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Final Report

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Chapter 1

Executive Summary

Introduction

During spring 1999 through spring 2002, we quantified how vegetation, fish, waterfowl, macroinvertebrates, and zooplankton responded to environmental pool management (EPM) of Pool 25, Mississippi River. The goal of EPM is to simulate a natural hydrograph by maintaining relatively low, stable water levels in the lower portion of the pool following drawdown in the spring (Figure 1-1; see Chapter 2 for formal definition). Mudflats exposed during EPM in the lower reach of the pool should produce abundant, non-persistent vegetation. Seed production from vegetation should in turn provide forage for many species of migratory birds. Vegetation inundated during fall and persisting through spring should also provide cover and nursery habitat for many fish species. In this chapter, we first summarize the major findings relative to our outlined objectives (see below). We then synthesize these results, providing explicit recommendations for EPM with the goal of enhancing waterfowl while maximizing the abundance and diversity of other waterbirds, fish, and invertebrates specifically in Pool 25, and generally in other pools of the Upper Mississippi River (UMR).

Objectives

- Quantify emergent vegetation response and estimate above ground seed biomass produced by EPM in the lower end of Pool 25, UMR.
- Quantify fish use of emergent vegetation created by EPM.
- Characterize fish use of residual vegetation produced by EPM.
- Characterize waterfowl use of EPM-created habitat.

- Determine effects of vegetation on water quality and zooplankton communities.
- Quantify aquatic macroinvertebrate abundance, diversity, and biomass, and assess how they are related to available vegetation and EPM.
- Evaluate the influence of timing, length, and severity of EPM drawdown on fish and invertebrates.

General EPM-related responses during 1999, 2000, and 2001

Water level, hydrology, and general vegetation responses differed among years as a function of the implementation of EPM (see Chapter 3). During summer 1999, water levels remained below the target elevation of EPM (432.0 ft) for 54 days; heavy rainfall necessitated full "tilting" of the pool for an extended time period to maintain a stable water level at the mid-pool control point at Mosier Landing (Figure 1-2, Table 1-1). The result was a large magnitude vegetation response comprised primarily of smartweeds, chufa (primarily red-root nutsedge), and millet. The hydrology and vegetation response of 2000 contrasted dramatically with that in 1999. Following a brief drawdown, water levels during summer 2000 were held by the U.S. Army Corps of Engineers near full pool (434 ft) for most of the summer to ensure adequate habitat for fish, resulting in the production of sparse amounts of vegetation (Figure 1-2, Table 1-1). In 2001, the summer drawdown regime resulted in a moderate amount of vegetation production relative to previous years (Figure 1-2, Table 1-1). During the three years of study, study plots contained high amounts of vegetation (1999), little to no vegetation (2000), and moderate amounts of vegetation (2001).

Emergent vegetation and above-ground seed biomass

Methods

Similar to previous research in Pool 25, vegetation composition and abundance were quantified along 11 transects during summer 2001 in the Batchtown area (lower Pool 25), three weeks post-drawdown (see Chapter 3). Transects were oriented perpendicular to the shore, following an elevation gradient relative to full pool. Seed biomass of common species was estimated following resumption of full pool during September 2001. All vegetation data were compared to those quantified during 1999 and 2000 for a related project (Dugger and Feddersen 2000).

Results

Vegetation differed in total abundance among years. During 2000, vegetation was virtually absent from the lower reach of Pool 25, with the exception of narrow margins of vegetation at some sites. In 2001, plant diversity was the highest since EPM was implemented. This high diversity can be attributed to the late and prolonged dewatering period that occurred this year relative to others (Figure 1-2).

Although vegetation was abundant during both 1999 and 2001, species composition differed between these years. In 1999, the assemblage was dominated by smartweed, millet, and chufa (i.e., red-root nutsedge). During 2001, smartweed was no longer the dominant plant species. Rather, red-root nutsedge, pigweed, and millet were dominant in the lower reach. The plant assemblage during 2001 was dominated by species that are generally favored by late, prolonged dewatering. The late drawdown that year likely reduced the success of smartweed.

Despite differences in plant composition between 1999 and 2001, the biomass of seeds produced was similar. Red-root nutsedge was the dominant seed producer during both years. Smartweed was relatively more important in 1999, whereas millet and sprangletop contributed more to seed production during 2001.

Woody species such as maple and willow were less abundant in 2001 than in 1999, suggesting that conditions favorable to germination of these species were absent in 2001. Because the woody species produced in 1999 were absent in 2001, these species do not appear to persist in the lower reach during multiple years, perhaps due to high water levels during fall or ice scour. During 2001 at lower elevations, river bulrush occurred. This species can form large, monotypic stands with low habitat quality and should be monitored closely.

In summary, variation among years in EPM led to differences in vegetation abundance and composition. During the two years when vegetation was produced, differences in composition did not affect seed biomass available for waterfowl. As such, managers have some flexibility in maintaining benefits for waterfowl while varying EPM to induce vegetation that maximizes macroinvertebrate and fish diversity. We will discuss this approach further in the following sections.

Fish use of emergent vegetation

Methods

We studied the fish response to EPM during 1999 through 2002 by quantifying fish use and selection of flooded vegetation during fall, and use of residual vegetation during spring by young-of-year (YOY) fishes (see Chapter 4). Study plots (one vegetated and one manipulatively devegetated) were established at each of four sites (Batchtown East, Batchtown West, Jim Crow, and Turner Island), and were sampled by seining during fall 1999, 2000, and 2001. We also sampled fish at side-channel/backwater, mid-pool and lower-pool sites contiguous with the main channel with a combination of seining, modified fyke nets, and cast nets.

Results

Fish responded dramatically to both environmentally induced hydrology and EPM. During fall 1999, 2000, and 2001, a total of 34,177 fish, encompassing 23 species, were collected in EPM-induced vegetation. The fish assemblage in the vegetation was dominated numerically by members of the minnow family (Cyprinidae) and the western mosquitofish, all of which likely play important ecological roles as prey for piscivores such as fish and waterbirds.

Across years, species richness and abundance of fishes were relatively high during 1999 and 2001, and low during 2000, when plots contained little vegetation. Some vegetated plots during 1999 contained low concentrations (< 3 mg 0_2/l) of dissolved oxygen, and these plots were dominated numerically by tolerant species (e.g., common carp and western mosquitofish). Sunfish abundance (i.e., bluegill and orangespotted sunfish) was low during fall 1999, when backwaters were isolated from the main channel for an extended period of time, but they were relatively abundant during 2000 and 2001, indicating the drawdown regime was not as harsh on the resident backwater fish fauna during those years. During 2000, species richness and abundance were relatively high in a narrow band (< 1 meter thick) of vegetation present at some sites; this sparse, shallow vegetation was used primarily by western mosquitofish.

The location and patch sizes of vegetation influenced fish. Fish species richness and abundance were higher in relatively smaller patches of vegetation associated with islands (Jim Crow and Turner Island) than in vegetation produced at sites within an extensive, shallow, backwater complex (Batchtown East and Batchtown West). The typically higher dissolved oxygen concentrations at the island sites adjacent to the channel likely enhanced fish assemblages residing there. The spotfin shiner, common carp, western mosquitofish, channel shiner, bullhead minnow, and bluegill were generally more abundant in vegetated plots than in devegetated plots. Conversely, the emerald shiner and orangespotted sunfish were more abundant in devegetated plots in 1999, when vegetation was dense, but they were abundant in both study plots in 2001, when vegetation density was moderate and dissolved oxygen concentrations were relatively high.

Vegetation induced by EPM is clearly a nursery habitat in Pool 25, because length-frequency histograms indicated most fish using the vegetation were YOY. During fall 1999, recently spawned individuals (< 20 mm) of the channel shiner and spotfin shiner selected vegetated plots over devegetated plots. Young-of-year western mosquitofish were more abundant in the vegetation during both fall 1999 and 2001. During spring 1999, 2000, 2001, and 2002, thirty-seven fish taxa from 11 families, primarily late larvae/early juvenile individuals, were collected in residual vegetation produced by EPM the preceding summer. Fish using the vegetation, comprised primarily of smartweed stalks, included YOY mooneye, silver chub, and blue sucker which are relatively rare fishes. Young of the commercially important buffalo species were also abundant in the residual vegetation.

Our comparison of mid-pool and lower-pool fish communities revealed similar densities and numbers of fish species. However, the relative composition of fish assemblages differed between reaches, mainly due to greater abundances of western mosquitofish in the lower pool.

In summary, the fish response to EPM is influenced by both the enhancement of shallow-water habitat through vegetation production and the underlying hydrology. Our data indicate that a major benefit of EPM-induced vegetation is that it provides nursery habitat for many fishes, particularly minnow species that spawn in late summer and fishes that spawn in spring. Summer drawdown regimes isolating backwaters to the point that water quality deteriorates will be detrimental to backwater obligate species such as sunfishes. The fish response to the EPM regime of 2001 indicates that a relatively benign drawdown that produces a moderate amount of vegetation will benefit most backwater fishes. Open patches dispersed within vegetation should create beneficial heterogeneity, enhancing both species that use edge areas and those that occur primarily in dense vegetation.

Waterfowl and waterbird responses

Methods

Following dewatering during late summer 2001, waterbirds were counted in the Turner Island and Batchtown area (see Chapter 3). Winter/spring waterfowl surveys occurred during spring 2001 and 2002. Results were compared to earlier work completed in 1999 (Dugger and Fedderson 2000).

Results

Although EPM-induced vegetation was abundant during summer 2001, few shorebirds were present at the Batchtown and Turner Island sites during this time. The low occurrence probably occurred because the timing of dewatering was after most had migrated through the area following spring and had not yet returned during the fall migration.

Residual vegetation as well as the sediment seed bank should be important factors attracting waterfowl during spring. During spring 2002, waterfowl were more abundant than during spring 2001. Low abundance of waterfowl in 2001 was largely due to the absence of dabbling ducks. In 2001, waterfowl were likely scarce because vegetation was absent the previous year (2000). Low water and ice during spring 2001 also probably contributed to low abundances.

Although higher than the preceding year, waterfowl abundance during spring 2002 was relatively low compared to spring 1999 and 2000, which also followed summers of abundant vegetation. High seed production during summer 2001 and stable water levels during spring 2002 should have attracted waterfowl. Several factors operating at a regional scale that influence the migratory patterns of species may have been responsible for this relatively lower occurrence of waterfowl. In addition, because vegetation did not persist above the waterline during spring 2002, waterfowl may have avoided the Batchtown area. Without further information, we do not recommend viewing the low waterfowl response during spring 2002 as an indication that habitat produced during summer 2001 was inferior relative to the other positive years. Confounding factors unrelated to EPM likely contributed to the low occurrence.

In combination with the previous study years, these data support the contention that EPM improves quality habitat for migrating birds. Plant species with known value to waterfowl continue to dominate the vegetation community, with no evidence of encroachment by woody species. Similar to managed moist-soil impoundments, common throughout the Midwest, by changing the timing, magnitude, and duration of the dewatering period in Pool 25 it appears managers can influence the vegetation structure and still produce seed biomass important to waterfowl. This fact provides some flexibility for how managers implement EPM. For example, altering drawdown characteristics to improve conditions for fish does not by definition have to lower the quality of the habitat for waterfowl

Water quality and zooplankton responses

Methods

Water depth, temperature, dissolved oxygen concentration, conductivity, turbidity, pH, and zooplankton species composition/density were quantified in study plots (one vegetated and one manipulatively devegetated) established at each of four sites (Batchtown East, Batchtown West, Jim Crow, and Turner Island) during fall 1999, 2000, and 2001 (Chapter 4). We also sampled zooplankton at side-channel/backwater mid-pool and lower-pool sites contiguous with the main channel (Chapter 5).

Results

Water depth was typically < 1 m and similar among all sites during the entire study, although water was generally deeper at the Batchtown sites. Temperature, conductivity, and pH often varied among sites and years. Dissolved oxygen (DO) concentration was lowest in vegetated plots, particularly during 1999. At one site (Batchtown East) DO concentration was chronically low (<3.0 mg/l) during much of 1999. Generally, DO concentration increased at sites through time as vegetation became less prevalent in the water column due to waves, wind, and senescence. With the exception of low DO concentration during the year of highest vegetation, all water quality parameters were well within the range of tolerances of river fishes.

Zooplankton abundance was higher in vegetated experimental plots in 1999 when vegetation was abundant, mainly due to the presence of chydorids and cyclopoida (including nauplii). No differences between plots occurred in 2000 or 2001. Zooplankton abundance was significantly higher in the lower pool sites throughout the study period, except for July when lower pool sites were dry. This difference likely occurred because the lower pool sites became more disconnected from the river and thus were more lentic in nature. Because zooplankton are energy-rich prey, particularly for YOY fish, their greater densities should facilitate growth, survival, and ultimately recruitment of many fish species that use vegetation in the lower pool as nursery habitat.

Macroinvertebrate response

Methods

Macroinvertebrates and benthic organic matter were quantified with a stove pipe corer in study plots (one vegetated and one manipulatively devegetated) established at each of four sites (Batchtown East, Batchtown West, Jim Crow, and Turner Island) during fall 2000 and 2001 (Chapter 5). We also sampled macroinvertebrates and organic matter at side-channel/backwater mid-pool and lower-pool sites contiguous with the main channel.

Results

Benthic organic matter is an important structural habitat and energy source for invertebrates in aquatic systems.   When comparing the vegetated and devegetated plots, benthic organic matter was higher with vegetation in 2000 and 2001, suggesting that greater food resources and habitat for invertebrates occurred there. Coarse fractions (coarse particulate organic matter; CPOM) were most variable and contributed most to total benthic organic matter pools. Because benthic organic matter appeared to decline in 2001 in the vegetation plots following the low-vegetation year of 2000, vegetation response during a given year appears to contribute to organic matter dynamics the next year.

Total macroinvertebrate density did not differ between vegetated and devegetated plots during 2000 and 2001. However, in 2001 but not 2000, total macroinvertebrate biomass was higher in vegetated plots. These differences between years in total biomass can be attributed to in the greater vegetation abundance in 2001 (i.e., vegetation enhanced total biomass). Focusing on specific taxa, no differences were detected in Oligochaeta abundance between vegetated and devegetated plots in 2000 and 2001, although Oligochaeta biomass was higher in vegetated treatments in 2001 but not 2000. Conversely, Chironomidae abundance and biomass were higher in devegetated treatments in 2001; no differences were detected in 2000. Three genera, *Chironomus*, *Polypedilum*, and *Tanytarsus* contributed most to this difference. Within sites, the total amount of vegetation during a year characterizes the macroinvertebrate community.

We also combined macroinvertebrate data between vegetated and devegetated plots and compared their total abundance between the years of low (2000) and high (2001) overall vegetation in the lower pool. Total macroinvertebrate density and biomass did not differ between years. Although Oligochaeta density and biomass did not differ between years, Chironomidae numbers and biomass were higher during the low vegetation year of 2000, and this followed patterns observed in experimental vegetation plots (e.g., more chironomids with less vegetation).

Comparing the mid-pool and lower-pool reaches, total benthic organic matter and coarse and fine fractions differed through the study period. Mid-pool values were ~2x less than lower pool values at times during the study, but there was no evidence that this adversely affected macroinvertebrates. Lower quantities probably occur at mid-pool due to higher flows and transport in these sites at full pool conditions. Differences in benthic organic matter and hydrology led to differences in macroinvertebrates between reaches as well. Relative to midpool, total macroinvertebrate abundance and biomass was significantly higher in the lower pool sites. This was primarily driven by the dominant macroinvertebrate groups, Oligochaeta and Chironomidae, which had significantly higher abundance and biomass in the lower pool sites relative to mid-pool ones.

Macroinvertebrate communities also differed between the lower pool and mid-pool sites. Mid-pool sites had ~14 taxa that were unique, many of them with relatively long, univoltine life cycles. In contrast, lower pool sites were dominated by polyvoltine taxa with short life cycles. This suggests the hydrologic stability and longer hydroperiod of mid-pool sites may be important for macroinvertebrate diversity there, even though lower pool sites may have higher abundance, biomass, and sometimes diversity (after long periods with water present).

In summary, lower pool sites appear to have more benthic organic matter and higher macroinvertebrate abundance and biomass. However, the lower pool sites are hydrologically more variable and thus macroinvertebrate diversity varies much more and there are much fewer larger, long-lived taxa than at mid-pool sites. Just after summer reflood, richness and diversity are lower in lower pool sites, but these values surpass those of mid-pool sites by the following spring.

Synthesis and Management Recommendations

Because hydrological conditions differed markedly among the three study years, we were able to effectively assess how vegetation, fish, waterfowl, and invertebrates respond to three, unique environmental- and EPM-induced scenarios that may typically arise in pools of the UMR (see Table 1-2 for a summary). In general, we found that EPM can generate mutually beneficial conditions for waterfowl and fish in the lower reaches of these systems. Although vegetation was only successfully produced during two years of the study, production of seed biomass remained high regardless of the hydroperiod and vegetation assemblage that arose. Hence, managers have the flexibility to choose an EPM-induced hydroperiod that generates a diverse and productive vegetation assemblage (i.e., less dominated by smartweed and high in invertebrate density and diversity) for fish while maintaining seed production and spring, residual vegetation for waterfowl.

Pool 25 is currently managed at a mid-pool control point that generates different hydrological conditions between the middle and lower reaches, during moderate to high discharge (Figure 1-3). As such, backwater habitats differ between these two reaches, with those at mid-pool being primarily flow-through and not influenced by EPM while those at lower pool are primarily back-fill habitats affected by EPM. Our research suggests that mid-pool water level management increases both fish and macroinvertebrate diversity at the pool scale in Pool 25 because the species within each reach are likely adapted to the local hydrology and a gradient of physical habitat conditions occurs between each reach. However, during years of high discharge (e.g., 1999) sustained drawdowns below 3 feet in the lower pool during spring through summer are required to maintain mid-pool water levels (Figure 1-3). Our research suggests that prolonged isolation of backwaters due to this practice will be detrimental to aquatic organisms in the lower pool. During these times, either allowing greater variance in water level limits at the mid-pool control point (currently not to exceed 1.75 feet) or switching the control point to Lock and Dam 25 will prevent water levels in the lower pool from reaching this deleterious point (e.g., maintain lower-reach water levels no less than 3 feet below full pool; Figure 1-3). Although water levels mid-pool would rise, the magnitude change would be similar to that in the lower pool if a mid-pool control point was maintained (Figure 1-3). We recommend that water level managers have the flexibility to adopt either a mid-pool or dam control point approach during spring through summer, depending on discharge conditions, to encourage maximum diversity of fish and invertebrates in Pool 25 and perhaps other UMR pools.

It has been recommended that, during fall through early spring, Pool 25 be managed consistently with a control point at Lock and Dam 25 to ensure stable water levels in the lower pool for both fish and waterfowl at high discharge (Figure 1-3). Although this approach will most likely be beneficial to the ecosystem in the lower pool, the relatively higher water levels that may occur at mid-pool (Figure 1-3) may reduce the retention of organic matter in off-channel habitats and alter the water quality characteristics (e.g., reducing the stratification of water temperature) of potential fish overwintering habitat. Before fully implementing this strategy, the potential benefits of improving habitat conditions in the lower pool must be weighed against deleterious effects of reducing stability of water levels at mid-pool reaches.

In the lower reaches of pools, maintaining heterogeneity of both hydrology and vegetation within individual sites should enhance the diversity of fish and invertebrates and should often improve water quality. Because vegetated and devegetated plots generated different fish and invertebrate assemblages, creating open patches either through mechanical clipping or broad-scale herbicide application will create diverse vegetation (rather than creating dense monocultures) and increase diversity within sites. Research in lakes suggests that maintaining 20-30% open patches within the vegetation should be optimal for maintaining diverse fish communities comprised of rapidly growing individuals. Similar management may improve fish success in the EPM-induced vegetation in lower pools. The most diverse vegetation with the least water quality problems arose during summer 2001, primarily due to the relatively late and moderate drawdown (i.e., successful implementation of EPM). Backwaters in lower Pool 25 become disconnected from the main channel at an elevation between 432 and 431 ft. Therefore, the EPM target of a drawdown no greater than 2 ft below full pool of 434 ft appears to be an adequate compromise between vegetation production and maintaining backwater fish faunas. Because water levels did fall 3 ft or greater below full pool during 2001 without obviously reducing the success of backwater fishes, reducing water levels beyond the target 2-foot limit for short periods (< 1 week) may not negatively affect fish assemblages.

Of course, the timing and rate of spring drawdown as well as the duration of EPM will vary among years (Table 1-2). Although our work demonstrates that biotic responses will follow suit, we caution that the responses we quantified were unreplicated through time. As such, we cannot directly tease apart the direct effects of EPM versus other potentially important environmental factors (i.e., surprises are likely still in store). For example, conditions for waterfowl appeared to be ideal during 2001, although waterfowl numbers were low. Unexplored factors contributed to waterfowl abundance that year and must be identified and understood to better predict responses of the entire ecosystem to EPM.

Although our 2001 data demonstrated that vegetation and fish are enhanced by EPM during summer and perhaps the following spring, long-term benefits to the fish and other aquatic components of Pool 25 are still unknown. To illustrate, we have established that EPM-induced vegetation provides important nursery habitat for fishes. But how this habitat contributes to survival, growth, fish cohort strength, and ultimately population size and stability is not well understood. General demographic research quantifying the abundance, size structure, and age structure of fish populations is required to answer these questions. Linkages among EPM-induced invertebrate assemblages and fish growth are also presently unclear and in need of investigation. Vegetation may provide shelter and increased survival for fish, but at the cost of density-dependent reductions in growth. Most of the fishes produced by EPM-induced vegetation serve as important prey for commercially and recreationally important fishes and many waterbirds. However, the relative importance of this flux of energy from the shallow, vegetated areas to these piscivores still needs to be assessed. Clearly, many questions about the ultimate effects of mid-pool control point management and EPM on pool ecosystem structure and function remain.

Environmental variation in precipitation and subsequent hydrology will dictate how EPM is conducted during each year. The only factor precluding EPM would be high discharge necessitating open river conditions. We suggest that inter-annual variability in water levels in the lower pool, with EPM-scenarios similar to 2001 occurring when possible, leads to a diversity of biotic responses, perhaps enhancing the overall diversity of the pool community. The added flexibility of (i) switching the control point to Lock and Dam 25 or (ii) allowing greater variation in water levels at mid-pool reaches during spring through summer when necessary (e.g., high discharge) will likely greatly enhance the ability to achieve EPM in the lower pool. Seed production seems to be robust to an array of hydrological conditions and resulting vegetation. As such, waterfowl should benefit from vegetation growth, perhaps regardless of the underlying hydrological regime. Conversely, years of low vegetation should foster open-water fish and invertebrate specialists while those of dense vegetation should favor backwater-dwelling aquatic organisms. Only with continued long-term monitoring of pool ecosystem responses to EPM will we be able to determine if these assumptions are valid. If the control-point is switched to Lock and Dam 25 when necessary, comparison of responses of vegetation, water quality, organic matter, aquatic organisms and water birds in both lower and mid-pool reaches to those before this strategy was implemented will allow us to assess its efficacy as a management tool.

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Table 1-1. Environmental pool management on Mississippi River Pool 25 during 1999, 2000, and 2001. Numbers represent the number of days 0.5, 1.0, 2.0, 3.0, and 4.0 ft equal to or below full pool. “Re-flood rate” is the difference in water levels between drawdown and 0.5 ft below full pool divided by the number of days between those water levels.

				Number of days below full pool (434 ft)					
Year	Initiated	EPM Ended	Total Days	≥0.5 ft	≥1.0 ft	≥2.0 ft	≥3.0 ft	≥4.0 ft	"re-flood rate"
1999	12-Jun	24-Aug	69	69	60	54	36	21	1.6 inches/day
2000	1-Jul	31-Jul	31	31	28	22	12	7	5.2 inches/day
2001	19-Jun	15-Aug	58	58	53	27	7	2	1.4 inches/day

Table 1-2. Summary of hydrological- and EPM-induced responses of Pool 25 Mississippi River during 1999, 2000, and 2001. MI = macroinvertebrates; ZP=zooplankton; YOY=young of year fish.

	Summer Draw-down (DD) and Hydrology				Responses				
Year	Timing of DD	Duration of DD	Intensity of DD	Late-summer Reflood Rate	Vegetation to Hydrology	Waterbirds[1,2] to Residual Vegetation	Invertebrates[2] to Vegetation	Fish to Vegetation	Water Quality to Vegetation
1999	Early	Long	High	Slow	-dense -low diversity -high seed biomass -high residual	-abundant	-high ZP -moderate MI density -high MI diversity	-high small, backwater fish -low sunfish - high YOY	-low dissolved oxygen
2000	Late	Short	Low	Rapid	-no vegetation produced	-abundant	-open-water ZP moderate -moderate MI density & diversity in open areas (Chironomidae)	-high sunfish	-high
2001	Late	Long	Moderate	Slow	-moderately dense -high diversity -high seed biomass -moderate residual	-scarce	-moderate ZP -moderate MI density -high MI biomass (Oligochaeta)	-high small, backwater fish -moderate sunfish -high YOY	-high

[1]Results are for spring of that year and reflect vegetation and seed production the preceding year.

[2]Results from 1999 and 2000 (for waterbirds) and 1999 (macroinvertebrates and zooplankton) are summarized in Dugger and Fedderson (2000) and are included for comparison.

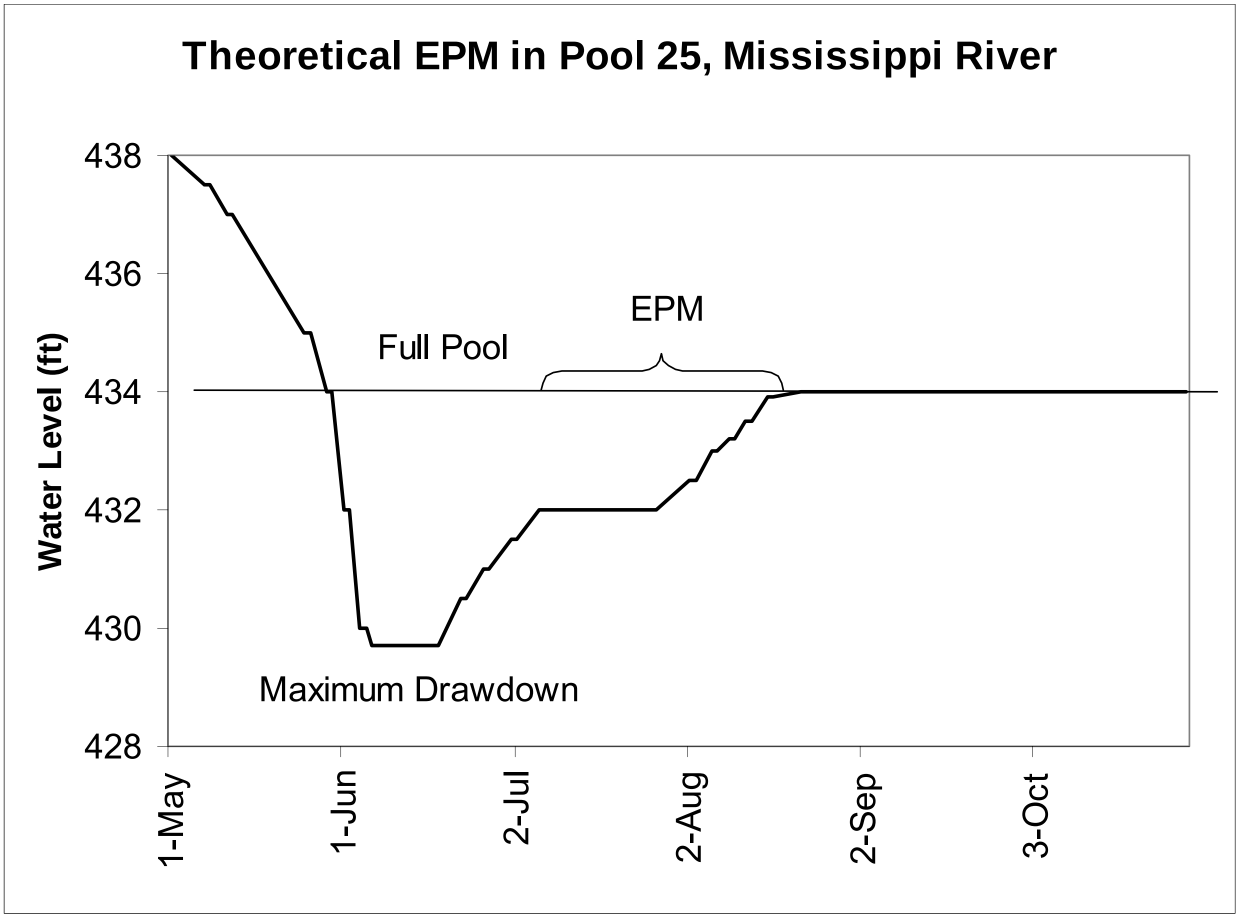

Figure 1-1. A theoretical depiction of Environmental Pool Management (EPM) in Pool 25, Mississippi River (developed in Sheehan et al. 2001).

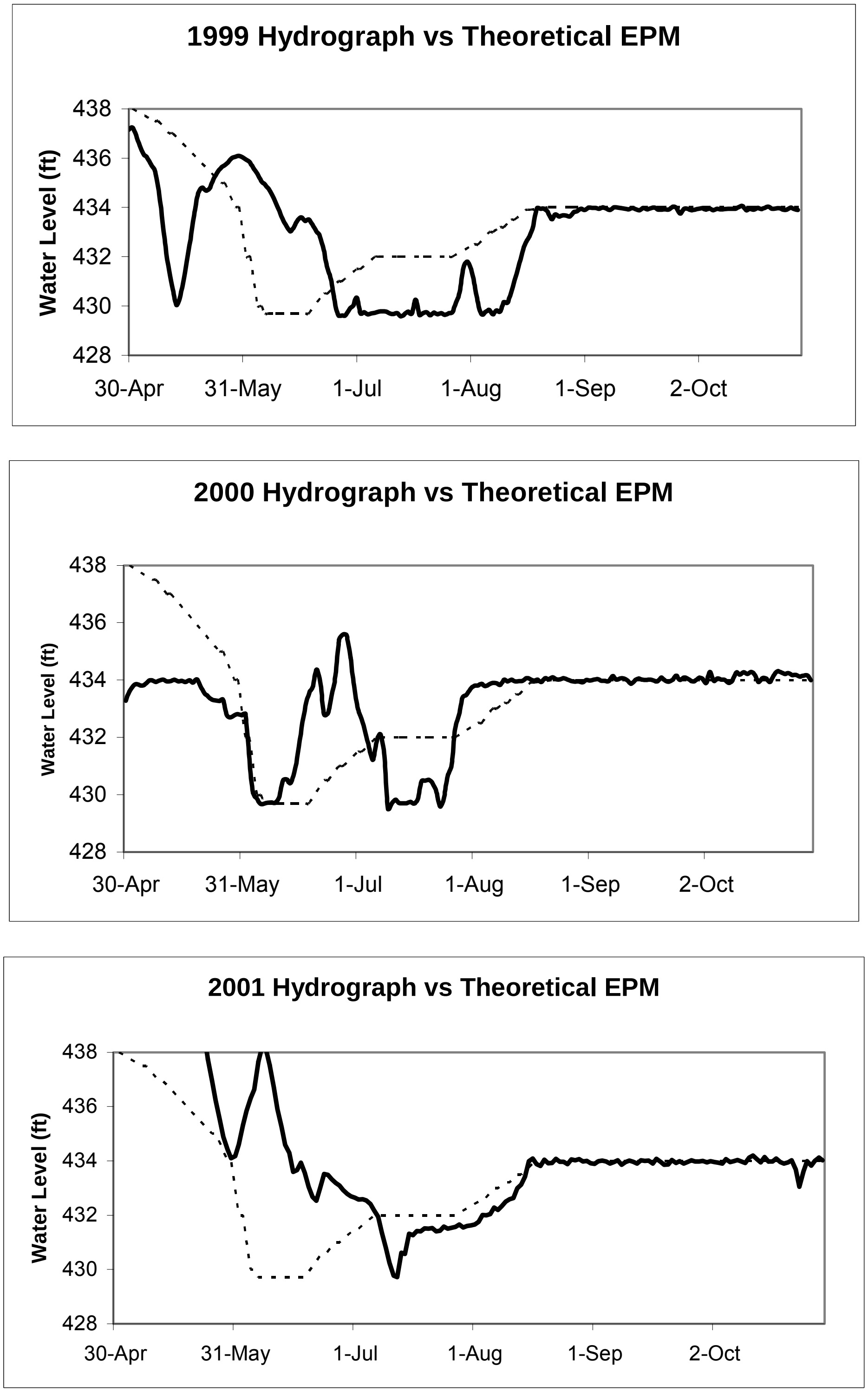

Figure 1-2. Water levels (ft) during 1999 (A), 2000 (B), and 2001 (C) (dark, solid lines) and theoretical, target Environmental Pool Management (EPM; dashed lines) in Pool 25 Mississippi River.

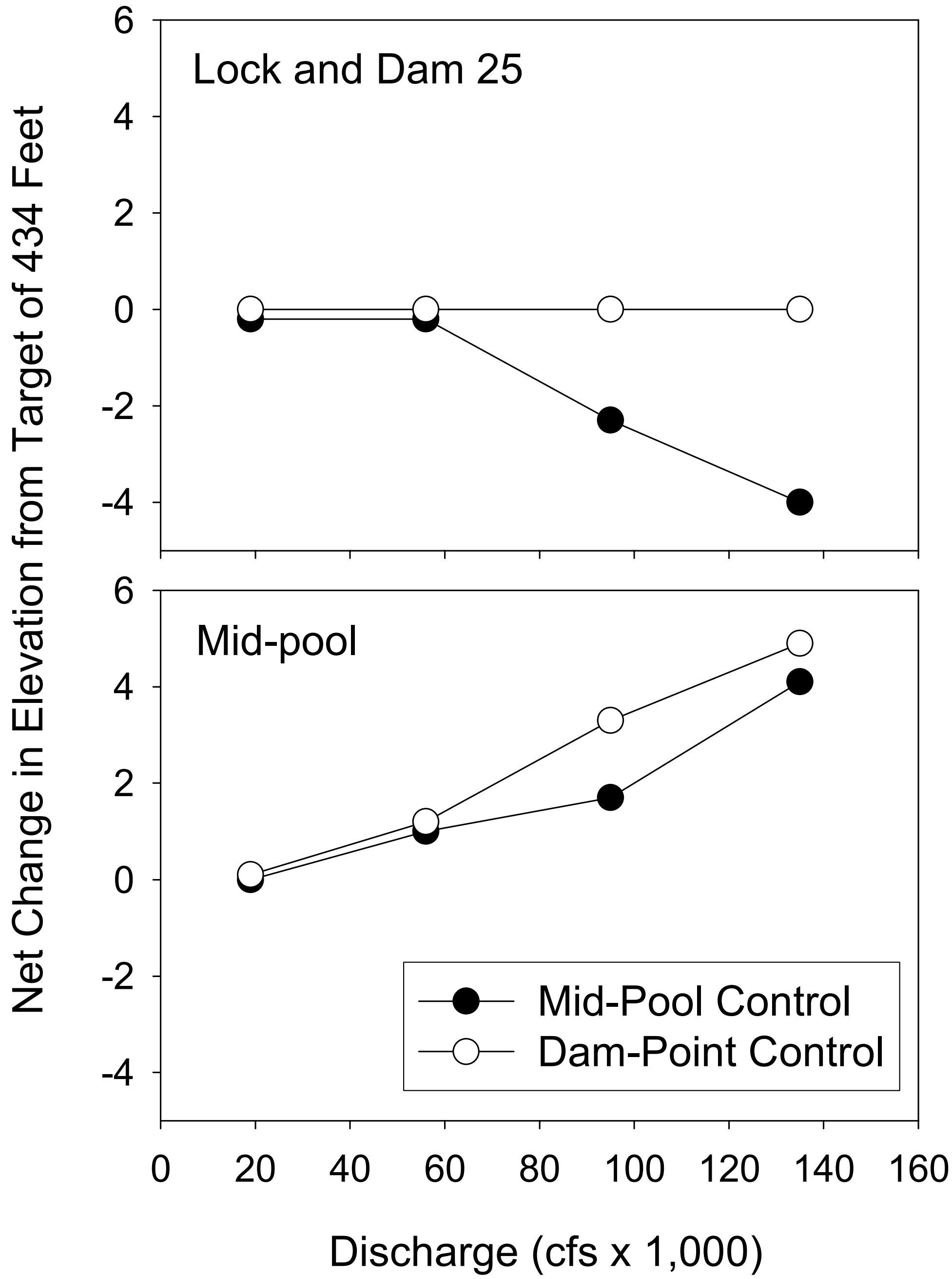

Figure 1-3. Predicted water level changes (deviation from full pool, 434 feet) at lower pool (Lock and Dam 25) and mid-pool (Mosier Landing, RM 260) reaches as a function of discharge and a water level control point at either mid-pool or at L&D 25 (predictions from modeling done in Wlosinski and Rogala 1996).

Chapter 2

Introduction

Water levels in Pool 25, Mississippi River, are currently managed at a mid-pool control point located near Mosier Landing at river mile 260.3 by the U.S. Army Corps of Engineers (USACE), St. Louis District. To maintain a 2.7-m navigation channel, water levels are managed between 434 - 437 ft at Mosier Landing and from 429.7 - 434 ft at Lock and Dam 25 over a specific range of discharges. During moderate flows, the pool becomes "tilted" when gates are lifted to maintain water levels at the mid pool control point. When discharge exceeds values manageable through operation of Lock and Dam 25 (often occurring during spring high water events) all gates at the dam are raised out of the water and the river is said to be at "open river." Spring flood waters may recede to an elevation of 429.7 at Lock and Dam 25. This elevation, also referred to as "maximum drawdown," is the maximum drop in water level that will still allow navigation in a 2.7-m channel (L&D 25 Water Control Plan). If the discharge continues to fall, the pool is regained based on discharge rates. Typically, the Corps starts to regain pool when the discharge causes the water level at Mosier Landing to fall below 437.0 feet.

Resource agencies recognize the need to work in conjunction with the USACE to improve hydrologic conditions for biota within the constraints of a multi-use system (Woltemade 1997). Given the real estate constraints that the St. Louis District operates under, the L&D has limited control over the timing of maximum drawdown during open river conditions. However, there is some flexibility in how water levels are managed during the return of the river to the target pool elevation following maximum drawdown. From 1994 to 1999, the time period conducive to water-level management ranged from approximately 38 to 57 days.

The operational goal of Environmental Pool Management (EPM) is to maintain relatively low, stable water levels in the lower portion of the pool, following maximum drawdown in the spring, in order to better simulate the natural hydrograph (Figure 2-1). Under some circumstances (e.g., high discharges), water levels in Pool 25 may descend to elevations greater than 2.0 feet below the target pool elevation, a consequence of the mid-pool control method that is not a goal of EPM. When implementing EPM, water levels typically are held 0.5 to 2.0 feet below the target pool elevation (434 ft in Pool 25) at the lock and dam for a minimum of 30 days (Atwood et al. 1996). Environmental Pool Management prolongs the dry phase during the summer growing season for nonpersistent wetland vegetation; the target time period for implementing EPM is between May 1 to July 30 (Atwood et al. 1996). Vegetation produced by EPM is primarily found in backwaters located in the lower reach of the pools. The target reflood rate (rate of return to full pool) of EPM is to not exceed 0.2 ft per day so that vegetation is not overtopped too quickly (Atwood et al. 1996). The St. Louis District implemented EPM in 1994 on Pools 24, 25, and 26. Early investigations of mudflats exposed via EPM showed lush production of nonpersistent wetland vegetation consisting mainly of millet, chufa, and smartweeds (Atwood et al. 1996).

