prey community. Overall, I would predict the habitat in the middle unit would support a larger and more diverse prey base for fishes, and that the fishes within that unit would generally ingest an increased diversity of prey taxa as well as consume more prey biomass.

Fish are often used as an indicator of habitat quality and are the primary assessment tool in the Index of Biological Integrity (Schneider 2002, Meloane et al. 2003, Ganasan and Hughes 1998). In this study, diet will be the key gauge of habitat quality. Diet was preferred over other techniques (such as stable isotope analysis, fish condition or an age-growth relationship) to assess habitat quality for several reasons. First, these units were open to the river for extended periods of time, allowing fish to immigrate and emigrate at will. For this reason, a study utilizing stable isotopes would have difficulty detecting trends within different habitats because the fish did not exclusively occupy that habitat. A study utilizing condition or growth would be useful in detecting differences several years after the project had been completed and the changes fully established but not necessarily during the course of these changes. Using diet allowed us to investigate differences in feeding habitat at the moment of capture.

Diet is often used to gauge differences in habitats. A study done by Persson and Hansson (1999) demonstrated how diet could be used to determine habitat preference in roach (*Rutilus rutilus*), perch (*Perca fluviatilis*) and bream (*Abramis brama*). Studies done by Werner, Mittelbach and others have shown that fish diet can be used to assess habitat selectivity (Mittelbach 1984, Power 1984, Werner et al. 1981, Werner et al. 1983, Werner and Hall 1979). Finally, fish diets have been used as an indicator of prey availability when sampling of the benthos is difficult and traditional techniques may be inadequate (Link 2004, Pinnegar et al. 2003).

My study assesses how diet of fishes varied between the middle and lower units of Swan Lake to gain insight into the effectiveness of the management practices applied to these units. To accomplish this I first examined the diets of five common riverine fishes during the summer and winter of 2005 to see if there were any differences in caloric intake, diet composition and diet richness between the units. The second portion focused on the vertical distribution of macroinvertebrates within the sediments of Swan Lake. Analysis of the diet data from the first portion of my study led me to hypothesize that soft sediments may provide a refuge to macroinvertebrates from benthivorous fishes. I tested this by developing a new sampling device capable of discerning the vertical distribution of macroinvertebrates in cores up to one meter.

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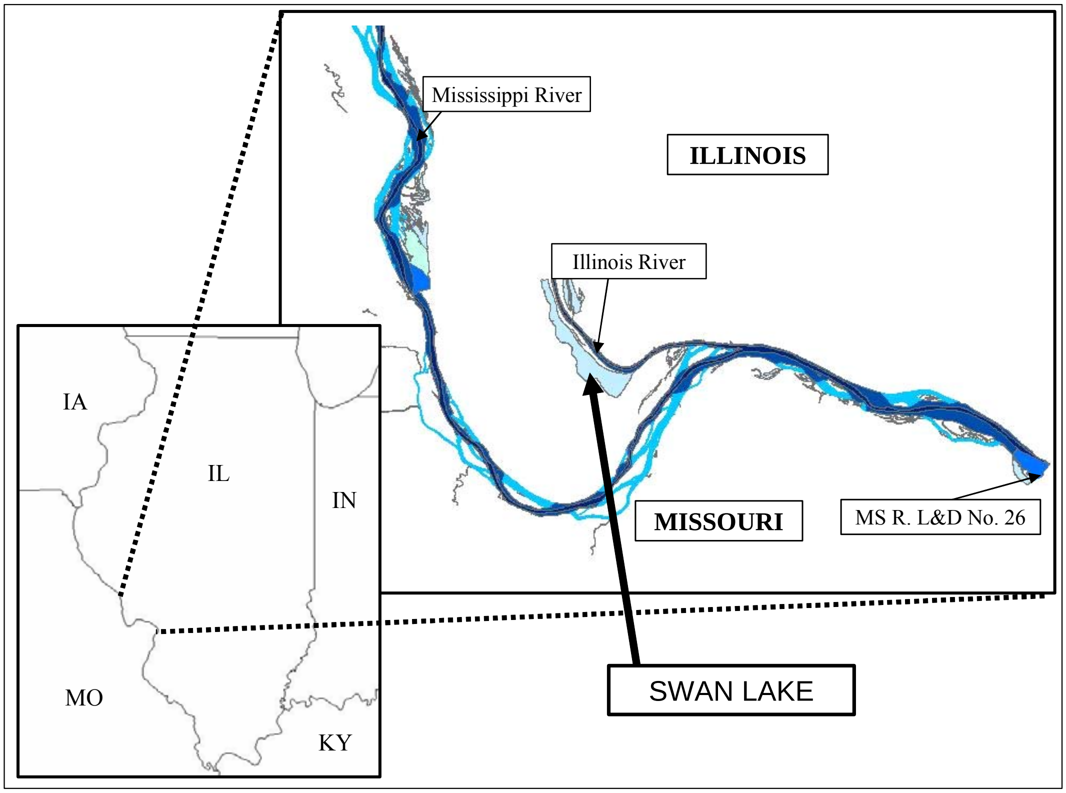

Figure 1.1: Location of Swan Lake, within the state of Illinois and within Pool 26 (inset).

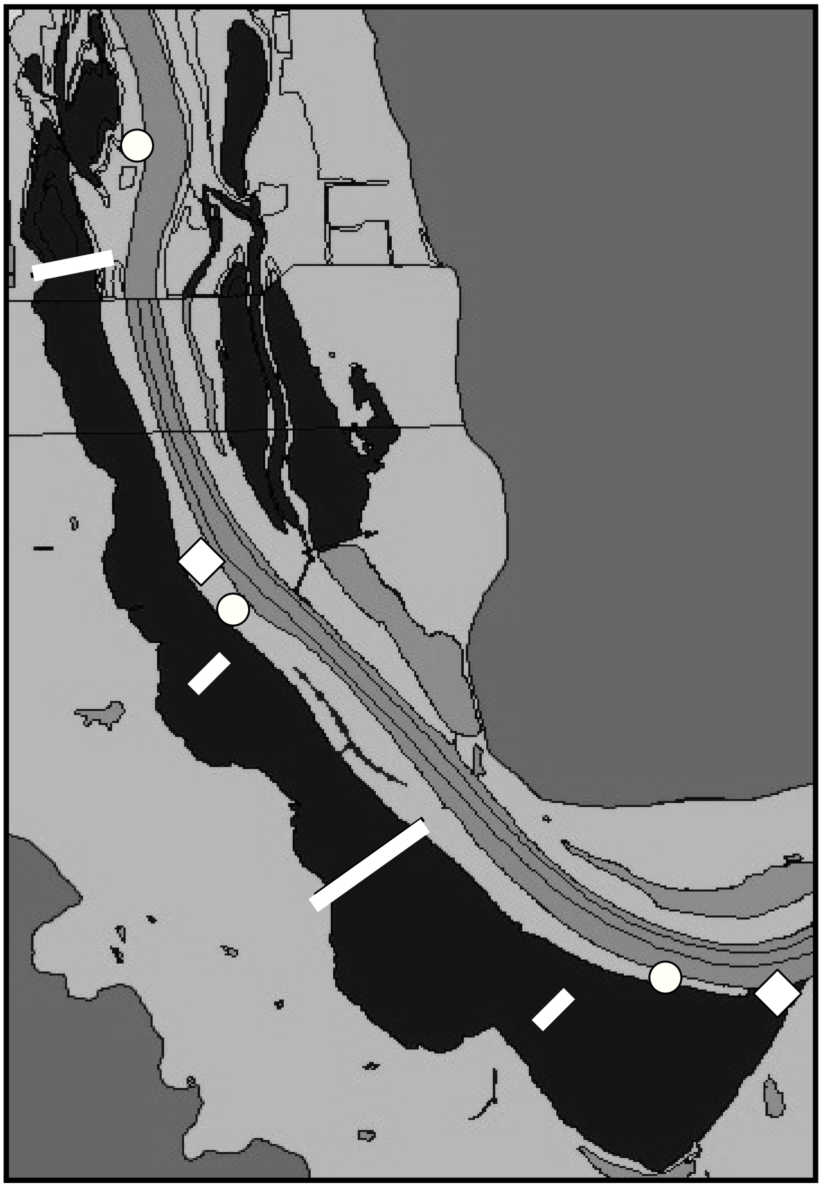

Figure 1.2: Improvement made to Swan Lake in the 2001 habitat rehabilitation and enhancement project. The large white lines represent levees that were installed to compartmentalize the lake, the small white lines are island chains, the circles indicate pumping structures, and the diamonds are stop log structures.

CHAPTER 2: VARIATION IN THE DIET OF FIVE COMMON RIVERINE FISHES UNDER DIFFERING MANAGEMENT STRATEGIES IN A BACKWATER LAKE

SUMMARY

Swan Lake, like many backwater lakes in temperate floodplain rivers, has been severely affected by sedimentation and loss of aquatic vegetation associated with river management and land use practices in the watershed. To compensate for these changes, a Habitat Rehabilitation and Enhancement Project (HREP) was completed that compartmentalized the lake into three units, allowing for drawdowns to compact sediments and the implementation of adaptive management techniques. The lower unit has been drawn down only once and has remained connected to the river at all other times. The middle unit has been drawn down several times and the connection to the river has generally been severed in the summer. My study examines the effect that these different management strategies have on the habitat for common riverine fishes. Diets of bluegill, common carp, white and black crappie, freshwater drum, and orangespotted sunfish were examined in the summer and winter of 2005. With a few exceptions, the differences found suggest that the middle unit management provided a better foraging habitat for fishes. Common carp ingested more calories in the summer and winter and more different prey items in the summer in the middle unit. After the middle unit drawdown, crappie ingested more calories in the middle unit than they did prior to the drawdown. While the management used in the middle unit appears to provide for better foraging habitat for fishes, improvements are still in early stages. Continued drawdowns on the middle unit and the addition of a drawdown regimen on the lower unit likely would further improve foraging conditions.

INTRODUCTION

Large river/floodplain ecosystems in temperate climates typically have a predictable annual hydrograph characterized by a high spring flood, a moderate fall flood and a summer low-water period (Sparks 1995, Junk et al. 1989). These fluctuations in water levels are vital to backwater lakes because the floods connect the backwaters to the river and the low water period exposes flats along the lake edges, compacting exposed sediments, decreasing turbidity and ultimately increasing the growth of aquatic vegetation (Theiling 1998, Sparks 1995, Middleton 2002). Critical habitats for fish and wildlife are created in these backwaters as emergent vegetation and submerged vegetation become established.

Damming and channelization of rivers severely alters the link between rivers and backwater lakes. In the 1930's, dams were installed on the Illinois River which maintained a nine foot commercial navigation channel and eliminated the summer low water period (Koel and Sparks 2002, Nelson et al. 1998 Sparks et al. 1998). Channelization throughout the watershed and draining of wetlands increased the rate and amount of sediments deposited into the Illinois River and its backwaters (Koel and Sparks 2002, Poff et al. 1997, Havera and Bellrose 1985). Most Illinois River backwaters have lost 30-100% of their early 1900's volume (Bhowmik 1993, Lee and Stall 1977). Increased sedimentation ultimately results in reductions or loss of macrophytes, including *Potamogeton* spp. and *Nelumbo lutea*, due to increased turbidity and lack of a firm rooting substrate (Bellrose et al. 1979, Middleton 2002, Reese and Lubinski 1983, Delong 2005).

To combat sedimentation and the loss of vegetation in backwater lakes and attempt to restore a more natural process, management strategies aimed at restoring low water periods and reducing sediment inputs are commonly implemented. One such strategy, termed “moist soil management,” converts backwater lakes into wetlands or marshes by isolating the lake from the river channel and then actively draining the lake (via pumping structures), allowing for vegetation growth (Havera et al. 1996, Havera and Bellrose 1985). Moist soil management is typically undertaken to improve habitat for migrating waterfowl that use the vegetation as a food and cover source, but does not allow riverine fishes access because connectivity with the river is lost except during extreme flooding events (Sparks et al. 1998), therefore opportunities for fishes for forage in these backwaters is reduced.

Swan Lake, a large backwater lake of the Illinois River (Figure 2.1), followed a trend similar to other Illinois River backwaters. It was once a highly productive habitat for fishes, aquatic vegetation and wildlife but sedimentation and lack of a summer dry period (due to river and land use practices) caused declines in biota (Theiling et al. 2000, USACE 1993). In response to degraded habitat conditions in Swan Lake, the United States Army Corps of Engineers (USACE) instituted a Habitat Rehabilitation and Enhancement Project (HREP). One of the main goals of the HREP was to restore aquatic macrophyte beds by reducing sediment inputs and compacting sediments through drawdowns. To reduce sediment inputs, a levee isolated Swan Lake from the river and connectivity was controlled through stop log structures. Compaction of the sediments was accomplished by installing pumping structures which allow for drawdowns, and additional levees were installed that compartmentalized the lake into management units (USACE 1993; Figure 2.2). Implementation of drawdowns allowed for the managers to mimic the pre-dam low water period and attempt to restore the natural compaction and hardening processes.

The division of Swan Lake allowed for management of the units independent from each other. The upper unit was previously isolated as part of another HREP and has been consistently managed by the Illinois Department of Natural Resources (IDNR) for waterfowl as a traditional moist soil unit (USACE 1993). In the other two units, the United States Fish and Wildlife Service (USFWS) have been experimenting with two different management strategies to find ways to gain the benefits of moist soil management while providing riverine fishes access to the lake (USACE 1993). The main variables being managed in these units are the percentage of time that the lake is connected to the river, and the frequency and aggressiveness of the drawdowns. This study focuses on the effects these management strategies have on fish habitat within the lower and middle units. Prior to the implementation of the management practices in the middle and lower units, these units were very similar in terms of sediment hardness and turbidity (Theiling et al. 2000). For this reason, any differences observed through this study will be assumed to result from differences in management practices rather than inherent differences between the units.

Fish diet was used as the key indicator of differences in habitat between the lower and middle units of Swan Lake. Diet was preferred over other techniques (such as fish condition, an age-growth relationship or stable isotope analysis) because it was sensitive to the changes within the time frame that was studies. A study using condition or growth would be useful in detecting differences several years after the project had been completed but may not be effective before changes were fully established. In addition, these units were open to the river for extended periods of time allowing fish to move in and out at will. For these reasons a study using stable isotopes would have difficulty detecting differences between units because the fish may not have exclusively occupied either unit long enough to develop distinct isotopic signatures. As opposed to the low sensitivity results that would have been obtained with growth, condition, or stable isotopes, diet provides for an instantaneous measure of habitat quality, increasing the probability of detecting differences that reflect the conditions in the two units. Persson and Hansson (1999) demonstrated how diet could be used to determine habitat preference in roach (*Rutilus rutilus*), perch (*Perca fluviatilis*) and bream (*Abramis brama*). Other studies have shown that fish diet can be used to assess habitat selectivity (Mittelbach 1984, Power 1984, Werner et al. 1981, Werner et al. 1983, Werner and Hall 1979). Finally, fish diets have been used as an indicator of prey availability where sampling of the benthos is difficult and traditional techniques may be inadequate (Link 2004).

Changes made through the HREP allowed for units of Swan Lake to be managed independently of one another. Since completion of the HREP in 2001 the middle unit has been continuously drawn down with intermittent periods of connection to the river, while the lower unit has only had one drawdown attempted and has been continuously connected. I assessed the effectiveness of management practices used in the middle and lower unit at improving habitat quality for fishes. I hypothesized that frequent drawdowns in the middle unit would harden the sediments, increase the amount of emergent vegetation and cause diversification in the prey base. For these reasons, fishes feeding in the middle unit would be expected to consume more calories and greater numbers of prey items than fishes in the lower unit. Overall, I expect the habitat in the middle unit would better support a prey base for riverine fishes and that the fishes within the middle unit would show greater prey richness and consume more calories.

METHODS

Study Site

Swan Lake is 1,175 hectare backwater lake of the Illinois River that reaches from Illinois River mile 5 to 13 with an average depth of 64 cm (Figure 2.1). Swan Lake once supported abundant of aquatic vegetation and diverse communities of fishes and wildlife; however, sedimentation caused declines in aquatic vegetation, unconsolidated sediments, and high turbidity, prompting the implementation of an HREP in 1995 (Theiling et al. 1991). The HREP created levees which separated the lake into three management units. Since completion of the HREP in 2001, some benefits due to the project have been observed. A lake wide (i.e. all units) drawdown in 2002 caused limited amounts of sediment compaction in all the units, but also killed a good portion of the fishes within the lake (J. Chick, personal communication). Since then regular partial summer drawdowns on the middle unit have allowed for the seasonal growth of emergent vegetation and further sediment compaction (J. Chick, personal communication). In the year of this study (2005) a major drawdown was implemented on the middle unit from June 10th to September 30th (Figure 2.3), while a connection between the backwater and the river was maintained to allow access by fishes for the remainder of the year (Figure 2.3). The lower unit was not pumped down and the connection between the backwater and the river was maintained from 2003 - 2005.

Fish Collection and Diet Analysis:

To assess ecological benefits caused by the HREP, staff at the Illinois Natural History Survey (INHS) conducted pre- and post- project monitoring of macrophytes, macroinvertebrates, fishes and water quality within the lake. Fishes were collected using tandem fyke nets, tandem mini-fyke nets and trammel nets. Tandem fyke and mini-fyke nets were two Wisconsin type nets tied at the leads and set at randomly chosen sites throughout the lake for 24 hours (Gutrenter et al. 1995). Tandem fyke nets were only set in water greater than 40 cm deep to ensure submersion of the throats. Trammel nets with 274 m by 1.8 m three panels were set at randomly chosen sites throughout the lake for 1 hour (Gutrenter et al. 1995). For this diet study, target fishes were collected from these sampling efforts and some supplemental sampling (this included hoop-net sampling and electrofishing). Total length (TL mm) was recorded for each fish captured. All non-target fishes were released and target fishes were kept on ice until stomach extraction. Wet mass (g) was later calculated for each target fish based on published length-mass relationship values (Anderson and Neumann 1996, Froese and Pauly 2006).