Many ecological benefits are expected from EPM. On a large scale, the management regime could provide system-wide benefits by consolidating substrates and re-establishing wetland biogeochemical processes. The Mississippi River is a major migratory route for waterfowl, and moist-soil plants provide food sources directly through seed and tuber production and indirectly by increasing invertebrate abundance (Fredrickson and Taylor 1982). Aquatic plants provide habitat for many Upper Mississippi River (UMR) fishes (Janecek 1988);

therefore, fish are expected to benefit from EPM. However, other than the response of plants (Atwood et al. 1996), very little data exist that evaluate biotic responses to EPM.

During 1998 through 2002, intensive evaluation of EPM in the lower reach of Pool 25 was conducted by personnel from the Southern Illinois University Cooperative Wildlife Research Lab, the Fisheries and Illinois Aquaculture Center, and the Southern Illinois University Department of Zoology (Figure 2-2). The goals of this research were to quantify plant responses, estimate above ground seed production, and measure invertebrate, waterbird, fish, and water quality responses to the two major attributes of EPM: the lowering of water levels and vegetation production. We also compared the aquatic communities in the lower reach of Pool 25 to those at mid-pool (Figure 2-3). Herein, we use terms such as "drawdown" and "drawdown regime" to include the overall water-level pattern during summer and subsequent vegetation production in a given year, and they are not necessarily referring to EPM. We make this distinction because some results are reported and discussed herein that were not due to the practice of EPM as currently defined. For example, due to high discharge throughout the summer of 1999, Pool 25 remained on tilt for an extended period of time. Nonetheless, water levels lowered and vegetation was produced (the basic components of EPM); therefore, biological information gathered in 1999 can still be used to evaluate EPM and assess/refine how it is practiced. Working within the premise of adaptive management, our primary objective was to monitor the biotic responses to drawdown regimes in Pool 25 to provide insights into how EPM can be implemented to maximize ecosystem benefits.

Our data collection corresponded to three very different drawdown regimes during summer of 1999, 2000, and 2001 (Figure 2-4; also see Table 1-1). Water levels remained ≥ 2.0 ft below full pool for 54 days and ≥ 4.0 ft below full pool for 21 days during summer of 1999.

This large magnitude and duration drawdown resulted in a high vegetation response. The drawdown regime of 2000 was a distinct contrast to that of 1999 in that water levels were allowed to return to full pool in late July following a 31-day drawdown period (to ensure adequate water levels for fish), and little to no vegetation remained following re-flood. Between mid June and mid August 2001, drawdown exceeded 1 foot for 48 days and 2 feet for 26 days; water levels were $\geq$ 3.0 ft below full pool for only 7 days. In our view, this drawdown regime closely resembled the criteria of EPM, and resulted in a moderate amount of vegetation compared to 1999 and 2000. Consequently, we studied three drawdown regimes that can be characterized according to vegetation production as exceptional (1999), minimal to none (2000), and moderate (2001).

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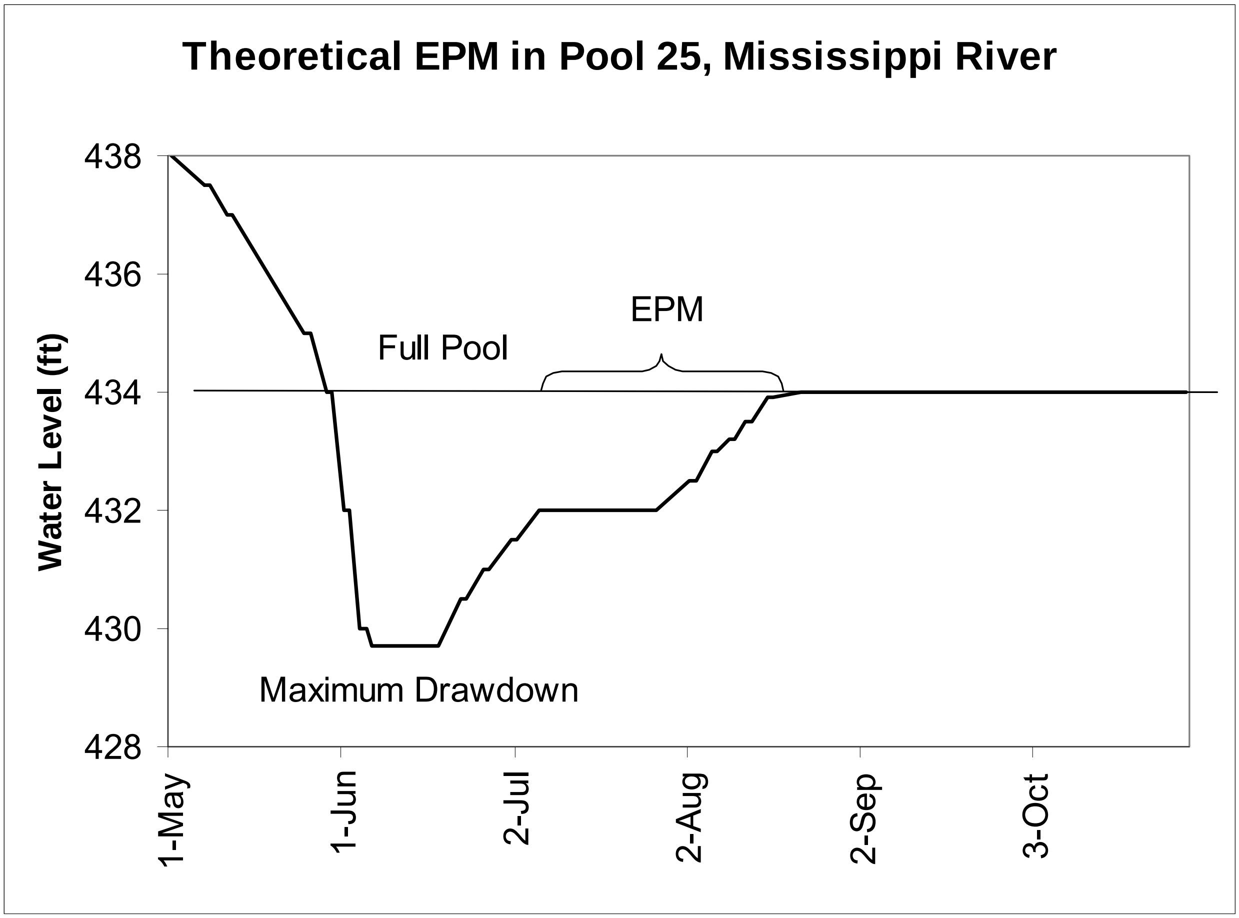

Figure 2-1. A theoretical depiction of Environmental Pool Management (EPM) in Pool 25, Mississippi River.

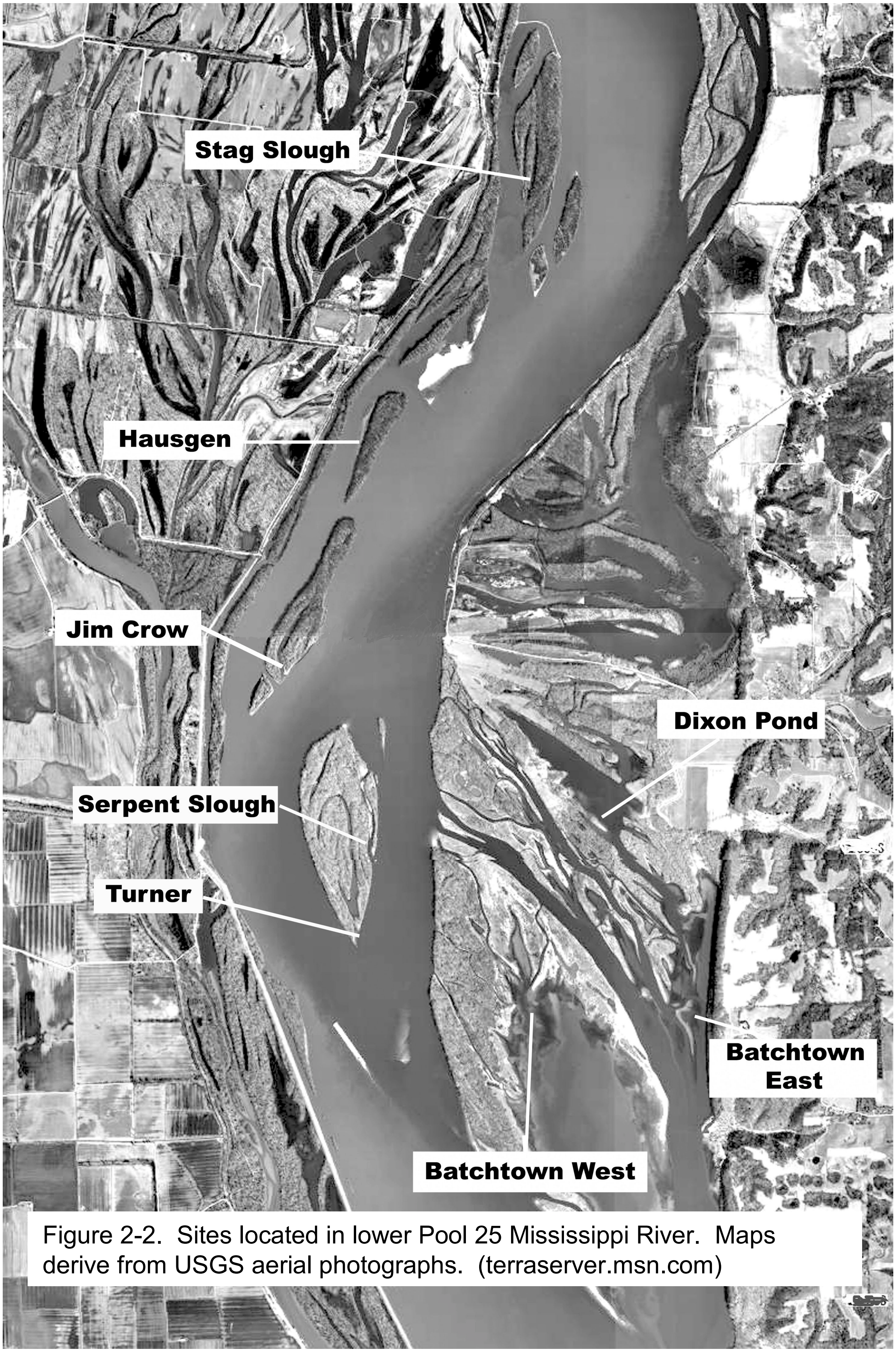

Figure 2-2. Sites located in lower Pool 25 Mississippi River. Maps derive from USGS aerial photographs. (terraserver.msn.com)

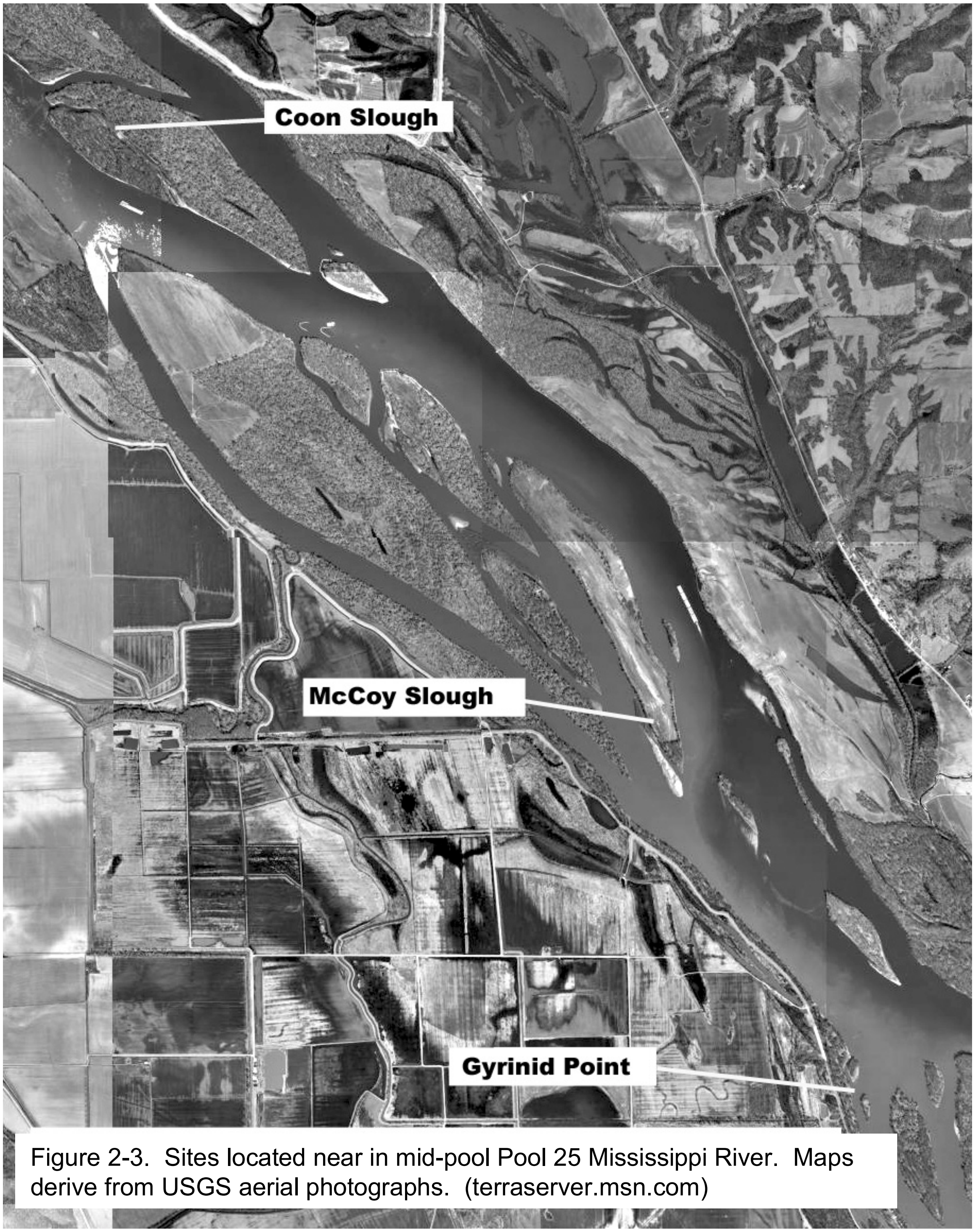

Figure 2-3. Sites located near in mid-pool Pool 25 Mississippi River. Maps derive from USGS aerial photographs. (terraserver.msn.com)

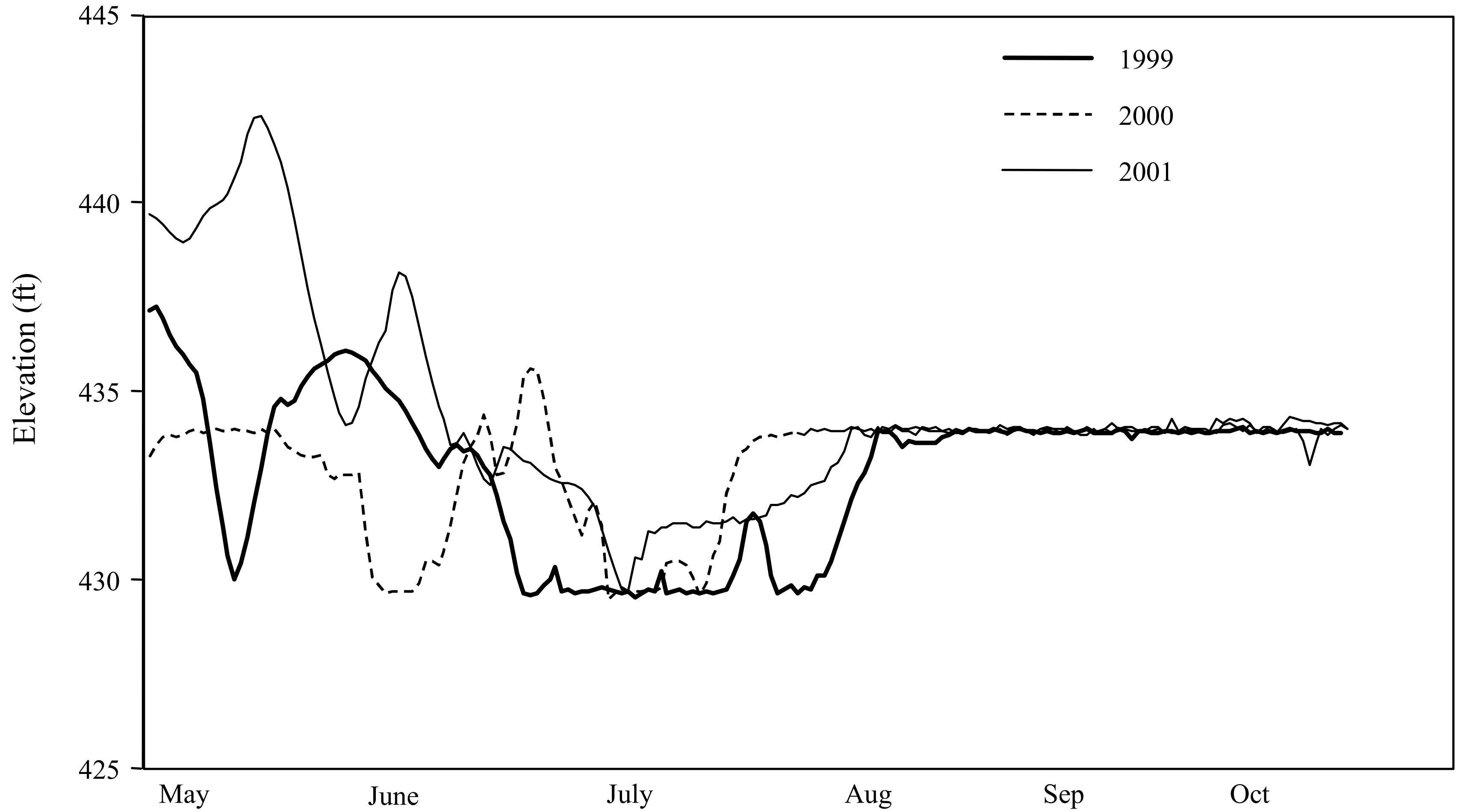

Figure 2-2. Water levels during 1999, 2000, and 2001 in Pool 25 Mississippi River (also see Figure 1-2).

Chapter 3

Waterbird Responses to Environmental Pool Management

Primary Principle Investigator: B.D. Dugger

Objectives

1. Quantify emergent vegetation response and estimate above ground seed biomass produced by EPM in the lower end of Pool 25, UMR.
2. Characterize waterbird use of EPM created habitat.

Methods

Plant Community Response

In 2001 we collected vegetation composition data along the same 11 transects established in the Batchtown area during 1999. We used a GPS to relocate the origin of each transect; each sample location was relocated using notes from 1999 that specified the distance of each plot from the transect origin. Data collection occurred during 1-3 August 2001, beginning 3 weeks post-drawdown. Transects were oriented perpendicular to the shoreline and followed the elevation gradient. A single 0.5-m^2 sample square was placed along the transect at locations that corresponded to 5, 20, 35, 50, and 75-cm water depth, relative to full pool (434.0 ft NGVD). In each square, we recorded occurrence and percent cover for each plant species. We also collected plant composition data in three 0.5-m^2 sample squares for each vegetated study plot included in the paired-plot field experiment designed to measure responses to EPM. Nomenclature follows Scott and Wasser (1980) and Mohlenbrock (1986). We used frequency of occurrence and percent cover to describe changes in community structure along the elevation gradient

(Daubenmire 1959). We used a Kruskal-Wallis nonparametric analysis of variance (ANOVA) to test for differences in percent cover related to elevation.

Seed Biomass

We estimated seed biomass of *Polygonum lapathifolium*, *Cyperus erythrorhizos*, *Leptochloa panicoides*, *Leersia oryzoides*, *Echinochloa crusgalli*, and *E. muricata* in the Batchtown area using techniques developed by Laubahn and Fredrickson (1992). This technique uses regression equations for each plant species or a group of 2 or 3 species, which is the case for *Echinochloa*, to estimate seed biomass from plant and seed head dimensions. We collected data on seed biomass on 19-20 September 2001, beginning approximately 3 weeks after normal pool elevation was resumed and after the dominant species could be differentiated and had set seed. Data were collected from 120 25x25-cm plots located randomly along the same transects used to collect plant species composition data. Number of stems and seed heads were recorded for each plant species rooted within the sampling frame. A representative plant for each species within the sampling column was chosen for measuring seed head and plant dimensions. We measured the straightened height of the plant (m), height of the seed head (cm) along the rachis from the lowest rachilla to the top of the straightened seed head, and base diameter of the seed head (cm) along the lowest seed producing rachilla (Laubahn and Fredrickson 1992).

Waterbirds

Shortly after dewatering in 2001, we conducted a waterbird count including the Batchtown area and Turner Island. The goal of the survey was to evaluate the use of exposed mudflats, which can provide important habitat for species like shorebirds. Shoreline habitats were counted from a boat where possible; the interior horseshoe area of Batchtown was surveyed on foot. Using binoculars and a 20-60x spotting scope, we recorded all waterbirds seen along the survey route.

Winter/spring waterfowl surveys and behavioral observations were conducted during February and March 2001 and 2002. Surveys were conducted from the bow of a boat in the main channel, side channel, and backwater areas downstream of Jim Crow Island and in Batchtown. Surveys of the slough on Jim Crow Island and the reservoir on Turner Island were conducted on foot. A route was chosen to minimize flushing birds to areas not yet surveyed. Wind speed (km/h), wind direction, air temperature (^{0}C), precipitation, and percent cloud cover (10% interval) were recorded prior to beginning surveys. Total number, species, and location (vegetation vs. open water) of waterfowl were noted during each survey period in 2001; however, residual vegetation was not detectable during 2002, so habitat could not be assigned.

For the 6 week survey period, we report the number of waterfowl-use days for dabbling ducks, diving ducks, and Canada geese (*Branta canadensis*). Waterfowl-use days were calculated by multiplying the mean waterfowl count of 2 consecutive surveys by the number of days between surveys then summing all weeks in the 6 week survey period. To test for guild-specific differences in waterfowl-use days between habitats, we used a two-tailed Mann-Whitney *U*-test with Normal Approximation and Continuity Correction.

We conducted behavioral observations to construct time-activity budgets of waterfowl during spring migration. Observations were conducted between sunrise and sunset (Central Standard Time) from duck blinds located throughout the study area using a 20-60x spotting scope. Individuals were selected for observation by aiming the spotting scope at the center of a flock and selecting the bird in the center of the field of view. Focal individuals were observed for 15-30 minutes with behavior recorded at 10-sec intervals. If the original bird swam out of view, before the end of the 30-min session, the observation was adjusted to the nearest neighbor of the same species and sex as the focal-individual (Losito et al. 1989). Behavioral categories included: feeding, comfort (preening, drinking, wing flapping, head shaking), locomotion (swimming, flying), agonistic (chasing, biting), courtship (including copulation), loafing (inactive and resting), and alert. All data were dictated into a portable microcassette recorder then sequentially transcribed to data sheets.

Results

Plants

We did not collect plant data during summer 2000; however, water levels during the 2000 growing season were variable (Fig. 3-1), and although low water conditions did occur during the growing season, the rise in water levels during June and July prevented successful establishment of emergent macrophytes (B. Dugger pers. obs.). In 2001, we detected 17 species of wetland plant. Species of sedge in the genus *Cyperus* were most common, occurring in 68.5% of samples, followed by pigweed (*Amaranthus sp*.) and millet (*Echinochloa crusgalli* and *E. muricata*; Table 3-1). Smartweeds (*Polygonum* sp.), the most common plant in 1999, were found in 30% of plots in 2001. Mean percent cover was independent of elevation for all species (*P*'s > 0.54, Table 3-2).

Including all species, our estimate for seed biomass was 3,336 $\pm$ 3,737 lbs/ac (Table 3-3). Excluding red-root nutsedge (*Cyperus erythrorhizos*), the estimate was 1,552 $\pm$ 1,739. Similar to 1999, the estimate for red-root seems overinflated. It is possible that morphological characteristics of the plant in Batchtown (plants were extremely large) fall outside the range of morphology measures used to develop the regression equations; thus biasing the estimate high. Estimates of sedge biomass were high in both 1999 and 2001; however, while *Polygonum* dominated in 1999, *Echinochloa* and *Leptochloa* (sprangletop) dominated in 2001.

Waterbirds

We found 7 species in the survey area during the summer waterbird survey; however, overall abundance was low (138 total individuals). Killdeer (*Charadrius vociferus*) were most common (105 individuals) followed by Great-blue Heron (*Ardea herodias*, 14), Pectoral Sandpiper (*Calidris melanotos*, 7), Canada Goose (*Branta canadensis*, 7), Great Egret (*Ardea alba*, 3), Ring-billed Gull (*Larus delawarensis*, 2) and Spotted Sandpiper (*Actitis macularia*, 1).

We conducted 3 surveys during the period 23 Feb – 13 Mar 2001 and 6 surveys during 16 Feb – 29 Mar 2002. We detected 9 and 16 species of waterfowl in 2001 and 2002, respectively (Table 3-4). Wigeon (*Anas Americana*), Gadwall (*A. strepera*), Northern Shoveler (*A. clypeata*), Ring-necked Duck (*Aythya collaris*), and Lesser Scaup (*A. affinis*) were absent in 2001 compared to 2002. Similar to 1999 and 2000, mallards (*Anas platyrhynhcos*) were most abundant in 2002. Common goldeneye (*Bucephala clangula*) and Common Mergansers (*Mergus merganser*) were the most abundant species in 2001. Total waterfowl counts were 1,706 and 7,847 and Duck Use Days (DUD) were 9,109 and 59,346 in 2001 and 2002, respectively (Table 3-5). Dabblers contributed most to DUD in 2002, whereas Divers accounted for most DUD during 2001 (Table 3-5). Total DUD in 2001 were the lowest of all years surveyed. In contrast, diving duck DUD in 2001was the second highest of the 4 years surveys were conducted. Comparing species composition with the diving guild among years, Common mergansers and goldeneye were the most abundant species in 2001; whereas in 2002 and previous years, fresh water diving ducks in the genus *Aythya* were most abundant.

Discussion

Hydrology

As summarized in Chapter 2, our data collection corresponded to three very different drawdown regimes during summer of 1999, 2000, and 2001 (Figure 3-1; also see Figure 2-4 and Table 1-1). Water levels remained ≥ 2.0 ft below full pool for 54 days and ≥ 4.0 ft below full pool for 21 days during summer of 1999. This large magnitude and duration drawdown resulted in a high vegetation response. The drawdown regime of 2000 was a distinct contrast to that of 1999 in that water levels were allowed to return to full pool in late July following a 31-day drawdown period, and little to no vegetation remained following re-flood. Between mid June and mid August 2001, drawdown exceeded 1 foot for 48 days and 2 feet for 26 days; water levels were ≥ 3.0 ft below full pool for only 7 days. In our view, this drawdown regime closely resembled the criteria of EPM, and resulted in a moderate amount of vegetation compared to 1999 and 2000. Consequently, we studied three drawdown regimes that can be characterized according to vegetation production as exceptional (1999), minimal to none (2000), and moderate (2001).

Vegetation

In 2000, low water did not persist long enough for emergent macrophytes to survive after reflooding. As a result, vegetation was absent from all but the highest elevations. Although circumstantial, failure of plants in 2000 does support the current EPM plan that specifies water level be stabilized for 30 days prior to reflooding. Plant diversity in 2001 was the highest recorded in pool 25 since USACE began implementing EPM. Differences between 2001 and work conducted by Wlosinski et al. (unpubl. data) in the mid 1990's are consistent with the later and more prolonged dewatering period in 2001; both hydrological characteristics promote species diversity (Fredrickson and Taylor 1982). Species diversity in 2001 was only slightly higher than 1999 (Feddersen 2001).

Although species richness was similar between 1999 and 2001, community composition differed between years. In 1999, the emergent macrophyte community was dominated (in descending order) by smartweeds, millet, and red-root nutsedge. In 2001, the plant community was dominated by red-root nutsedge, pigweed, and millet. Smartweed, which occurred in 93% of plots in 1999, only occurred in 27% of plots in 2001. Differences in plant species composition are consistent with between year differences in hydroperiod. Dewatering in 2001 started later and was more prolonged than 1999. This difference favors species that respond to later drawdowns like red-rooted nutsedge, and pigweed (Fredrickson and Taylor 1982). Polygonum can germinate under both early and late dewaterering conditions; however, it does best with early drawdowns and seed production is highest with early drawdowns.

Importantly, although plant species composition changed between years, the biomass of seeds produced for migratory birds was similar. Red-root nutsedge was the dominant seed producer in both years, but in 1999 smartweed produced biomass equal to the sedge. In 2001, smartweed seed production was much lower, but this was offset by large increases in production of millet and sprangletop seeds.

As in 1999, we failed to find evidence for stratification of plant species along the elevation gradient in Batchtown in 2001. We might have predicted better zonation in 2001 because the dewatering occurred over a longer time. Failure to find zonation may stem from the timing of our plant surveys, which occurred while plant species were relatively young. Data from this study on seed production compared with community composition at the time of our surveys suggests that the plant community may change significantly from early growth to maturity. Zonation patterns may have been more evident had our plant surveys occurred later in the growing season. We considered timing of surveys when we designed the study, but chose to sample plants relatively early because of uncertainty regarding the timing of reflood and the risk of not getting any data if reflooding occurred earlier than anticipated.

The reduction in smartweed abundance in 2001 was more apparent in the seed biomass estimates for the dominant species. Relative contributions of *Echinochloa* and *Leptochloa* to total seed production was higher in 2001 than would have been predicted from data on percent occurrence and percent cover (Tables 3-1,3-2,3-3). That is, although percent cover and percent occurrence of both these species was lower in 2001 than 1999, their contribution to fall seed production was much higher. One possible explanation is that both benefited from reduced competition for space with *Polygonum* (a broadleaf robust plant) and more young plants reached maturity and were able to set seed.

Decreased smartweed abundance in 2001 changed the structural characteristics of the plant community, which may have influenced the biota. Smartweed has robust stalks and large leaf surface area, and was frequently the only plant recognizable during waterfowl surveys in spring. The persistence of residual smartweed vegetation may have important consequences to aquatic organisms using Batchtown in spring. Furthermore, a decrease in the number of Polygonum leaves and stems may influence shading patterns, which may change solar energy inputs into the water column. If such changes occurred, this could change key water quality parameters and influence the structure and composition of the aquatic community.

Woody species (e.g., maple and willow) occurred less frequently in 2001 than 1999, suggesting germination conditions favorable to these species were not present in 2001. Furthermore, absence of woody species suggests that although favorable germination conditions may occur during some years (e.g., 1999), conditions may not be favorable for long term survival of these species throughout most of Batchtown. Fall high water and winter scouring are two factors likely leading to high mortality of woody species seedlings. River bulrush (*Scirpus fluviatilis*) did occur at lower elevations in 2001. Like woody species, river bulrush has the potential to form monotypic stands and lower habitat quality for many organisms. Continued monitoring of river bulrush is advised if monitoring of EPM continues.

Waterbirds

Use of mudflat habitats by migratory shorebirds will depend, in part, on the timing of habitat availability relative to migration chronology of each species. In summer 2001, the timing of dewatering was after most shorebirds had migrated through the area in spring, but prior to their arrival in August and September. As a result, few shorebirds were counted on the survey. However, variability in the timing of dewatering (either earlier or later), may provide substantial habitat benefits for shorebirds. Great-blue Herons (*Ardea herodea*) breed along the Mississippi River, and individuals were present during the summer survey. Waders like herons and egrets depend on aquatic vertebrates like fish and amphibians for survival and successful reproduction. Management activities that enhance the backwater fish community will benefit these species. Although difficult to quantify with a survey, Soras (*Porzana carolina*) appeared to use newly flooded vegetation extensively in fall. During September trips to the study area, it was common to induce a dozen birds to vocalize by clapping hands (B. Dugger pers. obs.). Rails are secretive species that frequently go unnoticed in traditional waterbird surveys. However, they use shallowly flooded emergent macrophytes during migration, thus EPM likely provides ideal habitat during fall migation.

Waterfowl numbers were markedly higher in spring 2002 than 2001. The sharp decline in DUD in 2001 was attributed to an almost complete absence of dabbling ducks (species in the genus *Anas*). Dabblers totaled 217,271, 168,442, and 55,670 DUD during 1999, 2000, and 2002; however only 1,392 during 2001. Although we conducted half the number of surveys in 2001 (which does influence DUD calculations), doubling the DUD to more closely match calculations from other years does not account for the difference in DUD between years. The sharp decline in dabbling duck numbers in spring 2001 is consistent with the lack of a vegetation response in summer 2000. Without seeds or extensive residual vegetation (which can benefit aquatic macroinvertebrates), the food resources required by species like Mallard and Northern Pintail were unavailable. However, this interpretation is confounded by the fact that periods of low water occurred during spring migration 2001 (Fig. 3-2). Additionally, cold temperatures caused some ice formation on what water remained. Both factors significantly reduced habitat availability in spring 2001; thus a decrease in dabbling duck numbers can not soley be attributed to low food production in summer 2000.

Diving ducks, which prefer open water habitats and use resources likely less influenced by the presence of emergent macrophytes, were more abundant in 2001 than 2002. The most common species (Common merganser and Common Goldeneye) wintered on the study area and their increased presence in 2001 may be attributable to colder winter temperatures that delayed the onset of spring migration. Consistent with this explanation, 88% of all Common mergansers and 96% of all goldeneyes were counted during the first survey week (Dugger unpubl. data). Considering Lesser scaup, Canvasback, Ring-necked Ducks, and Redhead separately, their abundance was down in 2001 like dabbling ducks.

The reasons for reduced waterfowl abundance in 2002 compared to 1999 and 2000 are less clear. Good seed production occurred during 2001 and water levels were relatively stable during spring 2002; both factors should have promoted use by waterfowl. However, vegetation established during the 2001 growing season did not persist above water in spring of 2002; if the presence of residual vegetation is a cue for habitat selection by migratory ducks, use of Batchtown could be lower than anticipated. Alternately, among year differences in fall use of Batchtown by ducks may have caused greater food depletion in fall of 2001 than 1999 or 2000, leaving fewer resources for spring migrants in 2002. More generally, the distribution of waterfowl during migration is influenced by a variety of local and regional factors (Havera 1999).   Species like Mallard and Pintail (the 2 most abundant ducks in our study) have flexible migration strategies that are influenced on a large scale by regional weather patterns. Thus, while local population size is determined in part by local habitat conditions, regional habitat availability (e.g., upper Midwest) as determined by rainfall or ice thaw patterns can be equally if not a more important factor.   Without further detailed investigations, we would not recommend viewing the waterfowl response in spring 2002 as an indication that habitat produced during summer 2001 were inferior to habitats produced in 1998 or 1999 (the growing seasons before springs of 1999 and 2000).

Because residual vegetation did not persist above the water line in 2002, we were unable to determine habitat use by birds (open water vs. residual vegetation) during our surveys.

However, general observation indicated the dabbling ducks were using the shallow water areas in Batchtown where residual vegetation likely occurred (underwater). We were unable to collect waterfowl behavior data during either spring. In 2001, this was caused by the almost complete lack of waterfowl. In 2002, although peak dabbling duck numbers did reach 1,000 birds, they were scattered in small groups throughout Batchtown. Efforts at observation were thwarted as observer disturbance (while trying to get close enough to birds to observe) caused birds to flush and relocate to other regions in the study area.

Conclusions

Data from all three years support the idea that EPM improves habitat quality for migratory birds. Plant species with known value to waterfowl continue to dominate the vegetation community, and there is no strong evidence suggesting encroachment by woody species has reduced coverage of herbaceous macrophytes. Similar to managed moist-soil impoundments, common throughout the Midwest, by changing the timing, magnitude, and duration of the dewatering period in Pool 25 it appears managers can influence the vegetation structure and still produce seed biomass important to waterfowl. This fact provides some flexibility for how managers implement EPM. For example, altering drawdown characteristics to improve conditions for fish does not by definition have to lower the quality of the habitat for waterfowl.

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Table 3-1. Percent occurrence of plant taxa in sample plots located along an elevation gradient (cm) relative to full pool (132.3 m. NGVD, n = 11 transects). Data are for the Batchtown area of Pool 25, Mississippi River, during summer 2001. Data for 1999 are provided for comparison.

	Elevation below full pool (cm)					Overall	
Taxa	5	20	35	50	75	2001	1999
Cyperus[c]	72.7	72.7	72.7	80.0	54.5	68.5	76.7
Amaranthus rudis	63.6	63.6	63.6	50.0	54.5	59.3	16.4
Echinochloa[b]	63.6	72.7	54.5	45.5	27.3	52.7	79.5
Polygonum[a]	27.3	18.2	36.4	20.0	36.4	27.8	93.2
Eragrostis hypnoides	19.2	19.2	19.2	10.0	36.4	20.4	4.1
Digitaria	27.3	27.3	9.1	9.1	0.0	14.5	--
Ipomea purpurea	18.2	9.1	18.2	9.1	9.1	12.7	6.9
Leptochloa panicoides	9.1	9.1	18.2	9.1	9.1	10.9	23.3
Ludwigia	--	--	19.2	9.0	18.2	9.1	--
Leersia oryzoides	9.1	9.1	9.1	9.1	9.1	9.1	20.6
Lindernia dubia	--	9.1	--	--	18.2	5.5	23.3
Scirpus	--	--	--	18.2	9.1	5.5	--
Xanthium strumarium	9.1	--	9.1	--	--	3.6	11.0

[a]Includes *Polygonum lapathifolium* and *P. pennsylvanicum*
[b]Includes *Echinochloa crusgalli* and *E. muricata*
[c]Includes *Cyperus esculentus* and *C. erythrorhizos*
[d]Includes *Populus* spp., *Acer* spp., and *Salix* spp.

Table 3-2. Mean percent cover (SE) of plants along an elevation gradient (cm) relative to full pool (132.3 m. NGVD) in Pool 25, Mississippi River, during summer 2001. Transects (n = 11) were oriented perpendicular to the shoreline. Mean values for 1999 provided for comparison.

	Elevation below full pool						
Taxa	5	20	35	50	75	Overall	1999
Cyperus	12.5(8.0)	9.0(4.1)	13.4(6.4)	8.1(4.4)	13.7(7.3)	11.3(2.7)	2.2
Amaranthus	7.3(5.8)	6.3(3.1)	9.5(3.9)	6.4(4.2)	3.8(2.6)	6.6(1.8)	7.9
Echinochloa	24.3(10)	13.9(6.1)	6.2(3.8)	4.4(2.4)	5.2(5.0)	10.8(2.9)	7.6
Polygonum	1.5(1.0)	2.1(1.8)	3.9(2.4)	2.4(2.0)	4.0(2.4)	2.8(1.3)	11.0
Eragrostis hypnoides	tr[d]	7.3(5.1)	2.5(2.3)	tr	2.9(2.2)	2.7(1.2)	2.5
Digitaria	3.4(3.2)	tr	tr	tr	--	tr	--
Ipomea purpurea	tr	tr	tr	tr	tr	tr	11.4
Leptochloa panicoides	tr	2.7(2.7)	tr	tr	tr	tr	3.3
Ludwigia	--	--	tr	tr	tr	tr	--
Leersia oryzoides	tr	7.3(7.3)	3.6(3.6)	tr	1.4(1.4)	2.6(1.6)	1.2
Lindernia dubia	--	tr	--	--	tr	tr	4.7
Scirpus fluviatilis	--	--	--	12.3(8.8)	7.3(7.3)	3.9(2.3)	--
Xanthium strumarium	tr	--	--	--	--	tr	5.1

[a]Includes *Polygonum lapathifolium* and *P. pennsylvanicum*
[b]Includes *Echinochloa crusgalli* and *E. muricata*
[c]Includes *Cyperus esculentus* and *C. erythrorhizos*
[d] tr = < 1.0%

Table 3-3. Estimated seed biomass (kg/ha) produced by moist-soil plants measured at Batchtown in Pool 25, Mississippi River, during summer 2001 [mean(SD)]. Seed biomass estimates were calculated using regression equations developed by Laubahn and Fredrickson (1992).

	Year	
Taxa	2001[a]	1999[b]
Cyperus erythrorhizos	1,783(2,868)	1,264(133)
Echinochloa	909(742)	114(21)
Leersia oryzoides	36(119)	12(5)
Leptochloa panicoides	486(953)	3(3)
Polygonum lapathifolium	120(239)	1,148(66)
Total	3,336	2,542

[a] *n* = 120 plots
[b] *n* = 232 plots

Table 3-4. Waterfowl-use days and their relative distribution (%) between vegetated and open water habitats for guilds of waterfowl (dabblers, divers, geese) surveyed weekly (n = 6, except 2001 n = 3) in the lower reach of Pool 25, Mississippi River, during late February through early April 1999 - 2002.

		Year			
Guild	Habitat	1999	2000	2001	2002
Dabbler	Vegetation	213,226 (98)	166,540 (99)	618 (44)	--
	Open Water	4,045 (2)	1,902 (1)	774 (56)	--
	Total	217,271	168,442	1,392	55,670 (100)
Diver	Vegetation	479 (5)	31 (1)	0 (0)	--
	Open Water	9,433 (95)	2,725 (99)	7,717 (100)	--
	Total	9,912	2,756	7,717	3,676 (100)
Geese	Vegetation	986 (79)	266 (69)	76 (69)	--
	Open Water	258 (21)	119 (31)	34 (31)	--
	Total	1,244	385	110	1,525 (100)

Table 3-4 cont…

		Year			
Guild	Habitat	1999	2000	2001	2002
Total Waterfowl	Vegetation	214,691 (94)	166,837 (97)	618 (7)	0 (0)
	Water	13,736 (6)	4,746 (3)	8,491(93)	59,346 (100)
	Total	228,427	171,583	9,109	59,346

Table 3-5. Total waterfowl counted during weekly surveys of vegetated and open water habitats in lower Pool 25, Mississippi River during late Feb - early Apr 1999, 2000, 2001, 2002.

Species	1999	2000	2001	2002
Branta canadensis	283	74	23	192
Anas platyrhynchos	18,812	13,169	270	2,948
Anas acuta	16,585	5,684	4	2,476
Anas americana	30	102	0	31
Anas strepera	105	1,234	0	154
Anas crecca	394	3,872	14	1,444
Anas clypeata	65	1,046	0	48
Anas discors	52	334	0	0
Anas rubripes	25	0	0	0
Aix sponsa	7	21	0	24
Mergus merganser	0	100	541	71
Lophodytes cucullatus	0	9	7	5
Aythya americana	85	200	0	0
Aythya collaris	405	1,429	0	21
Aythya valisineria	52	201	170	215
Aythya affinis	1,102	827	0	155
Bucephala albeola	0	11	13	14
Bucephala clangula	0	9	681	19
Oxyura jamaicensis	0	45	0	30
Total	38,002	28,292	1,706	7,847

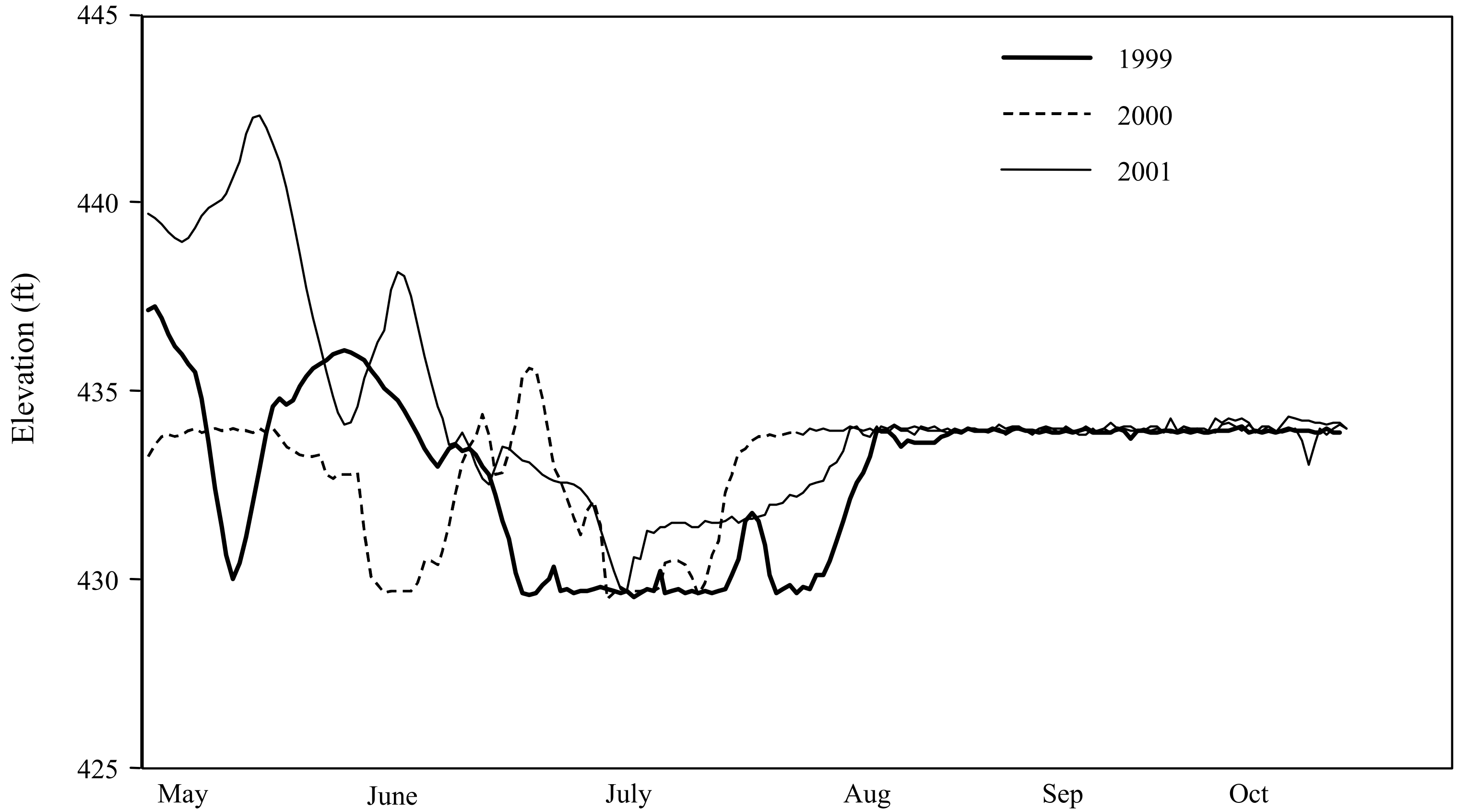

Figure 3-1. Hydrograph for lower Pool 25, Mississippi River from 1999 to 2001. Daily stages were obtained from Lock and Dam 25 (Upper) Winfield, MO.

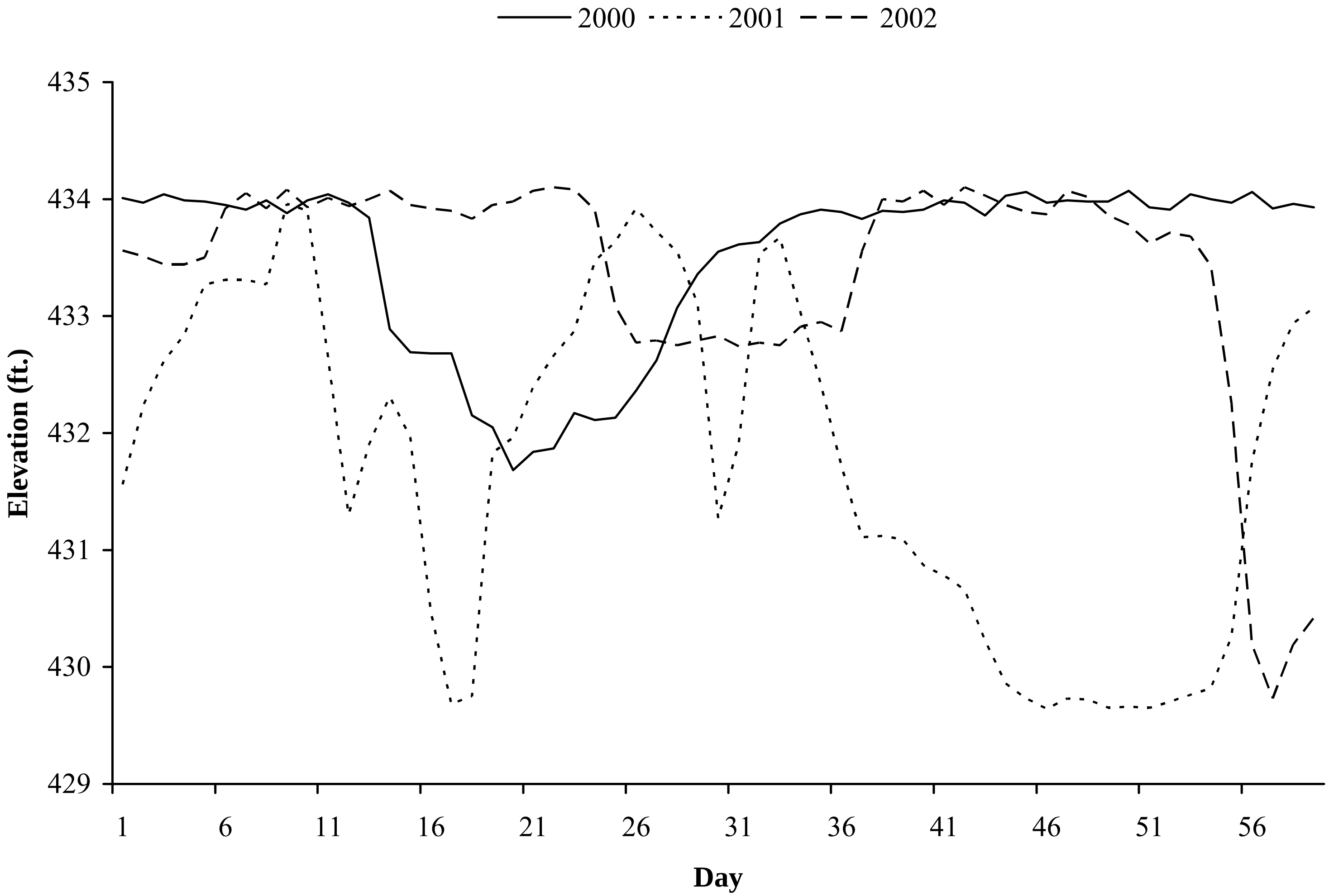

Fig. 3-2. Hydrograph for lower Pool 25, Mississippi River during springs 2000, 2001, and 2002. Day 1 = 15 February of each year; Day 58 = 15 April; Feb 29th 2000 deleted. Daily stages were obtained from Lock and Dam 25 (Upper) Winfield, MO.

Chapter 4

Fish and Water Quality Responses to Environmental Pool Management

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Introduction

Environmental Pool Management (EPM) prolongs the exposure of mud flats during the late spring/summer growing period to stimulate the growth of emergent vegetation (Chapter 2). Little is known about how fish respond to EPM. Benefits to fish are expected, as at least 84 species in the Upper Mississippi River reportedly use aquatic plants for reproduction, nursery habitat, cover, and feeding grounds (Janecek 1988). J. Wlosinski and B. Atwood analyzed seine samples taken in multiple habitat types from 1986 to 1996 in Pools 24, 25, and Melvin Price Pool, and concluded that maintaining lower water levels during the summer did not negatively affect small, nearshore fishes (pers. comm.). During fall 1997, fish were seined in vegetated and adjacent nonvegetated areas in Pools 24, 25, and 26 to examine fish use of emergent vegetation; this study documented use of the vegetation by 25 fish species, and indicated the vegetation was providing habitat for small forage fishes, particularly the emerald shiner, *Notropis atherinoides* (Heidinger et al. 1998).

As a continuation of the research effort initiated by Heidinger et al. (1998), we studied the fish response to EPM in Pool 25 during 1998 to 2002. Our specific objectives were 1) to quantify fish use of vegetation by comparing fish assemblage structure in the vegetation to that in adjacent, experimentally devegetated areas of similar water depth and velocity, 2) to determine how vegetation affects water quality, 3) to compare fish assemblages in backwaters located in the lower pool to fish assemblages in backwaters located at midpool that are not influenced by EPM, and 4) to determine if residual vegetation provides habitat for young-of-year (YOY) fishes during spring.

Methods

Fish and Water Quality Response to Flooded Vegetation in Fall

Study Sites - Reconnaissance during fall 1998 indicated most, if not all, vegetation produced by EPM was located in the lower impounded reach; therefore, all sampling occurred in the lower portion of Pool 25. During fall 1998, we chose four study sites based on evidence (presence of emergent vegetation) that the area was affected by EPM (see Figure 2-2, Chapter 2) (Table 4-1). Two sites (Batchtown West and Batchtown East) were established in an extensive, shallow backwater complex located in the Batchtown State Fish and Waterfowl Management Area, Calhoun County, Illinois. Historically, most of the EPM-induced vegetation in Pool 25 has occurred in the Batchtown area. Batchtown West was located in the northern end of a shallow, expansive bay characterized by soft substrates, and was more vulnerable than the other sites to wind-induced wave action. Batchtown East was situated near the limestone bluffs of the Illinois-side (eastern) river bank. In addition to Batchtown, relatively small acreages of vegetation were produced on islands near the main channel. Study sites were established on the downstream tip of Turner Island and within a shallow slough on Jim Crow Island. All four sites were sampled during fall 1999-2001. Additionally, another site in the Batchtown area (Dixon Pond) was sampled during fall 2000 and 2001, and a site on Hausgen Island was sampled during fall 2001.

We established two study plots (one vegetated and one devegetated) at each site. At Batchtown West, Batchtown East, Jim Crow, and Turner Island, the center of each study plot was marked with rebar during fall 1998. The rebar was used as a reference for locating plots and spatially standardizing sampling in subsequent visits. Devegetated plots were intended to simulate conditions in shallow backwater habitats without vegetation, and they were of similar depth (< 1.0 m) and water velocity (0 cm/sec) to vegetated areas. During summer of 1999 and 2001, sites were visited following initial exposure of the mudflats to delineate devegetated plots. At this time, coarse woody debris was removed from all plots. Plots to be devegetated were treated with Rodeo® herbicide applied with a backpack sprayer. Applications were made prior to re-flood and sufficient to eliminate vegetation from plots in 1999 and 2001; no herbicide application was made in 2000 because vegetation was absent (see below). We devegetated an area of 400 m^2 in reference to the center of each plot. Also, plots were devegetated out to the anticipated adjacent open water area so that water quality parameters (e.g., turbidity) would better reflect absence of vegetation. Devegetated plots were completely devoid of vegetation prior to re-flood. Distance between plots at a given site ranged from 10 to 30 m.

Hydrology during the summer drawdown period varied during our study, resulting in a different plant response to EPM each year. Therefore, the physical characteristics of our study plots (e.g., amount of vegetation, type of vegetation, and size of vegetated plots following re-flood) were different each year. Water levels in 1999 remained 0.5 ft below an elevation of 434 ft (full pool) for 69 days over the period from mid June to mid August and were greater than 2.0 ft below full pool for 54 days (see Figure 1-2, Table 1-1 in Chapter 1). This large magnitude drawdown resulted in an exceptional vegetation response. Study plots following re-flood in 1999 contained large amounts of vegetation, primarily smartweeds, and vegetated plot sizes were at least 400 m^2. In 2000, following a second maximum drawdown event in late June, water levels remained 0.5 ft below full pool for 31 days and below 432 ft for 22 days before returning to full pool in early August (Figure 1-2, Chapter 1). Mud flats were not exposed for a long enough period in the summer of 2000 to allow plant germination and/or enough growth to withstand inundation except at the highest elevations. No vegetation was present in study plots ("vegetated" or "devegetated"), following re-flood in 2000, at Batchtown East and Batchtown West; an exception was that a sparse amount of vegetation was present throughout both study plots at Jim Crow, and a narrow (1 m) band of inundated vegetation existed along the shoreline at Turner Island (vegetated plot size < 20 m^2). In summer of 2001, the drawdown lasted from mid June to mid August, and water levels remained 0.5 ft below full pool for 58 days and 2.0 ft below full pool for 27 days (Figure 1-2, Chapter 1). Following re-flood, vegetation was present in all vegetated plots except at Batchtown East. Vegetation in 2001 was not as dense as in 1999 and was primarily comprised of millet and chufa. During this study, the vegetation response to EPM was different across years and sites, and this variability was reflected in the fish response.

Fish Sampling in Experimental Plots - Preliminary sampling of sites during fall 1998 indicated the shallow, vegetated, backwater habitats were utilized primarily by very small (< 20 mm) fish. Therefore, we had to devise a method to effectively quantify small-bodied fish abundances in two structurally dissimilar habitats (vegetated and devegetated). Both pop nets and seining have been successfully used to take a quantified fish sample in shoreline zones with and without aquatic vegetation (Dewey et al. 1989, Killgore et al. 1989). We sampled study plots in 1999 using both pop nets and seining. Following our first year of data collection (1999), we evaluated the two sampling techniques in order to select the single best method to employ the next two years (2000 and 2001).

Gear evaluation/selection - We constructed twelve pop nets (a modified design from Dewey et al. (1989)) having a 1-m^2 buoyant frame of polyvinyl chloride pipe (3.18 cm diameter), an open bottom anchored on two sides with steel conduit pipe, and a mesh size of 1.6 mm. Pop nets were placed collapsed on the substrate (depth < 1 m), left overnight, and remotely triggered to "pop" the next day to collect fish in a 1-m^2 column of water extending from the bottom to the water's surface. Nets were harvested by pulling together the steel pipe which pursed the netting; nets were then lifted from the water and fish removed. These nets could be set and harvested effectively by two people. Plots at all sites were sampled with three popnets each on every sampling trip in fall 1999. Paired-plots at a given site were sampled on the same day/night interval.

In addition to pop nets, we sampled fish in plots with a 3.66-m seine (1.6-mm mesh). Two seine hauls/lifts, each 10 m long, were made in the center of devegetated plots (total area sampled = 72.2 m^2), and five kicksets were made in the center of vegetated plots (total area sampled = 72.2 m^2). The use of a series of stationary kicksets was the best method for sampling with a seine in the dense emergent vegetation. Kicksets were accomplished by holding the deployed seine stationary while one person "kicked" vigorously into the seine starting 4 m away. Fish collected by seining in a given plot were pooled together as one sample per plot. Fish collected by both pop nets and seining were fixed in 10% formalin and identified/enumerated in the laboratory

We used total numbers of fish (all dates and sites combined) during fall 1999 to evaluate our two sampling techniques (seining and pop nets) (Table 4-2 and 4-3). Overall rank of the seven most common fish species in the vegetated plots was significantly correlated between seine and pop net samples (N = 7; Spearman's $r_s = 0.82$; $P = 0.023$). In devegetated plots, concordance of ranks was not found in the seven most abundant species (N = 7; Spearman's r_s = 0.68; $P = 0.094$), but ranks were perfectly correlated (Spearman's $r_s = 1.0$) when the emerald shiner, *Notropis atherinoides*, and orangespotted sunfish, *Lepomis humilis*, were left out of the analysis. Pop nets were not as efficient at sampling the emerald shiner in devegetated plots (7-fold difference), probably because of a combination of their pelagic nature, schooling behavior, and larger size relative to other YOY (young-of-year) cyprinids in the habitats. Pop nets may have attracted YOY orangespotted sunfish by providing structure to a homogeneous habitat otherwise devoid of structure. In summary, seining and pop nets similarly quantified fish abundance in the vegetation, but seining was more effective in the devegetated plots. Seining generally captured more fish and more fish species in both vegetated and devegetated plots, and seven species were captured exclusively by the seine (Table 4-3). Therefore, we only used seining to sample fish during fall 2000 and 2001.

Standard sampling at all sites/plots, across all years (1999, 2000, and 2001), involved seining (72.2 m^2) the center of each study plot as previously described (i.e., hauls/lifts and/or kicksets). This standard approach facilitated comparisons of data across years, because samples were taken in the exact same location. Due to the dynamic nature of the physical structure of the plots across years, we also collected additional seine samples in 1999 and 2000 to better describe the fish response to EPM. In fall 1999, two seine hauls, each 10 m long, were made at the natural deep edge of the vegetation at Batchtown East and Batchtown West during five sampling trips. The seine was pulled parallel with the vegetated edge with one brail approximately one meter within the vegetation. Adjacent seine samples were taken in the deep portion of the devegetated plot on three sampling trips in 1999. The centers of all study plots in 2000 were devoid of vegetation (with the exception of Jim Crow); therefore, we collected additional seine samples (area sampled = 72.2 m^2) in the nearest band of inundated, shoreline vegetation at four sites (Turner Island - 3 samples, Batchtown East - 1 sample, Batchtown West - 1 sample, and

Dixon Pond - 2 samples). These additional samples were kept separate from fish collected directly within the center of the study plots.

All fish were fixed in 10% formalin in the field, and identified and counted in the laboratory. Total length (TL) was measured (nearest 1.0 mm) on up to 50 individuals of each species per sample. Individuals were classified as adults or YOY based on length-frequency histograms and total lengths reported in Becker (1983) and Pflieger (1997). Voucher specimens have been catalogued in the Southern Illinois University at Carbondale Fluid Vertebrate Collection.

Water Quality Sampling in Experimental Plots - Point-in-time measurements of major water quality variables (dissolved oxygen (DO), temperature, pH, conductivity, and turbidity) and water depth were made in each plot on each sampling trip in 1999, 2000, and 2001 between 0830 and 1600 hr. Dissolved oxygen concentration (accuracy = $\pm$ 0.2 mg/L) and temperature (accuracy = $\pm$ 0.2 ^{O}C) were quantified with a Yellow Springs Instrument (YSI) Model 95 digital meter at approximately 5 cm below the water's surface and 5 cm above the substrate if water depth exceeded 30 cm. A Hanna Instruments pHep®2 pocket-sized meter was used to measure pH ($\pm$ 0.1 pH). Dissolved ion concentration (µS/cm) was measured with a YSI Model 33 conductivity meter. Conductivity values were standardized to 25 ^{O}C according to Wetzel and Likens (1979). Conductivity and pH were measured at approximately 5 cm below the water's surface. A 10-ml water sample was taken in each plot, and turbidity determined in the laboratory with a Chemtrix Type-12 turbidimeter. A meter stick was used to measure water depth.

<u>Electrofishing</u> - Boat electrofishing (one pilot, one dip netter) was conducted within the large bay in Batchtown (in the proximity of Batchtown West) in mid October of 1998 - 2001. Electrical current was supplied by a 3-phase 5 KW generator producing 240 volts AC. Fish were netted

with a dipnet having a mesh size of 6.4 mm. Electrofishing was conducted for a total of 1 hr (sampling effort was 1.5 hr in 1998) along the deepest edge of the vegetation within the large bay in Batchtown, near Batchtown West (see Figure 2-2, Chapter 2). If vegetation was not present (e.g., in 2000), we sampled along the shoreline at a water depth similar to other years. One sample was taken each year at mid day in the 2nd-3rd week of October. Fish were identified, measured, counted, and released.

<u>Midpool/Lower Pool Comparison</u>

The comparison of fish communities in sites at midpool and lower pool was a mensurative experiment examining the fish response, over the same time period, in backwater habitats influenced by EPM (lower pool) and habitats not influenced by EPM (midpool). As a result of hydrologic operation of Pool 25 with a midpool control point, habitats near midpool were relatively unaffected by the summer drawdown, resulting in no emergent vegetation production in these habitats. Expansive, shallow backwaters did not exist at midpool. Therefore, we selected backwaters associated with islands (primarily sloughs) at both midpool and lower pool for this experiment. Three sites were established at midpool (McCoy Slough, Coon Slough, and Gyrinid Point; see Figure 2-3, Chapter 2), and four sites were established at lower pool (Turner Island, Jim Crow, Serpent Slough, and Stag Island Slough) (Table 4-4; also see Figure 2-2, Chapter 2). All sites were contiguous with the main channel at full pool elevation (434 ft).

Fish were sampled on three occasions: July 2001 (midpool: 25-26 July; lower pool: 27-28 July), October 2001 (midpool: 14-15 October; lower pool: 5-7 October), and April 2002 (midpool: 5-6 April; lower pool: 6-8 April). A representative, 30-m shoreline reach was established at each site. Fish were collected in the study reach using three sampling techniques:

seine (3.7 m long; 1.6 mm mesh), modified fyke net (box = 0.91 m by 1.83 m by 0.61 m; lead = 12.8 m; bar mesh = 9.5 mm), and cast net (2 m diameter; 1.0 cm mesh). Seining was conducted for three minutes (net-in-water time), and time was recorded with a stopwatch. A seine haul consisted of pulling the seine (6-8 m) from offshore straight in to the shore and lifted; we usually made 8-10 seine hauls in a study reach in three minutes. Fish collected by seining were fixed in 10% formalin in the field and identified/enumerated in the laboratory. One modified fyke net (Hubert 1996) was set overnight at each site per trip. Nets were set with the single lead running perpendicular from the shoreline to the rectangular frame positioned offshore. Ten throws of the cast net were made at each site/trip by the same person (SRA). Fish collected in fyke nets and via the cast net were identified/enumerated in the field and released. Habitat (depth, substrate type, and cover type) was measured using a point-transect method; habitat data were collected at five points on five transects (total data points = 25) within the 30-m study reach. Major water quality parameters were measured as previously described.

Residual Vegetation

Researchers suspect that residual vegetation produced by EPM during the previous fall will benefit fish by providing spawning and nursery habitat (Atwood et al. 1996), but data are lacking. We found that presence and amount of EPM-induced residual vegetation in backwaters during late spring/early summer was highly variable and unpredictable. In most cases, residual vegetation remaining through the winter and into spring was in the form of dead smartweed stalks. In spring of 1999 (8 June and 20 June), we sampled fish directly within residual vegetation at Batchtown East, Batchtown West, Turner Island, and Jim Crow. During late spring/early summer of 2000-2002, we studied spatial and temporal patterns of YOY fishes at 14 sites in lower Pool 25. There were seven samples during this time period where residual vegetation comprised greater than 50% of total available cover: 2000 - Little Stag Island (21 May), Little Hole Backwater (21 May and 24 June); 2001 - Church Slough Backwater (30 May and 16 June); 2002 - Hausgen Island (6 May and 22 May). All fish were collected with a 3.66-m seine (1.6 mm mesh). Fish were fixed in 10% formalin and identified/enumerated in the laboratory.

Data Analysis

The general statistical model for analyzing results of the paired-plot experiment was a three-way analysis of variance (ANOVA) with the three main factors being Plot (two levels: vegetated, devegetated), Year (three levels: 1999, 2000, 2001), and Site (four levels: Jim Crow, Turner Island, Batchtown East, Batchtown West). For each dependent variable, we first constructed the full factorial model (all possible interactions of the main effects) without the third-order interaction (Plot*Year*Site) because appropriate replication was lacking. Following rules outlined in Montgomery (1991), degrees of freedom and sums of squares of interaction effects were pooled with the error degrees of freedom and sums of squares when alpha was > 0.25, and a new model constructed. Three-way ANOVA models were used to test the null hypotheses that fish abundance, species richness, diversity, and water quality were equal among plots, years, sites, and interactions of these variables.

Fish abundance values represented mean number of fish per sampling trip captured by seining only. Species richness was the total number of species in a given plot and site within a year. Diversity of the fish community in a given plot and site within a year was represented by the Shannon diversity index (H') calculated using the following formula:

$$H' = -\sum p_i \ln p_i$$

where p_i is the proportional abundance of the ith species (n_i/N). This widely used diversity index is richness dominated and moderately sensitive to sample size, with values usually ranging between 1.5 and 3.5 (Magurran 1998). Prior to analysis of pH, we converted the values to hydrogen ion concentration as recommended by Wetzel and Likens (1979). Post-hoc testing was performed with the Tukey-Kramer Honestly Significant Difference test (Tukey HSD).

We also examined overall community similarity between samples (study plots, electrofishing samples, midpool/lower pool samples) with the Percent similarity index (PSI) and Spearman's rank correlation coefficient (r_s). Percentage similarity was calculated with the following formula:

$$P = \sum \text{minimum } (p_{1i}, p_{2i})$$

where P is the percentage similarity of sample 1 and 2 and p_{1i} and p_{2i} are the percentages of species i in samples 1 and 2, respectively. Percentage similarity index ranges from 0 to 100 with 100 indicating complete similarity (Krebs 1999). Spearman's (r_s) determined concordance of species ranks based on relative abundance (rank-order abundance) between two samples. This nonparametric correlation coefficient, which ranges from -1.0 to 1.0, is highly sensitive to sample size (number of species) and may perform better in low-diversity communities (Krebs 1999). Spearman's (r_s) was calculated including all species and including only common species. Using only common species is a more conservative approach since rare species will inflate the chance of finding a significant correlation. We defined "rare" species as those represented by < 10 individuals across samples. The Kolmogorov-Smirnov two-sample test was used to compare length distributions within selected fishes. In all tests, statistical significance was indicated by an alpha < 0.05, and marginal significance was recognized at alpha < 0.1.

Results

Paired-Plot Experiment

Water Quality - Water depth in the study plots was similar throughout the study (overall mean = 37.5 cm) (Table 4-5 and 4-6). Depth did, however, vary across sites (p<0.001) with deeper water at Batchtown East (55.7 cm) and Batchtown West (44.4 cm) than at Jim Crow (27.5 cm) and Turner Island (22.2 cm) (Tukey HSD). Water temperature (overall mean = 21.6 ^{O}C) and conductivity (overall mean = 415 μS/cm) did not differ between plots (Table 4-5 and 4-6). Significant variability in these quantities existed due to Year and Site (Table 4-5 and 4-6); however, the magnitude of differences in mean temperature (1 ^{O}C) and conductivity (22 μS/cm) were trivial from a biological standpoint. Values of pH ranged from 7.4 to 9.0 during the study and are within the tolerance range of riverine backwater biota (Table 4-6). Hydrogen ion concentration was marginally higher in vegetated plots (6.46 x 10^{-9} moles H^{+}/l) than in devegetated plots (4.3 x 10^{-9} moles H^{+}/l). There were also significant trends in pH due to Year and Site (Table 4-5), with pH being lower in 1999 and at Batchtown East (Tukey HSD). Turbidity ranged from 6 to 82 NTUs (Table 4-6), and was significantly higher in devegetated plots (32.9 NTUs) than in vegetated plots (25.6 NTUs) (Table 4-5 and 4-6). Turbidity also varied significantly across years and sites (Table 4-5), in which turbidity was highest in 1999 and lowest at Jim Crow (Tukey HSD).

Of the water chemistry parameters quantified, trends in dissolved oxygen were the most biologically relevant. Year, Plot, Site, Year*Plot, and Year*Site significantly affected dissolved oxygen concentration (Table 4-5 and 4-6). Dissolved oxygen was lowest in vegetated plots, particularly vegetated plots in 1999 (mean = 5.41 mg/l) (Tukey HSD). Among sites, lowest mean DO was at Batchtown East (mean = 7.06 mg/l) (Tukey HSD). However, it is of biological importance that DO values within the "Biotic crisis" range (Bain 1999) occurred. Dissolved oxygen values less than 3.0 mg/l were recorded in vegetated plots at Batchtown East, Batchtown West, and Turner Island in 1999 (Figure 4-1). At Batchtown East, low DO in the vegetated plot was a chronic problem throughout fall 1999, but DO increased over time at Batchtown West and Turner Island (Figure 4-1). The increase in DO was correlated with "laying over" of the vegetation due to a combination of wave and wind action and senescence.

Dissolved oxygen values < 4.0 mg/l were never found during the day in devegetated plots in 1999 or in any study plots in 2000 and 2001, including the shoreline vegetation sampled in 2000. In 2001, we determined if DO reached critically low concentrations during the night. On September 9 and October 7, 2001, we quantified DO in the study plots just prior to sunrise. On September 9, low DO (3.3 - 3.7 mg/l) occurred in the vegetated plots at Turner Island and Jim Crow. In an investigation at all sites, no evidence of critically low DO was found in vegetated (4.82 - 8.03 mg/l) or devegetated plots (8.29 - 13.8 mg/l) on October 7, 2001. In summary, low DO may have influenced fish use of vegetation in 1999. In 2001, dissolved oxygen concentration was probably only an issue for fish in the vegetation at night during August through September.

General Fish Assemblage Structure - A total of 41,065 fish, including 24 species, were collected in the backwater habitats of Pool 25 during fall of 1999, 2000, and 2001 (Tables 4-7, 4-8, and 4-9). Of this total, 34,177 fish (23 species) were collected within vegetation. The fish assemblage was dominated numerically by the minnow family Cyprinidae (12 species, 56% of total abundance) and Poeciliidae (1 species, 41% of total abundance). Two of the minnows were the exotic species common carp and grass carp. Based on length-frequency histograms and reported age/length relationships in other populations, a majority of fish collected in the vegetation were YOY (Table 4-10).

There was a significant (p<0.05) effect of Site, Year, and Site*Year on fish abundance (Table 4-11). Fish abundance was lowest in 2000 (Tukey HSD) when vegetation was not present in study plots (Figure 4-2A). Over all years and plots, more fish were collected at Jim Crow (517.2) and Turner Island (370.2) than at Batchtown East (28.6) and Batchtown West (71.5) (Tukey HSD). Higher numbers were collected at Batchtown East in 1999 (the only year vegetation was present at this site) than in 2000 and 2001 (Figure 4-3A). At Batchtown West and Turner Island, abundance tended to be higher in 2001, when vegetation production was moderate and DO was not limiting (Figure 4-3A). No statistical effect of Plot*Year on abundance was detected (p=0.12); however, overall mean abundance was approximately four and two times higher in vegetated plots than devegetated plots in 1999 and 2001, respectively (Figure 4-2A). General trends in fish abundance were higher numbers of fish at island sites and higher numbers in the vegetation, particularly in 1999 and 2001.

Species richness varied significantly (p<0.05) due to Site, Year, Plot*Site, and Site*Year (Table 4-11). Among years, richness was high in 1999 (8.4), moderate in 2001 (6.4) and low in 2000 (4.8) (Tukey HSD) (Figure 4-2B). Number of species was highest at Jim Crow (10.5), moderate at Turner Island (6.3) and Batchtown West (5.8), and lowest at Batchtown East (3.3) (Tukey HSD) (Figure 4-3B). At most sites, species richness was generally higher in 1999; however, species richness tended to be higher at Jim Crow in the two years (2000 and 2001) when the drawdown was not of such great magnitude and duration (Figure 4-3B). A significant Plot*Site effect occurred because richness tended to be higher in the vegetated plot at Batchtown West and Turner Island, but a higher number of species was found in the devegetated plot at

Batchtown East (Tukey HSD) (Figure 4-3B). In general, species richness was highest at most sites in 1999 and 2001, when vegetation was present in the study plots. Overall species richness was higher in the vegetated plot at two of four sites.

Plot, Site, and Year had no significant effect on diversity (Shannon's Index) (Table 4-11). Diversity was low (< 1.5) in all study plots across all years, probably due to the combination of relatively low species richness and high percent dominance of samples by one or two species. Year had a marginally significant effect (p=0.055) on similarity (PSI) of fish assemblages between study plots (Table 4-11). Similarity was very low in 1999 (26.6), high in 2000 when vegetation was absent (74.6), and moderate in 2001 (53.7).

Species-level Response - Significant concordance of ranks of the most common fish species was found between study plots across sites in 2000 (N = 8, $r_s = 0.95$, $P < 0.001$), but ranks were discordant (uncorrelated) between study plots in 1999 (N = 8, $r_s = 0.50$, $P = 0.207$) and 2001 (N = 10, $r_s = 0.56$, $P = 0.093$). The western mosquitofish, spotfin shiner, emerald shiner, and channel shiner were numerically dominant in both study plots in 2000 (Table 4-8). Vegetated plots in 1999 were numerically dominated by the channel shiner, western mosquitofish, and spotfin shiner, while the devegetated plots were dominated by the emerald shiner, western mosquitofish, and channel shiner (Table 4-7). The largest difference in ranks between plots in 1999 was that the emerald shiner was the most abundant species in devegetated plots, but it was the sixth most abundant in vegetated plots. In 2001, the spotfin shiner, emerald shiner, and channel shiner were abundant in both study plots, but the river shiner was relatively more abundant in devegetated plots (fourth ranked) than in vegetated plots (tenth ranked) (Table 4-9).

Low to moderate assemblage similarity (PSI) and Spearman's r_s indicated the fish response to vegetated and devegetated plots varied at the taxonomic (species) level in 1999 and 2001, but not in 2000 (no vegetation was present in plots). Also, patterns of assemblage similarity and concordance of ranks varied between sites within a given year (Tables 4-12 and 4-13). Therefore, we constructed a series of three-way ANOVA models to examine the effects of Year, Site, Plot, and interactions of these variables on abundance of the nine most common fish species.

Except for the channel shiner and orangespotted sunfish, abundance of most common species examined differed ($p < 0.10$) across years (Table 4-14). Underlying the yearly variation in abundance of each species were the individual responses to vegetation production and hydrology. Common carp, river shiner, and emerald shiner abundance was lowest in 2000, when the drawdown resulted in very little vegetation production and plots contained no physical structure (Figures 4-4A, 4-4B, and 4-5C). A similar response was observed in the western mosquitofish (low abundance in 2000) at all sites/plots except at Jim Crow, which accounted for 100% of the abundance in 2000 (Table 4-8). Conversely, abundance of orangespotted sunfish and bluegill tended to be low in 1999, but was high in 2000 and 2001 (Figure 4-6B and C). These sunfish species responded favorably to the higher water levels in 2000 and 2001 compared to 1999 (high magnitude and duration drawdown). In 2001, when the drawdown and vegetation response was moderate, abundance of the river shiner, bullhead minnow, spotfin shiner, and emerald shiner was high relative to other years (Figures 4-4B and C; 4-5A and C). During the three years of study, no common species experienced their lowest abundance in 2001.

There was evidence of selection of a particular study plot by some fish species. Abundance of the spotfin shiner, common carp, and western mosquitofish was higher ($p < 0.05$)

in vegetated plots (Table 4-14); this trend was evident for all three species in both 1999 and 2001, years when plots contained vegetation (Figures 4-5A, 4-4A, and 4-6A). Also, selection of vegetated plots was generally exhibited by the channel shiner in 1999 and 2001 (Figure 4-5B) and the bullhead minnow (Figure 4-5C) and bluegill (Figure 4-6C) in 2001. In some species, plot selection tended to be different between 1999 and 2001. For example, the emerald shiner (Figure 4-5C) and orangespotted sunfish (Figure 4-6B) were generally more abundant in devegetated plots in 1999, but in 2001, emerald shiner abundance was similar between study plots and orangespotted sunfish tended to be more abundant in vegetated plots. These two species may have been avoiding the dense vegetation (and low DO) characteristic of most sites in 1999. In summary, species responded differently to the paired-plot experiment due to differences in vegetation characteristics and hydrology among years and sites. To some extent, all nine species demonstrated a positive response to the vegetation. Even the emerald shiner, which tended to select devegetated plots, was only abundant in devegetated plots in 1999 and 2001, when vegetation was present in nearby plots.