I selected several fish species to study to cover the diversity of feeding guilds that exist within Swan Lake. These target fishes included: bluegill (*Lepomis macrochirus*), black and white crappie (*Pomoxis nigromaculatus* and *P. annularis*), common carp (*Cyprinus carpio*), freshwater drum (*Aplodinotus grunniens*), and orangespotted sunfish (*L. humilis*). Back and white crappie were grouped into one category (crappie) because in Swan Lake, the diets were not dissimilar (i.e. both species would be in the same feeding guild) and grouping them allowed me to increase the numbers of individuals examined and better describe the variation in their diet overall. Attempts were made to catch sufficient numbers of these fishes for a summer (June 1st – September 30th) and winter (November 15th – March 31st) analysis (Figure 2.3); however, not all fish species that were collected in the summer were available in sufficient numbers in the winter. All summer middle unit fishes were collected prior to the drawdown.

I enumerated items from the stomach and foregut only because this allowed for the detection of easily digested prey items, thus providing for an accurate assessment of diet composition (Suteka and Huusko 2000). Stomachs or foreguts (foregut was identified as the portion from the esophagus to the first bend of the intestine) were removed and preserved in a 10% buffered formalin solution. After extraction, stomachs and foreguts were dissected, thoroughly rinsed, filtered through a 35 µm sieve and placed into Petri dishes for sorting under a dissecting microscope. Contents were identified to the lowest taxonomic level possible given digestion of the item and counted.

For fishes thought to be zooplanktivorous (bluegill, crappie and orangespotted sunfish) the contents remaining after sorting were placed into a settling jar of known volume and a sub-sample of 5.25 mL was taken using a Henson-Stemple Pipette and transferred to a Ward Whipple Wheel for counting of zooplankton under a dissecting microscope. All large bodied zooplankton (including cladocerans and copepods) were enumerated and if more than 10 rotifers were observed, a second subsample of 1 mL was taken and rotifers were counted in a Sedgewick-Rafter Cell under a compound microscope. Cladocerans were identified to family, copepods to order and rotifers to genus because of digestion of the items.

Biomass was obtained for all prey items to allow for conversion to caloric values of the food items ingested. Calories were preferred over biomass because I felt that it gave a better sense of the differences in energy consumption between the units. All prey items other than zooplankton were individually weighed after 24 hours drying time at 70°C. For zooplankton a sample of 30 individuals of each group for each unit and season was measured for total length and those lengths were converted to dry mass using the regressions in McCauley (1971). Dry mass was converted to calories using the conversions outlined by Cummins and Wuycheck (1971). Total calories were calculated as the sum of the caloric values of each prey item found in the stomachs by an individual fish. My use of the term total calories should not be confused with a measure of total calories consumed over a specified measure of time.

Data Analysis

I tested for differences in the relative calories for each prey taxa among fish species and between units using analysis of similarity (ANOSIM). ANOSIM is a multivariate corollary to the univariate ANOVA that allows comparison of the degree of separation between groups. Using a Bray-Curtis similarity matrix, ANOSIM calculates an R statistic, which is a measure of the dissimilarity among groups relative to the dissimilarity within groups. Random permutations of the Bray-Curtis similarity matrix are used to determine a *P*-value based on the probability that a greater R statistic could be achieved from random combinations of the data (Clarke and Warwick 2001). An R close to 0 indicates that the differences between groups was slight while an R close to 1 indicates that the differences between groups were great (Clarke and Warwick 2001). Whereas the ANOSIM test of the differences among species might seem to be unnecessary because different species often consume different prey items, it was used to determine if the amount of diet overlap among species varied between units. Caloric values for the prey items were limited to those found in at least 5% of the fish, and these data were log transformed to reduce the effect of dominant prey items. Non-metric multi dimensional scaling (NMDS) was used to illustrate differences found in the ANOSIM. A measure of the adequacy of the NMDS fit is given as the stress value. A stress less than 0.2 is generally considered an adequate representation of the observations (Clarke and Warwick 2001). Both the ANOSIM and the NMDS were run using a Bray-Curtis similarity matrix and on the PRIMER-E v.5 statistical software (Clarke and Warwick 2001).

Similarity breakdown (SIMPER) was used to determine what prey items contributed to differences in the diet of fishes between the units. SIMPER calculates the dissimilarity between groups (in this case either differences between units or fish species) and how much an item contributes to the average dissimilarity; it also calculates the similarity within a group (Clarke and Warwick 2001). The same data matrix used in the ANOSIM was log transformed to reduce the effect of dominant species and analysis was conducted using PRIMER-E v.5 statistical software (Clarke and Warwick 2001).

To determine differences in total calories consumed between the two units, analysis of covariance (ANCOVA) was used (PROC GLM; SAS Institute 1999). Use of weight as a covariate was deemed essential because of the wide range in size of fishes examined. An analysis of variance (ANOVA) showed that there were no differences in the average mass of most fish species caught between the units. Carp, however, were the only species to show differences in mass between the units and later analysis showed that eliminating individuals to make the mass uniform did not alter the outcome of the test, so all fishes caught were included in the analyses conducted. The main effects tested were differences in units and fish species, and the covariate was wet mass of the fish. ANCOVA was first run with all of the co-variate interactions (i.e.: unit by mass, fish species by mass and unit by fish species by mass) then all non-significant co-variate interactions were removed so that the final model tested only differences in unit, fish species, unit by fish species and mass. To determine differences in diversity of the prey base between the units, an ANOVA was run on the number of different taxa consumed for just the fish that had prey items in their stomachs (PROC GLM; SAS Institute 1999). The main effects were differences in units, fish species and the interaction of unit with fish species. Total calories and wet mass of fishes was log transformed to make the data homoscedastic and to normalize the data distribution. Single degree of freedom post-hoc contrast statements were performed when main effects or the interaction was significant. For both the ANCOVA and the ANOVA the experimental unit to test the differences in the units was individual fish and significance was determined at a *P*-value of 0.05 or less.

RESULTS

I captured a total of 673 fish for diet examination; 439 in the summer period and 234 in the winter period. In the summer, the majority of the fishes examined were common carp (39%) followed by orangespotted sunfish, bluegill, drum and crappie. In the winter, 126 common carp and 108 crappie were examined; the remaining target fishes were not caught in sufficient numbers for examination of their diets. For all fishes captured for diet analysis, the division of catch between the lower and middle units represented a fairly even split (Table 2.1).

For some species the proportion of fishes feeding differed between the units, but other species showed little to no differences. Common carp in the summer differed dramatically between the units, with more feeding in the middle unit than in the lower unit, but in the winter, about 50% of the common carp did not feed in both the middle and the lower units (Table 2.1). Bluegill also differed in the summer with more fish feeding in the lower unit than in the middle unit (Table 2.1). Crappie, freshwater drum and orangespotted sunfish showed little variation between the units (Table 2.1).

A few dominant prey taxa comprised the diets of the fish and diet differed marginally between units but differences among species were greater (Figure 2.5; and Figure 2.7). ANOSIM showed significant differences between units for both the summer and the winter; but the very low R statistic shows these differences to be slight for both seasons (Table 2.2). SIMPER analysis showed that in the summer plant material contributed more to the diets of middle unit fishes than lower unit fishes, whereas corixids contributed more to diets in the lower unit than in the middle unit (Table 2.3; and Figure 2.5). ANOSIM also showed that in the summer diet composition differed among all species. Common carp diets were greatly different from crappie, bluegill and orangespotted sunfish whereas crappie and freshwater drum were less different (Table 2.2). In the winter, SIMPER analysis showed that fish (as a prey) and unknown crustaceans contributed more to the diets of middle unit fishes while zooplankton and crayfish contributed more to the diets of fishes in the lower unit (Table 2.3; and Figure 2.7). ANOSIM showed that in the winter the diet composition of common carp and crappie were statistically different and the R statistic showed this to be a moderately high difference (Table 2.2). Bluegill, crappie and orangespotted sunfish consumed a diversity of items other than chironomids including corixids, zooplankton, fish (as a prey) and other macroinvertebrates, the diets of common carp also included plant material whereas freshwater drum primarily consumed fish in addition to the chironomids (Table 2.4).

Diets of some fishes feeding in the middle unit were more diverse than those feeding in the lower unit, while other fish species showed little difference between the units (i.e. differences in the number of prey items ingested). For both the summer and winter the ANOVA model explained a significant portion of the data (summer: $F_{9, 421}$ = 22.09, $P < 0.001$, $R^2 = 0.26$; winter: $F_{3, 227} = 80.14$, $P < 0.001$, $R^2 = 0.54$). In the summer, the number of prey taxa ingested varied significantly among units, fish species, and there was a significant interaction between unit and fish species (Table 2.5). Common carp and orangespotted sunfish ingested more prey taxa in the middle unit than in the lower unit; whereas bluegill, crappie, and freshwater drum did not differ between the units (Table 2.5; Figure 2.8). A similar trend was observed for the winter with different numbers of prey items ingested between the units, fish species and there was a significant interaction between unit and fish species (Table 2.5). Crappie consumed more prey taxa in the middle unit than in the lower unit, whereas common carp did not differ between the units (Table 2.5; Figure 2.8).

Some of the species ingested more calories in one unit than the other, but patterns varied between seasons and among fishes. The ANCOVA model explained a significant portion of the variation in both the summer and the winter (summer: $F_{10, 420} = 16.24$, $P < 0.001$, $R^2 = 0.28$; winter: $F_{4, 226} = 9.52$, $P < 0.001$, $R^2 = 0.14$). In the summer, there were differences in total calories consumed between fish species and the interaction of unit and fish species was significant, but the main effect of unit was not significant nor was the covariate weight (Table 2.6). In the winter, the total calories consumed varied by unit and fish species, there was no interaction between unit and fish species, and the covariate weight was significant. In both seasons common carp ingested more calories in the middle unit than in the lower unit (Table 2.6; Figure 2.9). Crappie ingested more calories in the lower unit in the summer but more calories in the middle unit in the winter (Table 2.6; Figure 2.9). Bluegill, freshwater drum and orangespotted sunfish did not differ significantly between the units in total calories consumed (Table 2.6; Figure 2.9).

DISCUSSION

Overall, the middle unit of Swan Lake appeared to provide a better foraging habitat for common carp and crappie, and there was some evidence that the middle unit is beginning to provide better habitat for other fishes, such as orangespotted sunfish. The management practice used in the middle unit was aimed at consolidating sediments and promoting the growth of aquatic vegetation. Accomplishment of these goals appears to have begun. An overall measure of sediment flocculency indicates that the middle unit has firmer sediments than the lower unit (Dolan and Chick 2005) and the drawdowns have facilitated the growth of emergent vegetation in the late summer and fall (Dolan and Chick 2005). It could be expected that with continuation of these management practices the middle unit sediments would further harden and there would be the continuation of vegetation growth and possibly the establishment of submergent vegetation

The presumed effects of these management practices have begun to translate into differences in foraging habitat between the units. Crappie appears to best benefit from the seasonal growth of vegetation. Prior to the 2005 drawdown, crappie in the lower unit ingested more calories than those in the middle unit, but after the drawdown this pattern reversed, suggesting a better prey community in the middle unit after the drawdown. Common carp fed more often and ingested more calories in the middle unit than in the lower unit which could be because the firmer sediments better facilitates benthic feeding and vegetation provides for an additional food source. Persson and Svensson (2006) showed that benthic prey items co-existed with benthic predators by using the deeper sediments as a refuge. The results from this study could be applied to Swan Lake to possibly explain why common carp feeding in the lower unit appear to have some difficulty feeding.

The middle unit appears to provide better foraging habitat than the lower unit but how does that translate into meeting a fish's daily ration? Many papers have suggested methods to best study daily ration in the field (Bajkov 1935, Elliott and

Persson 1978). Most of these methods require knowledge of the fish's evacuation rate and the amount of food consumed over a specific time to ultimately determine the amount of food digested per unit of mass of the individual fish (Bajkov 1935, Elliott and Persson 1978). A conservative estimate of daily ration as biomass ingested per gram of body weight was calculated for comparison to the suggested requirement of 1-5% of their dry body mass in food a day for all fish species (Wang et al. 1998, Specziár, 2002). In this study, I did not expect to see a full daily ration in the foregut and stomach and because I did not know the time period over which prey were consumed. Therefore, the calculation I used did not account for food consumed by the fish that had already been digested or was in the lower digestive tract, food yet to be consumed by the fish, or the effect that specific abiotic and biotic factors have on the daily requirement for fishes and was a very conservative measure. Taking this into account, the middle unit, though improved, does not appear to provide ideal foraging habitat for most of the fishes examined (Figure 2.10). In particular, the amount of food found in common carp stomachs was a very small fraction of their daily ration.

The management used in either unit of Swan Lake has not fully restored this backwater. Although some of the species indicate that the middle unit provide for better foraging habitat, it is important to note these results are limited to only a few of the fish species. Of the five species we studied, only common carp and crappie showed difference between the units in total calories consumed and the ingestion of a more diverse prey base; bluegill and freshwater drum did not and orangespotted sunfish showed marginal benefits in the middle unit where they ingested a more diverse prey community. The measures of overall habitat (i.e. sediment hardness and vegetation growth) indicate that the conditions in the middle unit still need to improve while the lower unit needs substantial improvements. Establishment of seasonal growth of emergent vegetation is accomplished through the drawdowns but submersed vegetation has not been re-established (Dolan and Chick 2005). The drawdowns are effective at sediment consolidation on the mudflats (areas where water can be drawn off easily), but central portions of backwater lake basins remain covered with water year round and act as a sediment sink (Dolan and Chick 2005). These areas could cause increases in turbidity and possibly would not provide firm rooting for vegetation. Furthermore, establishment of submersed aquatic vegetation may require planting and protection from herbivores.

Continuing the management practices in the middle unit likely would further improve conditions, and continue to harden the sediments. Hardening of the sediments is the key to improving conditions because it would likely decrease turbidity and allow for the possible establishment of submersed aquatic vegetation. Submersed vegetation, unlike emergent vegetation can withstand both the low and high water periods, and vegetation is a key to increasing prey diversity within the lake. Furthermore, it is possible that the fishes utilizing Swan Lake, as well as the prey within the lake, simply need time to mature, to adapt to improved sediment and vegetation conditions before large differences are seen between the units and improvement were spread to more fishes.

The single drawdown within the lower unit in 2002 did not provide improved foraging habitat for fishes. Using the middle unit as a guide, however, foraging conditions could be improved by changing the management practices within the unit.

The lower unit, unlike the middle unit, is managed to maintain the riverine connection and to conduct periodic drawdowns. The middle unit has shown that annual partial drawdowns likely are required to maintain consolidated sediments. Furthermore, from a foraging perspective, it would appear that implementing an annual drawdown regimen would do nothing but benefit the fishes feeding within the unit as demonstrated by the effects of the drawdown we see in the middle unit. Successful backwater lake management balances the need for drawdowns with the need for river connectivity to best fulfill the population and community needs of fishes within the river.

While this study lacked replication of management practices, and inferences on the effect of management practices must be made with caution, it does have the benefit of assessing the effect of management on a whole lake scale. Conducting a whole lake study allowed determination of differences between the units as a whole, thus accounting for all unit-wide ecological effects on the diets of fishes. Furthermore, the INHS, as part of the Swan Lake HREP effects study, has obtained sufficient amounts of data suggesting that the middle unit and the lower unit were similar prior to the implementation of the management by the USFWS (Theiling et al. 1991). This pre-project information strengthens the inferences that the improvement of habitat is related to the management practices, rather than inherent differences between the units.

This study was successful at obtaining instantaneous data detecting differences in the foraging habitat between these two units. I demonstrated that management for riverine fishes is not necessarily as simple as allowing for connectivity throughout the entire season, but requires knowledge of historical backwater lake habitats and fish habitat usage. Furthermore, I showed that management for riverine fishes does not have to be exclusive of management for waterfowl. In the middle unit, the emergent vegetation is vitally important for migrating waterfowl and also to riverine fishes as it has diversified the diet and provided for increased caloric consumption for some fishes. Finally, the management used in the middle unit follows a historical pattern of water stage: being inundated in the spring and fall with a summer low water period in between these high water levels.

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TABLES AND FIGURES

Table 2.1: Summary of all target fish caught during the duration of the experiment. The number on the left of the species column indicates the total number caught for the group (Total Number) and the number on the right indicates the percentage of empty stomachs (% Empty). *nc* indicates that a particular species was not caught during that period. LU = lower unit, MU = middle unit, and Σ = Total across both units

		Bluegill		**Common Carp**		**Crappie**		**Freshwater Drum**		**Orangespotted Sunfish**		
		Total No.	**% Empty**	**Total No.**	**% Empty**	**Total No.**	**% Empty**	**Total No.**	**% Empty**	**Total No.**	**% Empty**	**Total**
Total	LU	41	0	130	65	81	2	25	32	48	15	325
	MU	42	36	135	36	77	3	36	25	58	5	348
	Σ	83	--	265	--	158	--	61	--	106	--	673
Summer	LU	41	0	64	75	24	0	25	32	48	15	202
	MU	42	36	75	25	26	0	36	25	58	5.2	237
	Σ	83	--	139	--	50	--	61	--	106	--	439
Winter	LU	*nc*	*nc*	66	54	57	4	*nc*	*nc*	*nc*	*nc*	123
	MU	*nc*	*nc*	60	48	51	4	*nc*	*nc*	*nc*	*nc*	111
	Σ	*nc*	*nc*	126	--	108	--	*nc*	*nc*	*nc*	*nc*	234

Table 2.2: ANOSIM results testing the differences between lake units, overall between fish groups and between individual fish groups for both the summer and winter.

Season	Comparison	R Statistic	P-Value
Summer	Lower vs. Middle	0.06	0.004
	Overall Species	0.33	<.001
	Common Carp vs. Crappie	0.61	<.001
	Common Carp vs. Bluegill	0.41	<.001
	Common Carp vs. Orangespotted Sunfish	0.39	<.001
	Bluegill vs. Freshwater Drum	0.38	<.001
	Crappie vs. Bluegill	0.36	<.001
	Common Carp vs. Freshwater Drum	0.30	<.001
	Freshwater Drum vs. Orangespotted Sunfish	0.29	<.001
	Crappie vs. Orangespotted Sunfish	0.28	<.001
	Orangespotted Sunfish vs. Bluegill	0.20	<.001
	Crappie vs. Freshwater Drum	0.18	<.001
Winter	Lower vs. Middle	0.14	<.001
	Overall Species	0.31	<.001
	Common Carp vs. Crappie	0.31	<.001

Table 2.3: Results from SIMPER analysis of the prey items contributing most to the differences in diets of fish caught in the lower unit and fish caught in the middle unit. LU AVG. ABUNDANCE = average relative abundance of prey item in lower unit fish, MU AVG. ABUNDANCE = average relative abundance of prey item in fish caught in the middle unit, CONTRIBUTION PERCENT = percent at which the prey item contributes to the difference between the units, Total = total contribution of all prey items to the fish diets.

		LU Avg. Abundance	**MU Avg. Abundance**	**Contribution Percent**
Summer	Chironomids	0.27	0.34	26
	Corixids	0.31	0.18	23
	Fish	0.15	0.12	14
	Zooplankton	0.09	0.09	10
	Other	0.10	0.06	9.2
	Plants	0.01	0.10	6.4
	Dipterans	0.03	0.03	3.7
	Total	0.96	0.92	92
Winter	Chironomids	0.37	0.39	28
	Zooplankton	0.18	0.07	13
	Fish	0.03	0.19	13
	Corixids	0.11	0.10	12
	Crayfish	0.14	0.00	8.7
	Crustaceans	0.00	0.08	5.1
	Plants	0.03	0.05	4.8
	Shrimp	0.03	0.02	3.3
	FN Clams	0.04	0.01	3.1
	Total	0.93	0.91	91

Table 2.4: Summary of the average caloric value of the dominant prey items ingested by species and within management unit. *nc* indicates that a particular species was not caught during that period. FWDM = freshwater drum, OSSF = orangespotted sunfish, LU = lower unit, MU = middle unit and -- indicates that the prey item was not found in the diet of that fish.

		Bluegill		**Common Carp**		**Crappie**		**Freshwater Drum**		**Orangespotted Sunfish**	
	Prey	**LU**	**MU**	**LU**	**MU**	**LU**	**MU**	**LU**	**MU**	**LU**	**MU**
Summer	**Chironomids**	20	463	32	49	84	3	149	232	3	6
	Corixidae	160	11	0.66	1	11	0.10	88	12	1	13
	Fish	--	--	--	--	366	173	959	536	3	2
	Zooplankton	0.04	0.12	--	0.01	0.54	0.21	--	--	0.50	0.17
	Plants	--	--	0.03	39	0.3	0.39	0.01	--	--	--
	Dipterans	--	0.40	--	--	14	30	11	17	--	--
Winter	**Chironomids**	*nc*	*nc*	173	609	24	14	*nc*	*nc*	*nc*	*nc*
	Zooplankton	*nc*	*nc*	0.01	--	1	0.59	*nc*	*nc*	*nc*	*nc*
	Fish	*nc*	*nc*	--	--	7	203	*nc*	*nc*	*nc*	*nc*
	Corixidae	*nc*	*nc*	0.75	4	2	27	*nc*	*nc*	*nc*	*nc*
	Crayfish	*nc*	*nc*	--	--	39	--	*nc*	*nc*	*nc*	*nc*
	Plants	*nc*	*nc*	2	16	0.02	--	*nc*	*nc*	*nc*	*nc*
	Shrimp	*nc*	*nc*	--	3	0.79	0.19	*nc*	*nc*	*nc*	*nc*
	FN Clams	*nc*	*nc*	3	0.14	--	--	*nc*	*nc*	*nc*	*nc*

Table 2.5: ANOVA results testing for differences in number of different taxa ingested between the two units, fish groups, the interactions and contrast statements testing for differences in number of different taxa ingested between the units for each species group.

SEASON	**R^2**	**SOURCE**	**DF**	**TYPE III SS**	**MSE**	**F-VALUE**	**Pr>F**	
Summer	0.262334	Unit	1	12.85	12.85	8.00	0.005	**
		Fish Group	4	164.98	164.98	25.66	<.001	**
		Unit x Fish Group	4	52.89	52.89	8.23	<.001	**
		Bluegill	1	1.00	1.00	0.62	0.431	
		Common Carp	1	78.91	78.91	49.10	<.001	**
		Crappie	1	0.03	0.03	0.02	0.901	
		Freshwater Drum	1	0.03	0.03	0.02	0.901	
		Orangespot Sunfish	1	7.31	7.31	4.55	0.034	**
Winter	0.538273	Unit	1	44.45	44.45	25.78	<.001	**
		Fish Group	1	394.60	394.60	228.89	<.001	**
		Unit x Fish Group	1	38.07	38.07	22.08	<.001	**
		Common Carp	1	0.13	0.13	0.08	0.781	
		Crappie	1	76.63	76.63	44.45	<.001	**

** = Significant at a *P*<0.05

Table 2.6: ANCOVA results testing for differences in total calories consumed between the two units, fish groups, the interactions, the co-variate weight and contrast statements testing for differences in total calories consumed between the units for each species group.

SEASON	R^2	SOURCE	DF	TYPE III SS	MSE	F-VALUE	Pr>F	
Summer	0.279	Unit	1	0.62	0.62	0.70	0.405	
		Fish Group	4	108.42	108.42	30.52	<.001	**
		Unit x Fish Group	4	29.02	29.02	8.17	<.001	**
		Weight	1	3.11	3.11	3.50	0.062	
		Bluegill	1	0.20	0.20	0.23	0.633	
		Common Carp	1	28.81	28.81	32.43	<.001	**
		Crappie	1	5.48	5.48	6.17	0.013	**
		Freshwater Drum	1	0.01	0.01	0.01	0.930	
		Orangespot Sunfish	1	2.20	2.20	2.47	0.117	
Winter	0.144	Unit	1	26.13	26.13	22.77	<.001	**
		Fish Group	1	14.56	14.56	12.68	<.001	**
		Unit x Fish Group	1	0.22	0.22	0.19	0.662	
		Weight	1	8.66	8.66	7.54	0.007	**
		Common Carp	1	16.19	16.19	14.10	<.001	**
		Crappie	1	9.64	9.64	8.40	0.004	**

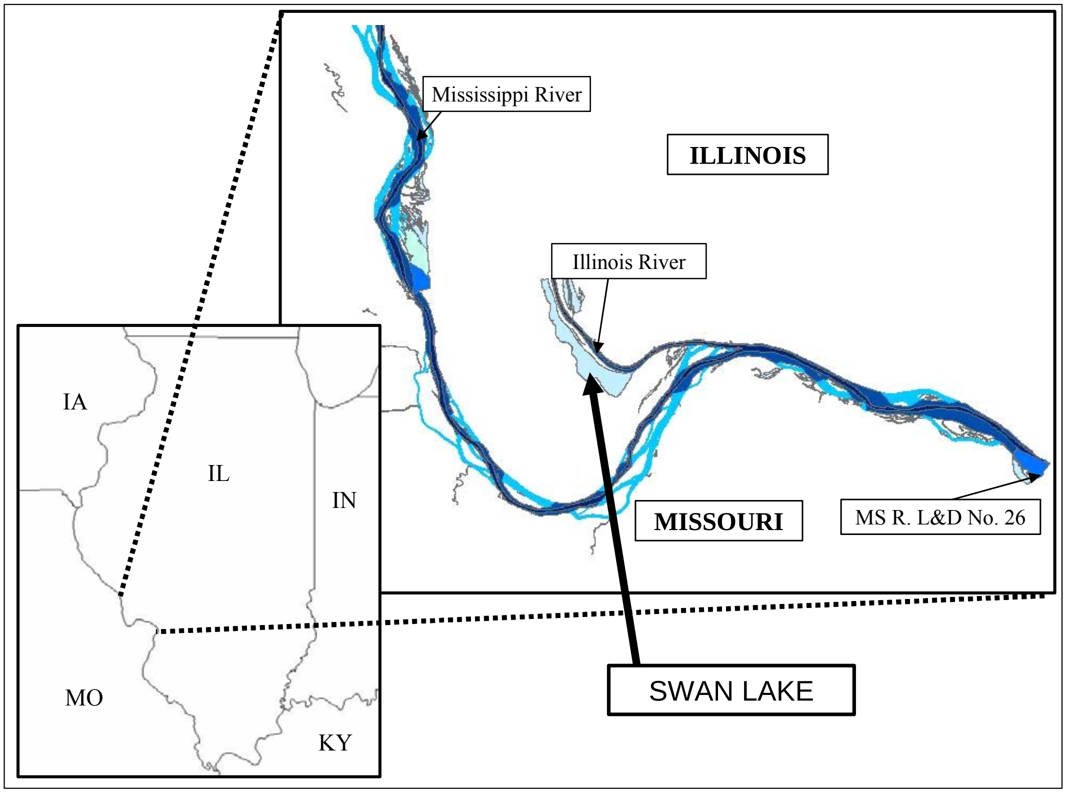

Figure 2.1: Location of Swan Lake, within the state of Illinois and within Pool 26 (inset).

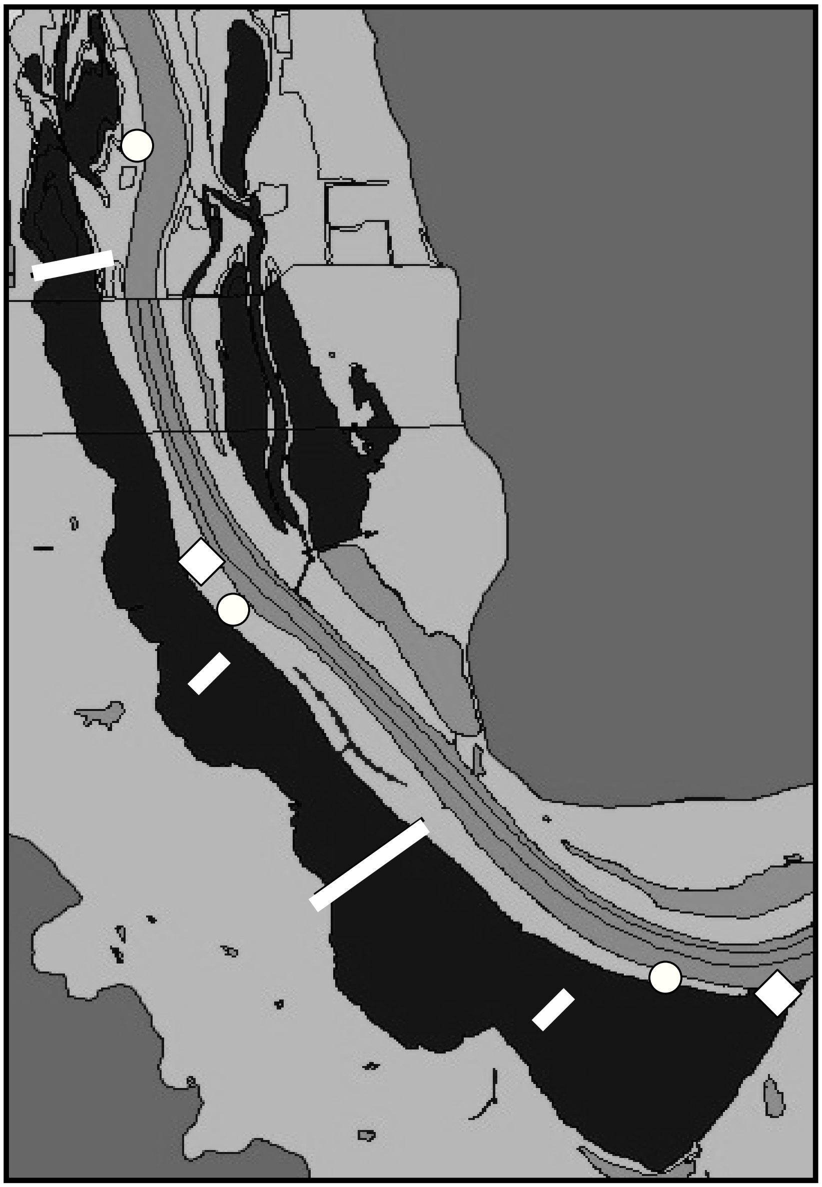

Figure 2.2: Improvement made to Swan Lake in the 2001 habitat rehabilitation and enhancement project. The large white lines represent levees that were installed to compartmentalize the lake, the small white lines are island chains, the circles indicate pumping structures, and the diamonds are stop log structures.

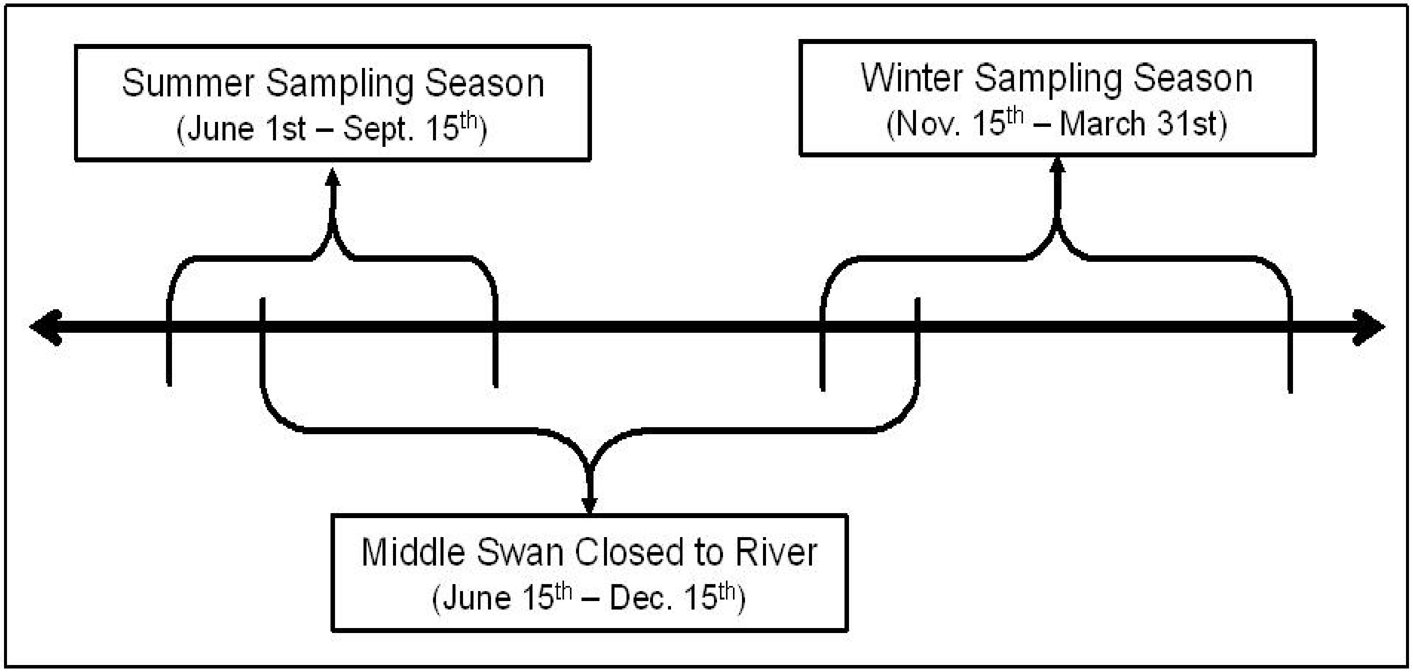

Figure 2.3: Timeline of the middle unit drawdown and sampling periods in the summer of 2005.

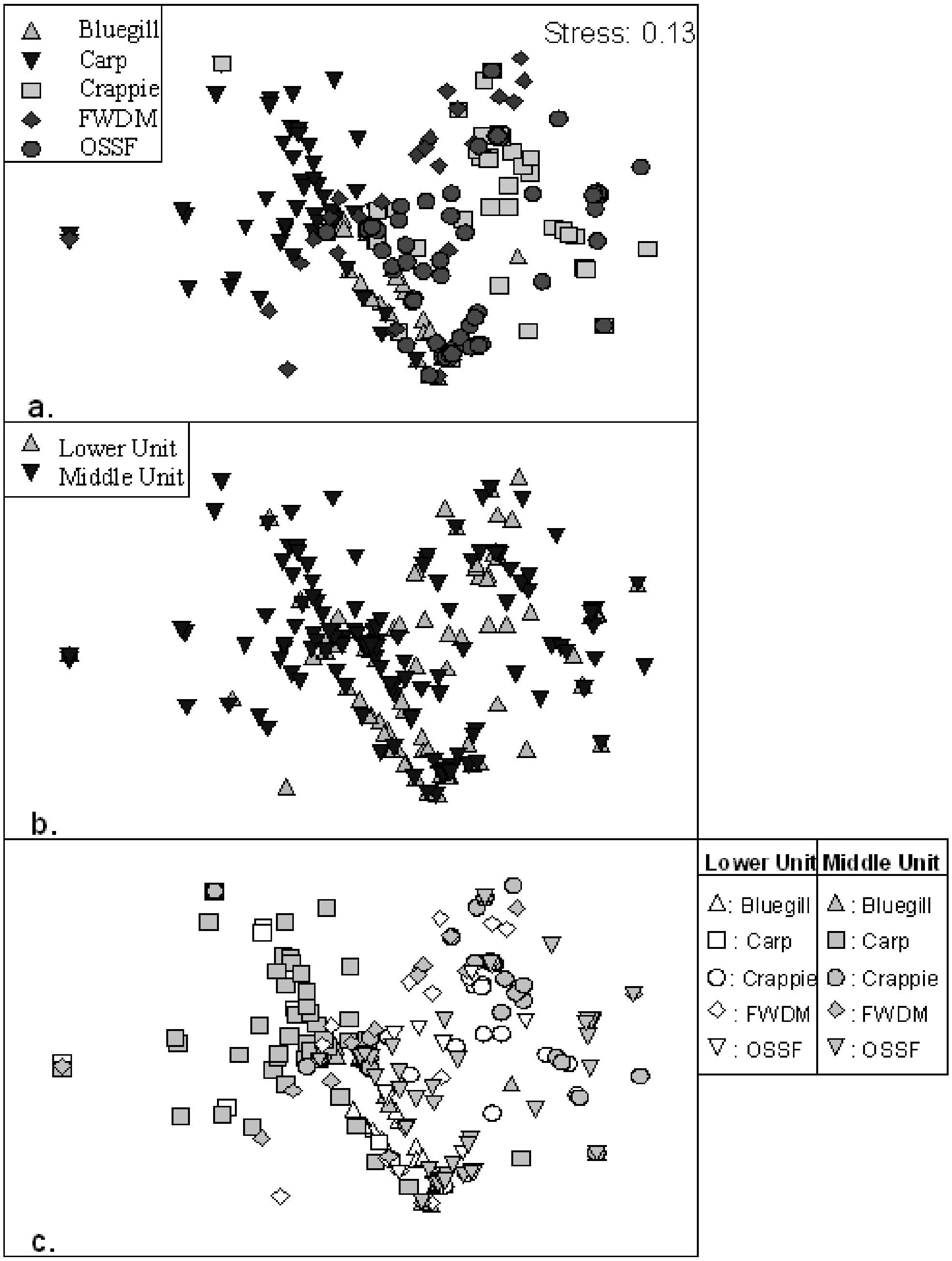

Figure 2.4: Non-metric multidimensional scaling plot for diet composition of a. bluegill, common carp, crappie, freshwater drum and orangespotted sunfish; b. lower unit and the middle unit; and c. interaction between fish group and unit for fish caught in Swan Lake in the summer of 2005. Symbols that are close together have greater similarity in diet than symbols that are further apart. Stress = 0.13.