Sample sizes were large enough for four species (emerald shiner, spotfin shiner, channel shiner, and western mosquitofish) to generate and compare length-frequency distributions between study plots. We found no significant difference in length-frequency distribution of emerald shiner between study plots in 1999 ($p>0.20$) or 2001 ($0.05<p<0.10$). A majority of emerald shiners collected in all plot/year combinations were 25-45 mm TL, indicating most fish using the shallow-water habitats were YOY probably spawned during early through mid summer (Figure 4-7). Length-frequency distribution of spotfin shiner was significantly different between study plots in 1999 ($p<0.001$) but not in 2001 ($0.10<p<0.20$). A similar pattern was observed for channel shiner where distribution of lengths between study plots differed in 1999 ($p<0.001$) but not in 2001 (p>0.20). The difference in 1999 for both species was due to the high relative abundance of YOY fish less than 20 mm TL present in the vegetated plots (Figures 4-8 and 4-9). Length-frequency histograms of western mosquitofish differed significantly between plots in both 1999 (p<0.001) and 2001 (p<0.001), the result of a preponderance of recently spawned fish (<20 mm TL) in the vegetated plots (Figure 4-10).

Additional Fish Assemblage Data

Due to the combination of dramatic interyear differences in EPM and variability in response of sites to EPM, our standard paired-plot sampling at the exact same location at 4 sites was not fully describing the fish response. Therefore, we opportunistically collected data in addition to our standard, paired-plot sampling in 1999 and 2000, and added sites to the experimental design in 2000 and 2001.

Vegetation edge in 1999 - Following re-flood during fall 1999, the Batchtown area contained a vast amount of inundated vegetation. Our fish collections within predetermined vegetated plots at Batchtown West and Batchtown East were well within (20-30 m) the interior of the dense vegetation. Fish and water quality were sampled at the deeper edge of these two sites because of the dramatic contrast in vegetation density and DO concentration between the interior of the vegetation. The edge habitat was approximately 20-30 cm deeper than the respective experimental plot. Of the major water quality parameters measured, only DO in the vegetated plot (interior) and vegetated edge were different. Unlike the respective vegetated plots, DO was never limiting at the vegetated edge at Batchtown East (4.68-7.88 mg/l) or Batchtown West (7.08-11.44 mg/l). Number of fish species (edge = 8.5; plot = 7.3) tended to be higher at the vegetated edge (8.5) compared with the respective vegetated (6.5) and devegetated

(8.0) plot (Table 4-7). Seine samples in the corresponding deep portion of devegetated plots contained fewer species (four-fold difference) than the vegetated edge (Table 4-7), indicating that increased water depth was not the primary factor driving the fish response to the vegetated edge.

Based on examination of rank-order abundance, the fish assemblage in the vegetated plot and vegetated edge was not correlated at Batchtown East (n = 10; Spearman's $r_s = 0.01$; $p = 0.984$) or Batchtown West (n = 10; Spearman's $r_s = 0.41$; $p = 0.277$) in 1999. At both sites, emerald shiner was more abundant and common carp and western mosquitofish were less abundant at the vegetated edge (Table 4-7). The largest difference in fish response between the interior of the vegetation and the vegetated edge was at Batchtown East, where DO was low in the interior of the vegetation throughout fall 1999 (Table 4-7). At Batchtown East and Batchtown West, sites that contained large amounts of vegetation in 1999, more fish species tended to use the edge of the vegetation, while the interior was used primarily by species tolerant of low DO (common carp and western mosquitofish).

Shoreline vegetation during 2000 - Following re-flood during fall 2000, there were only sparse amounts of vegetation available for fish due to drawdown hydrology that summer. Study plots generally contained no inundated vegetation, but plots at Jim Crow had very low densities of smartweed, millet, and rush. A narrow band (< 1 m thick) of inundated vegetation, comprised primarily of smartweeds, millet, and chufa did exist along the shoreline at Batchtown East, Batchtown West, and Turner Island. Maximum water depth within this band of vegetation was 25 cm. Therefore, vegetation production in 2000 was limited to only the very shallow land/water interfaces.

Mean fish abundance (2747.7 vs 53.0) and number of species (10.0 vs 2.6) was higher in the shoreline vegetation versus in the study plots (Figure 2A and B). Fish assemblage similarity was very low (0.3-8.6) between shoreline vegetation and respective study plots at Batchtown East, Batchtown West, and Turner Island (Table 4-12). Study plots at these sites were dominated numerically by the emerald shiner, while the western mosquitofish was very abundant (86% of total) in the shoreline vegetation (Table 4-8). The shoreline vegetation contained a relatively high number of species (Figure 4-2B); however, with the exception of the western mosquitofish, most species occurred at a lower density compared to density in vegetation in other years.

Additional study sites - During the second and third year of this study, two sites were added to the paired-plot experiment: Dixon Pond (2000-2001) and Hausgen Island (2001) (Table 4-1; Figure 2-2, Chapter 2). Dixon Pond, a backwater located within the Batchtown area, was added to provide additional information on the response of backwater fishes (e.g., sunfish) to EPM. Hausgen Island represented additional island fringe habitat similar to that of Turner Island. Data from these two sites were not included in previous statistical models. Similar to other sites, study plots during 2000 contained a different fish assemblage than the shoreline vegetation (PSI = 0.057) at Dixon Pond. The shoreline vegetation was dominated numerically by western mosquitofish, similar to other sites in 2000; however, abundance of bluegill and orangespotted sunfish was very high at Dixon Pond relative to other sites that year (Table 4-8). During 2001 at Dixon Pond, overall fish assemblages in study plots were moderately similar (PSI = 0.58), and ranks of species were marginally correlated (Table 4-13). Bluegill and orangespotted sunfish abundance tended to be higher (5-17 fold difference) in the vegetated plot (Table 4-9). Noteworthy was the large increase in abundance of the silverband shiner at Dixon Pond during 2001 relative to all other site/year combinations (Table 4-9).

At Hausgen Island during 2001, similarity of fish assemblages between study plots was relatively low (PSI = 0.42), and there was no strong evidence either way regarding rank-order abundance of species (Table 4-12). Abundance of spotfin shiner, bullhead minnow, and western mosquitofish tended to be higher in the vegetated plot, and there was some evidence the emerald shiner was selecting the devegetated plot (Table 4-9).

Electrofishing

We collected a total of 687 fish, including 21 species, during boat electrofishing in association with emergent vegetation produced in the large bay in Batchtown in October of 1998-2001 (Table 4-15). Total number of species captured was high in 1998 and 2001 (14 and 15, respectively), moderate in 1999 (8), and low in 2000 (4). Analysis of correlation of species' ranks (rare species excluded) revealed that samples were not concordant across years (N = 9; Kendall's $W = 0.003$; $p>0.50$). Primary differences between years was the high relative abundance of smallmouth buffalo in 1998 and 1999, and the high abundance of freshwater drum in 2000 (Table 4-15). Also, orangespotted sunfish and bluegill abundance was higher in 1998 and 2001 than in 1999 and 2000 (Table 4-15).

Midpool/Lower Pool Comparison

Fish collected by the three sampling techniques were pooled for each site/date for analysis and discussion. A total of 15,421 fish, including 34 species, were collected at midpool and lower pool sites during the sampling period (Tables 4-16, 4-17, and 4-18). Sampling during July 2001 was during the drawdown; whereas sites at midpool were still connected to the main channel during July, two of four sites at lower pool were completely dry. At the two sites containing water (Jim Crow and Stag Island Slough), the slough was isolated from the main channel and only a small, shallow (maximum depth = 9 and 17 cm, respectively) body of water remained. Rank-order abundance analysis (sites combined) revealed the fish community at midpool was not correlated with the fish community at lower pool (Jim Crow and Stag Island Slough only) (n = 10; Spearman 's $r_s = -0.36$; $p = 0.304$) in July 2001. The difference was due to the relative lack of minnow species and dominance of western mosquitofish in lower pool sites (Table 4-16). A large majority (79 - 98%) of spotfin shiners, river shiners, channel shiners, and bullhead minnows and a low percentage of emerald shiners (32%) collected in midpool sites were YOY less than 20 mm TL. This indicated midpool sites that remained contiguous to the main channel during summer were providing nursery habitat for many minnow species.

Sampling at midpool and lower pool during October 2001 and April 2002 was conducted near full pool elevation (434 ft); therefore, all sites contained water and were contiguous with the main channel at this time. In October 2001, all study reaches at lower pool sites contained emergent vegetation (percent coverage = 12 - 88%), but sites at midpool contained no vegetation. Fish assemblages at midpool and lower pool were similar during October 2001 based on analysis of ranks (n = 10; $r_s = 0.75$; $p = 0.013$). Minnows and sunfishes were relatively abundant at both locations during fall (Table 4-17). During April 2002, lower pool sites contained residual emergent vegetation (percent coverage = 4 - 64%), but sites at midpool did not. Rank-order abundance of species at midpool and lower pool were not correlated (n = 9; $r_s = 0.48$; $p = 0.187$). The most notable difference was the abundance of western mosquitofish at lower pool sites (second abundant) and their relative rareness at midpool sites (eighth abundant) (Table 4-18).

Residual Vegetation

Thirty-seven fish taxa from 11 families were collected in residual vegetation produced by EPM from 1999 to 2002 (Table 4-19). The family Cyprinidae was well represented with 16 species collected, three of which were exotic species (common carp, grass carp, and bighead carp). All but a few taxa collected in the residual vegetation were comprised primarily of late larvae and/or early juvenile individuals. Young of the mooneye, silver chub, emerald shiner, and logperch are not typically associated with vegetation in backwaters, but these fish were relatively abundant in our samples of the residual vegetation (Table 4-19). Three of the YOY species collected in the vegetation are considered "rare and uncommon" by the state of Missouri as of 2002: mooneye, silver chub, and blue sucker.

Discussion

During 1999, 2000, and 2001 in Pool 25, Mississippi River, different drawdown regimes resulted in hydrological and vegetation responses that affected fish assemblages and water quality. The high magnitude ($\geq$ 4.0 ft below full pool for 21 days) and duration (69 days $\geq$ 0.5 ft below full pool) drawdown of 1999 resulted in an exceptional vegetation response (primarily smartweed, chufa, and millet). In 2000, drawdown duration was shorter (31 days $\geq$ 0.5 ft below full pool) and the rate of re-flood (return to full pool) was high (9 days); this was done primarily to ensure that fish had adequate water levels. Subsequently, water levels remained relatively higher during summer 2000, and little to no vegetation was available for fish during fall. The drawdown regime of 2001 was intermediate relative to 1999 and 2000. In 2001, drawdown duration was relatively high ($\geq$ 0.5 ft below full pool for 58 days), but the magnitude ($\geq$ 3.0 ft below full pool for 7 days) was not nearly as great as in 1999, resulting in a vegetation community that was less dense and less dominated by smartweed compared to 1999 (see Chapter 3). Considering the range of possible drawdown regimes, we quantified the fish response near the extremes (1999: extremely high vegetation production; 2000: little to no vegetation production) and during an intermediate year (2001: moderate vegetation production).

Fish responded positively, in unique ways, to the drawdown regime in each year. The fish response cannot be generalized adequately by one single community metric (e.g., an increase or decrease in total abundance, richness, diversity, etc.). Rather, the species comprising the assemblage and their respective life histories and physiological tolerances must be considered. The two major outcomes of EPM are hydrological modification and resulting vegetation production, which influence fish assemblages (vegetation: Janecek 1988, Dibble et al. 1996; hydrology: Horwitz 1978, Poff and Allan 1995). We will explore how fish and water quality responded to these two outcomes of EPM, and conclude with an overall summary and management recommendations.

Vegetation

General response - Although water depth rarely exceeded 1.0 m and was occasionally less than 25 cm, an overwhelming number of fish used the inundated emergent vegetation during fall of 1999, 2000, and 2001. Most fish in the newly flooded vegetation were minnow species and western mosquitofish, probably using this habitat for cover and food. Minnow species provide important forage for piscivorous predators, including sport fishes and possibly waterbirds (e.g., herons). Western mosquitofish are denizens of shallow, vegetated backwaters, and are voracious consumers (capable of consuming 75% of body weight in food daily) of a wide range of prey, including dipteran larvae (Ross 2001). Through their feeding activity, mosquitofish and minnows are an important link between invertebrate production and the many higher-order consumers that eat them (e.g., fish, snakes, and birds; Becker 1983, Meffe and Snelson 1989).

Our paired-plot experiment was designed to not only quantify overall fish use of the vegetation, but to also determine the selectivity of specific species or life stages relative to shoreline areas without vegetation. Excluding data from Jim Crow, abundance and richness in study plots in 2000 (when no vegetation was present) was low compared to devegetated plots in 1999 and 2001, suggesting that fish assemblages in devegetated plots in 1999 and 2001 were not entirely representative of what the shoreline fish assemblage would be in the complete absence of vegetation. This "swamping" effect of devegetated plots during high vegetation years, due to their proximity to vegetation, demonstrates the importance of within-site habitat heterogeneity.

In the vegetation manipulation, we documented significant selection of the vegetation in three species (spotfin shiner, common carp, and western mosquitofish) and a trend for selection of the vegetation in the channel shiner, bullhead minnow, and bluegill. The emerald shiner was typically more abundant in devegetated plots, but abundance was generally low in study plots in 2000, indicating a general, negative response to complete absence of vegetation. Emerald shiner abundance was similar between study plots in 2001, when the vegetation was less dense than in 1999. In general, the emerald shiner tends to be an open-water pelagic fish (Becker 1983), and was attracted to the edge habitat created by devegetated plots.

Individuals of most species within the vegetation during fall were predominantly YOY. Recently spawned YOY western mosquitofish were more abundant in the vegetation versus devegetated plots. Also, YOY spotfin shiner and channel shiner selected vegetated plots over devegetated plots in 1999, and were also present in vegetation during 2001. Apparently, these

YOY fishes spawned late in the year benefited from the shallow, shoreline cover provided by emergent vegetation. The high invertebrate production (see Chapter 5), as a result of EPM-induced vegetation, should enhance overwinter survival of the late cohort of fishes produced during August and September by fueling growth and building energy reserves (Oliver et al. 1979, Cargnelli and Gross 1997). Enhancing survival of these prey fishes will ultimately benefit piscivores that often are of economic and conservation importance.

Variability in vegetation characteristics - During fall 1999, relatively large patches of vegetation occurred at Batchtown East and Batchtown West. Smartweed was abundant and persistent throughout the plots at both sites during fall 1999 (see Chapter 3). Low dissolved oxygen concentration (less than 3 mg/l) at both sites was in the "biotic crisis" range described by Bain (1999). Timing of sampling probably introduced some variation into DO measurements, but we sampled between the hours of 0830 and 1600. The lowest DO occurring at Batchtown East and Batchtown West was on a clear day at 1145 and 1600 hr., respectively; DO concentration should have been relatively high at these times if photosynthesis exceeded respiration. The low DO was most probably due to decomposition of emergent vegetation which swamped photosynthetic oxygen production. The dense vegetation also probably prevented wave action and subsequent atmospheric mixing, and it may have inhibited photosynthesis by phytoplankton (through shading) since DO was limiting.

Vegetated plots in Batchtown during 1999 were inhabited primarily by western mosquitofish and common carp, which are known to be relatively tolerant of low DO (Becker 1983). Although DO was chronically low at Batchtown East throughout 1999, it became adequate for fish (>5.0 mg/l) over time at Batchtown West (Bain 1999). However, the fish assemblage in Batchtown West did not change as DO concentration rose, suggesting additional factors were influencing fish use of the vegetation (e.g., vegetation composition or density). Stem density of smartweed was higher at Batchtown East and additional plant types not as resistant to inundation were a significant component of the plant community at Batchtown West. Open spaces created by the decomposition of plants less tolerant of inundation may explain why DO increased through time at Batchtown West.

Daytime dissolved oxygen was sufficiently high for fish during most of fall 1999 at Turner Island and Jim Crow. We did not quantify DO at night when it could have declined. Vegetation at these sites occurred in smaller patches than in Batchtown, and was comprised of a mix of plant species. Also, the vegetation at Turner Island was exposed to wave action that kept fresh, oxygenated water circulating. Due to the proximity of these sites to the main channel, vegetation patches were available to species that are typical in flowing water (e.g., channel shiner, spotfin shiner, and river shiner) as well as those in backwaters (e.g., western mosquitofish, and sunfish). Vegetation on Turner Island provided nursery habitat for numerous young channel shiners and spotfin shiners. Hence, the production of relatively small amounts of vegetation on islands likely provides habitat for a wider range of fish species than relatively isolated backwaters further from the main channel. Low DO concentrations may be less frequent in small patches of vegetation at exposed sites compared to large patches in shallow, low-gradient backwaters during years of high vegetation production (particularly smartweeds).

At Batchtown East and Batchtown West during fall 1999, more fish species used the deeper edge of vegetation patches compared to their interior. An additional four species were collected by boat electrofishing around the edge of the vegetation in Batchtown in 1999 that were not collected by seining. Many fish species congregate at edges of submergent vegetation. In particular, piscivorous fish use the edge as an ambush point (Killgore et al. 1989, Dibble et al.

1996). Piscivorous fish were not present, but minnow species and orangespotted sunfish tended to be more abundant at the vegetated edge compared to the interior.

Many organisms are attracted to edges (habitat transitions) because of the increase in heterogeneity caused by multiple habitat types in close proximity; this phenomenon is termed the "edge-effect" (Leopold 1933, Yahner 1988). The vegetated edge in Batchtown separated two relatively homogeneous environments: the open water and dense stands of smartweed. Unlike the interior of the vegetation, the edge offered cover and food without the problems of low DO and, potentially, too much structural complexity (e.g., high stem density may inhibit foraging). Our devegetated plots created additional edge, thereby attracting edge-dwelling species. To illustrate, the emerald shiner was the most abundant fish at both the vegetated edge and within the devegetated plot in Batchtown. The emerald shiner was very abundant in the vegetation in an earlier study (Heidinger et al. 1998), comprising 88% of fish captured; sampling effort in that study focused on the vegetated edge.

Increasing edge to benefit wildlife has been used by resource managers for the management of terrestrial game species (Leopold 1933). Investigators caution against the creation of too much edge because it could become a population sink, particularly for interior specialists (Yahner 1988). Increasing edge habitat in dense, homogeneous stands of emergent vegetation, such as existed in Batchtown in 1999, would probably benefit most fish by increasing foraging opportunities and potentially increasing DO concentrations within the vegetation. We increased edge through formation of our devegetated plots, which attracted some fish species that did not occur within the interior of the vegetation (e.g., orangespotted sunfish, emerald shiners, and brook silversides). Removal of vegetation and subsequent increases in edge already occur when duck hunters in the Batchtown area create open areas around duck blinds and cut boat lanes through the vegetation. During years of exceptionally high vegetation production, one management alternative to benefit fish is the creation of open areas within large patches of vegetation. For example, moderate level of vegetation removal (20-30%) increased fish growth and abundance in lakes (Trebitz et al. 1997, Olson et al. 1998).

Whereas there was a distinct contrast between the edge and interior during 1999 in Batchtown, we did not observe this contrast at sites during 2000 and 2001. At the pool scale, vegetation available for fish during fall 2000 and 2001 was much less compared to 1999, and patch size at a given site was much smaller. Dissolved oxygen concentration was always high/moderate during the daytime at all sites. During fall 2001, the contrast in fish sizes, abundance, and species composition between vegetated and devegetated plots was not as apparent as it was during 1999. For example, size distributions of channel shiner and spotfin shiner were similar between plots, with YOY less than 20 mm present in both. Emerald shiner was more abundant in vegetation during 2001 compared to other years, and it was equally abundant in both plots that year. Similarly, the orangespotted sunfish was abundant in devegetated plots in 1999, but their abundance was generally higher in vegetation in 2001. In a sense, vegetation produced in 2001 provided conditions similar to the edge habitat in 1999 (i.e., benign DO and less dense vegetation). Overall fish abundance and richness in the vegetation during 1999 and 2001 was similar, but diversity tended to be higher in 2001. In general, vegetation during 2001 provided benefits to fish similar to vegetation during fall 1999, with the primary difference being the vegetation in 2001 also provided relatively open-water habitat for sunfishes.

Relatively high fish abundance and species richness in the narrow band of shoreline vegetation during fall 2000 indicated that fish will use and likely benefit from the production of even very small amounts of vegetation. However, a vast majority of the fish were western mosquitofish, and most other species (sunfish being an exception) in the vegetation were less abundant compared to 1999 and 2001. The small amount of vegetation could probably not support high densities of a number of species, and the very shallow water favored the western mosquitofish which will generally outcompete other small-bodied fishes in marginal habitats.

Residual vegetation during spring - Many studies have demonstrated the benefits of living vegetation as habitat for fish (Janecek 1988), but the benefits and use of residual, annual vegetation during spring in the UMR or other systems is not well documented. We collected many YOY fishes at sites containing residual vegetation, including sport fish, commercially important species, and rare fishes. Most of the residual vegetation remaining was in the form of dead stalks of smartweed still attached to the substrate. The stalks, which at some sites formed a dense underwater network, could have provided direct spawning substrate for fish with adhesive eggs (e.g., gar and buffalo). Although all the leaves were gone, the remaining stalks offered shallow-water structure at water depths that otherwise would have contained no cover. This was particularly true at the Batchtown sites where no other form of mid-water cover was available. Also, the benefit of residual vegetation as littoral zone cover probably increases when water levels drop during spring, no longer inundating terrestrial vegetation. Residual vegetation could increase invertebrate abundance, and therefore food for fish, by providing cover, a direct food source, or by releasing nutrients once decomposition resumes. For example, we observed a YOY blue sucker directly feeding along a stalk of residual smartweed in the field; it was probably gleaning the biofilm on the vegetation.

From a management standpoint, it is important to understand the factors affecting how much residual vegetation remains following ice-out. Certainly the amount and composition of vegetation present going into the winter will be a factor. Smartweeds appear to be more tolerant of inundation than the other vegetation types and are more likely to be present following ice-out. Winter temperature is also probably important because decomposition rates increase with high temperatures (e.g., during a mild winter combined with fast rising spring temperatures). The majority of residual vegetation is likely lost to ice scour. Location is a factor because scouring due to thawing ice and open river conditions will affect some sites more than others. The amount of residual vegetation present during spring will probably be highest following summers of high smartweed production and winters with low water-level fluctuations.

Hydrology

Hydrology is one of the most important factors structuring fish communities in lotic systems (Horwitz 1978, Poff and Allan 1995). By influencing reproduction and recruitment processes, water-level manipulations (via midpool control point management and EPM) can affect the fish community composition of UMR pools, because fish species may respond differently to a particular hydrologic regime. The timing, rate, and duration of the late spring/early summer maximum drawdown (a result of midpool, control-point management) can significantly affect fish. Spring spawning species, already facing restricted access to quality floodplain habitat (Sheehan and Konikoff 1998), may suffer from a shortened spawning season if maximum drawdown occurs too early. Year-class strength may also be affected if maximum drawdown strands (e.g., through isolation) or forces newly hatched young from backwater nursery areas before they are fully prepared for life in river channel habitats.

During summer 1999, fish became isolated in Jim Crow slough following maximum drawdown. Fish species richness in this area declined from 23 species before drawdown to five species 49 days following isolation. Some of this decline was probably due to fish escaping the slough as water levels receded. Nonetheless, harsh conditions existed (e.g., high water temperatures and low water volume), and mortality ocurred (Sheehan et al. 2001). Other backwaters in lower Pool 25 probably responded similarly to Jim Crow in 1999 following drawdown. On 13 July 1999, many recently opened mussel shells (*Amblema*, *Quadrula*, and *Megalonaias*) were found scattered in one of the side channels traversing Batchtown. The exposed mussels appeared to have been easy prey for raccoons. Directly adjacent to the experimental plots at Batchtown West, we observed thousands of dead fish on 24 July 1999, which encompassed at least 11 species and was comprised of mostly YOY channel catfish and river carpsucker. The fish were associated with a shallow pool and probably died from the combined effects of extremely high midday temperatures and low DO concentration.

The summer hydrologic regime of 1999 was perhaps extreme compared to other years. Because of the combination of midpool, control-point management and elevated discharges upstream, Pool 25 was on tilt for most of the summer, resulting in extremely low water levels in the lower pool. Following maximum drawdown, water levels remained $\geq$ 2 ft below full pool (434 ft) for 54 days and $\geq$ 4 ft below full pool for 21 days. We have observed that at elevations below approximately 431 ft (3 ft below full pool), most backwaters in lower Pool 25 become isolated or completely dry. The fact that mussel beds containing relatively large, old individuals were exposed in Batchtown suggests the combined magnitude and duration of the low-water period that occurred in 1999 does not happen frequently.

Sunfish abundance in fall, primarily bluegill and orangespotted sunfish, should reflect overall backwater quality, because their abundance will be sensitive to water-level fluctuations, the absence of nursery habitat, and deteriorated water quality conditions (Kohler et al. 1993, Raibley et al. 1997). During fall 2000, mean number of sunfish captured per trip at a given site (mean = 9.11) was higher than during fall 1999 (mean = 0.88). The positive sunfish response to conditions during 2000 was most evident at Jim Crow where no sunfish were collected during fall 1999 but 175 were captured during 2000. Apparently, sunfish benefited from the drawdown regime of 2000 compared to the drawdown of 1999. Sunfish abundance remained relatively high during fall 2001. Boat electrofishing in Batchtown also suggested that sunfish abundance was low in 1999 (although much cover was present) compared to 1998 and 2001; electrofishing efficiency during 2000 was low, resulting in an overall low number of species due to the lack of structure/cover. Sunfish abundance patterns indicated the summer hydrologic regime of 1999 was not amenable to backwater fishes in comparison to 1998, 2000, and 2001.

Conclusions and Recommendations

All three drawdown regimes studied (1999, 2000, and 2001) affected fishes in unique and potentially beneficial ways. Numerous fishes, including the offspring of late-season spawning species, used the vast amounts of vegetation produced during summer 1999. During fall, fish were most abundant in relatively small patches of vegetation associated with islands and at the edge of the vegetation in large, expansive patches. Also, residual vegetation, as a result of the high amount of smartweed produced in 1999, provided nursery habitat for many YOY fishes during the following spring. Although very little vegetation was produced during summer 2000, the relatively high water levels resulted in a dramatic increase of sunfish compared to 1999. The moderate, relatively late drawdown and gradual rewatering characteristic of the water-level regime of 2001 enhanced vegetation production without compromising backwater fishes (e.g., sunfishes). Vegetation available during fall 2001 provided nursery habitat for many fishes, and low DO concentration and high vegetation density did not limit fish use of the habitat. Sunfish abundance during fall 2001 was relatively high. The fact that sunfish abundance during fall 2001 was similar between sites at midpool and lower pool further indicated that the summer drawdown regime of 2001 accomplished vegetation production without being detrimental to backwater faunas. Of the three drawdown regimes studied, the summer 2001 regime (moderate vegetation production) provided the most general benefits to fish.

From a fish perspective, we recommend that EPM targets drawdown regimes that produce ample amounts of vegetation without negatively affecting backwater fishes. Backwaters in lower Pool 25 become disconnected from the main channel at an elevation between 432 and 431 ft. Therefore, the EPM target of a drawdown no greater than 2 ft below full pool appears to be an adequate compromise between vegetation production and maintaining backwater fish faunas. Water levels did fall 3 ft or greater below full pool for 12 and 7 days in 2000 and 2001, respectively, without obviously reducing the success of backwater fishes. Due to management of Pool 25 with a midpool control point, high magnitude and duration drawdowns in the lower pool may be unavoidable during some years (e.g., during summers of high discharge as in 1999). Environmental Pool Management can be used in subsequent years to compensate negative impacts incurred during years such as 1999. This was demonstrated during 2000 when water levels were maintained near full pool, resulting in the apparent rebound of sunfish populations (a goal of water managers that year).

Within a given year, supplemental management tools can be used to maximize benefits to fishes. For example, an "irrigation event" (sensu Dugger and Feddersen 2000), where water levels are allowed to rise and inundate mudflats for a short time period, may be employed during a drawdown when backwaters have been isolated for a significant time period. Such an event could promote plant growth while rejuvenating conditions in backwaters. When large patches of vegetation are present or anticipated, open areas or lanes could be made in the vegetation. Open areas and lanes within large expanses of vegetation may help alleviate problems of low dissolved oxygen concentration. Also, by increasing the amount of edge habitat, more fish (e.g., emerald shiners) will have access to the vegetation.

The use and implementation of EPM to benefit fish will depend on the specific management objectives. In general, riverine fish diversity is positively correlated with habitat heterogeneity, including hydrological variability (within natural limits). At the local scale, habitat heterogeneity of shoreline zones is enhanced through vegetation produced by EPM, which can be further enhanced by the creation of additional edge habitat. The presence of contrasting backwater habitat types (midpool: primarily flow-through backwaters not influenced by EPM; lower pool: primarily back-fill habitats affected by EPM) increases fish diversity at the pool scale in Pool 25. Year-to-year variation in EPM (e.g., variability in timing, magnitude, and duration) will help maintain fish diversity (sensu Sparks 1995). We recommend that EPM and water-level management be implemented, spatially and temporally, to increase habitat heterogeneity and enhance fish diversity.

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Table 4-1.

Location of experimental plots at six sites in lower Pool 25, Mississippi River.

Site	Locality
Batchtown East	Pool 25, Mississippi River; approx. 0.5 mi North of boat ramp in Cockrell Hollow; Calhoun Co. Illinois; T12S, R2W, Sec 6; N39^{0}02.361 W90^{0}40.669; River Mile 244.5
Batchtown West	Pool 25, Mississippi River; in northend of large bay; Calhoun Co. Illinois; T12S, R2W, Sec 6; N39^{0}02.362 W90^{0}41.456; River Mile 244
Jim Crow	Pool 25, Mississippi River; slough on Jim Crow Island; Lincoln Co. Missouri; T50N, R3E, Sec 25; N39^{0}03.792 W90^{0}42.685; River Mile 246
Turner Island	Pool 25, Mississippi River; southern tip of Turner Island; Calhoun Co. Illinois; T12S, R2W, Sec 1; N39^{0}02.720 W90^{0}42.347; River Mile 244.5
Dixon Pond	Pool 25, Mississippi River; southern shoreline of Dixon Pond complex; Calhoun Co. Illinois; T12S, R2W, Sec 6; N39^{0}03.090 W90^{0}41.081; River Mile 245
Hausgen	Pool 25, Mississippi River; backwater shoreline of Hausgen Island; Lincoln Co. Missouri; T50N, R3E, Sec 24; River Mile 247

Table 4-2. List of scientific and common names of fishes collected in Pool 25, Mississippi River from 1999-2002.

Scientific Name	Common Name
Lepisosteus osseus	Longnose Gar
Lepisosteus platostomus	Shortnose Gar
Hiodon alosoides	Goldeye
Hiodon tergisus	Mooneye
Dorosoma cepedianum	Gizzard Shad
Campostoma anomalum	Central Stoneroller
Ctenopharyngodon idella	Grass Carp
Cyprinella lutrensis	Red Shiner
Cyprinella spiloptera	Spotfin Shiner
Hybognathus nuchalis	Mississippi Silvery Minnow
Hypophthalmichthys molitrix	Silver Carp
Hypophthalmichthys nobilis	Bighead Carp
Macrhybopsis storeriana	Silver Chub
Notemigonus crysoleucas	Golden Shiner
Notropis atherinoides	Emerald Shiner
Notropis blennius	River Shiner
Notropis dorsalis	Bigmouth Shiner
Notropis hudsonius	Spottail Shiner
Notropis shumardi	Silverband Shiner
Notropis stramineus	Sand Shiner
Notropis wickliffi	Channel Shiner
Phenacobius mirabilis	Suckermouth Minnow
Pimephales notatus	Bluntnose Minnow
Pimephales vigilax	Bullhead Minnow
Semotilus atromaculatus	Creek Chub
Carpiodes carpio	River Carpsucker
Carpiodes cyprinus	Quillback
Cycleptus elongatus	Blue Sucker
Ictiobus bubalus	Smallmouth Buffalo
Ictiobus cyprinellus	Bigmouth Buffalo
Ictiobus niger	Black Buffalo
Moxostoma sp.	Redhorse
Ictalurus punctatus	Channel Catfish
Gambusia affinis	Western Mosquitofish
Labidesthes sicculus	Brook Silverside
Morone chrysops	White Bass
Lepomis cyanellus	Green Sunfish
Lepomis gulosus	Warmouth
Lepomis humilis	Orangespotted Sunfish
Lepomis macrochirus	Bluegill
Micropterus salmoides	Largemouth Bass
Pomoxis annularis	White Crappie
Pomoxis nigromaculatus	Black Crappie
Etheostoma nigrum	Johnny Darter
Percina caprodes	Logperch
Percina shumardi	River Darter
Stizostedion canadense	Sauger
Stizostedion vitreum	Walleye
Aplodinotus grunniens	Freshwater Drum

Table 4-3.

Fish abundance and species richness in vegetated and devegetated plots based on collections using two sampling gears. Numbers are pooled from four sites in lower Pool 25, Mississippi River, and totaled over five sampling trips during fall 1999.

	Vegetated Plot		Devegetated Plot	
Species	Seine	Pop net	Seine	Pop net
Gizzard shad	2	1	2	0
Grass carp	196	15	24	3
Common carp	370	145	127	26
Spotfin shiner	1121	459	125	18
Emerald shiner	84	26	700	109
River shiner	52	33	3	0
Sand shiner	0	0	1	0
Silverband shiner	0	0	1	0
Channel shiner	2234	1027	423	120
Bluntnose minnow	1	0	0	0
Bullhead minnow	2	0	3	3
River carpsucker	0	0	3	0
Channel catfish	0	2	0	1
Western mosquitofish	2242	543	452	268
Brook silverside	0	0	6	2
Orangespotted sunfish	3	4	13	64
Bluegill	1	0	0	0
Green sunfish	2	0	0	0
Totals:				
Number of Species	13	10	14	10
Fish Abundance	6310	2255	1883	614

Table 4-4. Seven sites established in Pool 25, Mississippi River to study the biota in backwaters located at lower pool and midpool.

SITE NAME	LOCALITY
TURNER	Pool 25, Mississippi River; southern tip of Turner Island; Calhoun Co. Illinois;
(Lower pool)	T12S, R2w, Sec 1; N39º03.7920 W90º42.347, River Mile 244.
SERPENT SLOUGH	Pool 25, Mississippi River, slough on Turner Island; Calhoun Co. Illinois;
(Lower pool)	N39º23’00” W90º43’00”, River Mile 245.
JIM CROW	Pool 25, Mississippi River; slough on Jim Crow Island; Lincoln Co. Missouri;
(Lower pool)	N39º03.792 W90º42.685; River Mile 246.
STAG ISLAND	Pool 25, Mississippi River, slough on Stag Island, Lincoln Co. Missouri;
(Lower pool)	N39º05’529” W90º41’388”.
GYRINID POINT	Pool 25, Mississippi River, on cut between Missouri shore and Howard Island,
(Midpool)	Pike Co. Missouri; N39º15’505” W90º45’102”; River Mile 261.3.
COON SLOUGH	Pool 25, Mississippi River, slough on Coon Island, Calhoun Co. Illinois;
(Midpool)	N39º19’472” W90º48’818”; River Mile 267.5.
MCCOY SLOUGH	Pool 25, Mississippi River, slough on backwater side of McCoy Island,
(Midpool)	Calhoun Co. Illinois; N39º17’001” W090º46’121”; River Mile 263.5.

Table 4-5.

Three-way ANOVA results for water depth and water quality parameters in study plots at four sites in Pool 25, Mississippi River from 1999-2001. An asterisk indicates a significant result.

Parameter	Effect	*F* Ratio	*p*-value
Water Depth	Year	$F_{2,6} = 0.44$	0.664
	Plot	$F_{1,6} = 1.17$	0.32
	Year*Plot	$F_{2,6} = 1.18$	0.369
	Site	$F_{3,6} = 56.12$	< 0.001*
	Year*Site	$F_{6,6} = 2.55$	0.14
	Plot*Site	$F_{3,6} = 1.9$	0.23
Water Temperature	Year	$F_{2,11} = 56.99$	< 0.001*
	Plot	$F_{1,11} = 0.03$	0.86
	Site	$F_{3,11} = 31.37$	< 0.001*
	Year*Site	$F_{6,11} = 15.89$	< 0.001*
Dissolved Oxygen	Year	$F_{2,9} = 49.47$	< 0.001*
	Plot	$F_{1,9} = 11.32$	0.008*
	Year*Plot	$F_{2,9} = 6.0$	0.022*
	Site	$F_{3,9} = 13.79$	0.001*
	Year*Site	$F_{6,9} = 6.7$	0.006*
Conductivity	Year	$F_{2,8} = 26.92$	< 0.001*
	Plot	$F_{1,8} = 0.57$	0.472
	Site	$F_{3,8} = 13.45$	0.002*
	Year*Site	$F_{6,8} = 8.36$	0.004*
	Plot*Site	$F_{3,8} = 3.42$	0.073
Turbidity	Year	$F_{2,9} = 27.06$	< 0.001*
	Plot	$F_{1,9} = 6.66$	0.029*
	Year*Plot	$F_{2,9} = 2.61$	0.128
	Site	$F_{3,9} = 15.3$	< 0.001*
	Year*Site	$F_{6,9} = 6.66$	0.006*
pH	Year	$F_{2,11} = 4.18$	0.045*
	Plot	$F_{1,11} = 4.8$	0.051*
	Site	$F_{3,11} = 4.45$	0.028*
	Year*Site	$F_{6,11} = 2.39$	0.099

Table 4-6. Depth and water quality in study plots at sites in Pool 25, Mississippi River from 1999-2001. Values are means with $\pm$ upper and lower limits of the 95% confidence interval in parentheses. Minimum and maximum pH are provided.