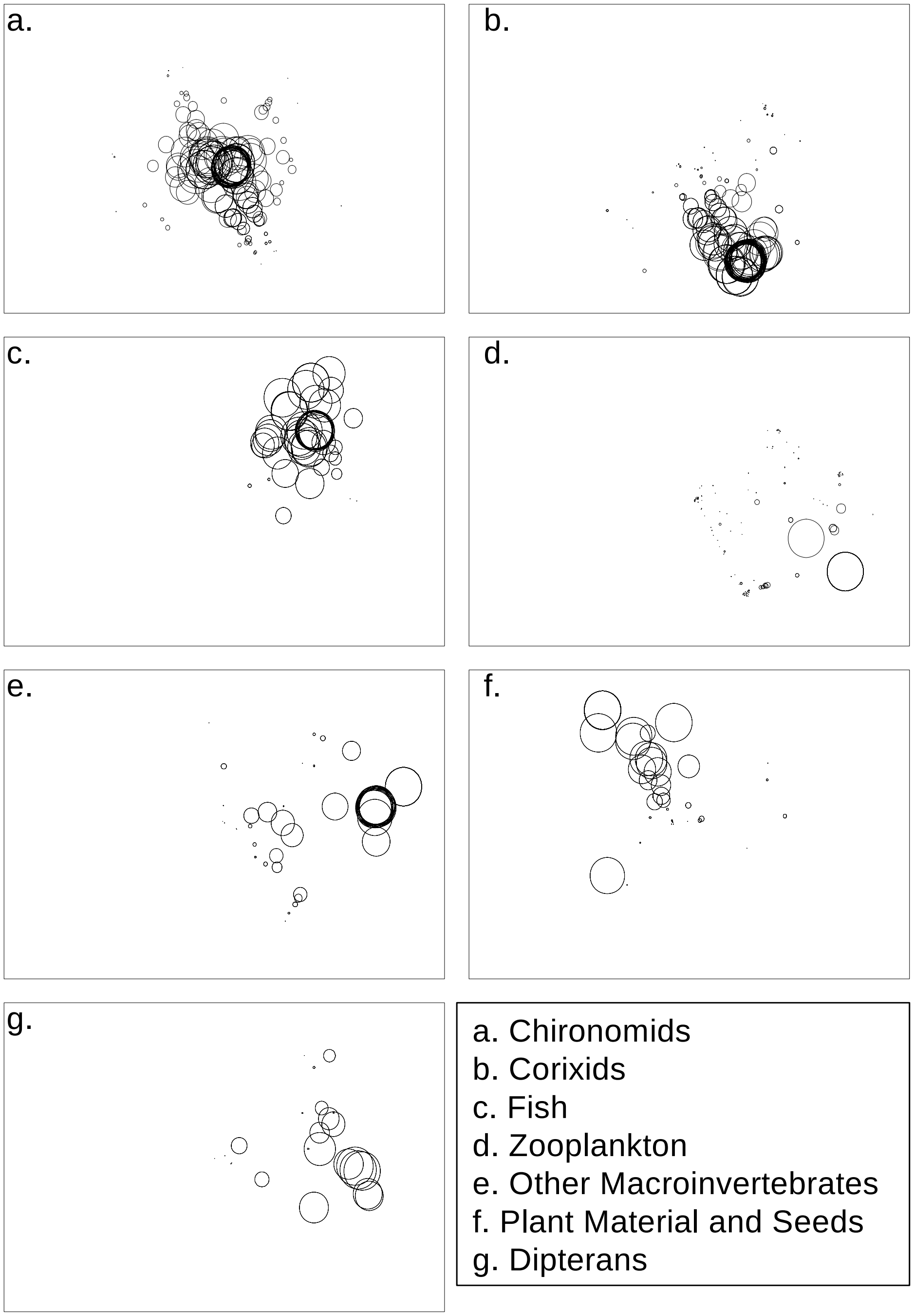

Figure 2.5: Bubble plots superimposed from the NMDS analysis of diet (Figure 2.3) representing the abundance of a. chironomids, b. corixids, c. fish, d. zooplankton, e. other macroinvertebrates, f. plant material and seeds and g. dipterans as prey items in the diet of fish caught in Swan Lake in the summer of 2005. Larger bubbles indicate a greater proportion of the prey item in the diet. Stress = 0.13.

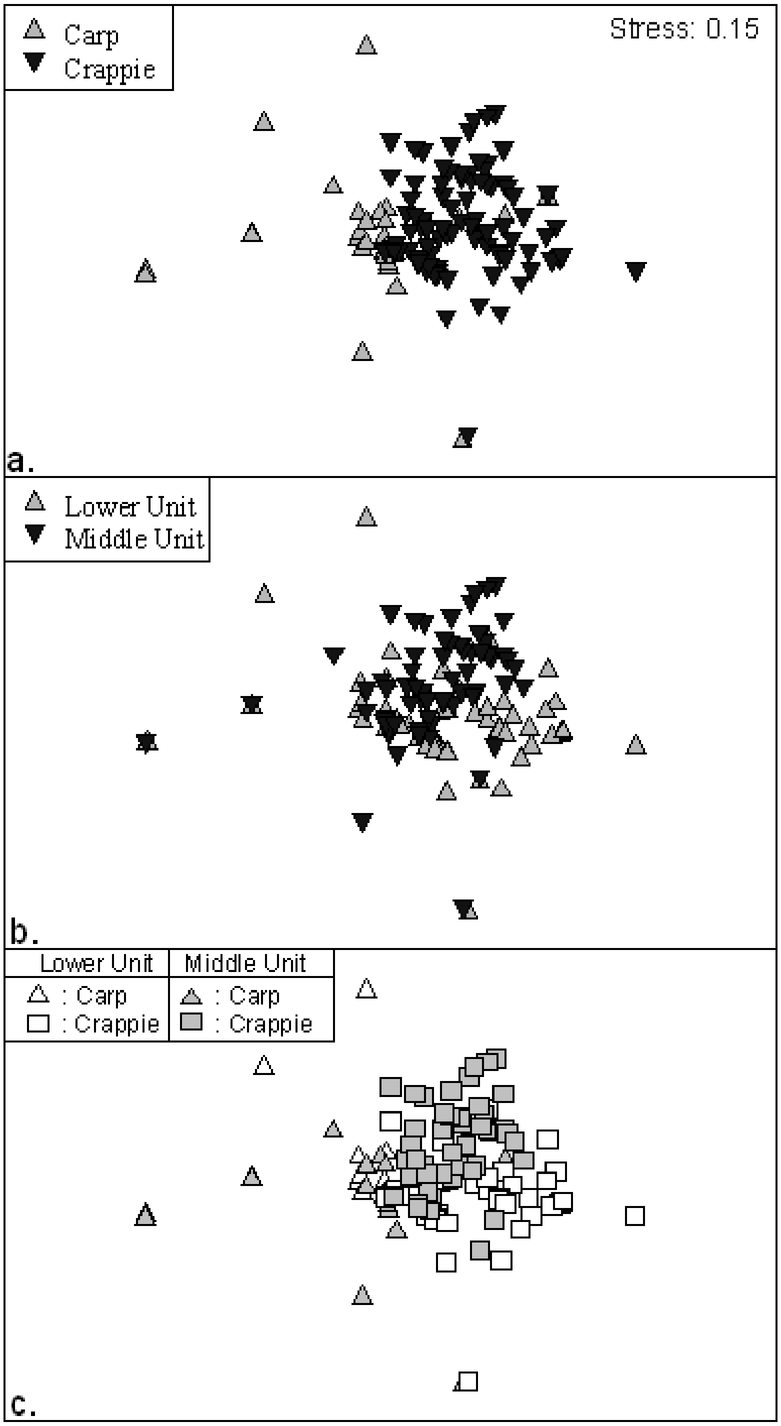

Figure 2.6: Non-metric multidimensional scaling plot for diet composition of a. common carp and crappie; b. lower unit and the middle unit; and c. interaction between fish group and unit for fish caught in Swan Lake in the winter of 2005. Symbols that are close together have greater similarity in diet than symbols that are further apart. Stress = 0.15

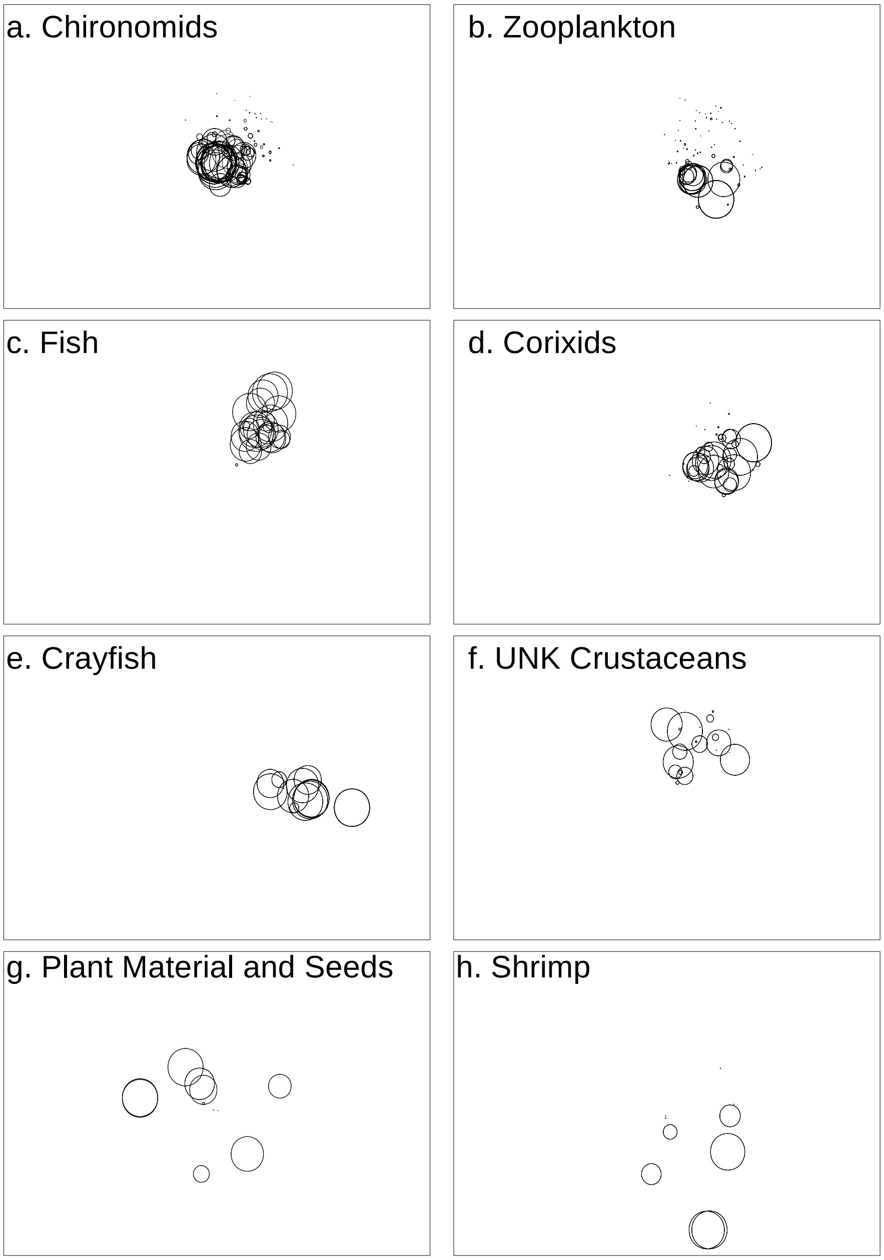

Figure 2.7: Bubble plots superimposed from the NMDS analysis of diet (Figure 2.5) representing the abundance of a. chironomids, b. zooplankton, c. fish, d. corixids, e. crayfish, f. unknown crustaceans, g. plant material and seeds and h. shrimp as prey items in the diet of fish caught in Swan Lake in the winter of 2005. Larger bubbles indicate a greater proportion of the prey item in the diet. Stress = 0.15

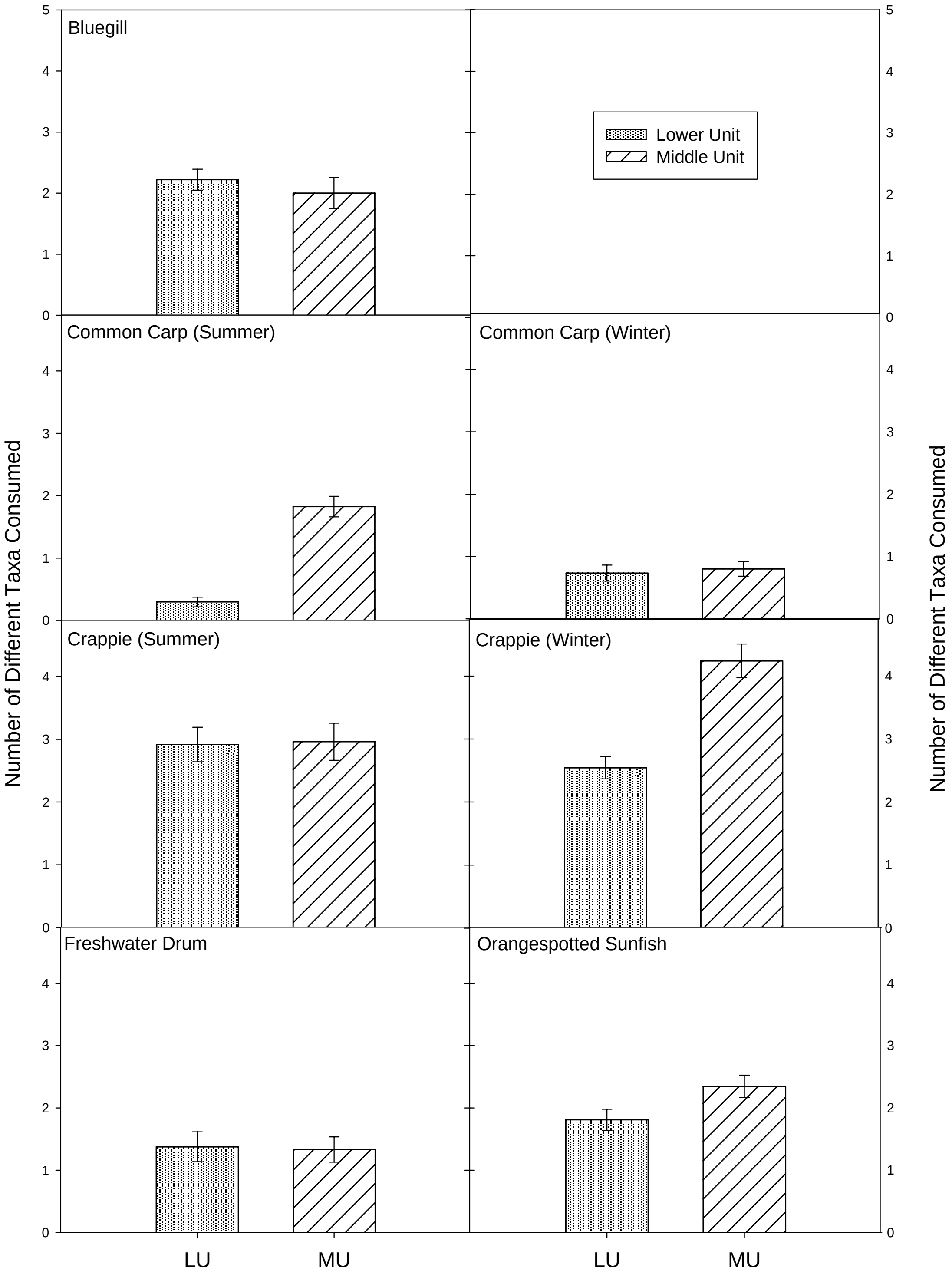

Figure 2.8: Mean of the number of different taxa consumed for each fish species in the lower (LU) and middle (MU) units. Error bars represent the standard error of the mean.