Site	Plot	Depth (cm)	Temp (^{O}C)	DO (mg/L)	Cond (25 ^{O}C) (µS/cm)	Turbidity (NTU)	pH
1999							
Batch. West	Veg	44.4 (3.7)	22.0 (4.0)	5.9 (3.2)	410.0 (23.1)	61.6 (34.5)	7.9-8.7
	∅	42.0 (5.0)	22.2 (5.0)	8.1 (1.6)	412.0 (27.7)	64.6 (23.4)	8.1-8.7
Batch. East	Veg	53.5 (4.2)	20.6 (4.2)	2.5 (0.9)	359.4 (66.2)	17.5 (15.5)	7.4-8.0
	∅	55.2 (2.4)	21.2 (3.8)	5.6 (1.5)	368.2 (66.5)	48.9 (21.9)	7.8-8.4
Jim Crow	Veg	27.3 (1.7)	23.1 (5.4)	8.7 (2.6)	403.6 (43.3)	26.7 (16.3)	8.2-9.0
	∅	28.5 (2.6)	23.2 (5.9)	10.2 (2.7)	415.2 (35.0)	56.9 (33.0)	8.0-8.7
Turner	Veg	24.8 (5.5)	21.9 (4.8)	6.7 (3.8)	406.0 (39.0)	67.4 (24.8)	7.8-8.8
	∅	27.4 (4.8)	21.4 (4.6)	9.0 (2.0)	382.0 (47.5)	81.6 (30.9)	8.3-8.8
2000							
Batch. West	Veg	46.3 (4.0)	20.7 (3.8)	11.0 (2.0)	376.0 (34.7)	20.8 (9.2)	8.4-9.0
	∅	42.3 (2.4)	20.8 (3.6)	11.2 (2.0)	376.7 (33.4)	26.5 (45.2)	8.3-9.0
Batch. East	Veg	53.7 (8.8)	20.5 (4.7)	8.6 (1.0)	370.0 (28.9)	25.8 (9.2)	8.0-8.8
	∅	56.6 (7.2)	20.4 (4.9)	8.9 (0.9)	371.7 (54.6)	22.7 (14.0)	7.9-8.8
Jim Crow	Veg	23.0 (1.6)	21.1 (7.1)	10.9 (5.8)	378.3 (64.3)	14.3 (3.0)	8.0-9.0
	∅	26.5 (2.4)	20.9 (6.1)	11.3 (3.0)	373.0 (56.4)	15.4 (6.2)	8.1-8.9
Turner	Veg	28.7 (7.0)	21.8 (10.2)	10.3 (7.3)	377.0 (46.4)	28.3 (20.4)	7.8-9.0
	∅	22.5 (14.3)	21.7 (17.5)	11.0 (6.8)	380.0 (95.5)	20.1 (12.3)	8.1-8.9
Dixon Pond	Veg	45.0 (8.5)	21.8 (4.9)	12.0 (2.9)	360.0 (42.4)	29.2 (42.2)	8.5-9.0
	∅	43.7 (7.6)	21.7 (4.8)	12.1 (2.8)	358.3 (30.4)	23.5 (23.7)	8.5-9.0
2001							
Batch. West	Veg	47.5 (3.6)	22.6 (10.7)	13.2 (1.1)	406.7 (93.9)	21.3 (17.1)	8.5-9.4
	∅	44.3 (5.6)	22.7 (10.6)	13.8 (2.1)	406.3 (91.8)	21.7 (15.6)	8.6-9.3
Batch. East	Veg	57.3 (4.0)	21.6 (8.0)	10.7 (2.1)	388.0 (66.6)	38.3 (17.2)	8.1-9.4
	∅	58.3 (2.4)	21.2 (8.2)	10.8 (2.0)	389.3 (68.8)	47.1 (25.0)	8.2-9.3
Jim Crow	Veg	29.7 (5.6)	21.6 (10.6)	9.9 (0.4)	392.3 (107.1)	6.2 (2.3)	8.1-9.0
	∅	30.5 (8.3)	21.9 (10.3)	10.9 (1.8)	398.0 (96.6)	11.7 (2.2)	8.2-8.8
Turner	Veg	23.3 (4.9)	22.2 (4.7)	10.8 (6.7)	408.0 (43.4)	36.9 (18.1)	7.7-8.8
	∅	11.7 (2.4)	22.3 (5.6)	11.3 (4.6)	397.7 (38.6)	49.9 (45.0)	8.3-8.5
Dixon Pond	Veg	20.7 (3.7)	23.5 (9.6)	8.8 (0.9)	401.3 (73.8)	36.9 (22.2)	8.0-8.6
	∅	22.5 (2.1)	23.2 (9.4)	9.1 (1.6)	399.0 (68.8)	26.6 (18.1)	7.8-8.6
Hausgen	Veg	10.7 (2.4)	20.1 (7.3)	9.5 (2.7)	364.0 (36.1)	34.1	8.0-8.3
	∅	13.3 (4.6)	20.7 (6.2)	11.3 (2.6)	379.3 (27.0)	31.2 (8.4)	8.0-8.5

Table 4-7. Species abundance and richness in vegetated (Veg) and devegetated (∅) plots and in Edge habitat (Batchtown West and Batchtown East only) at four sites in Pool 25, Mississippi River. Numbers represent pooled seine and popnet samples totaled over five sampling trips during fall 1999. Edge samples corresponded to the deeper portions of the study plots.

	Batch. West				Batch. East				Jim Crow		Turner		Totals
Species	Veg	V.Edge	∅	∅Edge	Veg	V.Edge	∅	∅Edge	Veg	∅	Veg	∅	
Gizzard shad	1	1	0	0	0	0	0	0	0	0	2	2	6
Grass carp	0	0	0	0	0	0	0	0	211	27	0	0	238
Red shiner	0	0	0	0	0	0	0	0	0	0	0	0	0
Spotfin shiner	75	47	5	0	57	353	26	0	61	88	1387	24	2123
Common carp	285	2	3	0	87	0	0	0	84	149	59	1	670
Golden shiner	0	1	0	0	0	0	0	0	0	0	0	0	1
Emerald shiner	30	95	78	19	0	400	400	3	5	56	75	275	1436
River shiner	1	0	0	0	0	3	0	0	1	3	83	0	91
Silverband shiner	0	0	0	0	0	0	1	0	0	0	0	0	1
Sand shiner	0	0	0	0	0	1	0	0	0	1	0	0	2
Channel shiner	1	10	18	0	0	5	22	2	102	414	3158	89	3821
Bluntnose minnow	0	0	0	0	0	1	0	0	1	0	0	0	2
Bullhead minnow	0	0	0	0	0	0	2	0	0	1	2	3	8
River carpsucker	0	0	1	0	0	1	1	0	0	1	0	0	4
Bigmouth buffalo	0	0	0	0	0	0	0	0	0	0	0	0	0
Smallmouth buffalo	0	0	0	0	0	0	0	0	0	0	0	0	0
Channel catfish	0	0	0	0	0	0	0	0	0	0	2	1	3
Western mosquitofish	230	12	1	0	201	4	1	0	2262	718	92	0	3521
Brook silverside	0	0	0	0	0	0	8	0	0	0	0	0	8
Green sunfish	2	0	0	0	0	0	0	0	0	0	0	0	2
Orangespotted sunfish	2	21	61	0	2	2	5	1	0	1	3	9	107
Bluegill	0	0	0	0	0	0	0	0	0	0	1	0	1
Lepomis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Largemouth bass	0	0	0	0	0	0	0	0	0	0	0	0	0
Freshwater drum	0	0	0	0	0	0	0	0	0	0	0	0	0
Number of Species	9	8	7	1	4	9	9	3	8	11	11	8	19
Number of Individuals	627	189	167	19	347	770	466	6	2727	1459	4864	404	12045
Shannon Index	0.52	0.60	0.52	N/A	0.43	0.36	0.27	N/A	0.30	0.58	0.40	0.41	

Table 4-8. Species abundance and richness in vegetated (Veg) and devegetated (∅) plots and in shoreline vegetation (Shore) at five sites in Pool 25, Mississippi River. In general, both study plots contained no vegetation which was confined to a narrow band along the shoreline at most sites. Numbers represent pooled seine samples totaled over three sampling trips during fall 2000.

	Batch. West			Batch. East			Jim Crow		Turner			Dixon Pond			Totals
Species	Veg	∅	Shore	Veg	∅	Shore	Veg	∅	Veg	∅	Shore	Veg	∅	Shore	
Gizzard shad	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Grass carp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Red shiner	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Spotfin shiner	0	1	89	0	0	279	291	689	0	1	335	0	0	18	1703
Common carp	0	0	1	0	0	1	11	11	0	0	4	0	0	0	28
Golden shiner	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Emerald shiner	7	29	4	3	1	23	72	75	22	171	54	137	5	3	606
River shiner	0	0	1	0	0	0	7	2	0	0	38	0	0	0	48
Silverband shiner	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sand shiner	0	0	0	0	0	0	7	3	0	0	0	0	0	0	10
Channel shiner	2	0	4	0	1	135	212	402	7	68	89	0	0	1	921
Bluntnose minnow	0	0	0	0	0	2	2	0	0	0	0	0	0	0	4
Bullhead minnow	0	0	0	0	0	14	4	0	0	0	2	1	0	4	25
River carpsucker	0	0	0	0	0	0	0	0	0	0	5	0	0	0	5
Bigmouth buffalo	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Smallmouth buffalo	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Channel catfish	0	0	0	0	0	0	1	0	0	0	2	0	0	0	3
Western mosquitofish	0	0	3036	0	0	2968	999	1279	0	0	1126	0	0	2333	11741
Brook silverside	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Green sunfish	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
Orangespotted sunfish	2	1	1	0	0	13	31	60	0	0	9	1	0	170	288
Bluegill	0	0	0	0	0	4	30	49	2	0	1	7	2	215	310
Lepomis sp.	0	0	0	0	0	0	0	4	0	0	0	0	0	0	4
Largemouth bass	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1
Freshwater drum	0	0	0	0	0	0	0	2	0	0	1	0	0	1	4
Number of Species	3	3	8	1	2	9	12	11	3	3	13	4	2	8	18
Number of Individuals	11	31	3137	3	2	3439	1667	2577	31	240	1667	146	7	2745	15703
Shannon Index	.39	.12	.07	N/A	N/A	0.24	0.55	0.57	.33	.27	.45	.12	.26	.24	

Table 4-9. Species abundance and richness in vegetated (Veg) and devegetated (∅) plots and in shoreline vegetation (Shore) at six sites in Pool 25, Mississippi River. Numbers represent pooled seine samples totaled over three sampling trips during fall 2001.

	Batch. West		Batch. East			Jim Crow		Turner		Dixon Pond		Hausgen		Totals
Species	Veg	∅	Veg	∅	Shore	Veg	∅	Veg	∅	Veg	∅	Veg	∅	
Gizzard shad	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Grass carp	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Red shiner	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spotfin shiner	46	4	0	0	537	162	273	848	63	198	254	1095	153	3633
Common carp	16	0	0	0	0	237	2	3	0	1	0	10	0	269
Golden shiner	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Emerald shiner	560	69	6	6	258	261	372	680	1027	166	237	120	562	4324
River shiner	0	0	0	0	9	11	55	38	8	0	0	27	53	201
Silverband shiner	0	0	0	2	3	0	0	0	0	160	42	0	0	207
Sand shiner	0	0	0	0	1	0	1	0	0	0	0	0	3	5
Channel shiner	43	3	0	8	214	751	32	445	58	78	22	452	337	2443
Bluntnose minnow	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bullhead minnow	1	0	0	0	25	14	4	8	0	98	4	26	3	183
River carpsucker	0	0	0	0	0	0	0	3	0	0	0	0	0	3
Bigmouth buffalo	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Smallmouth buffalo	0	0	0	0	0	1	0	0	0	0	0	0	0	1
Channel catfish	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Western mosquitofish	4	0	0	0	359	179	1	49	0	391	78	439	1	1501
Brook silverside	0	0	0	0	0	0	0	0	0	0	2	0	0	2
Green sunfish	0	0	0	0	0	16	0	0	0	0	0	0	0	16
Orangespotted sunfish	6	10	0	0	29	45	0	1	0	102	21	1	0	215
Bluegill	5	1	0	0	19	64	4	0	0	105	6	0	0	204
Lepomis sp.	0	0	0	0	0	0	69	0	0	0	45	0	0	114
Largemouth bass	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Freshwater drum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Number of Species	8	5	1	3	10	11	10	9	4	10	9	8	7	17
Number of Individuals	681	87	6	16	1454	1741	813	2075	1156	1300	711	2170	1112	13317
Shannon Index	.31	.32	N/A	N/A	.68	.74	.58	.55	.20	.85	.63	.56	.51	

Table 4-10.

Total lengths (mm) of fish species collected in study plots (vegetated and devegetated plots combined) at sites in Pool 25, Mississippi River during fall 1999, 2000, and 2001. Values represent means with minimum and maximum values in parentheses.

Species	1999	2000	2001
Gizzard shad	78.3 (65-100)	---	---
Grass carp	35.8 (24-71)	---	---
Red shiner	---	46.0	---
Spotfin shiner	20.7 (9-46)	27.9 (12-62)	28.9 (13-66)
Common carp	20.8 (9-50)	47.2 (25-79)	20.8 (14-48)
Emerald shiner	38.2 (12-63)	40.5 (18-79)	33.6 (15-65)
River shiner	20.2 (12-52)	26.6 (11-57)	32.4 (12-57)
Silverband shiner	26.0	---	30.0 (19-47)
Sand shiner	54.0	25.1 (15-41)	36.2 (25-42)
Channel shiner	18.7 (8-41)	24.5 (12-40)	22.3 (11-48)
Bluntnose minnow	42.0	41.5 (37-48)	---
Bullhead minnow	33.2 (30-36)	37.0 (22-49)	36.6 (22-53)
River carpsucker	72.7 (51-110)	36.4 (26-55)	43.0 (33-51)
Bigmouth buffalo	---	---	113.0
Smallmouth buffalo	---	---	101.0
Channel catfish	---	55.3 (54-57)	---
Western mosquitofish	18.7 (8-39)	23.6 (8-41)	22.5 (10-43)
Brook silverside	61.8 (55-66)	---	58.0 (56-60)
Green sunfish	28.0 (26-30)	26.0	24.5 (18-32)
Orangespotted sunfish	17.8 (11-38)	32.8 (18-58)	27.7 (13-58)
Bluegill	70.0	33.7 (20-65)	26.5 (10-54)
Lepomis sp.	---	14.0	18.4 (13-23)
Largemouth bass	---	165.0	---
Freshwater drum	---	66.0 (45-89)	---

Table 4-11. Results of three-way ANOVA models (N = 24) examining the effects of Plot, Site, Year, and interaction effects on fish abundance, species richness, Shannon index, and plot similarity in Pool 25, Mississippi River during fall 1999, 2000, and 2001. Results are from the final model after pooling. The analysis describes results from data in Figure __ (sites: BW, BE, JC, and TU). Statistical significance ($p < 0.05$**) and marginal significance ($p < 0.1$*) are indicated.

Dependent Variable	Effect	*F* Ratio	*p*-value
Fish Abundance	Plot	$F_{1,9} = 0.75$	0.409
	Site	$F_{3,9} = 33.13$	< 0.001**
	Year	$F_{2,9} = 15.12$	0.001**
	Plot*Year	$F_{2,9} = 2.71$	0.12
	Site*Year	$F_{6,9} = 5.00$	0.016**
Species Richness	Plot	$F_{1,8} = 0.82$	0.392
	Site	$F_{3,8} = 52.40$	< 0.001**
	Plot*Site	$F_{3,8} = 11.59$	0.003**
	Year	$F_{2,8} = 35.07$	< 0.001**
	Site*Year	$F_{6,8} = 8.40$	0.004**
Diversity (H')	Plot	$F_{1,10} = 0.57$	0.469
	Site	$F_{2,10} = 2.13$	0.17
	Year	$F_{2,10} = 1.26$	0.326
	Plot*Year	$F_{2,10} = 2.07$	0.177
Plot Similarity (PSI)	Year	$F_{2,6} = 4.87$	0.056*
	Site	$F_{3,6} = 1.22$	0.382

Table 4-12. Similarity of fish assemblages between study plots (∅ = devegetated plot) and additional samples, based on the Percentage Similarity Index (PSI), at study sites in Pool 25, Mississippi River during 1999, 2000, and 2001.

Site	Comparison	PSI
1999		
Batchtown West	Veg Plot : ∅ Plot	10.3
	Veg Plot : Veg Edge	24.7
Batchtown East	Veg Plot : ∅ Plot	6.4
	Veg Plot : Veg Edge	17.1
Jim Crow	Veg Plot : ∅ Plot	60.0
Turner Island	Veg Plot : ∅ Plot	29.8
2000		
Batchtown West	Veg Plot : ∅ Plot	66.8
	Veg Plot : Shoreline	0.3
Batchtown East	Veg Plot : ∅ Plot	50.0
	Veg Plot : Shoreline	0.7
Jim Crow	Veg Plot : ∅ Plot	86.8
Turner Island	Veg Plot : ∅ Plot	93.4
	Veg Plot : Shoreline	8.6
Dixon Pond	Veg Plot : ∅ Plot	5.6
	Veg Plot: Shoreline	5.7
2001		
Batchtown West	Veg Plot : ∅ Plot	88.8
Batchtown East	Veg Plot : ∅ Plot	37.5
	Veg Plot : Shoreline	17.7
Jim Crow	Veg Plot : ∅ Plot	35.9
Turner Island	Veg Plot : ∅ Plot	43.9
Dixon Pond	Veg Plot : ∅ Plot	58.5
Hausgen Island	Veg Plot : ∅ Plot	41.6

Table 4-13.

Correlation analyses comparing the rank-order abundance of species collected in vegetated and devegetated plots at each site in Fall 1999 and 2001 in Pool 25, Mississippi River. Correlations were calculated using all species present and excluding rare species. An asterisk denotes a significant correlation in fish community structure between vegetated and devegetated plots.

Site		N	Spearman r_s	*p* - value
1999				
Batchtown West	All	10	0.35	0.326
	Rare excluded	6	-0.71	0.111
Batchtown East	All	10	-0.32	0.359
	Rare excluded	5	-0.72	0.172
Jim Crow Island	All	12	0.83	0.001*
	Rare excluded	6	0.43	0.396
Turner Island	All	11	0.32	0.331
	Rare excluded	7	0.16	0.728
2001				
Batchtown West	All	8	0.73	0.039*
	Rare excluded	5	0.40	0.505
Jim Crow	All	11	0.46	0.158
	Rare excluded	7	-0.11	0.819
Turner Island	All	9	0.84	0.004*
	Rare excluded	5	0.80	0.104
Dixon Pond	All	10	0.85	0.002*
	Rare excluded	7	0.68	0.094
Hausgen Island	All	9	0.62	0.074
	Rare excluded	7	0.57	0.180

Table 4-14. Three-way ANOVA (N = 24) results for nine species collected in study plots at four sites in Pool 25, Mississippi River during 1999, 2000, and 2001. Statistical significance (p<0.05**) and marginal significance (p<0.1*) are indicated.

Species	Effect	*F* Ratio	*p*-value
Spotfin shiner	Plot	$F_{1,6}$ = 7.63	0.033**
	Site	$F_{3,6}$ = 31.12	< 0.001**
	Plot*Site	$F_{3,6}$ = 4.6	0.054**
	Year	$F_{2,6}$ = 10.12	0.012**
	Plot*Year	$F_{2,6}$ = 4.58	0.062*
	Site*Year	$F_{6,6}$ = 10.53	0.006**
Common carp	Plot	$F_{1,17}$ = 10.25	0.005**
	Site	$F_{3,17}$ = 5.21	0.01**
	Year	$F_{2,17}$ = 6.62	0.008**
Emerald shiner	Plot	$F_{1,9}$ = 4.73	0.058*
	Site	$F_{3,9}$ = 9.62	0.004**
	Year	$F_{2,9}$ = 8.54	0.008**
	Plot*Year	$F_{2,9}$ = 3.5	0.076*
	Site*Year	$F_{6,9}$ = 2.4	0.115
River shiner	Plot	$F_{1,8}$ = 1.17	0.311
	Site	$F_{3,8}$ = 6.33	0.017**
	Plot*Site	$F_{3,8}$ = 2.2	0.166
	Year	$F_{2,8}$ = 3.72	0.072*
	Site*Year	$F_{6,8}$ = 2.14	0.158
Channel shiner	Plot	$F_{1,17}$ = 0.00	0.999
	Site	$F_{3,17}$ = 16.95	< 0.001**
	Year	$F_{2,17}$ = 1.88	0.183
Bullhead minnow	Plot	$F_{1,9}$ = 3.28	0.103
	Site	$F_{3,9}$ = 4.68	0.031**
	Year	$F_{2,9}$ = 4.46	0.045**
	Plot*Year	$F_{2,9}$ = 3.38	0.08*
	Site*Year	$F_{6,9}$ = 2.26	0.131
Western mosquitofish	Plot	$F_{1,9}$ = 19.45	0.002**
	Site	$F_{3,9}$ = 24.15	< 0.001**
	Year	$F_{2,9}$ = 4.69	0.04**
	Plot*Year	$F_{2,9}$ = 5.26	0.031**
	Site*Year	$F_{6,9}$ = 2.62	0.094*
Orangespotted sunfish	Plot	$F_{1,9}$ = 0.09	0.773
	Site	$F_{3,9}$ = 4.12	0.043*
	Year	$F_{2,9}$ = 0.03	0.967
	Plot*Year	$F_{2,9}$ = 2.23	0.164
	Site*Year	$F_{6,9}$ = 3.01	0.067*
Bluegill	Plot	$F_{1,11}$ = 1.78	0.208
	Site	$F_{3,11}$ = 12.96	0.001**
	Year	$F_{2,11}$ = 4.97	0.029**
	Site*Year	$F_{6,11}$ = 4.00	0.023**

Table 4-15.

Fish collected by boat electrofishing in the Batchtown State Wildlife Management Area 1998-2001, Pool 25, Mississippi River. Numbers are based on 1.5 hrs of electrofishing in 1998 and 1 hr of effort in 1999-2001. Sampling was conducted at mid day in October of all years.

Common Name	1998 (fish/hr)	1999 (fish/hr)	2000 (fish/hr)	2001 (fish/hr)
Skipjack herring	0	0	0	1
Gizzard shad	96	141	113	17
Grass carp	0	0	0	4
Spotfin shiner	0	0	0	1
Common carp	11.3	7	6	2
Silver chub	0	0	0	1
Emerald shiner	3.3	0	6	17
Bullhead minnow	0	0	0	3
River carpsucker	8	14	0	18
Smallmouth buffalo	13.3	13	0	0
Bigmouth buffalo	0.7	1	0	0
Black buffalo	2.7	6	0	1
Redhorse	1.3	0	0	0
Channel catfish	1.3	1	0	0
Brook silverside	0	0	0	5
White bass	0.7	0	0	1
Bluegill	2.7	0	0	10
Orangespotted sunfish	2.7	1	0	9
Warmouth	0.7	0	0	0
Largemouth bass	0	0	0	3
Freshwater drum	1.3	0	66	0
Number of Taxa:	14	8	4	15

Table 4-16. Fish collected with three sampling gears (seine, cast net, and modified fyke net) at sites in Pool 25, Mississippi River during July 2001. Three sites (MS, CS, GP) were located at midpool and four sites (JC, TU, STS, SEP) at lower pool.

	Midpool			**Lower pool**			
Species	MS	CS	GP	JC	TU	STS	SEP
Longnose gar	1	--	--	4	--	--	--
Shortnose gar	--	--	2	--	--	--	--
Gizzard shad	6	--	2	9	--	70	--
Red shiner	--	--	--	--	--	--	--
Spotfin shiner	1174	914	1	64	--	2	--
Common carp	--	--	1	--	--	--	--
Mississippi silvery minnow	--	--	--	--	--	--	--
Silver carp	--	--	--	--	--	--	--
Silver chub	6	--	--	--	--	--	--
Golden shiner	--	--	--	1	--	--	--
Emerald shiner	24	323	80	--	--	--	--
River shiner	40	23	--	--	--	--	--
Spottail shiner	5	--	2	--	--	--	--
Silverband shiner	--	--	--	--	--	--	--
Sand shiner	--	--	--	--	--	--	--
Channel shiner	418	66	--	--	--	--	--
Bluntnose minnow	--	--	--	1	--	--	--
Bullhead minnow	58	43	--	--	--	3	--
River carpsucker	--	--	--	--	--	--	--
Quillback	--	--	--	--	--	--	--
Carpsucker sp	--	--	--	--	--	--	--
Smallmouth buffalo	--	1	--	4	--	3	--
Bigmouth buffalo	1	--	--	10	--	4	--
Buffalo sp	--	--	--	--	--	5	--
Western mosquitofish	4	7	--	654	--	289	--
Brook silverside	--	--	2	1	--	1	--
White bass	2	--	--	--	--	--	--
Green sunfish	--	--	1	1	--	--	--
Orangespotted sunfish	4	7	1	36	--	--	--
Bluegill	--	1	2	10	--	14	--
Largemouth bass	--	--	--	4	--	--	--
White crappie	--	--	--	--	--	1	--
Black crappie	--	1	1	1	--	--	--
Sauger	1	--	--	--	--	--	--
Walleye	--	1	--	--	--	--	--
Freshwater drum	--	1	--	--	--	--	--
Number of species	14	12	11	14	(Dry)	9	(Dry)

Table 4-17. Fish collected with three sampling gears (seine, cast net, and modified fyke net) at sites in Pool 25, Mississippi River during October 2001. Three sites (MS, CS, GP) were located at midpool and four sites (JC, TU, STS, SEP) at lower pool.

	Midpool			Lower pool			
Species	MS	CS	GP	JC	TU	STS	SEP
Longnose gar	--	--	--	--	--	--	--
Shortnose gar	--	--	--	--	1	--	--
Gizzard shad	11	1	--	--	--	2	--
Red shiner	--	2	--	--	--	--	--
Spotfin shiner	1840	180	38	11	446	32	56
Common carp	--	--	--	--	1	3	--
Mississippi silvery minnow	--	--	--	--	--	--	--
Silver carp	--	--	--	1	--	--	--
Silver chub	--	--	--	--	--	--	--
Golden shiner	--	--	--	--	--	--	--
Emerald shiner	1264	87	90	60	160	26	31
River shiner	33	--	1	1	26	5	29
Spottail shiner	--	--	--	--	--	--	--
Silverband shiner	--	--	1	--	--	--	--
Sand shiner	1	--	--	--	--	--	--
Channel shiner	752	180	58	127	261	619	415
Bluntnose minnow	--	1	--	--	--	--	--
Bullhead minnow	105	170	7	--	4	125	141
River carpsucker	--	--	--	--	6	--	1
Quillback	--	--	--	1	--	--	--
Carpsucker sp	1	--	--	--	--	--	--
Smallmouth buffalo	--	--	--	--	--	--	--
Bigmouth buffalo	2	--	--	--	--	--	--
Buffalo sp	--	--	--	--	--	--	--
Western mosquitofish	81	12	27	486	224	205	287
Brook silverside	--	1	5	--	--	--	--
White bass	--	--	--	--	3	--	--
Green sunfish	--	--	1	95	--	--	--
Orangespotted sunfish	91	50	--	105	2	17	36
Bluegill	2	12	--	3	--	29	2
Largemouth bass	--	--	1	--	--	--	--
White crappie	3	--	--	--	--	3	--
Black crappie	1	--	--	--	--	1	--
Sauger	--	--	--	--	--	--	--
Walleye	--	--	--	--	--	--	--
Freshwater drum	4	--	--	--	--	1	--
Number of species	15	11	10	10	11	13	9

Table 4-18. Fish collected with three sampling gears (seine, cast net, and modified fyke net) at sites in Pool 25, Mississippi River during April, 2002. Three sites (MS, CS, GP) were located at midpool and four sites (JC, TU, STS, SEP) at lower pool.

	Midpool			**Lower pool**			
Species	MS	CS	GP	JC	TU	STS	SEP
Longnose gar	--	--	--	--	--	--	--
Shortnose gar	--	--	--	--	--	--	--
Gizzard shad	1	17	--	--	3	25	3
Red shiner	--	--	--	--	--	--	--
Spotfin shiner	3	135	2	5	13	--	6
Common carp	--	--	--	--	--	--	1
Mississippi silvery minnow	--	20	1	--	--	--	--
Silver carp	--	--	--	--	--	--	--
Silver chub	--	--	--	--	--	--	--
Golden shiner	--	--	--	--	--	--	--
Emerald shiner	19	851	40	1	11	1	288
River shiner	--	--	5	--	11	3	3
Spottail shiner	--	--	--	--	--	--	--
Silverband shiner	--	--	--	--	--	--	--
Sand shiner	--	1	--	--	--	--	--
Channel shiner	1	14	35	22	26	8	26
Bluntnose minnow	--	--	--	--	--	--	--
Bullhead minnow	4	12	--	--	2	15	1
River carpsucker	--	--	--	--	--	1	2
Quillback	--	--	--	--	1	--	--
Carpsucker sp	--	--	--	--	--	--	--
Smallmouth buffalo	--	--	--	--	--	--	--
Bigmouth buffalo	--	--	--	--	--	--	--
Buffalo sp	--	--	--	--	--	--	--
Western mosquitofish	--	6	--	91	--	31	4
Brook silverside	2	--	--	--	--	--	--
White bass	--	1	--	--	2	2	1
Green sunfish	--	--	--	--	--	--	--
Orangespotted sunfish	--	1	--	5	--	4	--
Bluegill	--	--	--	--	--	--	--
Largemouth bass	--	4	--	--	--	--	--
White crappie	--	--	--	--	--	--	--
Black crappie	--	1	--	--	--	1	--
Sauger	--	--	--	--	--	--	--
Walleye	--	--	--	--	--	--	--
Freshwater drum	--	--	--	--	1	--	--
Number of species	6	12	5	5	9	10	10

Table 4-19. Fish associated with EPM-induced residual vegetation at eight sites in lower Pool 25, Mississippi River. Batchtown West (BW), Batchtown East (BE), Jim Crow (JC), and Turner Island (TU) were sampled in spring of 1999. Little Stag (LS) and Little Hole (LH), Church Slough (CS), and Hausgen (HA) were sampled in spring of 2000, 2001, and 2002, respectively. Numbers represent YOY fish unless separated by a colon (YOY:Adult).

Species	**BW**	**BE**	**JC**	**TU**	**LS**	**LH**	**CS**	**HA**
Shortnose gar	-	-	-	-	-	-	2	-
Gar species	-	1	27	6	-	5	-	-
Goldeye	-	-	-	5	-	-	-	-
Mooneye	8	2	13	9	-	-	53	-
Gizzard shad	1:1	15	745	10	8	-	-	-
Central stoneroller	-	-	-	-	-	-	-	3
Grass Carp	-	-	-	-	-	1	-	-
Spotfin shiner	0:5	0:25	0:11	0:34	-	0:76	0:13	0:49
Common carp	-	-	14	-	-	-	1	0:2
Mississippi silvery minnow	1	-	-	-	-	-	-	-
Bighead carp	-	-	12	-	-	-	-	-
Silver chub	-	-	70	65	1	-	-	-
Golden shiner	-	-	-	-	-	2	-	-
Emerald shiner	0:9	4:21	25:6	47:31	-	4:1	0:24	0:30
River shiner	-	-	1	-	-	-	-	0:14
Bigmouth shiner	-	-	-	1	-	-	-	-
Spottail shiner	-	-	1	3	-	-	-	-
Channel shiner	0:11	0:12	0:17	0:40	-	-	-	0:87
Suckermouth minnow	-	1	-	1	-	-	-	-
Bullhead minnow	0:1	0:3	-	0:1	-	-	1	0:29
Creek chub	-	1	-	-	-	-	-	-
Carpsucker species	-	-	4	1	-	-	-	-
Blue sucker	-	2	-	1	18	-	-	-
Smallmouth buffalo	-	-	-	-	-	5	7	-
Bigmouth buffalo	-	-	-	-	-	-	19	-
Buffalo species	-	1	28:1	4	2432	15	9	-
Redhorse species	-	-	1	7	-	-	-	-
Channel catfish	-	-	0:1	-	-	-	-	-
Western mosquitofish	1	-	107	3	1818	1029	2	-
White bass	-	-	20	13	1	-	-	-
Orangespotted sunfish	-	4	0:3	0:1	-	-	0:3	0:3
Bluegill	-	-	0:3	-	0:2	119:6	0:7	0:7
Lepomis species	-	-	-	-	-	195	-	-
Largemouth bass	-	-	-	-	-	2	2	-
Black crappie	-	-	-	-	-	22	-	-
Johnny darter	1	-	1	-	-	-	-	-
Logperch	-	-	12	29	-	-	-	-
River darter	1	-	-	-	24	-	-	-
Pikeperch species	-	-	1	1	2	-	-	-
Freshwater drum	-	2	311	163	-	-	1	0:1

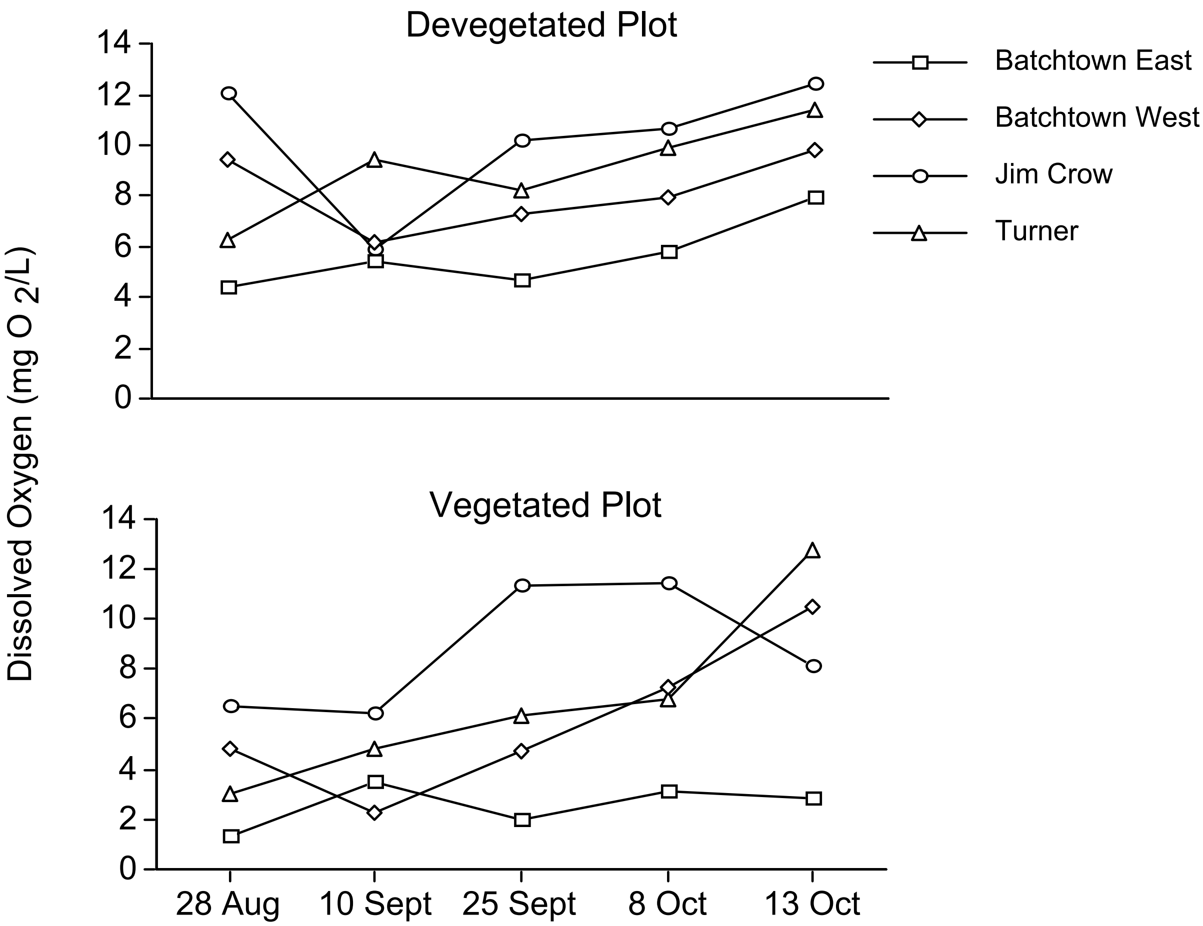

Figure 4-1. Dissolved oxygen concentration measured in devegetated and vegetated plots in 1999 on five dates at four sites in Pool 25, Mississippi River.

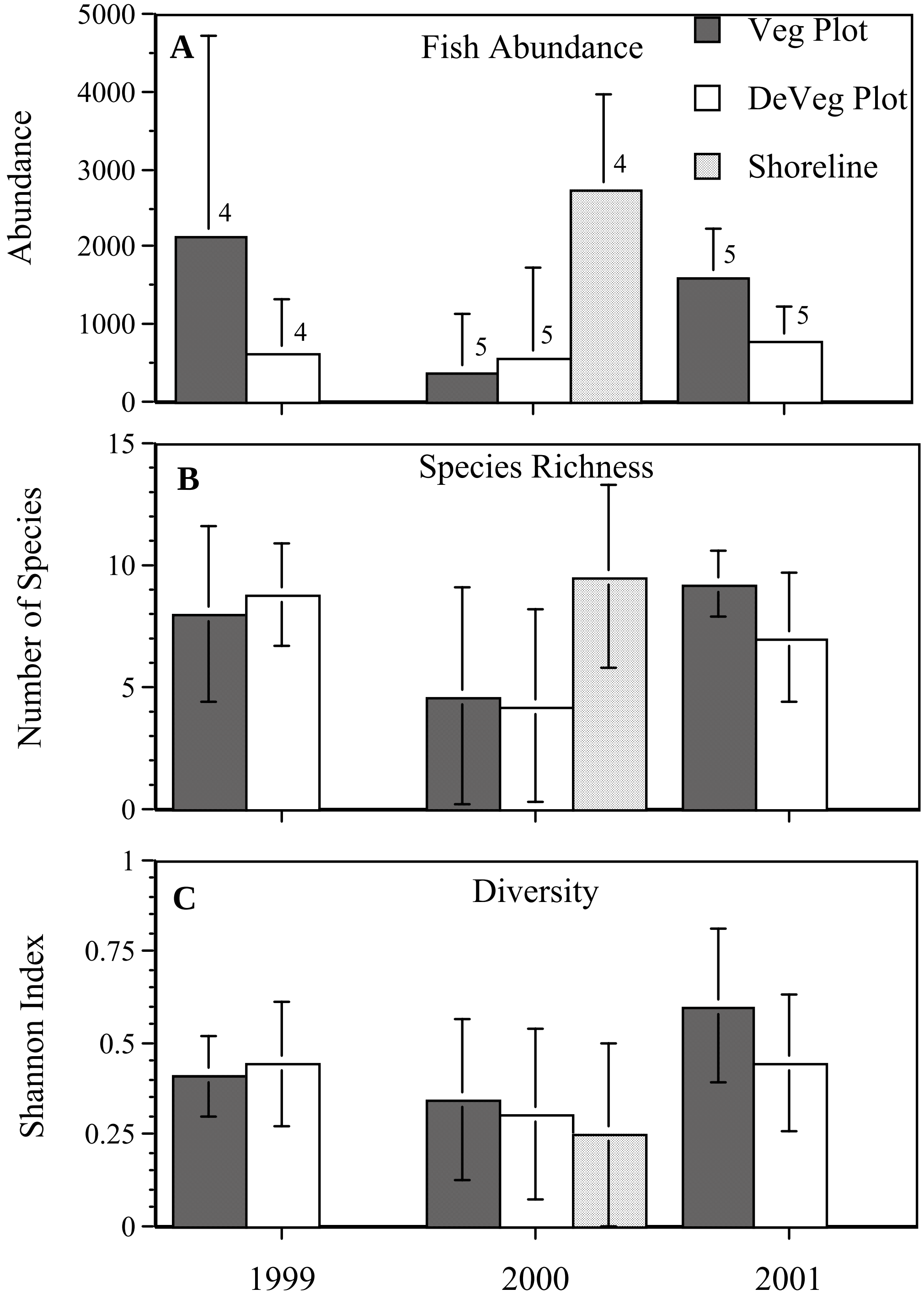

Figure 4-2. Mean (95% C.I.) fish abundance, species richness, and Shannon Index in study plots and shoreline station at sites in Pool 25, Mississippi River during 1999, 2000, and 2001. Sample sizes are above bars.

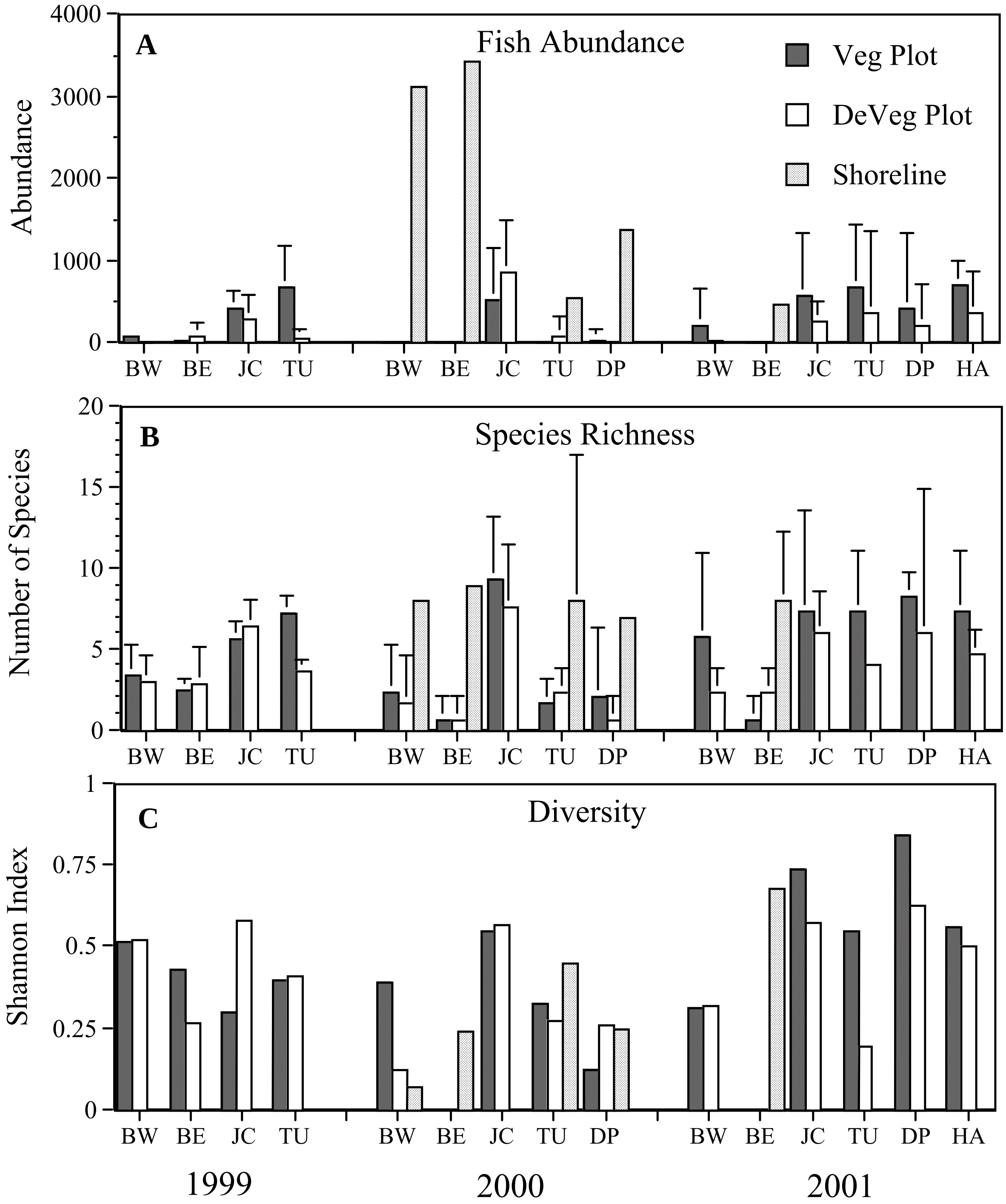

Figure 4-3. Mean (95% CI) fish abundance and species richness and total Shannon Index by site in study plots and shoreline station in Pool 25, Mississippi River during 1999, 2000, and 2001.

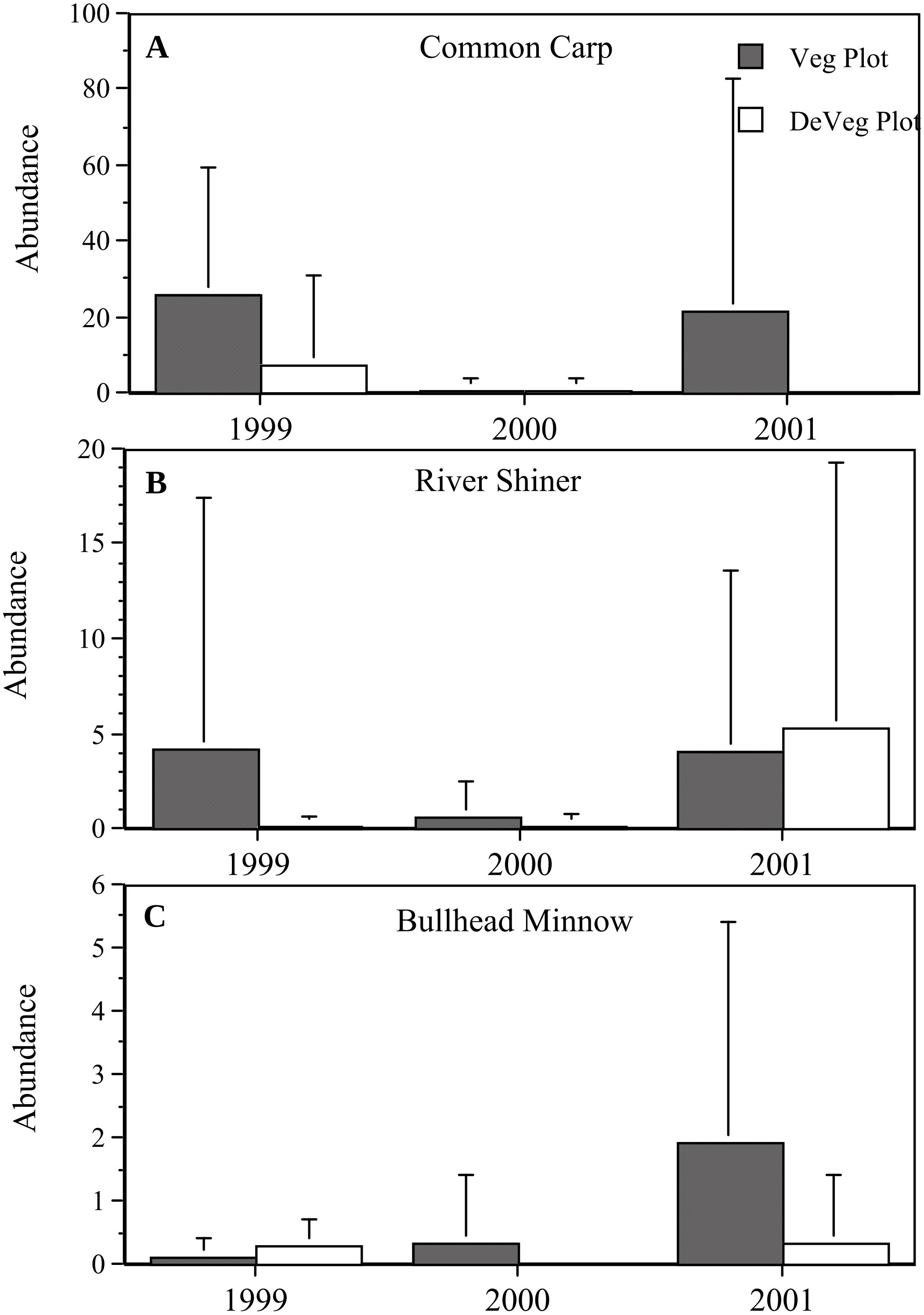

Figure 4-4. Mean (+ 95% CI) abundance of the common carp, river shiner, and bullhead minnow in study plots in Pool 25, Mississippi River during 1999, 2000, and 2001 at four sites (BE, BW, JC, and TU). Mean abundance of the bullhead minnow in devegetated plots in 2000 was zero.

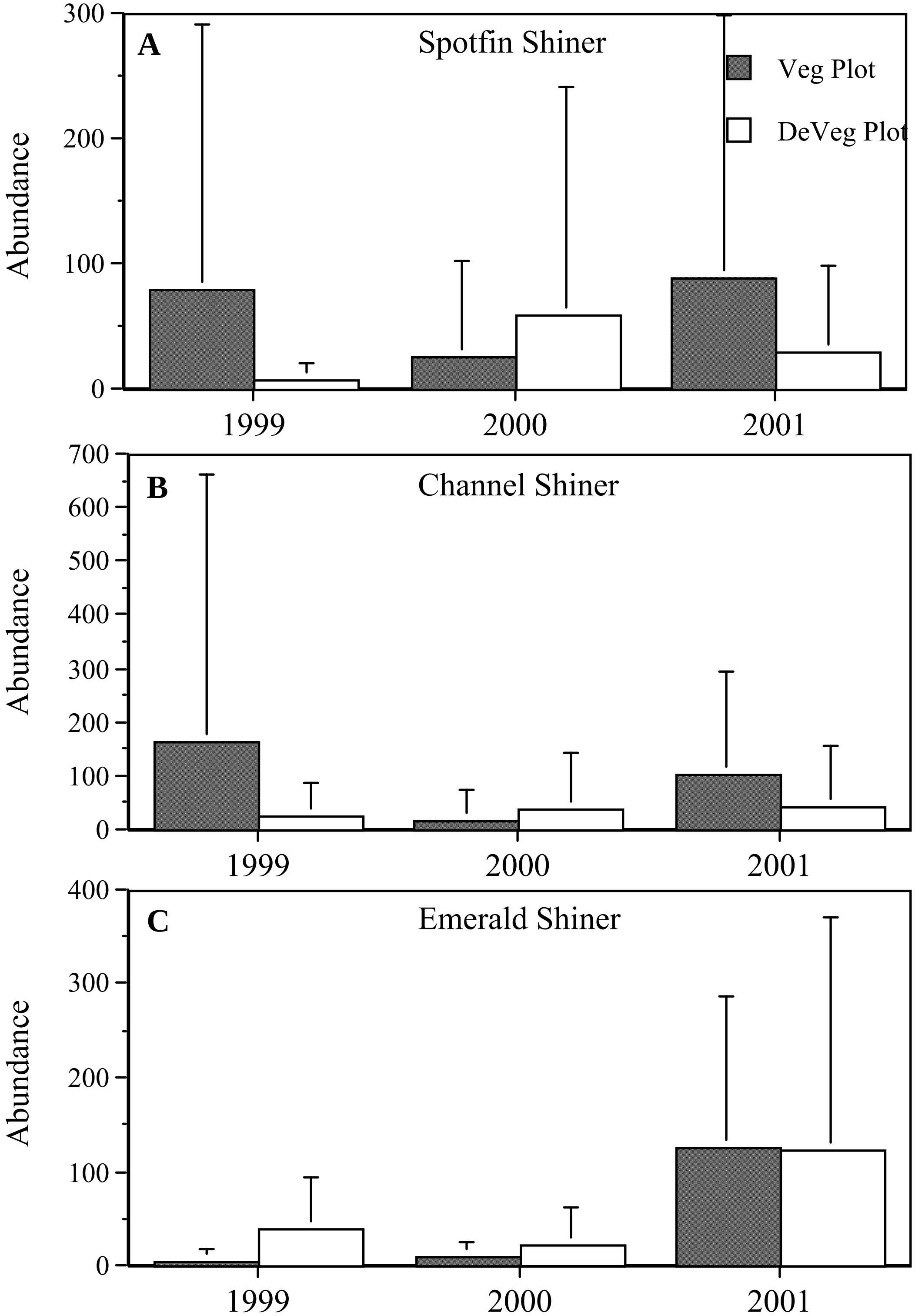

Figure 4-5. Mean (+ 95% CI) abundance of the spotfin shiner, channel shiner, and emerald shiner in study plots in Pool 25, Mississippi River during 1999, 2000, and 2001 at four sites (BE, BW, JC, and TU).

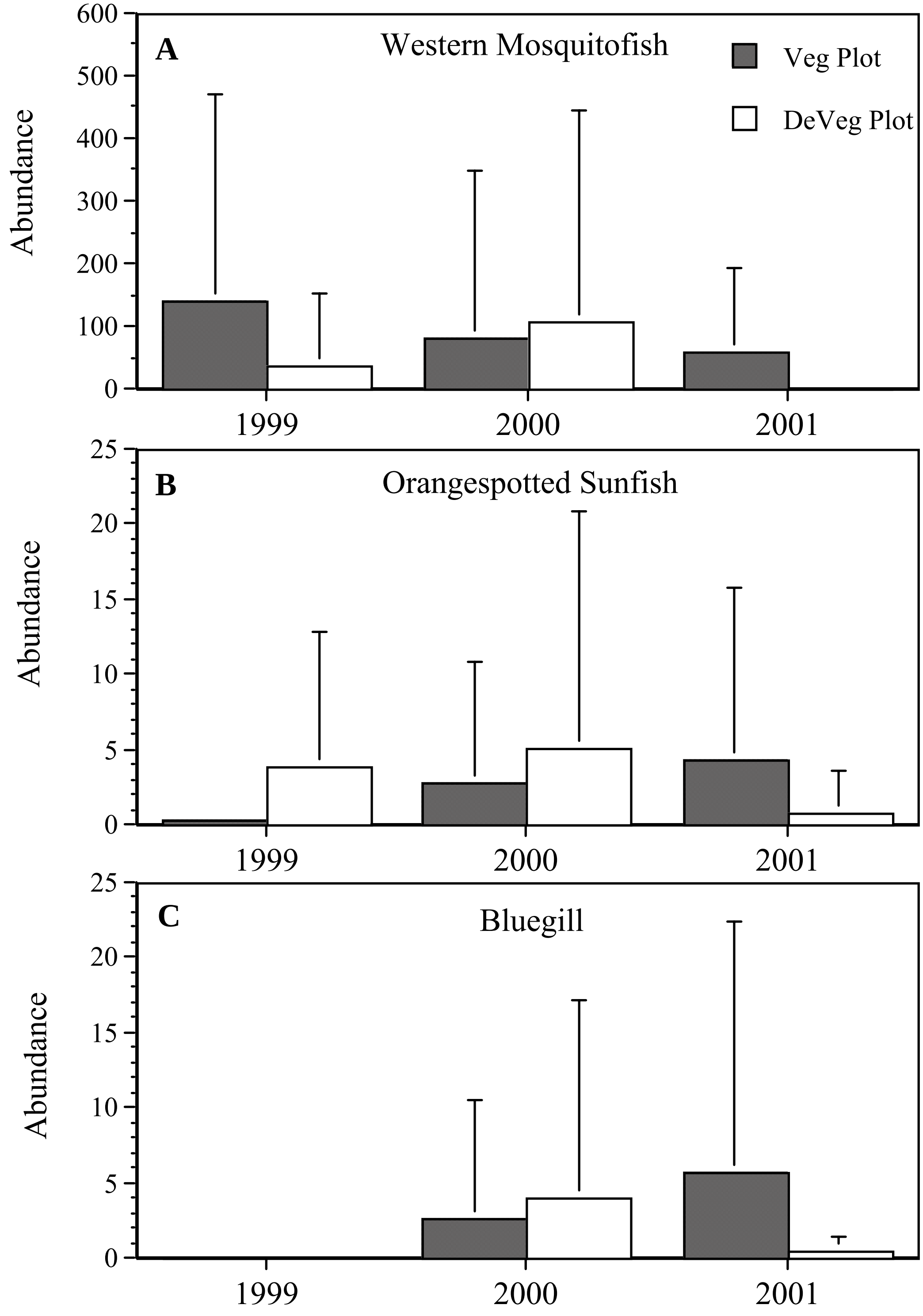

Figure 4-6. Mean (+ 95% CI) abundance of the western mosquitofish, orangespotted sunfish, and bluegill in study plots in Pool 25, Mississippi River during 1999, 2000, and 2001 at four sites (BE, BW, JC, and TU). Mean abundance of the western mosquitofish in devegetated plots during 2001 was < 1.0. Mean abundance of bluegill in 1999 was < 1.0 and zero in vegetated and devegetated plots, respectively.

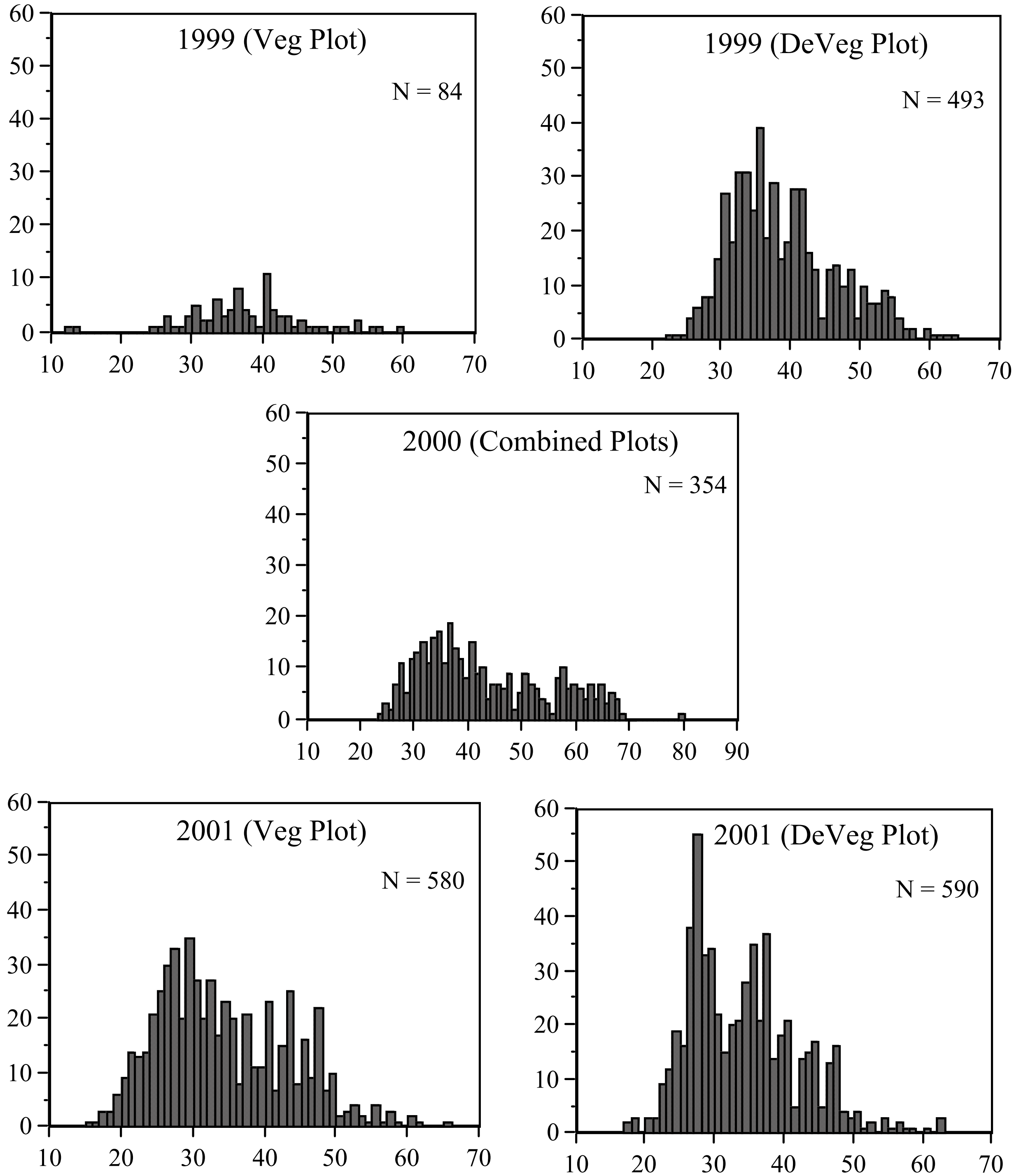

Figure 4-7. Length (mm)(x-axis) - frequency (y-axis) histograms of the emerald shiner in study plots in Pool 25, Mississippi River from 1999-2001.

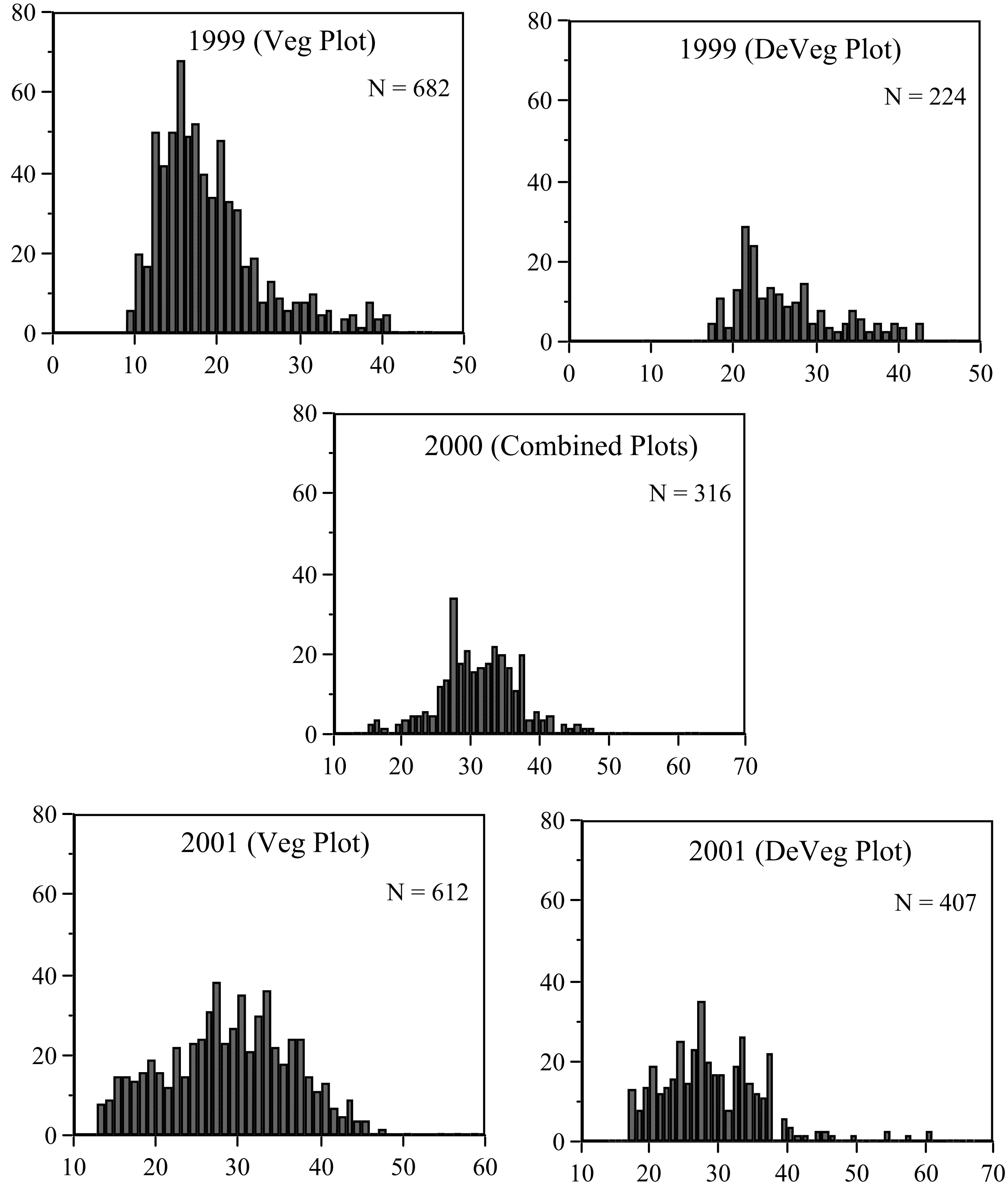

Figure 4-8. Length (mm)(x-axis) - frequency (y-axis) histograms of the spotfin shiner in study plots from sites in Pool 25, Mississippi River from 1999-2000.

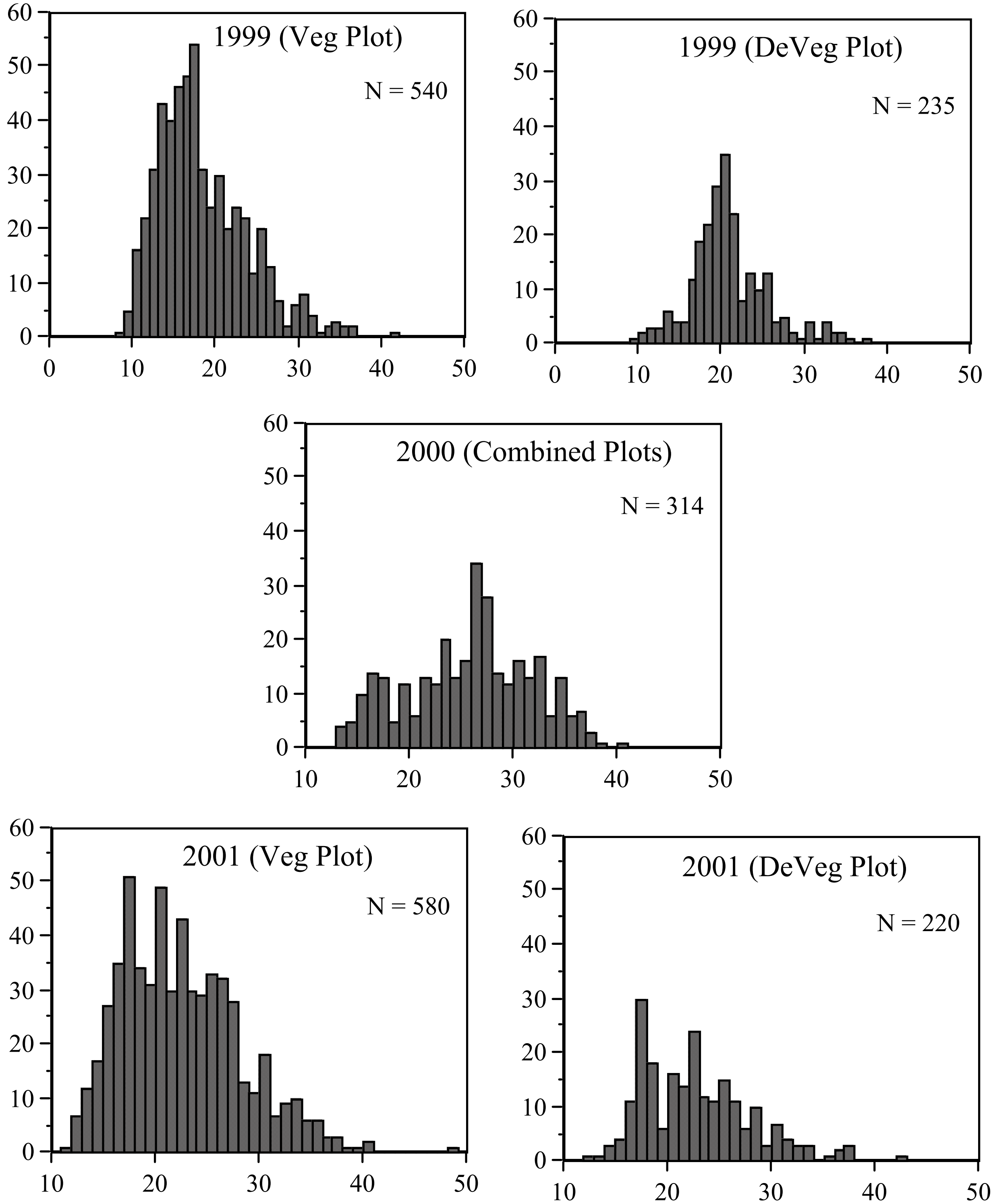

Figure 4-9. Length (mm)(x-axis) - frequency (y-axis) histograms of the channel shiner in study plots from sites in Pool 25, Mississippi River from 1999-2001.

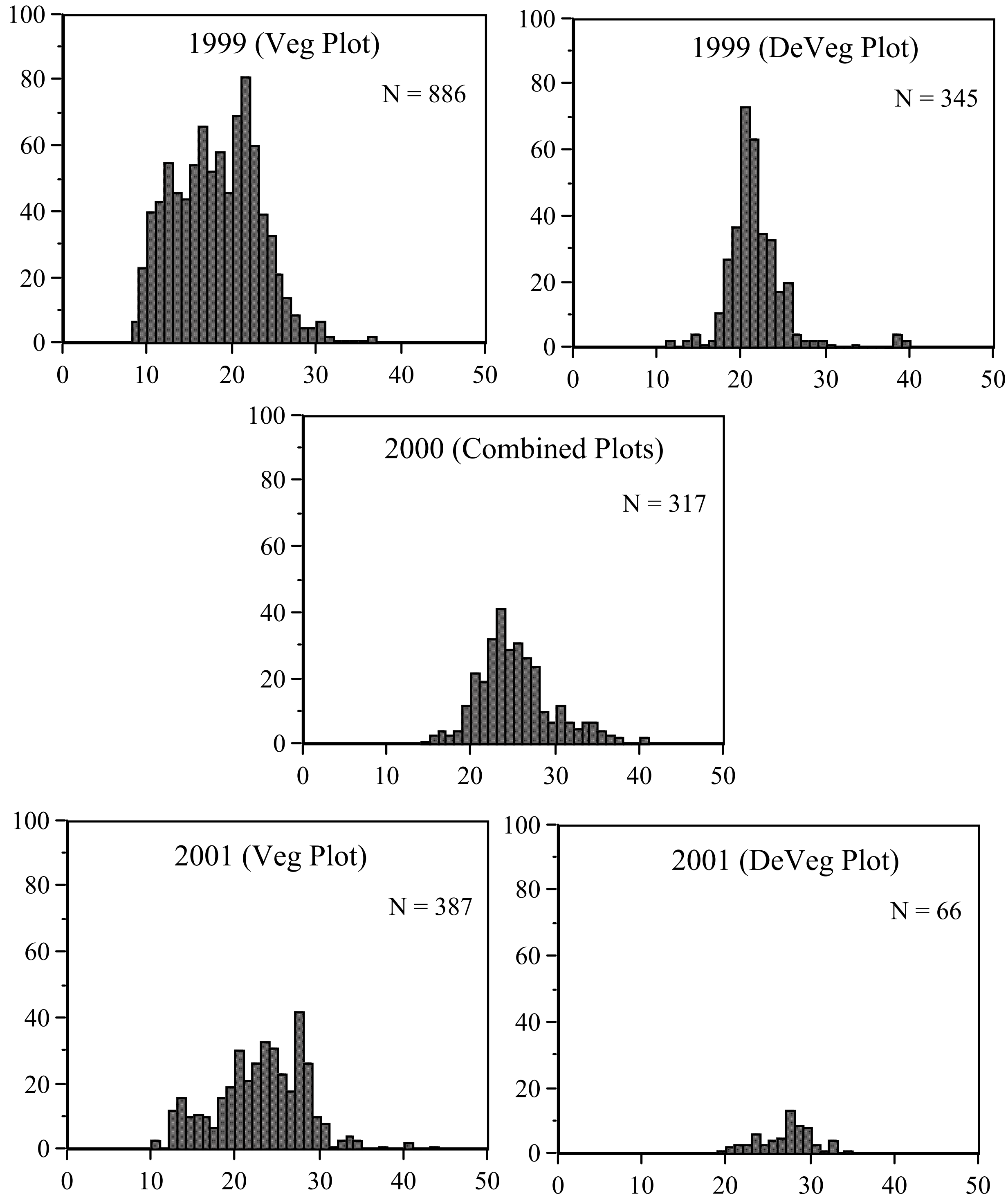

Figure 4-10. Length (mm)(x-axis) - frequency (y-axis) histograms of the western mosquitofish in study plots from sites in Pool 25, Mississippi River from 1999-2001.

Chapter 5

Macroinvertebrate and Zooplankton Responses

To Environmental Pool Management

Primary Principle Investigators: M.B. Flinn and M.R. Whiles

Objectives

1. Quantify invertebrate and organic matter responses to vegetation produced by EPM both through experimental manipulation and examining interannual differences in vegetation production.

2. Quantify differences in organic matter and abundance and diversity of invertebrates between mid-pool and lower-pool reaches of Pool 25.

3. Quantify zooplankton responses to vegetation produced by EPM.

Methods

Invertebrate and organic matter responses to EPM and emergent vegetation

Five and 6 sites during 2000 and 2001, respectively, were selected in lower Pool 25 for examining invertebrate and organic matter responses to the presence of vegetation (see Figure 2-2, Chapter 2). Jim Crow, Turner, Batchtown West, Batchtown East, and Dixon Pond were used in 2000 and Hausgen was added to increase replication during 2001. At each site, a paired-plot design consisting of a 400-m^2 vegetated plot and adjacent 400-m^2 devegetated plot were established. Vegetated plots were undisturbed, whereas a backpack sprayer and the herbicide

Rodeo® were used to remove vegetation from devegetated plots during the summer drawdown period. Plots were marked with rebar stakes and colored flagging and the same plots were used during both study years (except Hausgen which was added in 2001).

During 2000, there was a minimal vegetation response because there was only a very brief (~31 days) summer drawdown. Therefore, experimental devegetation plots were not treated with herbicide during 2000. Most sites had no differences between vegetation treatments during 2000, with exception of Jim Crow, which had a narrow band of vegetation along the shoreline in the vegetated plot. In contrast, a moderate vegetation response was observed in 2001 and devegetated plots were thus treated with herbicide during the summer drawdown period to remove vegetation. Because there was a vegetation response in 2001, we emphasize results from the 2001 study year for assessing responses to emergent vegetation.

Three 314-cm^2 stove pipe core samples were collected in each vegetated and devegetated plot after late summer/fall re-flooding in 2000 and 2001. Exact sample locations were chosen in a stratified random manner to account for the depth gradient (shallow, middle, deep) across the plots. For each sample, the corer was plunged through the water and into the substrate. All water in the core was then bailed and substrates were removed down to a depth of ~10-cm. The entire contents of each benthic core (water and substrates) were emptied into a plastic five-gallon bucket, elutriated, and rinsed through a 250-µm sieve in the field. Material retained on the sieve was placed in plastic bags and preserved in 10% formalin solution with phloxine-b dye added to aid sorting.

Organic matter - Core samples were used to estimate benthic organic matter standing stocks in vegetated and devegetated plots. In the laboratory, samples were sieved into fine particulate organic matter (FPOM = < 1mm > 250 µm) and coarse particulate organic matter

(CPOM = > 1mm) fractions. Following macroinvertebrate removal (see below) fractions were then dried for a minimum of 48 hours at 50º C, weighed, ashed at 500º C for 1-2 hours, and then reweighed to estimate ash-free dry mass (AFDM). Data were standardized to m^{-2} prior to statistical analyses.

Macroinvertebrates - In the laboratory, samples were processed to remove all macroinvertebrates by washing through nested sieves (1 mm and 250 µm mesh sizes). Invertebrates in coarse fractions (>1 mm) were sorted and removed by eye and fine fractions (<1mm >250 µm) were processed under a dissecting microscope. Fine fractions were occasionally subsampled (up to 1/16 of total) when they were large and/or contained large numbers of invertebrates. Macroinvertebrates were identified to the lowest practical taxonomic level (usually genus) according to Merrit and Cummins (1996) and Smith (2001) and measured (total body length). Taxon-specific length-mass regressions (Benke et al. 1999, Stagliano et al. 1998, Schoener, 1980, Bottrell et al. 1976, Rogers et al. 1976) were used to estimate biomass (dry mass [DM]) based on body length. Abundance and biomass data were standardized to m^{-2} prior to statistical analyses.

Zooplankton- Zooplankton communities were sampled in each vegetated and devegetated plot using a 9-cm diameter littoral sampling tube (Pennak 1962). Three samples were collected in each plot after late summer/fall re-flood (September 12 and October 14, 1999; September 30 and October 21, 2000; September 9 and October 6, 2001). Samples collected in the same year were combined in analyses. Thus, there were six samples collected in each treatment per year. Samples collected in the tube were rinsed through 80-µm Nitex® mesh, placed in 100-ml plastic containers, and preserved in 5% buffered formalin. In the laboratory, samples were rinsed through an 80-µm Nitex® mesh sieve, placed under a dissecting microscope, and counted and identified to the lowest practical taxonomic level (usually family) using Smith (2001). Raw abundance data were converted to individuals/L based on the original volume of water sampled with the tube.

Statistical analyses- Organic matter, macroinvertebrate, and zooplankton data from each year were analyzed separately using one-way analysis of variance blocked over sites. The JMP® statistical software package (SAS Institute, 2001) was used for all analyses. Values were transformed (log [x+1], or arcsine where appropriate) to normalize data and reduce heteroscedasticity prior to analysis. We tested for significance at α =0.05 and considered p values of 0.05-0.1 marginally significant because of replication constraints and the inherent variability of these types of field data sets.

<u>Response to hydrologic fluctuations: midpool versus lower pool</u>

Three sites at midpool (Coon Slough, McCoy Slough, and Gyrinid Point) and four sites at lower pool (Jim Crow Slough, Stag Slough, Serpent Slough, and Turner Tip) (see Figures 2-2, 2-3, Chapter 2) were chosen to compare macroinvertebrate and zooplankton responses to increased hydrologic variability associated with lower pool habitats during 2001-2002. Representative off-channel habitats that were contiguous with the main channel during full pool conditions were chosen for sites at midpool and lower pool. At each site, a 30-m representative linear study reach was designated on one shoreline.

Organic matter and macroinvertebrates- Benthic core samples were collected in July and September 2001 and April 2002. On each sampling date, three 314-cm^2 benthic core samples were collected and processed for organic content and macroinvertebrates using the same procedures as outlined above for experimental vegetation plots. Samples were not collected at the Serpent Slough and Turner Tip sites (both lower pool) during July 2001 because they were completely dry. As a result, it was assumed there were no aquatic macroinvertebrates present and these sites received 0 values for this date in statistical analyses. For organic matter, Serpent Slough and Turner Tip sites were not included in analyses for July 2001 because samples were not collected when the sites were dry and organic matter values were unknown. Samples collected in the vegetated treatment at Turner Tip (lower pool) for the vegetation study (see above) were used for September 2001 values for that site.

Zooplankton - Zooplankton were sampled at each site using a Wisconsin style zooplankton net (11 cm diameter) fitted with 80-µm Nitex® mesh. Three 4-m hauls were performed perpendicular to shore at mid-depth at each site in September 2001 and April 2002. In July 2001, zooplankton were collected at all midpool sites but only one lower pool site, Stag Slough, had sufficient water to sample zooplankton. Because there was no, or very little (e.g., wet mud) water present at the other lower pool sites in July, it was assumed there were no zooplankton present and they received 0 values in statistical analyses. Zooplankton samples were processed and analyzed in the same fashion as outlined above for experimental vegetation plots. Zooplankton net sampling efficiency was not calculated during our study; comparisons of our data with other studies should consider the net efficiency for our study ~50-70% (Gehringer 1968).

Statistical analyses- For midpool vs. lower pool comparisons, data were analyzed using a repeated measures analysis of variance (ANOVA). Values were log (log [x+1]) transformed to normalize data and reduce heteroscedasticity prior to analysis. Where appropriate, F values and degrees of freedom were Huynh-Feldt corrected (Milliken and Johnson 1992). All procedures were performed using JMP® statistical software (SAS Institute, 2001).

Results

<u>*Responses to Emergent Vegetation*</u>

Organic matter **-** Differences in benthic organic matter in vegetated and devegetated treatments were evident during both years, even though there was no vegetation response to the limited drawdown during 2000. In 2000, there was significantly higher total organic matter in the vegetated plots (p = 0.008), primarily because the coarse fraction was significantly higher in the vegetated plots (p = 0.004) (Figure 5-1, Table 5-1). There was no difference in the fine fraction in 2000. In 2001, total organic matter in vegetated plots was somewhat higher in vegetated plots (p = 0.125), and coarse organic matter was marginally significantly higher in vegetated plots (p = 0.087). Vegetated treatments contained 25% more total organic matter and 75% more coarse fraction organic matter during 2001, but there was no difference in fine particulate organic matter between treatments in 2001.

Macroinvertebrates- During both study years, there were no significant differences in total macroinvertebrate abundances between treatments (Figure 5-2, Table 5-2), but there were differences among individual taxa. Oligochaetes were the most abundant macroinvertebrate sampled in both 2000 and 2001 (Figure 5-3, Table 5-3). However, no significant differences in Oligochaeta total abundance were observed between treatments during the two years. Although not different in 2000, Chironomidae abundance was ~2.5x higher in devegetated treatments compared to vegetated treatments in 2001(p = 0.003, Figure 5-4, Table 5-4).

Total macroinvertebrate biomass was also similar between vegetated and devegetated treatments during 2000. During 2001, however, vegetated plots had ~2x higher macroinvertebrate biomass than devegetated plots (p = 0.001) (Figure 5-5, Table 5-5). The difference in total biomass during 2001 was primarily because of significantly higher Oligocheata biomass in vegetated plots during 2001 (p = 0.009); Oligocheata biomass was similar between treatments during 2000 (Figure 5-6, Table 5-6). In contrast to oligochaetes, Chironomidae biomass was ~5x higher in the devegated plots during 2001 (p = 0.012) (Figure 5-7, Table 5-7) owing to higher biomass of *Chironomus, Polypedilum,* and *Tanytarsus*.