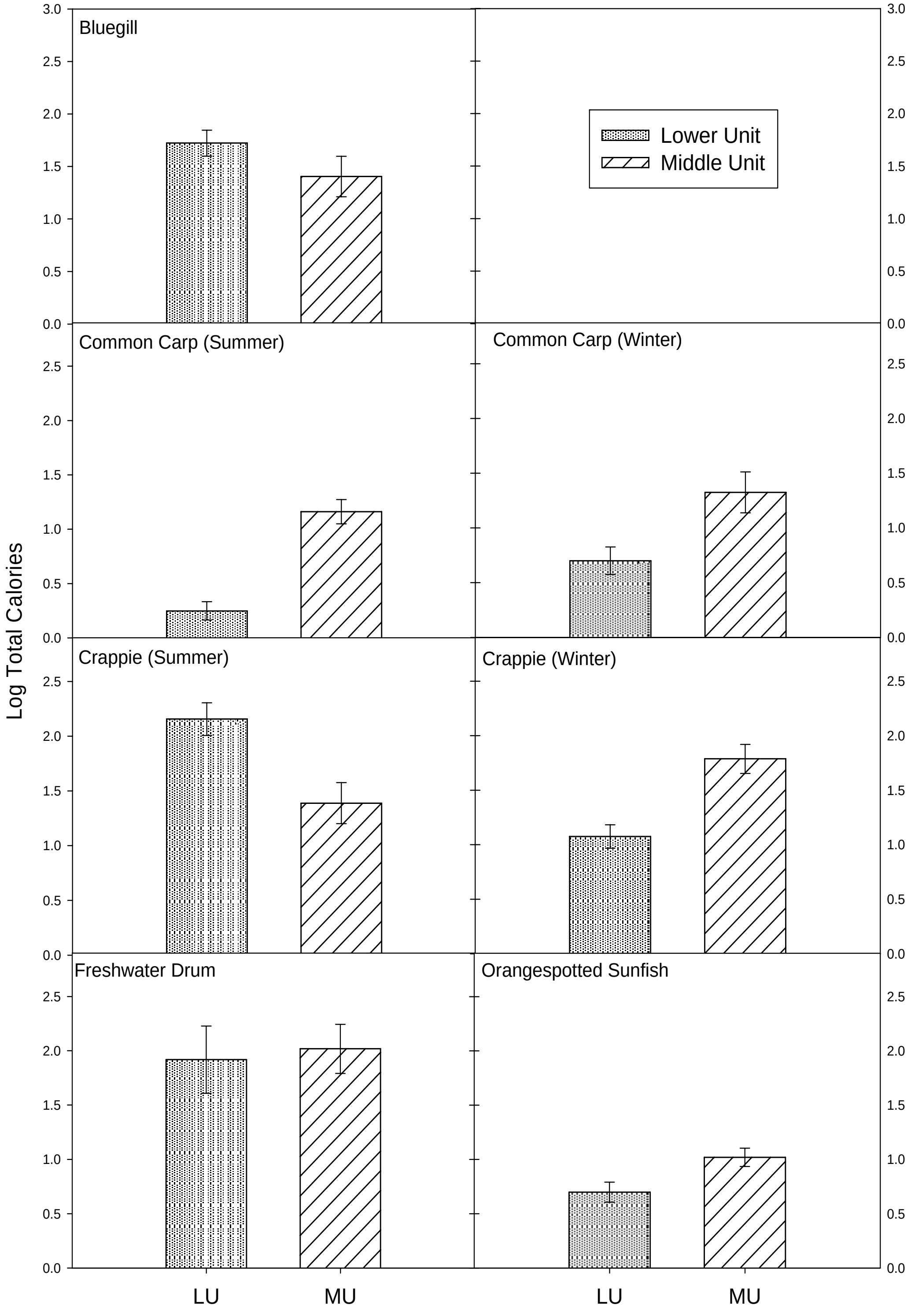

Figure 2.9: Mean of the total calories (log transformed) consumed for each fish species in the lower (LU) and middle (MU) units for each species. Error bars represent the standard error of the mean.

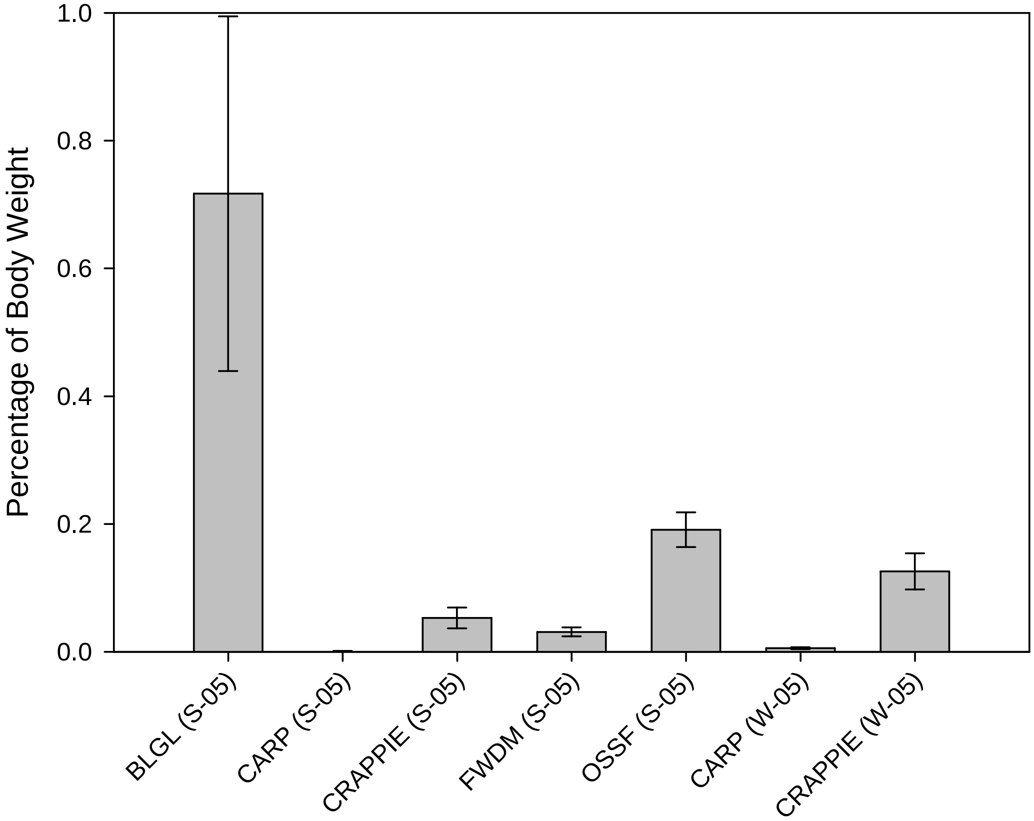

Figure 2.10: Conservative estimation of daily ration as the average percent of body weight consumed for each fish species within the middle unit. Error bars represent the standard error of the mean, BLGL = bluegill, CARP = common carp, FWDM = freshwater drum and OSSF = orangespotted sunfish.

CHAPTER 3: THE RELATIONSHIP BETWEEN THE FORAGING SUCCESS OF A BENTHIVOROUS FISH AND THE VERTICAL DISTRIBUTION OF MACROINVERTEBRATES IN A BACKWATER LAKE OF THE ILLINOIS RIVER

SUMMARY

Predator, prey, and structural complexity studies in freshwater systems typically focus on vegetation as the key structural component. However, there are many other elements that could offer structural complexity. This study examines the potential role that sediments play in offering refuge for macroinvertebrates from benthivorous fishes in Swan Lake. As part of a Habitat Rehabilitation and Enhancement Project, the lake was compartmentalized allowing for different management strategies to be implemented in two lake units. This has caused a divergence in the sediment hardness between these two units: the middle unit is on average harder than the lower unit. For this study I determined if there were differences in the diet of common carp between the middle and lower units and if any differences could be explained by examination of the macroinvertebrate community. Common carp in the middle unit fed more often and consumed more calories than common carp in the lower unit. This difference was severe, 75% of the common carp in the lower unit had no food in their stomachs, whereas 25% of the fish in the middle unit had no food in their stomachs. When this comparison was limited only to those common carp that fed, there was an equivalent amount of calories consumed. This suggested a patchy prey source in the lower unit; a few common carp in the lower unit were able to find suitable foraging habitat and feed well, whereas most common carp apparently did not locate suitable foraging habitat. PONAR samples demonstrated that there were little differences in biomass of

macroinvertebrates between the units, but core sampling showed that there were differences in the amount of prey that was available for fish to feed upon. In the lower unit prey was inaccessible to fishes because they were in the deeper parts of the sediment (i.e., from 10 to 80 cm deep); whereas a greater proportion of macroinvertebrates were available in the upper 10 cm of sediments in the middle unit where fishes could consume them. Differences in the depth distribution of macroinvertebrates appear to be a function of sediment hardness, with a deep refuge for macroinvertebrates only available in soft sediment sites.

INTRODUCTION

The relationships between predators, prey, and habitat structural complexity have been extensively studied in ecology (Gotceitas and Colgan 1990, Nelson 1979, Heck and Thomas 1981). Many fishes alter their behavior in response to predation risk. For example, in the presence of largemouth bass (*Micropterus salmoides*) bluegill (*Lepomis macrochirus*) either seek refuge in vegetative zones (Werner et al. 1983) or school if no vegetation is present (Savino and Stein 1982). Werner et al. (1983) showed that bluegill generally inhabit the more energetically profitable open pelagic zones of the lakes when largemouth bass are not present. High levels of structural complexity cause a decrease in predator capture efficiency, whereas predators are capable of consuming high numbers of prey when structural complexity is low (Savino and Stein 1982). This finding led researchers to suggest that intermediate amounts of structural complexity would yield the greatest amount of predator growth and reproduction, because predators would have moderate levels of capture success but would be unlikely to over-graze their prey resources (Cooper and Crowder 1982, Savino and Stein 1982, Gotceitas and Colgan 1990, Heck and Thomas 1981).

While most studies have focused on the role of vegetation in providing structural complexity, many other habitat elements can play a role (Heck and Crowder 1991). Woody debris, coral reefs and rocky intertidal spaces have all been shown to provide refuge from predators (Sass et al. 2006, Coull and Wells 1983, Angfermeier and Karr 1984). However, one lesser studied element is the role of sediments. In stream systems, macroinvertebrates may use the hyporheic zone as a refuge during low water or dry periods (Williams and Hynes 1977, Griffith and Perry 1993). However, relatively few studies have assessed whether sediments offer refuge from predation in a similar manner to vegetation and other structurally complex habitats. Two exceptions to this are studies done by Peterson (1982) on clam predation by whelks (*Busycon* spp.) in marine systems and an experimental study done by Persson and Svensson (2006) on the vertical distribution of benthic macroinvertebrates in response to fish predation in freshwater ponds in southern Sweden. Peterson (1982) found that the density of the shallow burrowing clam (*Chione cancellata*) declined in response to whelk predation; whereas the density of the slightly deeper burrowing clam (*Mercenaria mercenaria*) was unchanged. This suggested that either the higher density of plant roots found at deeper depths, or the depth itself provided a refuge to the deeper burrowing clam. The study by Persson and Svensson (2006) illustrates that some macroinvertebrate groups co-exist with benthic predators by surviving in the deeper parts of the sediments and that benthic predation affected large and immobile prey.

In the fall of 2000 a Habitat Rehabilitation and Enhancement Project (HREP) was completed on Swan Lake, a large backwater lake of the Illinois River, which had goals of reducing sedimentation and increasing the growth of vegetation (USACE 1993, Theiling et al. 2000). The HREP compartmentalized the lake into units which allowed for the implementation of management practices in one unit, independent of the others. The differing management practices used in the middle and lower units has caused a divergence in the ecological characteristics of the units (Dolan and Chick 2005). A previous study suggested that the middle unit provided better foraging habitat than the lower unit (Chapter 2). While there were significant differences in foraging habitat between the units observed for a few of the species studied, the most consistent results were from the examination of common carp diets. Common carp (*Cyprinus carpio*) fed more often in the middle unit and appeared to consume more total calories (Chapter 2). Furthermore, the different management practices have caused changes in the sediment conditions. The middle unit, on average, had an average sediment flocculency of 23.4 (± 1.4 SE) whereas the lower unit had an average reading of 50.0 (± 1.3 SE; Dolan and Chick 2005).

I examined the role that flocculent sediments play in the vertical distribution and availability of benthic prey items within Swan Lake. Due to the differences in sediment conditions it will be possible to determine if softer sediments allow for a deep refuge for macroinvertebrates. I hypothesize that in hard sediment sites there will not be a deep refuge and the majority of the macroinvertebrate biomass will be in the upper portions of the sediments. While in soft sediment sites there will be a deep refuge and the majority of the macroinvertebrate biomass will be in the lower or deeper portions of the sediment. The middle unit, which has an abundance of hard sediment sites, will offer more sites where macroinvertebrates will be available; whereas the lower unit, where the majority of the sediment is soft, will offer less sites where the macroinvertebrates are available. To test this hypothesis, I first will examine the diets of carp in the middle and lower units. Common carp was used as the representative benthivore because it abundant in Swan Lake allowing me to obtain a sample size large enough to detect differences between the units. PONAR sampling was conducted in the same season as the stomachs were collected so I was able to determine if any differences we observed within the common carp diets were the result of macroinvertebrate biomass difference between the units. Finally, I also used core sampling to determine if the macroinvertebrates were using deep sediments as a refuge from benthic predation.

METHODS

Study Site

Swan Lake is a 1175 hectare backwater lake of the Illinois River that extends from Illinois River mile 5 to 13 with an average depth of 64 cm (Chapter 2). Although Swan Lake once supported an abundance of aquatic vegetation and diverse communities of fishes and wildlife, sedimentation caused increases in unconsolidated sediments and turbidity, causing declines in aquatic vegetation. This habitat degradation prompted the implementation of an HREP in 1995 (Theiling et al. 2000). The HREP created levees that separated the lake into three management units and allowed for different management strategies to be implemented in different units

(Chapter 2). The different management strategies have created different sediment conditions between the units: the lower unit is softer and contains fewer areas with hard sediment compared to the middle unit which is harder and contains fewer soft sediment areas (Dolan and Chick 2005).

Fish Collection and Diet Analysis:

To assess ecological benefits caused by the HREP, staff at the Illinois Natural History Survey (INHS) conducted pre- and post-project monitoring of macrophytes, macroinvertebrates, fishes and water quality within the lake. Using techniques outlined by Gutrenter et al. (1995), fishes were collected using tandem fyke nets, tandem mini-fyke nets and trammel nets. Common carp were caught during the summer of 2005 (June 1st – September 30th). Procedures for foregut collection and diet determination are detailed in Chapter 2. Briefly, we removed the foregut from the fish and preserved them in 10% formalin solution. These foreguts were then rinsed and dissected in the laboratory. All prey items were identified to the lowest taxonomic level possible and were weighed to obtain a measure of mass which was converted to calories according to Cummins and Wuycheck 1971.

Macroinvertebrate Collection and Analysis:

Macroinvertebrates were sampled using two methods. The first method sampled benthic macroinvertebrates from 20 sites from the lower and middle units of Swan Lake by use of a PONAR grab (524 cm^2) from May to June of 2005. Sites were randomly chosen using a random number generator and a grid system overlain a map of Swan Lake. Samples were collected and rinsed through a 0.5 mm sieve and preserved in 10% buffered formalin (Dolan and Chick 2005). Macroinvertebrates were identified to the lowest taxonomic level possible and measured for length. The length measurements were converted into biomass by use of regressions published by Benke et al. (1999) and Nalepa and Quigley (1980).

For the second method, macroinvertebrates were randomly sampled from 11 hard and 11 soft sediment sites in both the lower and middle units, using a customized core sampler (Figure 3.1) in March of 2006. This core sampler was a 1 meter long clear acrylic tube fitted with a removable one-way valve on top which created suction and held the flocculent sediments in the core as it was extracted. I designed this sampler with the largest inner diameter practical (10.16 cm, sampling area = 105.68 cm^2) to maximize the number of macroinvertebrate taxa collected with each sample. Sediment hardness was previously measured in Swan Lake by use of a penetrometer which measures how far five pounds would sink a pole into the sediments (Dolan and Chick 2005). Sites were classified as "hard" if the penetrometer had a reading of 0 – 30 cm and "soft" if the penetrometer reading was greater than 30 cm. A map of the hard and soft sediment sites in the Swan Lake was made using penetrometer data collected from randomly chosen sites and Spatial Analysis in Arcview-GIS 3.3 Samples were collected to a depth indicative of their sediment hardness: at each site a penetrometer reading was taken and the core was sunk approximately 5 – 10 cm beyond that measurement. Once the sample was collected, the one-way valve was removed and a plunger was used to extract the core dividing it into 5 cm sections. Each 5 cm section was filtered through a 0.425 mm sieve and all contents were rinsed into a jar, preserved in 10% buffered formalin for identification in the lab. Macroinvertebrates were identified and enumerated to the lowest taxonomic level possible. Biomass was then obtained for all taxa groups by drying at 70°C until a constant mass was obtained (about 24 hours).

Data Analysis:

I used Analysis of Variance (ANOVA) to test for differences in calories consumed by all common carp and, in a separate analysis, tested for differences by just those common carp that had food items in their stomachs. In these tests, the experimental unit was each individual fish whose diet was examined. The response variable used was calories, which was log transformed to make these data homoscedastic and normal. I also used ANOVA to determine differences in the biomass of invertebrates collected via PONAR sampling between units. In this test the experimental unit was each PONAR sample that was collected (20 for each of the units). Separate analyses were run for each of five major taxa, biomass for each taxa was log transformed to make these data homoscedastic and normal and a Bonferroni Correction was applied that reduced the critical alpha level from 0.05 to 0.01.