Macroinvertebrate community metrics also reflected differences between treatments. During 2001, diversity was significantly higher in the vegetated plots (H′=1.02) compared to devegetated plots (H′=0.75) (p = 0.006) (Table 5-8). Dominance, calculated as co-dominance by Oligochaeta and Chironomidae, was significantly higher in devegetated plots (0.90) compared to vegetated (0.73) in 2001 ($p<0.0001$) (Table 5-8). Taxonomic richness was similar between plot types during both years (Table 5-8).

Zooplankton - Total zooplankton abundance in 1999 was 165% higher in the vegetated plots than the devegetated plots (p = 0.043) (Figure 5-8, Table 5-9), mostly due to the higher abundance of Chydoridae (p = 0.049), Sididae (p = 0.029) and cyclopoid copepods (p = 0.056) in vegetated plots. However, during 2000 and 2001 there were no significant differences in total zooplankton abundance or individual taxa abundance between treatment types.

Responses to hydrology: midpool versus lower pool

Organic matter - Total ($p<0.0001$), coarse ($p<0.0005$), and fine ($p<0.0001$) fractions of particulate organic matter differed significantly between midpool and lower pool sites during the summer 2001 re-flood – spring 2002 study period (Figure 5-9, Table 5-10). In July 2001, there was 319% more total organic matter at midpool sites compared to those in the lower pool. Conversely, in September 2001 and April 2002, the lower pool sites had approximately 2x more organic matter than midpool sites. Both CPOM and FPOM fractions followed these general trends**.**

Macroinvertebrates **-** Total macroinvertebrate abundance differed significantly between midpool and lower pool sites during the summer 2001 re-flood – spring 2002 study period (p = 0.023) (Figure 5-10, Table 5-11). The largest difference between midpool and lower pool sites was observed in July 2001 and April 2002, when total macroinvertebrate abundances were 230% and a 134% higher in lower pool sites, respectively. Total biomass was also significantly different (p = 0.026) and followed the same trend as abundance; lower pool sites had 216% and 323% higher biomass during July 2001 and April 2002, respectively (Figure 5-11, Table 5-12). During September 2001, both total macroinvertebrate abundance and biomass did not differ greatly between midpool and lower pool sites.

Of individual taxa, Oligocheata abundance was significantly different between midpool and lower pool sites (p 0.011), and this difference was most pronounced in April 2002 when lower pool sites had ~20% higher oligochaete abundance compared to midpool sites (Figure 5-12, Table 5-13). Oligochaeta biomass was also significantly different (p =0.001), and was ~9x higher at midpool sites in July 2001 (Figure 5-13, Table 5-14). However, Oligocheata biomass was 140% higher in the lower pool in April 2002.

Chironomidae abundance showed a similar pattern as oligochaetes, with a ~5x higher abundance at midpool sites compared to lower pool sites in July 2001, near equal abundances in midpool and lower pool sites in September 2001, and then ~1.5x higher abundance in lower pool sites in April 2002 (p =0.002) (Figure 5-14, Table 5-15). Chironomidae biomass was also significantly different between sites (p = 0.002) (Figure 5-15, Table 5-16), with the biggest difference, ~8.5x higher biomass in lower pool sites, in April 2002.

Macroinvertebrate community metrics also reflected differences between pools during the fall 2001 re-flood – spring 2002 study period. Significant differences in Shannon diversity (p = 0.003), were most evident in July 2001, where midpool diversity (H′=1.4) was ~2x higher than lower pool (H′=0.67) (Table 5-17). Subsequent sample dates showed more similar diversity values in September 2001 and by April 2002 diversity was higher in the lower pool sites (H′=1.05) compared to midpool (H′=0.85). Richness followed the same trend; in July richness was ~1.3x higher in midpool sites, in September there was no difference, and in April richness was ~1.5x higher in lower pool sites (Table 5-17). Dominance, calculated as co-dominance by Oligocheata and Chironomidae, was significantly different (p = 0.001) between midpool and lower pool sites. Dominance at midpool was ~1.25x higher than lower pool sites in July 2001 and April 2002, but this trend was reversed in September 2001 (Table 5-17).

***Zooplankton* -** Zooplankton abundance was significantly different between midpool and lower pool sites during the summer 2001 re-flood – spring 2002 study period (p = 0.032) (Figure 5-16, Table 5-18). Differences were most pronounced in September 2001, when abundance at midpool sites was ~2.5x higher than lower pool. In April 2002, abundance was relatively higher at both lower pool and midpool sites compared to other sample dates. Of zooplankton taxa, Cyclopoida (p = 0.032) and immature copepods (nauplii) (p = 0.004) differed most between pools. These two groups had lowest abundance in July 2001 and then gradually increased until April 2002.

Discussion

We analyzed the 2000 and 2001 study years separately because of substantial differences in hydrology and drawdown periods between the 2 years. As a result of hydrologic differences, associated vegetation responses were also quite different, with very little response during the short drawdown period (28 days at 1 ft below full pool) of 2000 and a more substantial response during the longer drawdown period of 2001 (48 days at 1 ft below full pool) (see Figure 1-2, Table 1-1, Chapter 1). Because we were interested in responses to vegetation, and very little emergent vegetation was produced during 2000, we emphasized results from 2001 for interpreting macroinvertebrate and zooplankton responses to vegetation treatments. Likewise, we assumed that the general lack of differences in the communities between vegetated and devegetated plots during 2000 was related to the lack of vegetation. For zooplankton, we also include samples collected during 1999, a year with the strongest vegetation response observed yet during EPM studies in pool 25 (Chapter 3).

Benthic organic matter in vegetation manipulation plots

The distribution of benthic organic matter can be an important determinant of invertebrate diversity and productivity (e.g., Egglishaw 1964, Silver et al. 2000, Baer et al. 2001), and some important patterns were evident during this study. Despite a lack of vegetation response in 2000, differences in benthic organic matter were evident during this year. Most likely, this was a result of residual organic matter from the strong vegetation response in these treatments during 1999 (Chapter 3). This is a potentially important result because it suggests that vegetation responses during a given year may influence food and habitat resources for invertebrates and other groups during subsequent years. However, we did not observe substantial differences in invertebrate communities between vegetated and de-vegetated treatments during 2000, indicating the residual organic matter was not having a substantial influence on invertebrate distributions, at least at the scale of our examination. This was most likely because benthic organic matter does not offer the structure and complexity of standing emergent vegetation, and thus may not have as much influence on the distribution of invertebrates. Further, since collector-gatherers dominated the invertebrate communities in these habitats, differences would likely be most evident where differences in FPOM, their primary food source, were evident (Whiles and Wallace 1995, Wallace and Webster 1996). Throughout this study, FPOM showed the least variability between treatments. In fact, average combined treatment FPOM values differed by less than 1.0 g AFDM/m^2 between 2000 and 2001, evidence that perhaps FPOM is less spatially and temporally variable in this system compared to larger particles.

Macroinvertebrate responses to vegetation

Macroinvertebrate responses to vegetation varied greatly between study years. In 2000, total macroinvertebrate abundance and biomass did not differ across treatments and no differences in individual taxa, including Chironomidae and Oligochaeta that dominated communities, were observed. We hypothesize that this lack of differences during 2000 was related to the lack of a vegetation response, even though there were some differences in benthic CPOM between treatments.

In contrast to 2000, differential responses to the presence of vegetation were evident among major invertebrate groups, although total values still did not differ across treatments. In 2001, total macroinvertebrate abundance and Oligochaeta abundance were similar across treatments. Since the oligochaetes in our samples are primarily benthic and are collector gatherers, and benthic FPOM resources were similar between treatments, this was not a surprising result. During 2001, oligochaetes outnumbered other taxa by more than 10 to 1. Hence, detecting a difference in total macroinvertebrate abundance was difficult due to the fact that other taxa made up such a small proportion of the total. Despite the similarity in abundances, Oligochaeta biomass was significantly higher in vegetated plots in 2001, indicating larger oligochaetes were present in the vegetated treatments. This may be a result of taxonomic differences in oligochaete communities in the two plot types, a factor we did not examine. Another possibility is that oligochaetes could live longer, and thus grow larger, in vegetated sites due to fewer disturbances and reduced predation. Becket et al. (1992) suggested open areas (no vegetation) of lentic littoral zones were areas of increased disturbance and higher predator efficiency. The root systems of aquatic macrophytes have also been shown to oxidize sediments (Chen and Barko 1988), and an increase in oxygen in the sediments could also shift the distribution of oligochaete taxa and/or size classes.

Chironomidae communities during 2001 were dominated by three genera, *Chironomus, Polypedilum,* and *Tanytarsus.* These three genera have been associated with a variety of habitats (Epler, 1999), and perhaps a species level preference for devegetated areas may explain the differences we observed. The possible effects of predation by other macroinvertebrates in the vegetation seem unlikely since most of these chironomid taxa burrow in the sediment, while most macroinvertebrate predators found in the vegetated plots were clingers or climbers associated with the vegetation. Fish and other vertebrate predators may also have impacted chironomid abundance, but this explanation also seems unlikely because it has been shown that vertebrate predator effectiveness is often lower in vegetated areas due to interference. For example, under laboratory conditions, Hershey (1985) showed that fishes that foraged in vegetated areas consumed about half as many chironomid larvae as those that foraged over bare sediments.

Differences in community metrics between plots (e.g., Shannon diversity) suggest that the vegetation was providing a structural habitat for macroinvertebates that was absent in devegetated treatments during 2001. The vegetated treatments had higher numbers of clinger-climber taxa such as Odonata (~9x higher), *Berosus* (Coleoptera: Hydrophilidae) larvae (~4x higher), and *Callibaetis* (Ephemeroptera: Baetidae) (39x higher). The increased abundance of these taxa undoubtedly contributed to community evenness and thus the lower dominance value in the vegetated treatment during 2001. Although these taxa were not dominant, their response is important because they represent increased diversity of prey for higher trophic levels, including prey that are available in different microhabitats (e.g., on vegetation in the water column rather than buried in sediments like oligochaetes) and they represent different size classes of prey.

Although we treated plots as replicates, important differences among plots became evident during this study and may have influenced our results. For example, Jim Crow slough had consistently high taxonomic richness in the vegetated treatment (mean 15.6) compared to other vegetated treatments (mean Dixon=13.3; Hausgen=11; Batch East=9.3). This site appeared unique and in many analyses it disproportionately influenced results. One unique characteristic of Jim Crow Slough compared to the other sites is that it becomes disconnected from the main channel for an extended period of time during the summer (~30-60 days) under EPM water level regulation. Unless high water events inundate Jim Crow Slough during the summer, it may go through periods of occasional drying. If not too frequent or severe, these drying events could be classified as moderate disturbances that limit biotic interactions and lead to increased richness and diversity (e.g., Connel 1978, Whiles and Goldowitz 2001).

Zooplankton responses to vegetation

Zooplankton abundance varied less than expected from 1999-2001. In 2000 and 2001, little or no differences were found in total zooplankton abundance or individual taxa. As discussed above, the vegetation responses differed between the three years sampled and zooplankton abundance appeared to follow vegetation responses. For example, in 2000, when there was no vegetation response, there were no differences in zooplankton abundance. In 2001, there was a moderate vegetation response and a slight difference in the zooplankton abundance in treatments was evident. Finally, the 1999 vegetation was quite strong and this was the only year in which there was a statistical difference in zooplankton abundance between treatments (higher in vegetated plots).

Based on water quality data, the difference in zooplankton abundance observed in 1999 might seem unexpected. Dissolved oxygen was often low in the vegetated treatments during 1999 because shading effects from the heavy vegetation canopy limited light penetration and presumably photosynthetic activity (Chapter 4). However, in 1999 we did record lower levels of turbidity in the vegetated treatment (Chapter 4), and it has been shown that suspended fine particulates like clay and silt may impede zooplankton feeding if they overwhelm algal particles in the water (Threlkeld 1986, Kirk 1992). Additionally, Chydoridae (Cladocera) and Cyclopoida (Copepoda) were the two most abundant taxa and were responsible for differences between plots. Chydorids are typically not planktonic, but crawl along surfaces where they scrape or filter detrital particles (Fryer 1968). Thus, the observed shading effects of the 1999 vegetation may not have affected chydorids. Likewise, many cyclopoid copepods are carnivorous (Fryer 1957) and thus would also be unaffected by low phytoplankton abundance.

Responses to hydrologic variability: midpool vs. lower pool

The midpool and lower pool regions of Mississippi River Pool 25 are characterized by different hydrologic regimes. The midpool sites we chose for this study were similar to the lower pool sites during full pool, but the two sites experience different hydrologic dynamics as the pool is drawn down. The midpool sites undergo periods of lower water during drawdown, but rarely, if ever go dry, whereas lower pool sites are subject to more extremes and can go dry during drawdown. Hence, hydroperiods at the midpool sites are generally longer, with almost continuous water throughout the year, whereas the hydroperiods at the lower pool sites are shorter, and these sites often experience periods without water. For this reason, there is a distinct difference between the vegetation responses at midpool sites versus lower pool sites, with a much higher response at lower pool. Even with drawdown events associated with EPM, water levels at midpool are minimally influenced. The midpool sites generally have steeper sided banks and therefore a comparable change in water level inundates or exposes far less shoreline and mudflat habitat at midpool compared to lower pool, which in turn can influence development of vegetation. Our comparison of midpool and lower pool backwater sites provides further information on the effects of environmental pool management by comparing sites directly targeted by EPM (lower pool) with sites that are less affected (midpool).

Benthic organic matter dynamics varied significantly between midpool and lower pool sites during our study. Organic matter resources have been shown to be important to macroinvertebrates, and differences in the availability and distribution of this potentially limiting resource could influence macroinvertebrate communities (e.g., Egglishaw 1964, Silver et al. 2000, Baer et al. 2001). A noticeable difference was that midpool benthic organic matter values were generally less variable across sample dates (mean ~220 g AFDM/m^2 +/- 15%) compared to lower pool sites (mean ~310 g AFDM/m^2 +/- 76%). This result was likely due to differences in hydrology (e.g., periods of deposition associated with low water and periods of scour associated with high water and connectivity to the river) between the two regions of the pool.

During September 2001 and April 2002, lower pool sites had ~2x more benthic organic matter compared to midpool sites. This difference may be due to greater amounts of deposition at the lower pool sites that become more lentic and disconnected from the river for longer periods than midpool. Also, the general lack of inundated moist soil vegetation at midpool compared to lower pool sites may have contributed to this pattern. Vegetation produced *in situ* at lower pool sites appeared to contribute greatly to benthic organic matter pools there, whereas midpool organic matter resources had more woody debris and tree leaves in the CPOM fraction. Thus, the source of coarse organic materials at midpool sites was less clear than for lower pool, but may be coming from upstream sources or lateral inputs from trees and other floodplain vegetation (e.g., Vannote et al. 1980, Junk et al. 1997).

Macroinvertebrates responded differently to lower pool and midpool hydrology. Macroinvertebrate communities were both more abundant and higher in biomass in the lower pools throughout the study period. This response was driven by a few dominant taxa in the lower pool, the oligochaetes and the Chironomidae genera *Chironomus*, *Polypedilum* and *Tanytarsus*, all of which may prefer the more depositional nature of the lower pool sites. The exceptionally large difference in total macroinvertebrate abundance and biomass in July 2001 may have been exaggerated due to concentration in the lower pool sites that had water and were sampled, Jim Crow Slough and Stag Slough. Water in both sites during this period was confined to a narrow strip (<1m) of very shallow water (<5cm) in the middle of the site, and remaining macroinvertebrates were likely becoming increasingly concentrated. This may have also increased the chance of catching rare or uncommon species and thus inflated richness and diversity values at these sites during this period.

An important result of this study that was not evident from richness or diversity values was the differences in macroinvertebrate community composition between the midpool sites and lower pool sites. There were 13 unique taxa (three Ephemeroptera families, one Odonata genus, one Trichoptera genus, one Coleoptera family, and 7 Diptera genera) collected at midpool that were not collected at lower pool sites. Many of these taxa have longer life cycles (e.g., univoltine: one generation per year) that require longer periods of inundation, and thus they may not survive at lower pool sites because of more frequent and intense dry periods.

The difference in hydroperiod between midpool sites and lower pool sites may shape macroinvertebrate communities in different ways. The longer hydroperiods and more stable conditions at midpool sites likely favors longer-lived taxa, but the more frequent hydrologic disturbances at the lower pool sites could enhance diversity by limiting biotic interactions (e.g., Connel, 1978). Whiles and Goldowitz (2001) showed that backwater wetland sites of the Platte River that had intermediate hydroperiods (250-300 days/year), and thus intermediate levels of drying disturbance, had higher richness than sites with longer or shorter hydroperiods. Conversely, it has been shown in other studies that freshwater habitats that are least disturbed can harbor higher invertebrate richness (Death and Winterborne 1995), and timing since the last disturbance is most important. In the case of pool 25, the timing of the drawdown may be an important factor affecting richness and diversity of macroinvertebrates in pool 25. Since the drawdown usually begins in early summer (June), many univoltine macroinvertebrate taxa have already completed the aquatic portion of their life cycle and emerged as adults. Macroinvertebrates that have shorter bivoltine or polyvoltine (i.e., two or more generations per year) life cycles are likely affected by dry periods and experience reduced abundance, but these types of invertebrates are quick to recolonize and build up populations following disturbance (e.g., Gray 1981, Molles 1985, Wallace 1990, Whiles and Wallace 1995).

The mix of hydrologically different habitats (e.g., midpool vs. lower pool) that results from managing pool 25 with a midpool control point may be an optimal situation. Sites at midpool provide habitats with longer hydroperiods and thus have more longer-lived and larger-bodied taxa, while sites at lower pool provide habitats with relatively shorter hydroperiods and more emergent vegetation that may enhance macroinvertebrate abundance, biomass, and production. Combined on a larger scale, this hydrologic habitat diversity likely enhances macroinvertebrate abundance and diversity at the landscape scale.

The more lentic conditions of lower pool sites appeared to favor zooplankton communities, at least by April 2002. For example, at Jim Crow Slough and Stag Slough, which are the most disconnected and lentic of all sites we examined, zooplankton abundances were ~10x and ~2.5x greater, respectively, than those of other sites during this study. However, the lack of water in lower pool sites during July 2001, and the rather short wet period from re-flood to the September 2001 sampling date may have limited development of zooplankton communities in summer-fall. During July 2001, only one sample was taken in the lower pool sites because most sites were dry and thus had no zooplankton in them. However, all midpool sites had water in July. Hence, the more lentic lower pool sites appear to support higher abundance of zooplankton during most of the year, exclusive of long drawdown events and subsequent recovery periods. Sites at midpool may not be optimal habitat for zooplankton because they periodically experience flow through them which may flush zooplankton.

The results of this study suggest numerous benefits of EPM. In particular, EPM effectively provides a heterogeneity of habitats with different hydrologic regimes within the same pool, and is thus compatible with management goals to maximize available food for waterfowl and fish because on a landscape scale macroinvertebrate diversity and abundance are enhanced. While midpool and lower pool may be different in terms of hydrology and macroinvertebrate communities, combined they provide increased macroinvertebrate resources in terms of diversity, abundance, and biomass.

We hypothesize that midpool and lower pool habitats represent hydrologic extremes of a gradient that exists between the two points. This may be important and deserves further investigation. For example, the possibility of a gradient of hydrologic conditions from midpool to lower pool could serve a dual purpose of further enhancing the range of hydrologic habitats within the pool and thus macroinvertebrate and zooplankton diversity. Organisms requiring more stable habitats with longer hydroperiods will likely show increasing abundance in off-channel habitats as midpool is approached, while management practices that use dry periods to enhance emergent vegetation, invertebrate abundance and biomass, and waterfowl use will be more effective as lower pool is approached.

Though the management goals of EPM do not include macroinvertebrate and zooplankton community responses *per se*, these groups have a substantial role in a successful ecosystem management plan. Freshwater macroinvertebrates and zooplankton are a well-documented food source to higher trophic levels including fish and birds, and this includes emergent adult insects that are important in riparian food webs (e.g., Orians 1964, Hynes 1970, Orians1980, Street 1977, Sjöberg and Danell 1982, Gray 1993, Cox and Kadlec 1995). Therefore, the primary goals of EPM to increase vegetation and seed biomass also provide the additional benefit of enhancing macroinvertebrate and zooplankton resources.

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Table 5-1. Benthic organic matter for six devegetated (deveg) and vegetated (veg) sites on Pool 25, Mississippi River in fall of 2000 and 2001. Numbers represent dry weight of ash free organic matter (g AFDM / m^2). The Hausgen site was added in 2001.

		2000						2001					
		DEVEG			VEG			DEVEG			VEG		
Site	Core	Total	CPOM	FPOM	Total	CPOM	FPOM	Total	CPOM	FPOM	Total	CPOM	FPOM
Batch East	1	281	190	91	586	294	292	475	176	299	409	177	231
Batch East	2	459	243	216	572	412	160	358	141	217	530	301	229
Batch East	3	652	399	254	550	395	155	229	109	121	414	160	255
Batch West	1	101	45	55	374	179	195	459	247	213	385	237	148
Batch West	2	66	23	43	172	127	45	441	277	165	300	171	129
Batch West	3	72	26	46	476	326	150	390	211	179	447	339	108
Dixon	1	164	83	81	189	56	133	308	83	224	236	95	141
Dixon	2	288	128	160	207	67	140	150	48	102	204	33	171
Dixon	3	254	109	144	216	85	131	209	58	150	215	50	165
Jim Crow	1	149	70	78	387	260	127	179	106	73	323	173	150
Jim Crow	2	356	290	66	446	327	119	226	78	147	231	138	93
Jim Crow	3	267	186	82	1,062	718	344	500	334	166	173	102	71
Turner	1	236	122	115	110	99	11	62	14	48	846	510	336
Turner	2	232	110	123	574	336	238	127	25	102	305	182	123
Turner	3	289	133	155	786	486	300	35	9	26	404	202	202
Hausgen	1	-	-	-	-	-	-	250	199	51	45	14	30
Hausgen	2	-	-	-	-	-	-	100	44	56	234	185	49
Hausgen	3	-	-	-	-	-	-	98	55	42	164	111	53
Mean g AFDM / m^2		**258**	**144**	**114**	**447**	**278**	**169**	**255**	**123**	**132**	**326**	**177**	**149**
Standard Error		39	27	16	67	47	24	36	23	18	42	28	19

Table 5-2. Total macroinvertebrate abundance (no./m^2) at devegetated (deveg) and vegetated (veg) sites in Mississippi River pool 25 during fall of 2000 and 2001. Values are presented by individual sample (core). The Hausgen site was added in 2001.

		2000		2001	
Site	**Core**	**Deveg**	**Veg**	**Deveg**	**Veg**
Batch East	1	21,888	16,384	17,216	55,424
Batch East	2	29,248	29,088	21,504	28,896
Batch East	3	23,008	23,520	20,992	28,256
Batch West	1	31,712	24,096	35,136	19,296
Batch West	2	22,976	31,584	14,560	8,480
Batch West	3	48,896	38,400	31,232	62,304
Dixon	1	25,856	31,200	146,176	179,904
Dixon	2	84,672	67,776	94,368	98,080
Dixon	3	39,264	24,128	66,528	89,760
Jim Crow	1	21,600	99,968	74,176	24,448
Jim Crow	2	41,696	73,568	45,056	38,720
Jim Crow	3	9,952	11,904	34,336	33,856
Turner	1	19,072	41,984	27,200	20,032
Turner	2	40,992	24,000	27,008	32,608
Turner	3	25,056	26,400	28,160	8,160
Hausgen	1	-	-	27,904	23,968
Hausgen	2	-	-	19,008	54,496
Hausgen	3	-	-	28,288	81,248
Mean abundance		**32,393**	**37,600**	**42,158**	**49,330**
Standard error		4,580	6,266	7,930	9,933

Table 5-3. Total macroinvertebrate biomass in 6 devegetated (deveg) and 6 vegetated (veg) sites in Mississippi River pool 25 in fall of 2000 and 2001. Numbers represent total macroinvertebrate dry mass (DM)/m^2 and are presented by individual sample (core). The Hausgen site was added in 2001.

		2000		2001	
Site	Core	**Deveg**	**Veg**	**Deveg**	**Veg**
Batch East	1	489	691	1,546	921
Batch East	2	1,067	497	788	612
Batch East	3	1,256	637	1,499	778
Batch West	1	1,069	989	606	1,255
Batch West	2	647	397	193	997
Batch West	3	1,935	1,541	407	1,396
Dixon	1	1,003	797	2,478	3,375
Dixon	2	1,160	2,014	1,226	1,209
Dixon	3	2,482	807	733	1,747
Jim Crow	1	1,028	1,817	2,903	5,895
Jim Crow	2	5,845	2,649	2,779	4,317
Jim Crow	3	272	3,968	1,874	6,367
Turner	1	2,430	974	268	1,745
Turner	2	3,404	1,164	572	2,049
Turner	3	1,193	1,295	253	810
Hausgen	1	-	-	600	1,358
Hausgen	2	-	-	1,986	2,364
Hausgen	3	-	-	490	5,056
Mean Biomass		**1,685**	**1,349**	**1,178**	**2,347**
Standard Error		368	246	212	435

Table 5-4. Oligocheata abundance (no./m^2) at 6 devegetated (deveg) and 6 vegetated (veg) sites in Mississippi River pool 25 during fall of 2000 and 2001. Numbers are presented by individual sample (core). The Hausgen site was added in 2001.

		2000		2001	
Site	Core	Deveg	**Veg**	**Deveg**	**Veg**
Batch East	1	8,864	13,408	13,440	31,232
Batch East	2	22,208	7,648	16,064	19,872
Batch East	3	19,584	15,104	10,400	15,008
Batch West	1	20,768	18,592	26,848	14,112
Batch West	2	18,304	12,416	12,896	7,680
Batch West	3	27,552	26,144	26,016	44,672
Dixon	1	22,944	24,064	115,136	102,784
Dixon	2	68,672	50,464	73,632	32,608
Dixon	3	36,096	18,816	53,696	47,072
Jim Crow	1	12,480	63,808	64,000	14,848
Jim Crow	2	20,704	56,864	34,112	27,360
Jim Crow	3	4,320	11,200	30,080	23,392
Turner	1	16,768	36,224	26,656	14,464
Turner	2	35,904	21,664	25,216	31,680
Turner	3	16,832	21,536	24,352	6,080
Hausgen	1	-	-	26,432	17,408
Hausgen	2	-	-	13,312	40,000
Hausgen	3	-	-	17,216	44,960
Mean abundance		**23,467**	**26,530**	**33,861**	**29,735**
Standard Error		3,917	4,497	6,342	5,261

Table 5-5. Oligocheata biomass for 6 devegetated (deveg) and vegetated (veg) sites on Pool 25, Mississippi River in fall of 2000 and 2001. Numbers represent biomass of dry weight of individuals (mg DM / m^2) derived from length-mass equations. The Hausgen site was added in 2001.

		2000		2001	
Site	Core	**Deveg**	**Veg**	**Deveg**	**Veg**
Batch East	1	385	584	1,159	549
Batch East	2	750	433	662	295
Batch East	3	731	550	928	567
Batch West	1	880	696	251	1,245
Batch West	2	444	244	130	971
Batch West	3	843	1,103	188	687
Dixon	1	779	439	1,260	1,574
Dixon	2	830	1,255	908	692
Dixon	3	2,246	552	512	1,362
Jim Crow	1	419	1,223	775	3,037
Jim Crow	2	1,328	1,612	988	1,808
Jim Crow	3	237	1,126	1,550	1,172
Turner	1	1,041	755	239	852
Turner	2	2,271	872	518	527
Turner	3	635	933	209	607
Hausgen	1	-	-	404	968
Hausgen	2	-	-	1,375	742
Hausgen	3	-	-	228	724
Mean Biomass		**921**	**825**	**683**	**1,021**
Standard Error		157	98	107	151

Table 5-6. Chironomidae abundance at six devegetated (deveg) and vegetated (veg) sites in Pool 25, Mississippi River during fall of 2000 and 2001. Numbers represent Chironomidae in each benthic stovepipe core (no. / m^2). The Hausgen site was added in 2001.

		2000		2001	
Site	Core	**Deveg**	**Veg**	**Deveg**	**Veg**
Batch East	1	6,432	1,696	2,112	3,616
Batch East	2	7,040	2,112	3,136	3,616
Batch East	3	3,328	1,216	4,672	4,544
Batch West	1	2,240	5,472	1,632	64
Batch West	2	3,360	3,808	1,408	0
Batch West	3	1,952	4,000	512	0
Dixon	1	2,592	7,136	22,048	2,496
Dixon	2	7,776	6,976	14,432	2,112
Dixon	3	2,624	3,552	5,984	2,624
Jim Crow	1	8,352	15,808	3,136	1,440
Jim Crow	2	17,344	7,232	3,296	4,352
Jim Crow	3	992	320	1,600	1,888
Turner	1	1,408	4,640	544	1,920
Turner	2	3,776	1,824	1,536	0
Turner	3	7,104	4,832	3,808	1,664
Hausgen	1	-	-	544	1,888
Hausgen	2	-	-	2,720	32
Hausgen	3	-	-	8,128	32
Mean abundance		**5,088**	**4,708**	**4,514**	**1,794**
Standard Error		1,083	975	1,302	366

Table 5-7. Chironomidae biomass for 6 devegetated (deveg) and vegetated (veg) sites in Pool 25, Mississippi River in fall of 2000 and 2001. Numbers represent biomass of dry weight of individuals (mg DM / m^2) derived from length-mass equations. The Hausgen site was added in 2001.

		2000		2001	
Site	Core	**Deveg**	**Veg**	**Deveg**	**Veg**
Batch East	1	100	49	257	274
Batch East	2	204	64	122	257
Batch East	3	507	85	270	193
Batch West	1	188	246	248	6
Batch West	2	112	153	63	0
Batch West	3	231	324	92	0
Dixon	1	210	358	275	603
Dixon	2	287	689	229	120
Dixon	3	229	243	167	162
Jim Crow	1	382	546	56	11
Jim Crow	2	317	207	31	41
Jim Crow	3	14	8	13	20
Turner	1	74	161	29	34
Turner	2	1,130	128	53	0
Turner	3	453	218	44	195
Hausgen	1	-	-	8	32
Hausgen	2	-	-	49	1
Hausgen	3	-	-	239	1
Mean Biomass		**296**	**232**	**125**	**108**
Standard Error		69	48	24	37

Table 5-8. Macroinvertebrate community metrics for devegetated (deveg) and vegetated (veg) sites in Pool 25, Mississippi River in fall of 2000 (n=5) and 2001 (n=6). Numbers represent mean (+/- S.E). Numbers with asterisks are significantly different within year at p<0.05.

	2000				**2001**			
	DEVEG		VEG		DEVEG		VEG	
Richness	9.667	(0.797)	8.733	(0.733)	9.722	(0.690)	10.222	(0.931)
Shannon-Weiner Index	0.884	(0.098)	0.762	(0.061)	0.751*	(0.080)	1.016*	(0.083)
Dominance	0.875	(0.040)	0.841	(0.050)	0.899*	(0.018)	0.725*	(0.036)

Table 5-9. Zooplankton abundance at six devegetated (deveg) and vegetated (veg) sites in Pool 25, Mississippi River during fall of 1999, 2000 and 2001. Numbers represent zooplankton in each integrated water column sample (no./L). Dixon Pond was added in 2000 and Hausgen was added in 2001.

		1999		2000		2001	
Site	**Sample**	**Deveg**	**Veg**	**Deveg**	**Veg**	**Deveg**	**Veg**
Batch East	1	2.68	7.02	0.39	3.59	6.90	4.53
Batch East	2	1.02	5.15	0.46	0.61	5.05	3.56
Batch East	3	3.62	10.69	0.91	0.46	5.28	3.96
Batch West	1	0.40	0.53	0.74	0.78	0.87	1.71
Batch West	2	0.44	0.67	0.26	1.23	0.73	1.54
Batch West	3	0.44	4.77	0.62	0.32	0.36	1.31
Dixon Pond	1	-	-	2.41	3.45	4.09	6.21
Dixon Pond	2	-	-	4.09	3.45	12.27	3.07
Dixon Pond	3	-	-	5.24	2.74	4.01	2.75
Hausgen	1	-	-	-	-	0.67	0.51
Hausgen	2	-	-	-	-	0.39	0.63
Hausgen	3	-	-	-	-	2.28	0.24
Jim Crow	1	6.13	3.49	3.36	0.16	4.04	4.60
Jim Crow	2	5.50	1.57	1.62	1.73	9.51	2.49
Jim Crow	3	2.85	3.29	2.23	3.61	5.97	7.78
Turner	1	0.16	1.49	0.13	0.00	1.18	0.31
Turner	2	0.13	0.24	0.47	1.20	0.94	3.93
Turner	3	0.33	0.13	0.07	0.16	2.52	2.61
Mean Abundance (no. / L)		**1.97**	**3.25**	**1.53**	**1.57**	**3.73**	**2.87**
Taxa							
Copepoda							
Cyclopoida		0.93	5.05	2.53	2.17	9.31	10.02
Calanoida		0.51	0.34	1.08	0.49	3.28	0.92
Harpacticoid		0.03	0.06	0.12	0.86	0.17	0.00
Nauplii		17.49	24.31	7.55	8.23	16.47	10.12
Cladocera							
Bosminidae		0.07	0.03	1.30	1.26	0.33	0.29
Chydoridae		0.06	0.72	1.25	1.29	0.55	0.66
Daphnidae		0.00	0.04	0.20	0.08	0.22	0.47
Macrothricidae		0.20	0.83	0.32	0.39	0.12	0.25
Moinidae		0.02	0.00	0.12	0.49	6.38	4.18
Sididae		0.46	1.17	0.86	0.38	0.42	1.81
Mean Abundance (no. / L)		**2.09**	**3.06**	**1.42**	**1.50**	**3.10**	**2.08**

Table 5-10. Benthic Organic matter in midpool (MP, n=3) and lower pool (LP, n=4) sites during 2001 and 2002. Values are ash-free dry mass (AFDM)/m^2 +/- std. error of total, coarse (CPOM) and fine (FPOM) particulate organic matter.

			July 2001				September 2001				April 2002		
Site	Pool	Core	Total	CPOM	FPOM		Total	CPOM	FPOM		Total	CPOM	FPOM
Jim Crow	LP	1	71	24	47		124	72	52		526	218	307
Jim Crow	LP	2	97	50	47		404	280	124		141	43	97
Jim Crow	LP	3	82	31	51		381	284	96		318	122	196
Serpent	LP	1	-	-	-		263	167	96		424	205	219
Serpent	LP	2	-	-	-		252	102	150		601	436	166
Serpent	LP	3	-	-	-		160	97	64		416	224	192
Stag	LP	1	122	45	77		191	111	80		633	408	225
Stag	LP	2	34	16	17		1,054	667	387		485	225	260
Stag	LP	3	34	14	20		145	69	77		1,350	1,036	315
Turner	LP	1	-	-	-		846	510	336		187	16	171
Turner	LP	2	-	-	-		305	182	123		281	44	236
Turner	LP	3	-	-	-		404	202	202		390	63	326
Mean (g AFDM / m^2)	**LP**		73	30	43		377	229	149		479	253	226
Standard error			14	6	9		83	54	31		90	81	20
McCoy	MP	1	96	13	83		207	45	162		97	32	65
McCoy	MP	2	205	57	148		70	18	52		172	60	113
McCoy	MP	3	74	14	60		77	30	48		200	96	103
Coon	MP	1	208	108	100		100	7	93		455	296	159
Coon	MP	2	96	10	86		138	71	67		183	86	97
Coon	MP	3	242	66	175		217	133	84		178	18	160
Gyrinid	MP	1	218	47	172		234	75	159		355	107	247
Gyrinid	MP	2	437	276	161		301	123	178		153	50	103
Gyrinid	MP	3	525	322	203		340	151	188		385	223	162
Mean (g AFDM / m^2)		***MP***	233	101	132		187	73	115		242	108	134
Standard error			52	39	17		32	18	19		41	31	18

Table 5-11. Total macroinvertebrate abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent abundance of invertebrates (no. / m^2).

Site	Pool	Core	**July 2001**	September 2001	**April 2002**
Jim Crow	LP	1	47,968	39,168	196,960
Jim Crow	LP	2	27,520	22,336	65,184
Jim Crow	LP	3	53,216	11,520	98,176
Serpent	LP	1	0	26,848	125,664
Serpent	LP	2	0	22,688	68,096
Serpent	LP	3	0	18,208	177,248
Stag	LP	1	193,824	16,896	123,456
Stag	LP	2	36,416	50,112	137,920
Stag	LP	3	35,328	16,832	72,032
Turner	LP	1	0	20,032	53,888
Turner	LP	2	0	32,608	68,896
Turner	LP	3	0	8,192	53,984
Mean Abundance	**LP**		**32,856**	**23,787**	**103,459**
Standard error			15830	3432	14061
Coon	MP	1	6,560	19,392	175,168
Coon	MP	2	12,768	10,240	111,520
Coon	MP	3	10,208	18,496	274,016
Gyrinid	MP	1	17,504	2,752	10,112
Gyrinid	MP	2	18,752	20,896	5,152
Gyrinid	MP	3	4,672	9,664	16,224
McCoy	MP	1	10,560	22,912	37,024
McCoy	MP	2	10,464	50,112	57,536
McCoy	MP	3	37,312	43,936	10,528
Mean Abundance	MP		**14,311**	**22,044**	**77,476**
Standard error			3249	5209	31039

Table 5-12. Total macroinvertebrate biomass in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Values are mg dry mass (DM) of total macroinvertebrates/m^2.

Site	Pool	Core	July 2001	September 2001	April 2002
Jim Crow	LP	1	4,057	2,236	3,967
Jim Crow	LP	2	3,001	4,104	4,862
Jim Crow	LP	3	17,674	1,193	3,048
Serpent	LP	1	0	2,726	4,383
Serpent	LP	2	0	2,740	2,383
Serpent	LP	3	0	863	5,295
Stag	LP	1	2,003	285	10,219
Stag	LP	2	2,138	1,288	7,204
Stag	LP	3	1,834	1,247	3,196
Turner	LP	1	0	1,297	3,752
Turner	LP	2	0	2,866	6,370
Turner	LP	3	0	431	5,730
Mean Biomass	LP		**2,559**	**1,773**	**5,034**
Standard error			1433	333	625
Coon	MP	1	274	804	3,702
Coon	MP	2	651	335	1,567
Coon	MP	3	1,196	1,927	3,892
Gyrinid	MP	1	1,123	414	1,464
Gyrinid	MP	2	1,123	2,787	76
Gyrinid	MP	3	363	713	1,426
McCoy	MP	1	1,107	477	5,364
McCoy	MP	2	2,087	2,034	3,549
McCoy	MP	3	2,760	8,554	1,990
Mean Biomass	**MP**		**1,187**	**2,005**	**2,559**
Standard error			266	868	551

Table 5-13. Oligochaeta abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent abundance of oligocheates (no. / m^2).