For the macroinvertebrates collected through core sampling, raw biomass was converted into relative biomass for each individual taxonomic group by dividing the raw biomass of the taxa within a layer by the total biomass for those taxa in the core. Relative biomass was used because the timing of the macroinvertebrate sampling did not correspond to the diet analysis. Therefore, relative abundance relative to depth was more appropriate for testing the hypothesis than actual biomass. Common carp are reported to feed to a depth of 7 cm (Chapman and Fernando 1994), so the differences in biomass in the upper 10 cm between the middle and lower unit were determined. An ANOVA was used to test for differences in the total biomass of all macroinvertebrates and multivariate analysis of variance (MANOVA) was used to test for shifts among taxonomic groups. The main effects tested in the ANOVA and MANOVA were: unit, sediment type, and the interaction between unit and sediment type. I used the Wilk's Lambda (λ) test statistic to determine significance for the MANOVA. To further examine the MANOVA results, separate ANOVAs were used to test for differences caused by the main effects of each specific taxonomic group and because five separate tests were run a Bonferroni Correction was applied reducing the critical alpha from 0.05 to 0.01. The relative biomass in the upper 10 cm for each taxonomic group was arcsine transformed to normalize the data and correct for heteroscedasticity.

RESULTS

Fish Diet Analysis

During the summer of 2005, common carp feeding in the middle unit appeared to have found suitable foraging habitat more often than those feeding in the lower unit. In the middle unit 75% of the common carp had food in their guts whereas in the lower unit only 25% did. The composition of the common carp diet focused on chironomids in both units and plant material and seeds in the lower unit (Chapter 2). There was a difference in total calories consumed between the units ($F_{1, 134} = 39.54$, $P < 0.001$, $R^2 = 0.23$) with fish feeding in the middle unit consuming more total calories than those feeding in the lower unit (Figure 3.2). When the total calories consumed were compared after eliminating common carp with empty stomachs, the ANOVA model was not significant ($F_{1, 68} = 3.56$, $P = 0.063$, $R^2 = 0.05$) but total calories consumed in the middle unit was only slightly greater than the lower unit (Figure 3.2). The fact that the few common carp able to find prey in the lower unit consumed similar amounts of calories as common carp in the middle units suggests an extremely patchy distribution of prey in the lower unit.

PONAR and Core Sampling

Both the PONAR sampling and the core sampling revealed that about 95% the benthic macroinvertebrate community biomass was made up of chironomids, leeches, oligochaetes and fingernail clams; all other macroinvertebrates were lumped into a group called “other macroinvertebrates”. Chironomids accounted for the majority of the biomass obtained by both sampling methods in both units (Figure 3.3; Figure 3.4) and core sampling only found leeches in the middle unit. (Figure 3.4). However, core sampling detected more taxa per sample than the PONAR sampling (Figure 3.5).

PONAR sampling done in the same season as the analysis of the common carp diet demonstrated no differences in biomass for the dominant macroinvertebrate taxa. Chironomids ($F_{1,37} = 0.02$, $P = 0.882$, $R^2 = 0.01$), leeches ($F_{1,37} = 3.50$, $P = 0.069$, $R^2 = 0.08$), oligochaetaes ($F_{1,37} = 4.95$, $P = 0.032$, $R^2 = 0.12$) and other macroinvertebrates ($F_{1,37} = 2.86$, $P = 0.099$, $R^2 = 0.07$) showed no statistical difference between the middle and lower units in terms of biomass. The exception to this was fingernail clams ($F_{1,37} = 12.26$, $P = 0.001$, $R^2 = 0.24$), which had a higher biomass in the lower unit than the middle unit (Figure 3.3).

Vertical Distribution of Macroinvertebrates

Core samples showed that the depth distribution of macroinvertebrates varied among taxonomic groups and between units, but all macroinvertebrates appear to use the deeper sediments. Chironomids, fingernail clams and other macroinvertebrates were found as deep as 80 cm in the sediment, leeches were found as deep as 70 cm and oligochaetaes were found 40 cm deep (Figure 3.6). The middle unit had more relative biomass in the upper 10 cm of the core for chironomids, leeches and other macroinvertebrates. For oligochaetaes and fingernail clams, however, the mean biomass in the upper portion of the middle and lower unit cores showed little difference (Figure 3.6). The lower unit generally had higher relative biomass in the deeper sediments than the middle unit. This was especially true for oligochaetaes, which were only found in the upper layer in the middle unit but in the lower unit were found much deeper (Figure 3.6).

The relative biomass of all macroinvertebrates in the upper 10 cm differed among unit and sediment type. The ANOVA model used to test for differences in unit and sediment type explained a significant portion of the variation ($F_{3,40} = 8.52$, $P = 0.001$, $R^2 = 0.39$). There was a higher proportion of macroinvertebrates in the upper 10 cm in the middle unit than the lower unit ($F_{3,40} = 7.57$, $P = 0.009$; Figure 3.7), in hard sediment sites than soft sediment sites ($F_{3,40} = 8.14$, $P = 0.007$; Figure 3.7), and the interaction unit by sediment was also significant ($F_{3,40} = 6.29$, $P = 0.016$; Figure 3.8). The biomass of macroinvertebrates in the upper 10 cm did not differ between hard and soft sediment sites of the lower unit, but in the middle unit there was more relative biomass in the upper 10 cm of hard sediment sites than soft (Figure 3.8).

The relative biomass of the five major taxonomic groups in the upper 10 cm differed between units and sediment types. Relative abundance of the five macroinvertebrate taxa varied among units (λ: $F_{5,36} = 8.93$, $P < 0.001$; Figure 3.9), and sediment type (λ: $F_{5,36} = 4.83$, $P = 0.002$; Figure 3.10), and there was a significant interaction between unit and sediment type (λ: $F_{5,36} = 4.95$, $P = 0.002$. For chironomids, leeches and other macroinvertebrates the ANOVA models used to interpret the MANOVA results explained a significant portion of the variation. Differences in biomass between units were mainly the result of leeches, which had more biomass in the upper 10 cm in the middle unit than the lower unit (Table 3.1; Figure 3.9). Similarly, differences in biomass between sediment types were the result of chironomids, leeches and other macroinvertebrates, all of which were more abundant in the upper portion of the core in hard sediment sites than soft sediment sites (Table 3.1; Figure 3.10). For both oligochaetes and fingernail clams, the ANOVA model did not explain a significant portion of the variation (Table 3.1). Chironomids and leeches had significant interaction terms (Table 3.1), likely accounting for the significant interaction term in the MANOVA. Chironomids had greater relative biomass in the upper 10 cm of middle unit hard sites than lower unit hard sites whereas differences among units for soft sites were less distinct (Figure 3.11). Leeches were not present in the upper 10 cm of sediment in the lower unit (Figure 3.11).

DISCUSSION

Common carp appeared to encounter better foraging habitat in the middle unit than in the lower unit. More common carp were observed with food in their stomachs in the middle unit than in the lower unit and those feeding in the middle unit consumed considerably more total calories than those feeding in the lower unit. One potential explanation for these differences would be reduced prey biomass in the lower unit. However, this explanation appears unlikely. PONAR samples revealed little statistical difference in macroinvertebrate abundance (although the means appear to be different) and composition between units. Additionally, if the lower unit simply had lower prey abundance, we probably would expect that common carp in the lower unit would have consumed less total calories, but not the dramatic difference in the number of empty stomachs. In contrast, common carp captured in the lower unit that had prey in their stomachs (i.e.: had successfully located prey) consumed similar amounts of calories as common carp in the middle unit. These patterns do not suggest a reduced prey base; rather, they suggest a lack of sites with prey available to feed on.

A potential explanation for the lack of prey availability in the lower unit was found when the vertical distribution of macroinvertebrate taxa was examined. The majority of the macroinvertebrate biomass in the lower unit was likely inaccessible to the feeding common carp because much of the prey base occurred at depths greater than 10 cm. In the middle unit the majority of the biomass was in the top 10 cm of sediments, and was therefore more likely to be available for benthivorous fishes (assuming they can only feed to a depth of 10 regardless of the sediment characteristics). Core sampling demonstrated that in hard sediment sites the majority of the biomass was in the upper portions of the core but in the soft sediment sites the majority of the biomass was in the lower portions of the core. This explains the trends observed: in the lower unit there is preponderance of soft sediment, whereas in the middle unit sediment sites are typically hard (Figure 3.12).

A classic theory in ecology is that without a refuge from predation, prey populations would go extinct (Gause 1934, Huffaker 1958). Studies in aquatic systems have demonstrated that fish predators foraging in habitats with low structural complexity can greatly reduce abundance of their prey (Cooper and Crowder 1979, Savino and Stein 1982). Due to the lack of submersed vegetation population, woody debris and other features that would add structural complexity in the lower unit, but the persistence of both fishes feeding and macroinvertebrate biomass there must be another element offering refuge. I hypothesize that flocculent sediments can offer refuge to macroinvertebrates from fish predators. The use of deep sediments as a refuge has been studied in marine systems (Blundon and Kennedy 1982, Virstein 1977, Wilson 1991), but understudied in freshwater systems as my study in only the second to address this issue (Persson and Svenson 2006).

The major unanswered question of this study is: what is the cause of the vertical distribution of the macroinvertebrates? One possibility is that it is the result of active predator avoidance by the macroinvertebrate. The movement of prey to avoid predation is well documented in fish (Turner and Mittelback 1990) and invertebrates (Stich and Lambert 1981, Zaret and Suffern 1976). However, Persson and Svensson (2006) concluded that the "…vertical response was the result of foraging behavior of the [benthivore] not the anti-predator behavior of the benthos." A second possibility is that the benthivorous fishes crop the biomass down in the upper layers of the sediment. However, this was unlikely because of the high percentage of common carp that were unable to feed (i.e.: empty stomachs) in the lower unit. Furthermore, there were relatively similar numbers of common carp captured in the middle and lower units (Dolan and Chick 2005), so we would expect to see similar vertical distributions of macroinvertebrates between the units if direct consumption was the cause of the pattern.

Another possibility is that there is no active movement and passive sinking in the extremely flocculent sediments results in the observed vertical distribution.

Core sampling effectively documented the vertical distribution of macroinvertebrates, which likely provides the mechanisms explaining the differences in common carp foraging success between the middle and lower units. While both PONAR sampling and core sampling were effective at determining overall community composition, the actual depth sampled by PONAR grabs is unknown. The PONAR is a large dredge-like device that samples the entire gradient regardless of depth (Thrush 1991, Lee 1996). At extremely flocculent sites, the loose sediment would push out of the mesh on top of the PONAR and the sampler would continue until the friction between the dredge and the sediment stopped its descent. This depth was unknown, so the sampler could have collected from only the top 10 cm or the upper 80 cm. The core sampler, on the other hand, was effective at sampling both the entire community and documenting the vertical distribution of macroinvertebrates within the core. This opens up a new area for exploring the ecology of benthic macroinvertebrates and associations with benthic feeding fishes.

The result of the vertical distribution of macroinvertebrates within the hard and soft sediments does not give exact causation to the differences in fish diets, simply a possible solution to the differences. I chose to report macroinvertebrate biomass obtained through cores sampling as relative biomass, rather than actual biomass, for two reasons. First, vertical distribution patterns were my main concern and relative biomass is a more appropriate way to compare these patterns between units and sediment types. Additionally, the timing of the core sampling did not overlap with the diet analysis.

Therefore, any differences in actual biomass among units or sediment type would not necessarily reflect the conditions at the time fish diet was assessed.

The results of this study show another possible detriment to sedimentation and sediment flocculency in backwater lakes. Excessively flocculent sediment causes a deep sediment refuge for burrowing macroinvertebrates. The management practice used in the middle unit of Swan Lake is aimed at sediment consolidation and these areas of hard sediments provide for a benthic prey source, thus the higher percentage of common carp feeding and the increased consumption of calories (Figure 3.11). The management of the lower unit focuses on river connection and was much less successful at consolidation of sediments, resulting in very few areas likely to be suitable for foraging by benthivorous fishes (Figure 3.11).

The results from this study have implications for foraging by both benthic fishes and waterfowl. There have been very few studies that measure the depth at which a benthivore will feed to in the sediments. However, if most other native benthivores (i.e.: smallmouth buffalo (*Ictiobus bubalus*), black buffalo (*I. niger*) and channel catfish (*Ictalusus punctatus*)) feed to a depth similar to common carp, then the majority of the prey will be inaccessible to them. Furthermore, the possibility exists that in backwater lakes affected by sedimentation, diving waterfowl, such as the lesser scaup (*Aythya affinis*), would be forced to feed benthically. If prey items are distributed deeper than the feeding range of these ducks, then a similar situation would present itsef to the waterfowl (R. Smith Personal Communication). This study has obvious broader implications to ecological issues such as predator, prey and structural complexity interactions and also to backwater lake degradation studies. To my knowledge, this is the second study to investigate the vertical distribution of macroinvertebrates in freshwater systems, and my results suggest this is an important area for further investigation.

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TABLES AND FIGURES

Table 3.1: ANOVA results from test of Unit, Sediment and Unit*Sediment (Unit*Sed). R2 = Overall goodness of fit for the taxa test, Model F-Value = Overall model F statistic, Model *P* Value = overall model *P* Value, Test = main effect tested, Type III Sums of Squares, Mean Square Error, F-Value and Pr>F all refer to individual main effect tested.

Prey Item	**R^2**	**Model F-Value**	**Model *P* Value**	**Test**	**Type III Sums of Square**	**Mean Square Error**	**F-Value**	**Pr>F**	
Chironomids	0.34	6.78	0.001	Unit	0.267	0.267	1.38	0.2472	
				Sediment	1.433	1.43	7.40	0.0096	**
				Unit * Sed	1.88	1.88	9.72	0.0034	**
Leeches	0.60	19.97	<.0001	Unit	7.08	7.081	37.22	<.0001	**
				Sediment	1.44	1.45	7.62	0.0087	**
				Unit * Sed	1.45	1.45	7.62	0.0087	**
Oligochaetes	0.04	0.62	0.60	Unit	0.328	0.327	0.61	0.4404	
				Sediment	0.021	0.021	0.04	0.8442	
				Unit * Sed	0.805	0.805	1.49	0.2292	
FN Clams	0.14	2.25	0.09	Unit	0.840	0.840	2.56	0.1173	
				Sediment	0.759	0.759	2.31	0.1361	
				Unit * Sed	0.060	0.060	0.18	0.6716	
Other MI	0.18	2.89	0.04	Unit	0.348	0.348	1.22	0.2762	
				Sediment	2.43	2.43	8.50	0.0058	**
				Unit * Sed	0.021	0.021	0.07	0.7867	

** = Significant with an alpha = 0.01 or less.

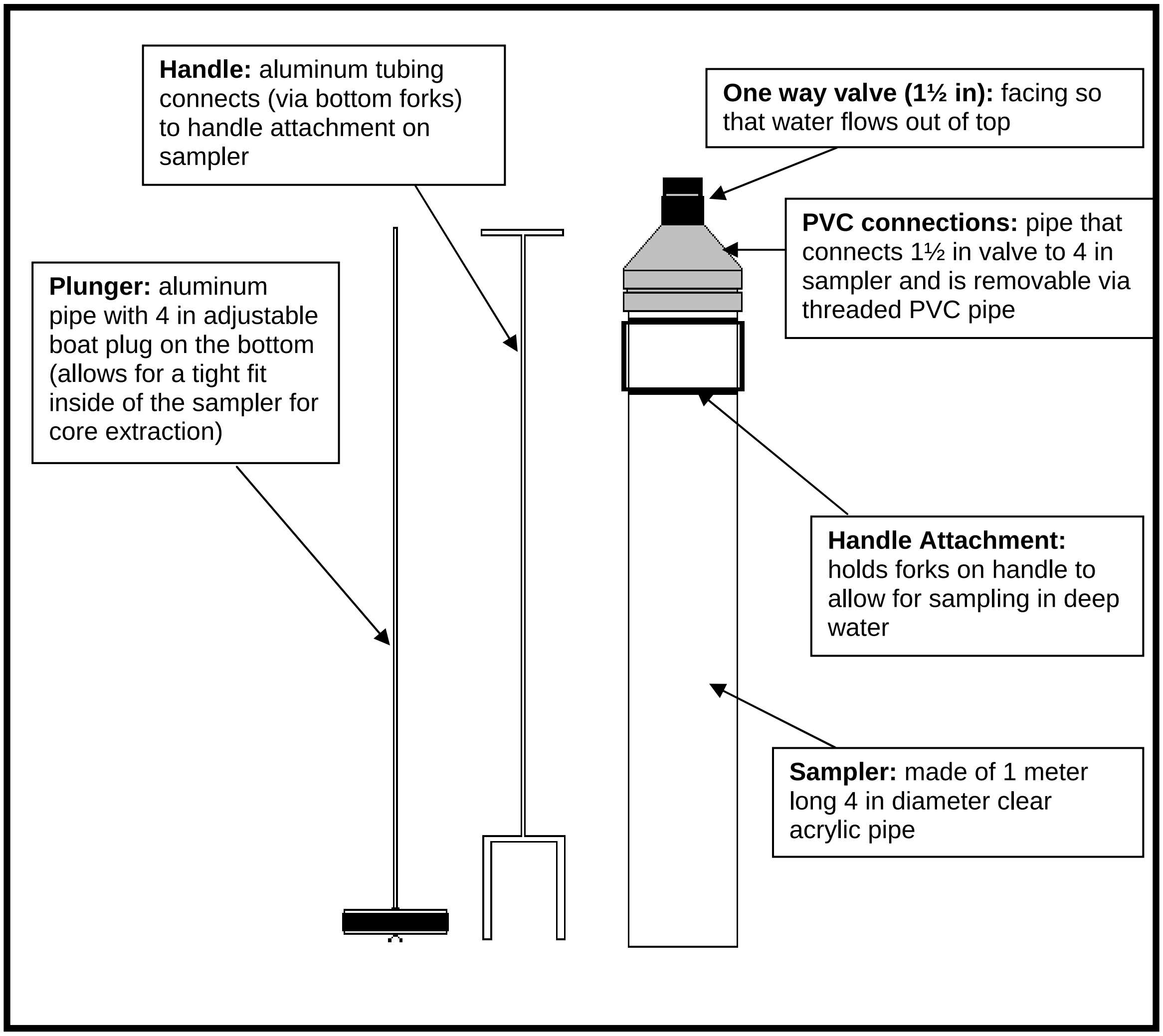

Figure 3.1: Diagram of core sampler (drawing is not to scale).

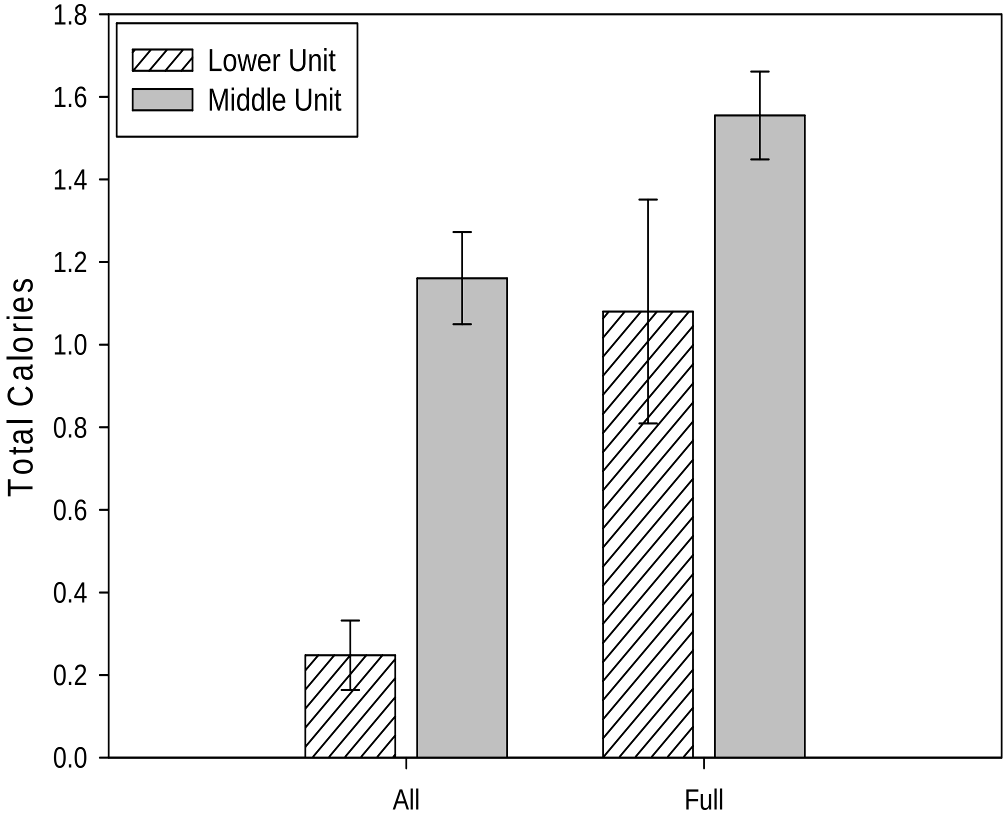

Figure 3.2: Average of the total calories consumed (log transformed) for common carp feeding in the lower and middle units of Swan Lake in 2005. The 2 bars of the left (All) are the average of the total calories consumed for all carp whose diets were examined and the 2 bars on the right (Full) are just the carp that had items in their stomachs. Error bars represent standard error of the mean.

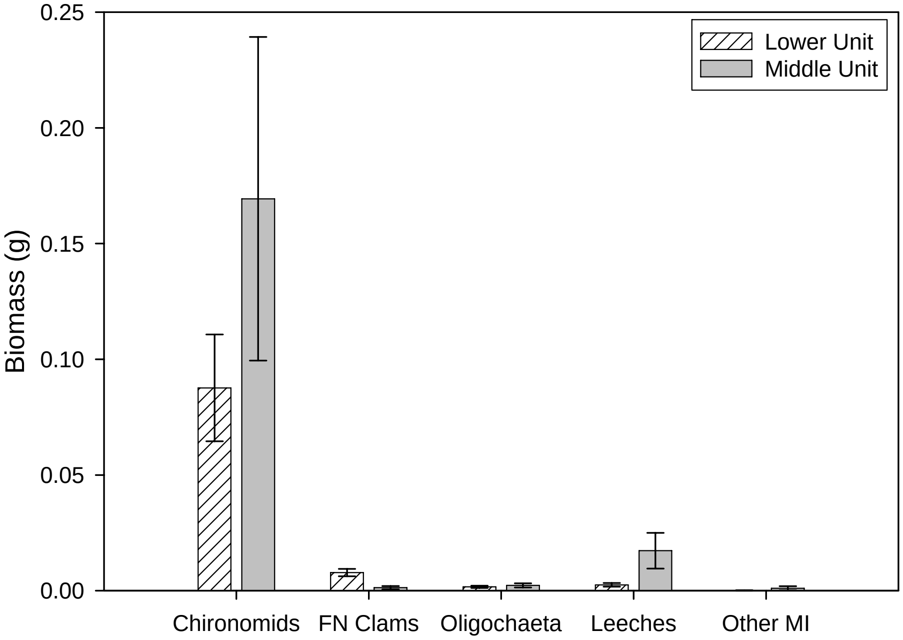

Figure 3.3: Biomass of macroinvertebrate groups obtained through PONAR sampling. FN Clams = fingernail clams, Other MI = other non-dominant macroinvertebrates and error bars are the standard error of the mean.

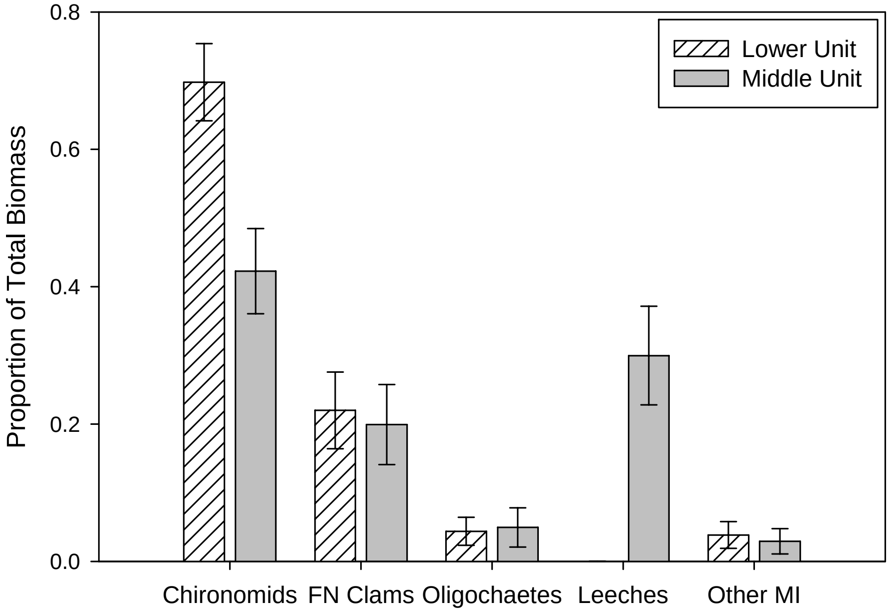

Figure 3.4: Proportion of total core biomass made up of specific taxa in the middle and lower units. FN Clams = fingernail clams, Other MI = other non-dominant macroinvertebrates and error bars represent the standard error of the mean.

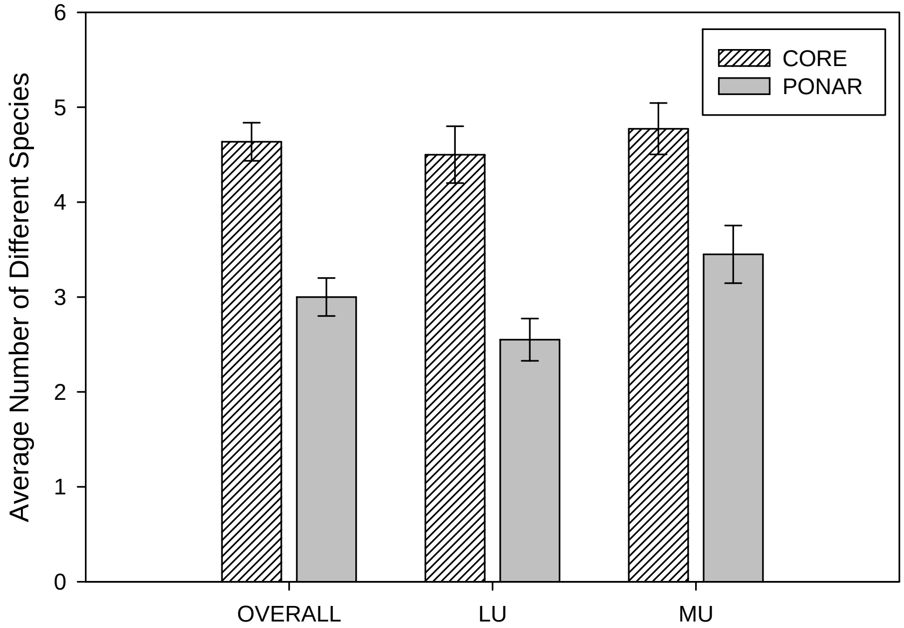

Figure 3.5: Species richness compared between core samples and PONAR samples. LU = lower unit and MU = middle unit, error bars are the standard error of the mean.

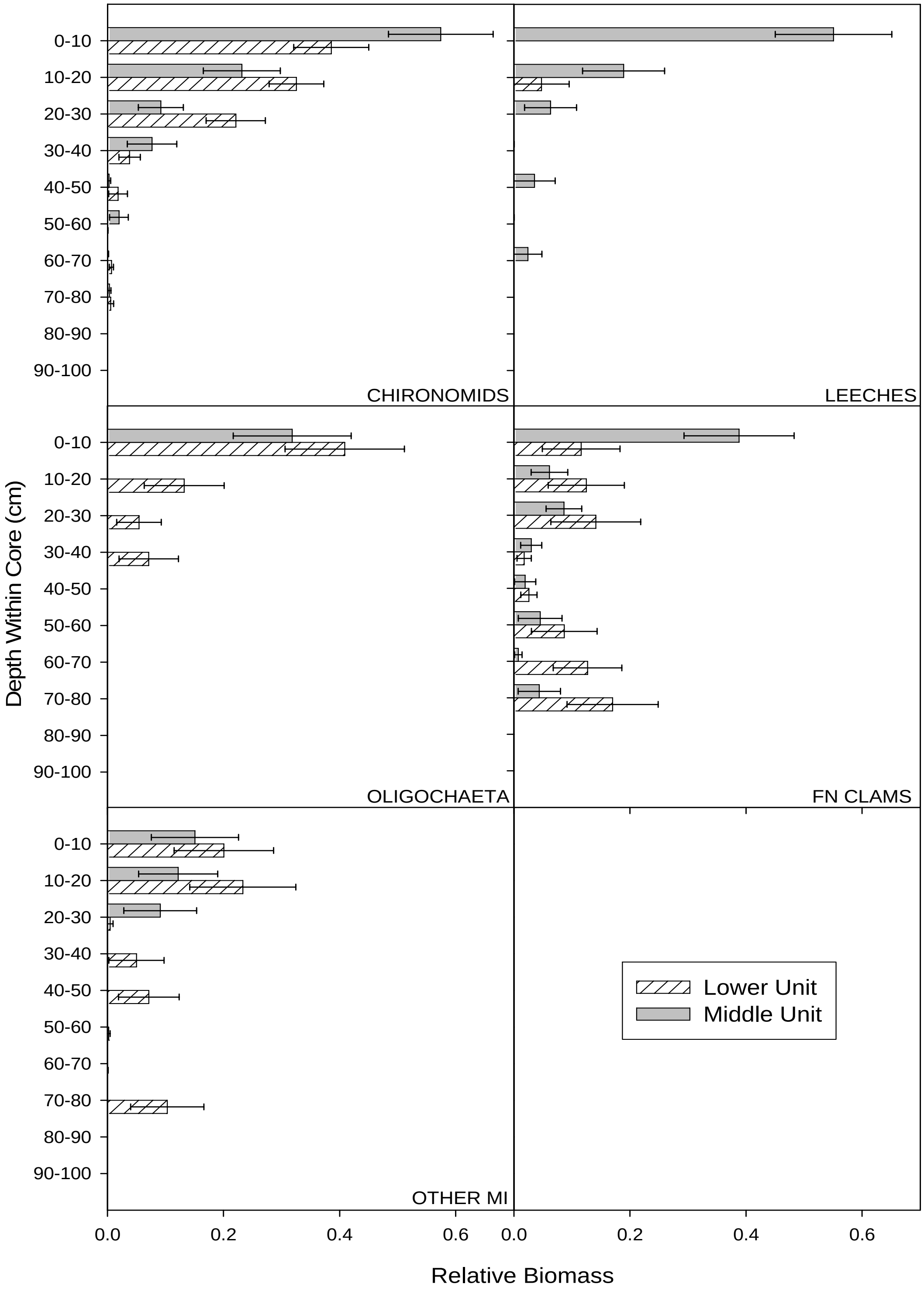

Figure 3.6: Average distribution of relative biomass for each macroinvertebrate taxa within the sediment core of the lower and middle units. FN Clams = fingernail clams, Other MI = Other non-dominant macroinvertebrate taxa and error bars are the standard error of the mean.

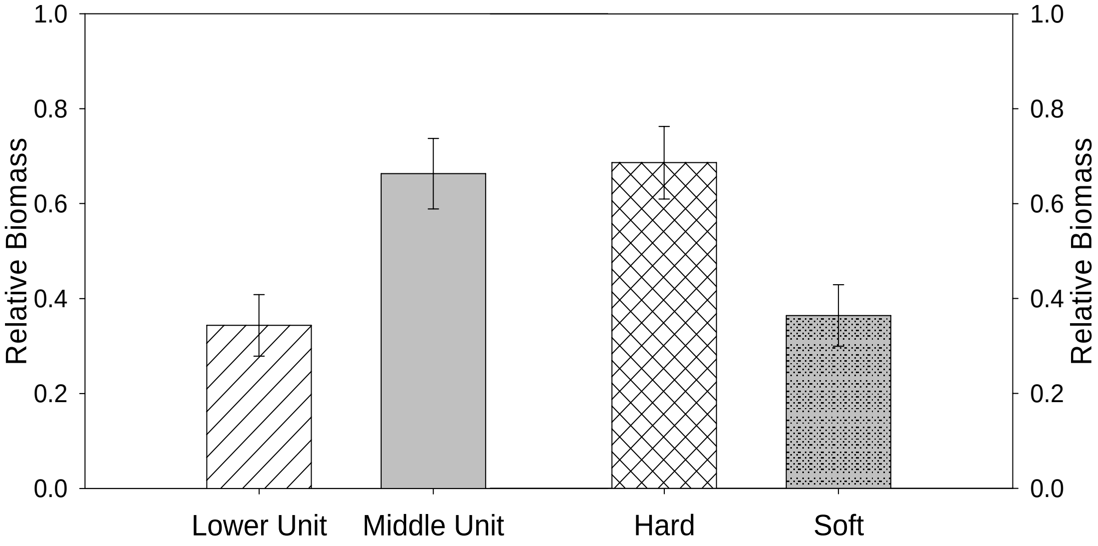

Figure 3.7: Relative biomass in the upper 10 cm compared between the lower and middle units and hard and soft sediment sites. Error bars represent the standard error of the mean.

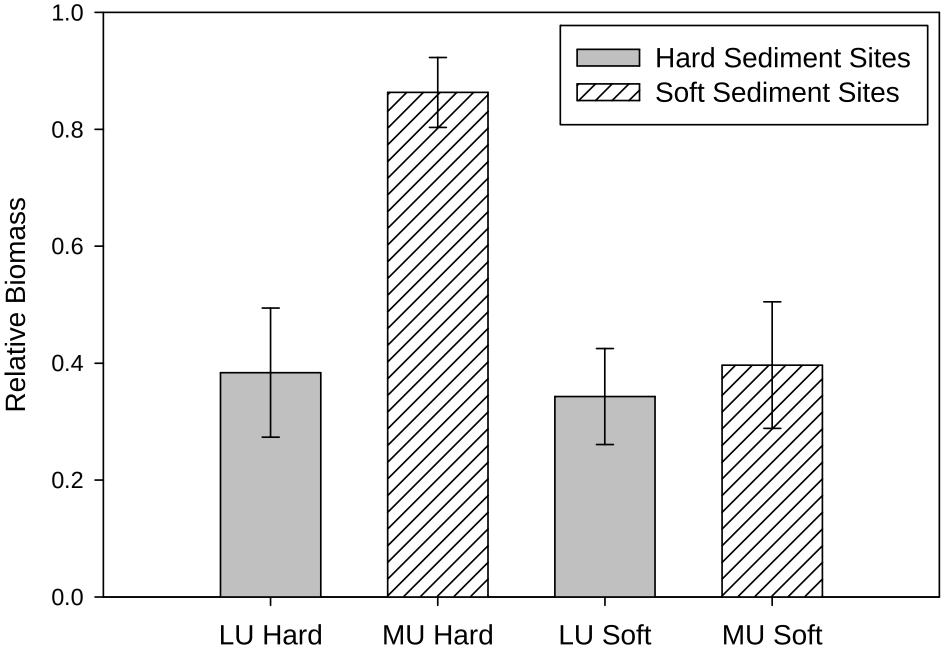

Figure 3.8: Relative biomass in the upper 10 cm compared between sediment sites within units (i.e The interaction unit by sediment type). LU = lower unit, MU = Middle unit and the error bars represent the standard error of the mean.

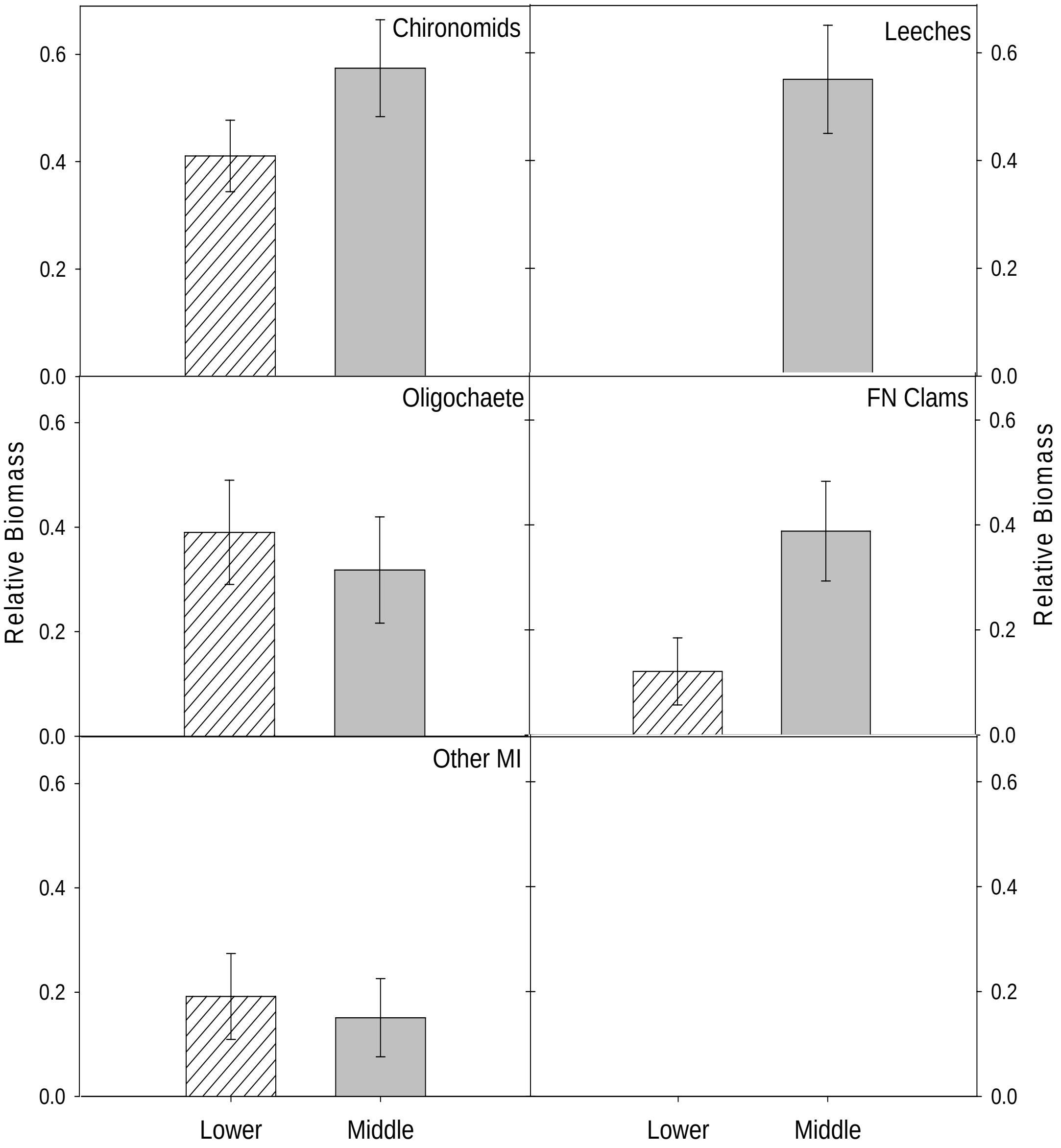

Figure 3.9: Average relative biomass of macroinvertebrate taxa within the upper 10 cm of the sediment core in the lower and middle units. FN Clams = fingernail clams, Other MI = other non-dominant macroinvertebrates found and error bars represent standard error of the mean.

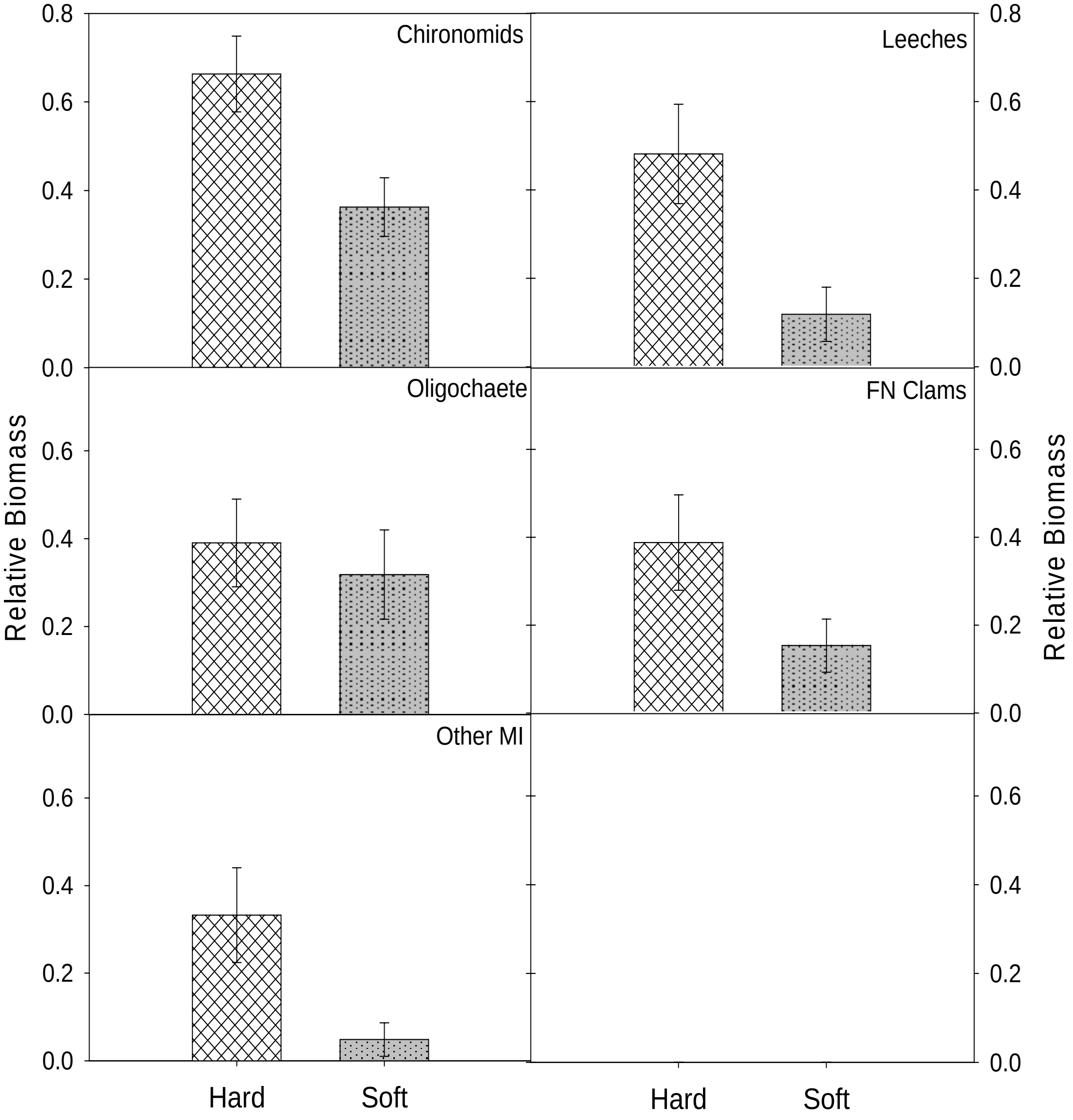

Figure 3.10: Average relative biomass of invertebrate taxa within the upper 10 cm of the sediment cores of hard and soft sediment sites. FN Clams = fingernail clams, Other MI = other non-dominant macroinvertebrates found and error bars represent standard error of the mean.

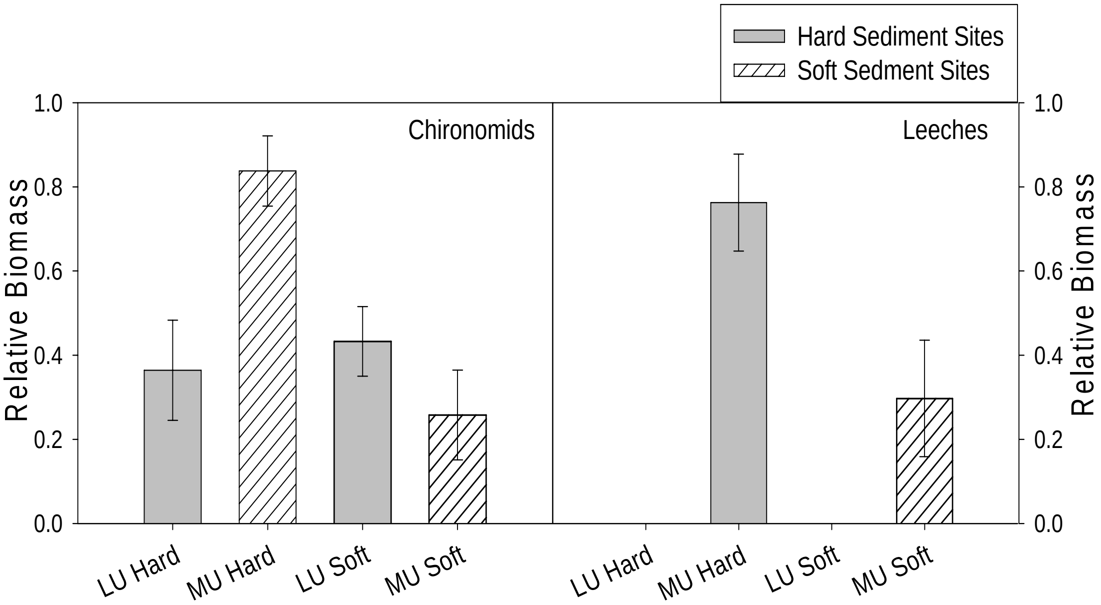

Figure 3.11: Average relative biomass in the upper 10 cm compared between lower and middle unit hard and soft sites for chironomids and leeches only. The grey bars are the hard site comparison and the white lined bars are the soft sediment site comparison. Error bars represent standard error of the mean.

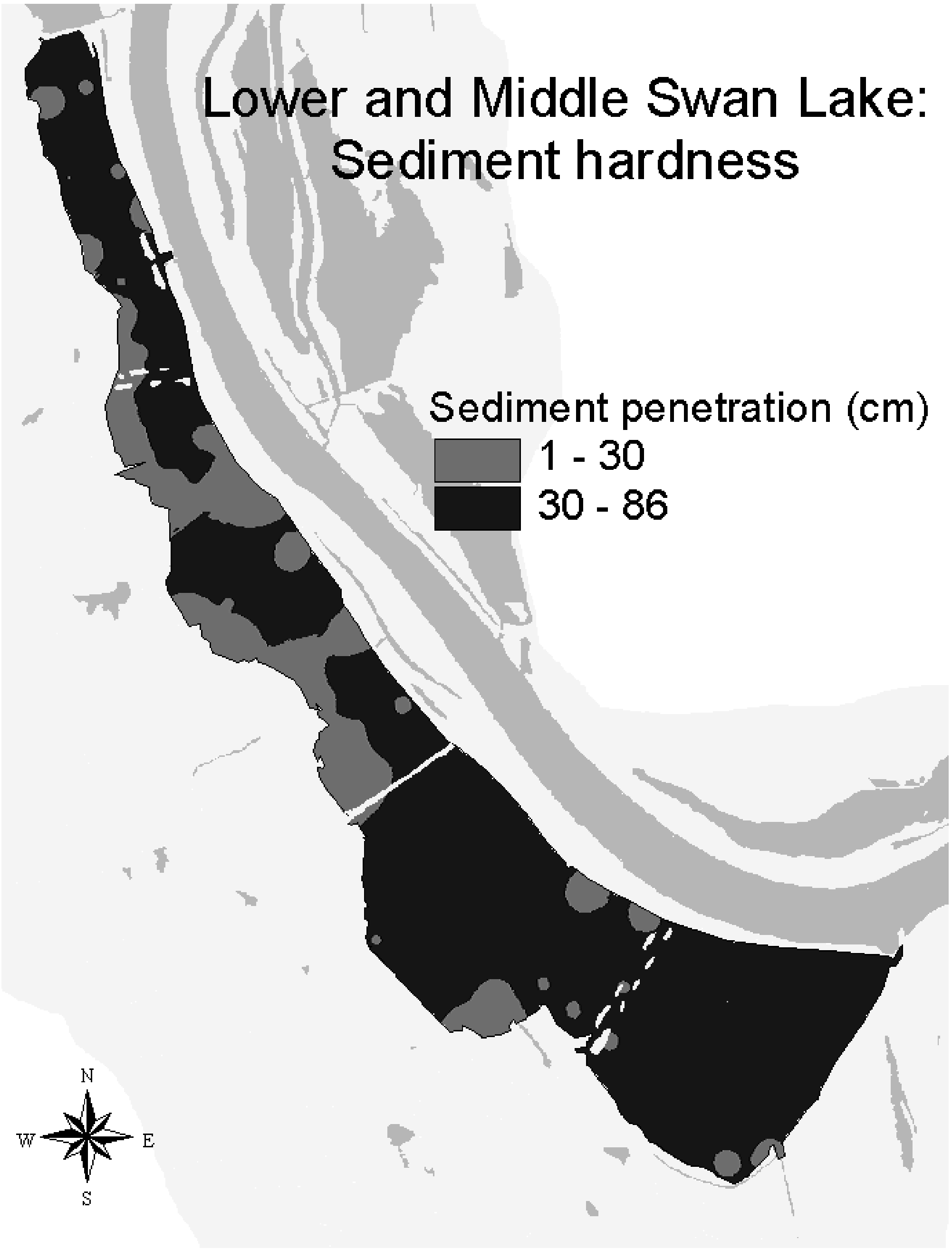

Figure 3.12: GIS map of the sediment condition in the lower and middle units of Swan Lake. The black represents sites with sediment flocculency readings of 30 cm and higher and the dark grey represents sites where sediment flocculency measures 30 cm and less.

CHAPTER 4: CONCLUSION

The problems in Swan Lake are common issues in backwater lakes along the highly modified Illinois and Mississippi rivers. Sediment pollution in backwater lakes causes the loss of critical habitat for fishes and wildlife, and is the crux of the endangerment of many native species and the decline of natural resources in these rivers. With a demand to manage riverine systems to maintain anthropogenic and ecological needs, modifications to common management practices are critical. Studies are needed to determine the effectiveness of these modified management practices. The results of this study have direct implications to management practices on backwater lakes, and also provide new ecological insights.

The analysis of the diet of riverine fishes allowed for an assessment of the alternative management practices used in the lower and middle units. The middle unit combines the sediment consolidation offered by typical moist soil management while maintaining a portion of the river backwater lake connection offered by unmodified systems; whereas the management in the lower unit is aimed at maintaining river backwater lake connection over sediment consolidation. The separation of fish diet between the middle and lower units was not dramatic. Fishes feeding in both units relied on similar prey species (chironomids), and for 3 of the 5 fishes studies there were no differences observed between units in total calories consumed or number of different prey items ingested. Nevertheless, I found a few differences that suggest that the management practice used in the middle unit, though in their infancy, is beginning to modify this unit for the better. Common carp (*Cyprinus carpio*) consistently consumed more calories in the middle unit than in the lower unit, and consumed a greater number of prey taxa in the summer in the middle unit. After the drawdown of the middle unit during the summer of 2005, crappie (both black crappie, *Pomoxis nigromaculatus* and white crappie, *P. annularis*) ingested more total calories than they did prior to the drawdown, suggesting a direct benefit from this management practice. These results, coupled with post-project monitoring in the units (i.e.: Dolan and Chick 2005), suggest that the management practice used in the middle unit could ultimately provide for better foraging habitat for riverine fishes.

Further investigation of the diets of common carp and the analysis of sediment core samples provided insight into an understudied element of predator, prey and structural complexity interactions. Whereas studies of these interactions are widespread in ecology for structural elements such as vegetation, woody debris, coral reefs and rocks (See: Heck and Crowder 1991 and Helfman 1986), studies on the role that flocculent sediments play in providing a vertical refuge from predation are limited in freshwater systems (Persson and Svensson 2006). This study revealed that common carp consumed more total calories and fed more often in the middle unit than in the lower unit. When this comparison was limited to only those fish that fed, common carp in the middle and lower unit consumed equivalent amounts of food. This result suggested an extremely patchy prey base in the lower unit, but standard PONAR grab samples of macroinvertebrates yielded little insight into the source of this patchiness.

Macroinvertebrate core samples revealed that flocculent sediment provided a deep sediment refuge to some of the macroinvertebrate taxa groups. This result appears to explain the source for the patchy prey base in the lower unit. Very few areas in the lower unit had hard sediments, and the preponderance of soft-flocculent sediment suggest that most common carp in the lower unit were foraging in areas that offered a vertical refuge to macroinvertebrates. In contrast, the middle unit had many more areas with hard sediments, likely increasing the vulnerability of macroinvertebrates to common carp predation. Problems with excessive sedimentation are issues not only in backwater lakes, but in any system where sediments are allowed to flow uninhibited into waters and there is not a mechanism for drying and compacting the sediments.

The phenomenon of loss of habitat through excessive sedimentation and lack of a summer drying period is not exclusive to Swan Lake, and the solutions that the managers employed to resolve these issues may be instructive for many other systems. With a combination of adaptive management and openness to the results of studies that evaluate the effectiveness of their management, there is greater hope for the continued recovery of degraded habitat within Swan Lake. The results obtained by this study, as well as the Illinois Natural History Survey's (INHS) post-project monitoring, show the improvements the middle unit management practices have over practices adopted for the lower unit. Managers of Swan Lake have been receptive to these results and have modified their management practices to include more aggressive drawdown on the lower unit. Continuation of focused research studies and ecological monitoring are vital for continued assessment the management practices.

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