Site	Pool	Core	July 2001	September 2001	**April 2002**
Jim Crow	LP	1	8,768	34,560	138,464
Jim Crow	LP	2	12,320	10,848	41,984
Jim Crow	LP	3	16,480	3,584	70,560
Serpent	LP	1	0	18,528	67,808
Serpent	LP	2	0	20,416	36,288
Serpent	LP	3	0	11,552	133,536
Stag	LP	1	11,392	12,128	70,432
Stag	LP	2	6,848	32,544	95,264
Stag	LP	3	6,688	12,928	44,896
Turner	LP	1	0	14,464	49,376
Turner	LP	2	0	31,680	61,568
Turner	LP	3	0	6,080	46,304
Mean Abundance	LP		**5,208**	**17,443**	**71,373**
Standard error			1732	3003	9907
Coon	MP	1	2,944	17,120	118,016
Coon	MP	2	7,808	5,600	74,208
Coon	MP	3	8,640	6,720	244,224
Gyrinid	MP	1	1,600	1,888	6,304
Gyrinid	MP	2	3,616	13,344	2,560
Gyrinid	MP	3	1,792	8,512	13,408
McCoy	MP	1	5,280	16,000	34,944
McCoy	MP	2	7,456	36,800	39,840
McCoy	MP	3	7,968	32,384	8,288
Mean Abundance	**MP**		**5,234**	**15,374**	**60,199**
Standard error			940	4006	26238

Table 5-14. Oligochaeta biomass in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent dry weight of oligochaetes (mg DM/m^2) derived from length-mass equations.

Site	Pool	Core	July 2001	**September 2001**	**April 2002**
Jim Crow	LP	1	161	832	2,409
Jim Crow	LP	2	185	132	2,101
Jim Crow	LP	3	125	56	1,954
Serpent	LP	1	0	1,178	3,066
Serpent	LP	2	0	1,371	1,656
Serpent	LP	3	0	656	4,826
Stag	LP	1	477	131	2,750
Stag	LP	2	327	978	2,328
Stag	LP	3	265	1,011	1,227
Turner	LP	1	0	404	3,316
Turner	LP	2	0	1,375	4,841
Turner	LP	3	0	228	5,243
Mean Biomass	LP		**128**	**696**	**2,977**
			46	143	385
Coon	MP	1	217	244	2,843
Coon	MP	2	376	280	1,209
Coon	MP	3	1,104	1,180	3,366
Gyrinid	MP	1	39	224	254
Gyrinid	MP	2	1,033	796	30
Gyrinid	MP	3	303	410	1,411
McCoy	MP	1	972	355	5,352
McCoy	MP	2	1,331	1,344	3,402
McCoy	MP	3	2,591	817	1,670
Mean Biomass	**MP**		**885**	**628**	**2,171**
			262	141	572

Table 5-15. Chironomidae abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent abundance of Chironomidae individuals (no. / m^2).

Site	Pool	Core	July 2001	**September 2001**	**April 2002**
Jim Crow	LP	1	800	1,664	6,528
Jim Crow	LP	2	736	1,408	7,808
Jim Crow	LP	3	1120	320	1,024
Serpent	LP	1	0	5,824	16,480
Serpent	LP	2	0	1,120	11,200
Serpent	LP	3	0	6,528	16,832
Stag	LP	1	32	1,632	5,792
Stag	LP	2	672	1,344	12,992
Stag	LP	3	1,472	1,664	9,408
Turner	LP	1	0	1,920	2,400
Turner	LP	2	0	0	5,120
Turner	LP	3	0	1,664	3,392
Mean Abundance	LP		**403**	**2091**	**8,248**
Standard error			153	577	1515
Coon	MP	1	1,696	736	32,096
Coon	MP	2	2,560	1,984	4,192
Coon	MP	3	512	4,320	8,576
Gyrinid	MP	1	512	224	768
Gyrinid	MP	2	2,720	736	416
Gyrinid	MP	3	544	160	128
McCoy	MP	1	1,856	2,080	0
McCoy	MP	2	544	1,696	0
McCoy	MP	3	6,048	1,248	0
Mean Abundance	**MP**		**1,888**	**1,465**	**5,131**
Standard error			600	428	3506

Table 5-16. Chironomidae biomass in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent dry weight of Chironomidae individuals (mg DM / m^2) derived from length-mass equations.

Site	Pool	Core	July 2001	September 2001	April 2002
Jim Crow	LP	1	89	53	205
Jim Crow	LP	2	63	17	1,710
Jim Crow	LP	3	28	7	444
Serpent	LP	1	0	143	999
Serpent	LP	2	0	31	646
Serpent	LP	3	0	101	389
Stag	LP	1	191	39	5,764
Stag	LP	2	20	133	2,949
Stag	LP	3	382	39	302
Turner	LP	1	0	34	434
Turner	LP	2	0	0	71
Turner	LP	3	0	195	451
Mean Biomass	LP		**64**	**66**	**1,197**
Standard error			33	18	476
Coon	MP	1	17	46	766
Coon	MP	2	22	28	129
Coon	MP	3	7	102	270
Gyrinid	MP	1	4	14	82
Gyrinid	MP	2	33	49	11
Gyrinid	MP	3	10	4	1
McCoy	MP	1	15	67	0
McCoy	MP	2	51	26	0
McCoy	MP	3	36	72	0
Mean Biomass	**MP**		**22**	**45**	**140**
Standard error			5	10	84

Table 5-17. Community metrics in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent mean +/-(S.E.).

	Pool	July 2001		September 2001		April 2002	
Richness	LP	7.00	(2.16)	10.92	(0.76)	14.00	(1.41)
	MP	9.33	(0.58)	10.89	(1.18)	9.22	(1.21)
Shannon	LP	0.67	(0.23)	1.00	(0.12)	1.05	(0.11)
	MP	1.40	(0.12)	0.99	(0.12)	0.85	(0.11)
Dominance	LP	0.60	(0.12)	0.78	(0.08)	0.55	(0.08)
	MP	0.70	(0.06)	0.62	(0.12)	0.68	(0.09)

Table 5-18. Zooplankton abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent abundance of zooplankton (no. / L).

Site	Pool	July 2001	September 2001	April 2002
Jim Crow	LP	0.00	0.18	5.99
Serpent Slough	LP	0.00	0.05	0.39
Stag Slough	LP	0.02	0.59	0.95
Turner Tip	LP	0.00	0.28	0.37
Mean Abundance (no. / L)	**LP**	**0.00**	**0.28**	**1.92**
Standard error		0.00	0.07	0.79
Coon Slough	MP	0.08	1.60	2.93
Gyrinid Point	MP	0.15	0.18	0.52
McCoy Slough	MP	0.11	0.26	1.24
Mean Abundance (no. / L)	**MP**	**0.11**	**0.68**	**1.57**
Standard error		0.03	0.31	0.58

Table 5-19. Zooplankton abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Numbers represent abundance of zooplankton (no. / L).

	Lower Pool			Midpool		
Taxa	July 2001	September 2001	April 2002	July 2001	September 2001	April 2002
Copepoda						
Cyclopoida	0.00	1.42	2.92	0.35	0.37	1.92
Calanoida	0.01	0.26	0.04	0.01	0.42	0.01
Harpacticoida	0.00	0.01	0.73	0.00	0.00	0.05
Nauplius stage	0.01	1.32	19.31	0.76	7.10	16.70
Cladocera						
Bosminidae	0.00	0.00	0.02	0.03	0.05	0.06
Daphnidae	0.00	0.22	0.00	0.01	0.00	0.01
Chydoridae	0.00	0.00	0.00	0.00	0.19	0.01
Macrothricidae	0.00	0.00	0.07	0.00	0.01	0.02
Moinidae	0.01	0.01	0.00	0.15	0.00	0.00
Polyphemidae	0.00	0.00	0.00	0.00	0.00	0.00
Sididae	0.00	0.07	0.00	0.05	0.01	0.00
Ostracoda	0.02	0.00	0.00	0.00	0.00	0.00
Mean Abundance (no. / L)	**0.00**	**0.28**	**1.92**	**0.11**	**0.68**	**1.57**
Standard error	0.00	0.07	0.79	0.03	0.31	0.58

Figure 5-1. Benthic organic matter in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n=5 of each in 2000, n=6 of each in 2001). Bars represent mean organic matter (g ash-free dry mass [AFDM]/m^2) +/- 1 standard error for total, coarse (CPOM), and fine (FPOM) particulate fractions. Different letters over values indicate significant differences between total organic matter (p=0.009) and CPOM (p=0.004) during 2000. Different letters over 2001 Total organic matter (p=0.125) and CPOM values indicate a marginally significant difference (p<0.087).

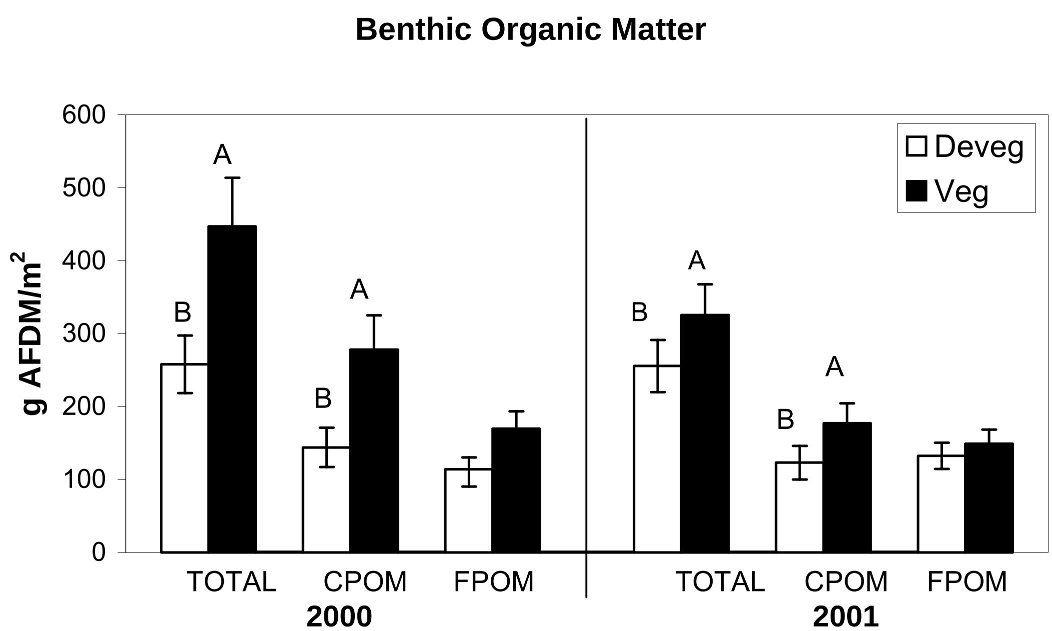

Figure 5-2. Total abundance of macroinvertebrates in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n=5 of each in 2000, n=6 of each in 2001). Bars show mean total abundance (number of individuals x 1000/m^2) +/- 1 standard error.

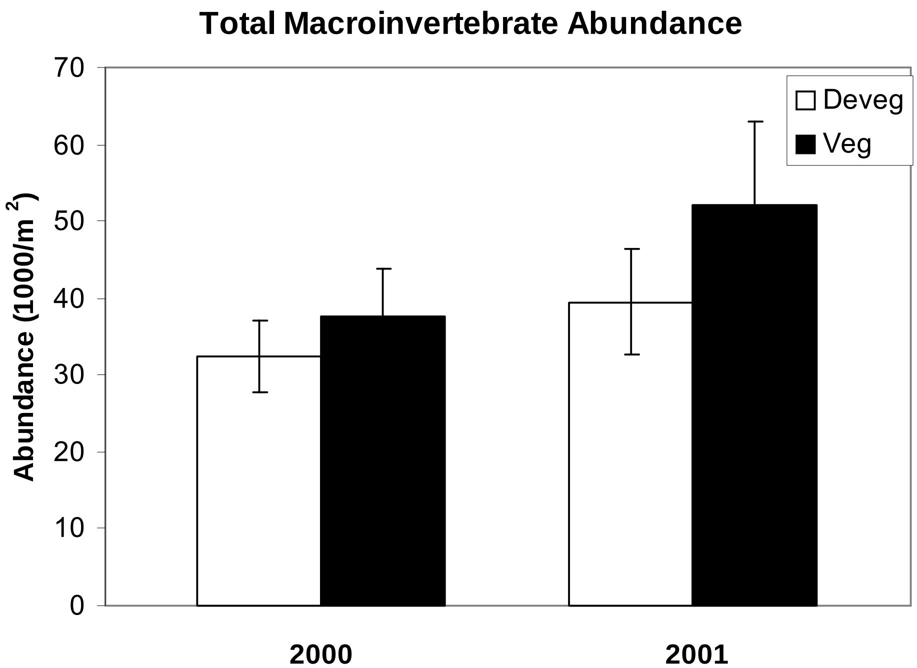

Figure 5-3. Total biomass of macroinvertebrates in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n=5 of each in 2000, n=6 of each in 2001). Bars represent mean total invertebrate dry mass (DM) +/- 1 standard error. Different letters above 2001 values indicate a significant difference between treatments (p=0.005).

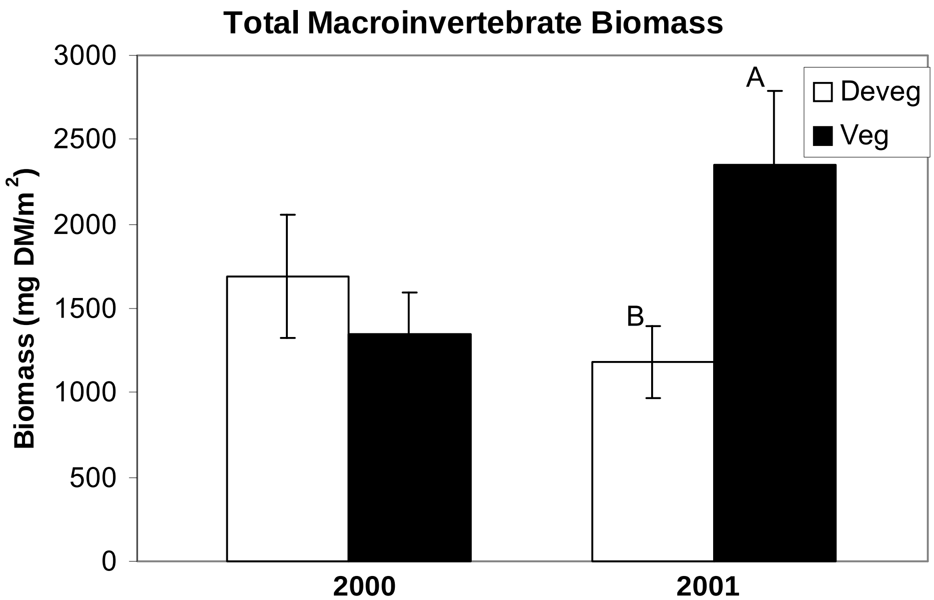

Figure 5-4. Oligochaeta abundance in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n=5 of each in 2000, n=6 of each in 2001). Bars represent mean Oligochaeta abundance (number of individuals x 1000/m2) +/- 1 standard error**.**

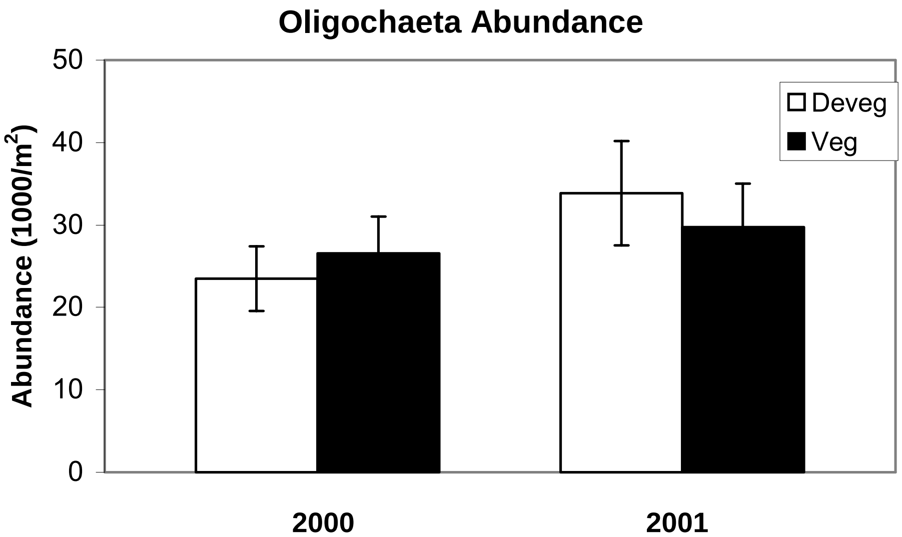

Figure 5-5. Oligochaeta biomass in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n=5 of each in 2000, n=6 of each in 2001). Bars represent mean Oligochaeta dry mass (DM) +/- 1 standard error. Different letters over 2001 values indicate there was a significant difference (p=0.009) between treatments.

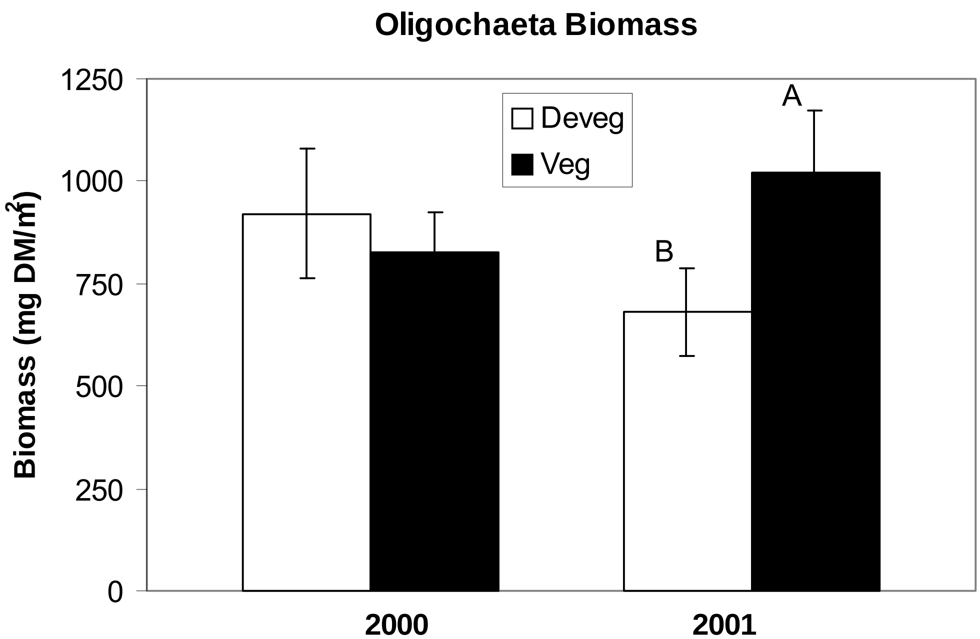

Figure 5-6. Chironomidae abundance in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n=5 of each in 2000, n=6 of each in 2001). Bars represent mean Chironomidae abundance (number of individuals x $1000/m^2$) +/- 1 standard error. Different letters over 2001 values indicate there was a significant difference (p=0.003) between treatments.

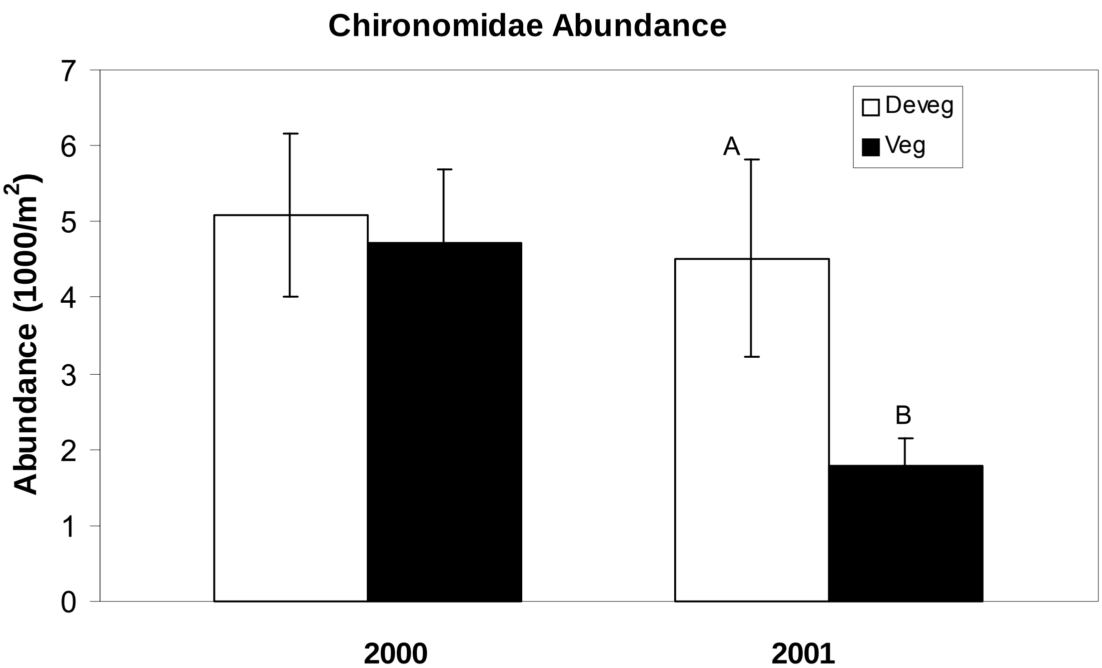

Figure 5-7. Chironomidae biomass in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n=5 of each in 2000, n=6 of each in 2001). Bars represent mean Chironomidae biomass (mg DM/m2) +/- std. error. Different letters over 2001 values indicate there was a significant difference (p=0.013) between treatments.

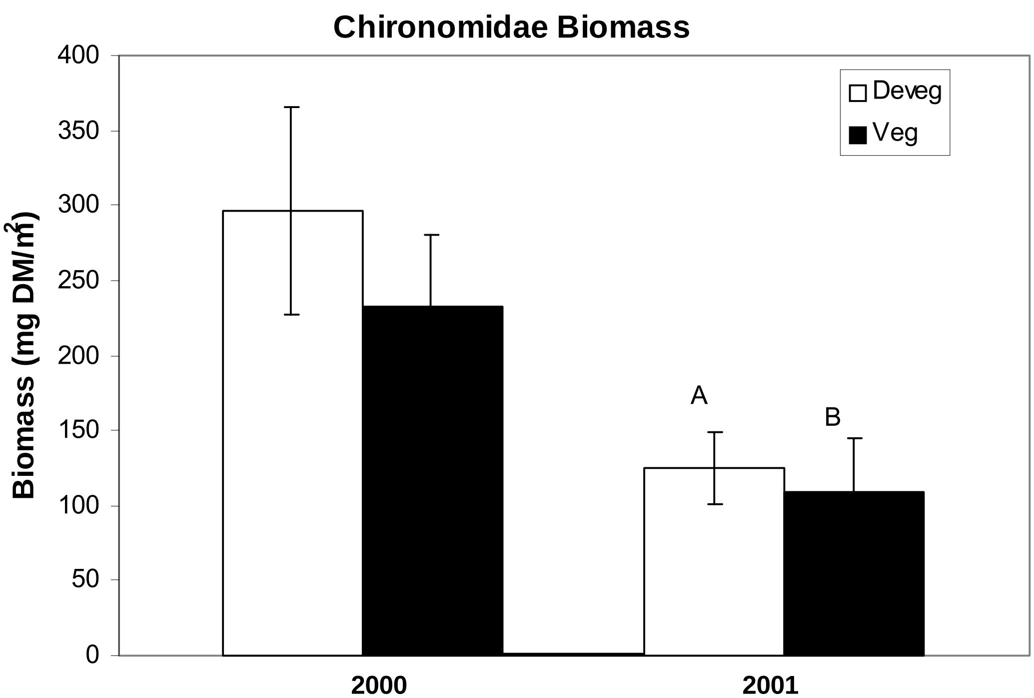

Figure 5-8. Zooplankton abundance in vegetated (veg) and devegetated (deveg) sites in Pool 25 of the Mississippi River (n= 4 of each in 1999, n=5 of each in 2000, n=6 of each in 2001). Bars represent mean zooplankton abundance (number of individuals/L) +/- 1 standard error. Different letters over 1999 indicate a significant difference between treatments (p=0.043).

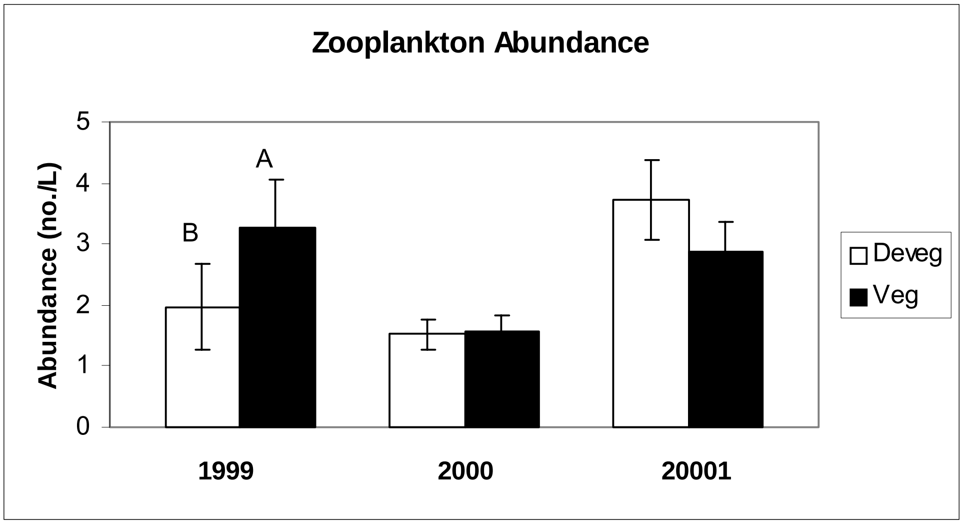

Figure 5-9. Benthic organic matter in midpool (MP, n=3) and lower pool (LP, n=4) sites during 2001 and 2002. Values are ash-free dry mass (AFDM)/m^2 +/- std. error of total, coarse (CPOM) and fine (FPOM) particulate organic matter.

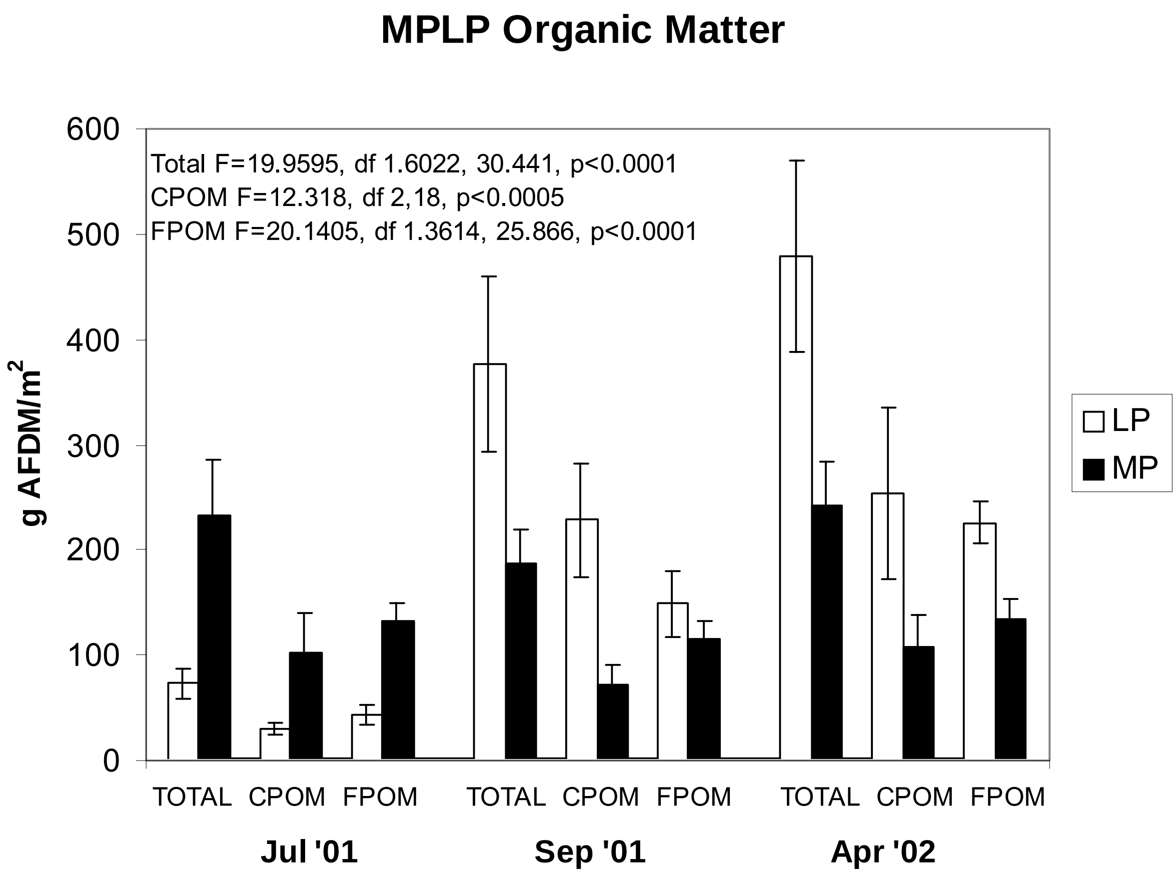

Figure 5-10. Total macroinvertebrate abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Bars represent abundance of Macroinvertebrates x 1000/m^2 +/- 1 standard error.

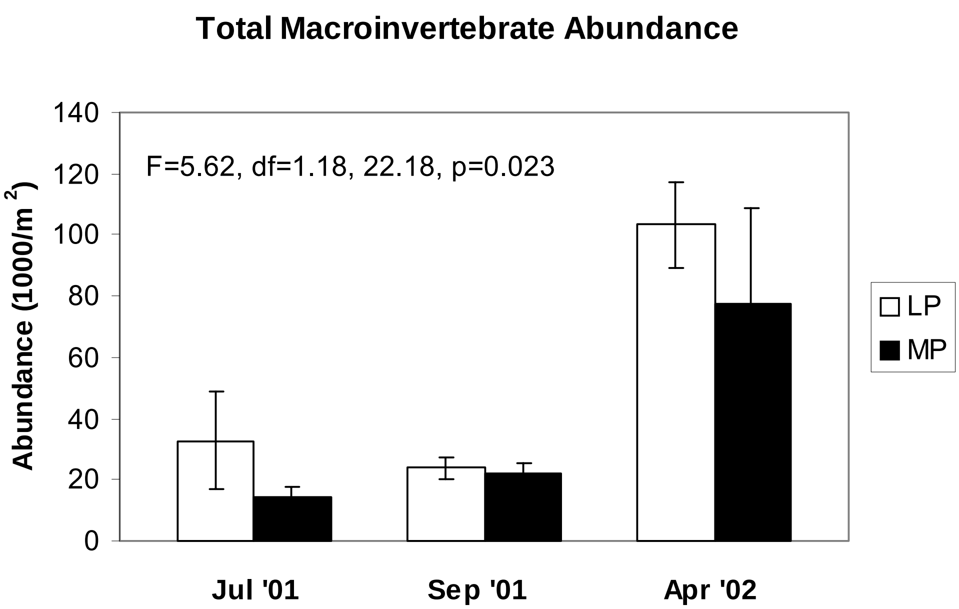

Figure 5-11. Total macroinvertebrate biomass in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Bars represent mg dry mass (DM) of total Macroinvertebrates x 100/m^2 +/- 1 standard error.

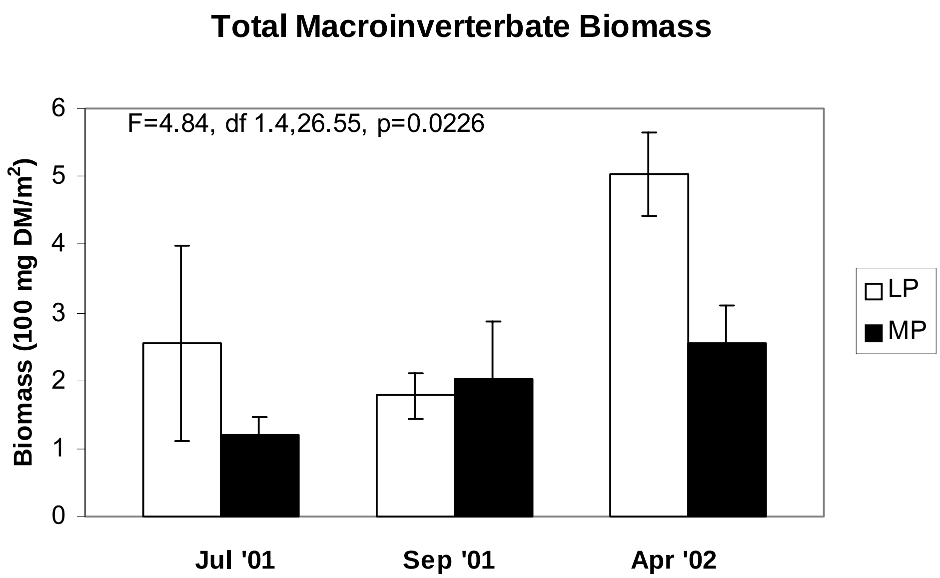

Figure 5-12. Oligochaeta abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Bars represent Oligochaeta abundance x 1000/m^2 +/- 1 standard error.

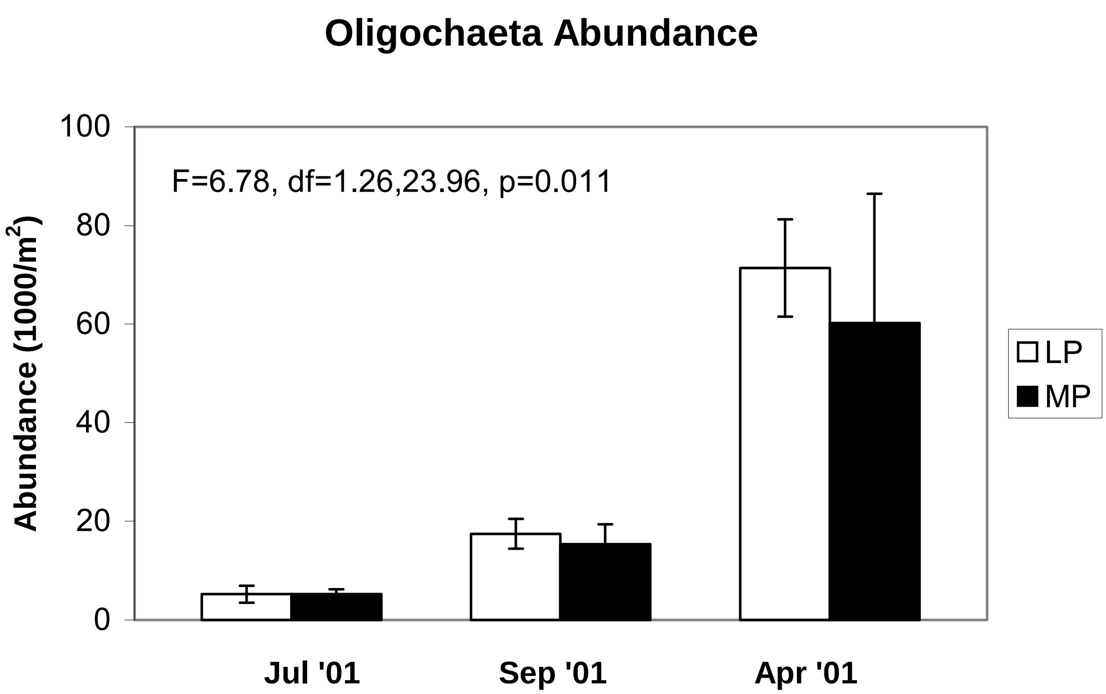

Figure 5-13. Oligochaeta biomass in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Values are mg dry mass (DM) Oligochaeta x 100/m^2 +/- 1 standard error.

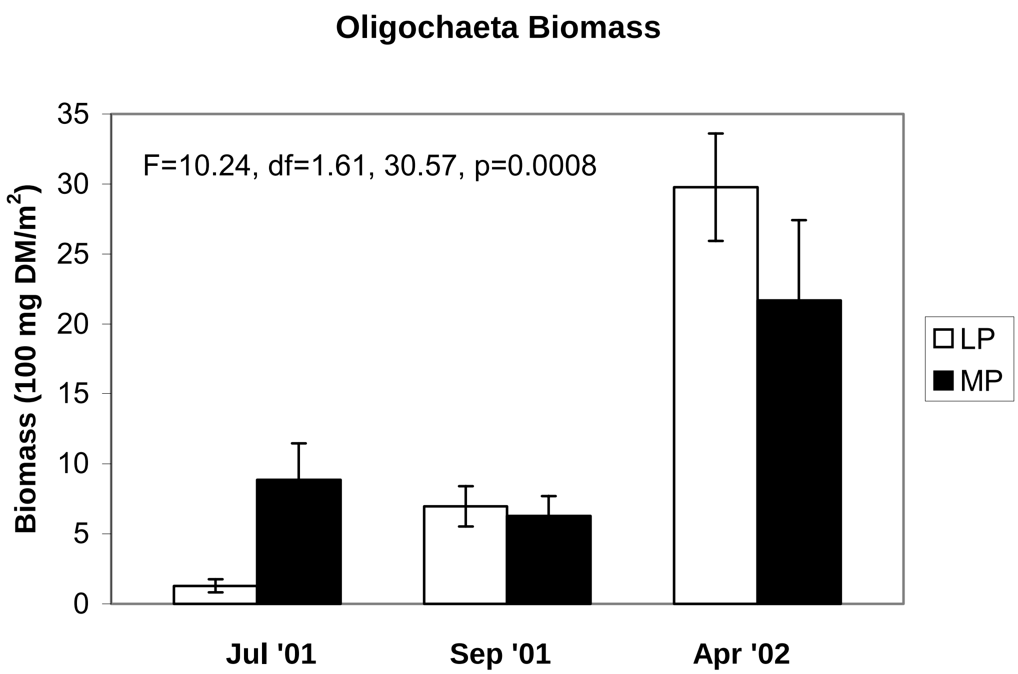

Figure 5-14. Chironomidae abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Bars represent Chironomidae abundance x $1000/m^2$ +/- 1 standard error.

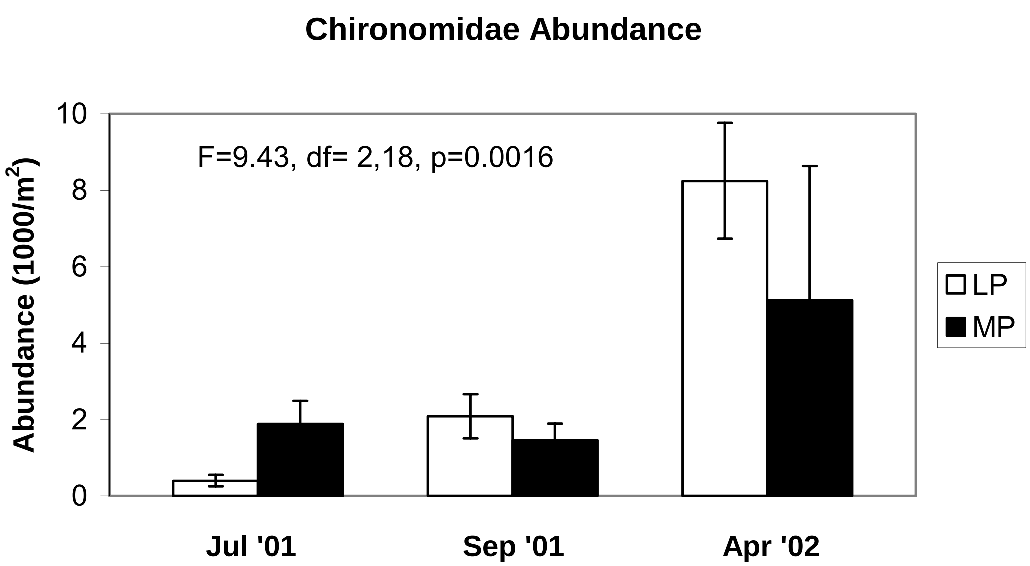

Figure 5-15. Chironomidae biomass in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Values are mg dry mass (DM) Chironomidae x 100/m^2 +/- 1 standard error.

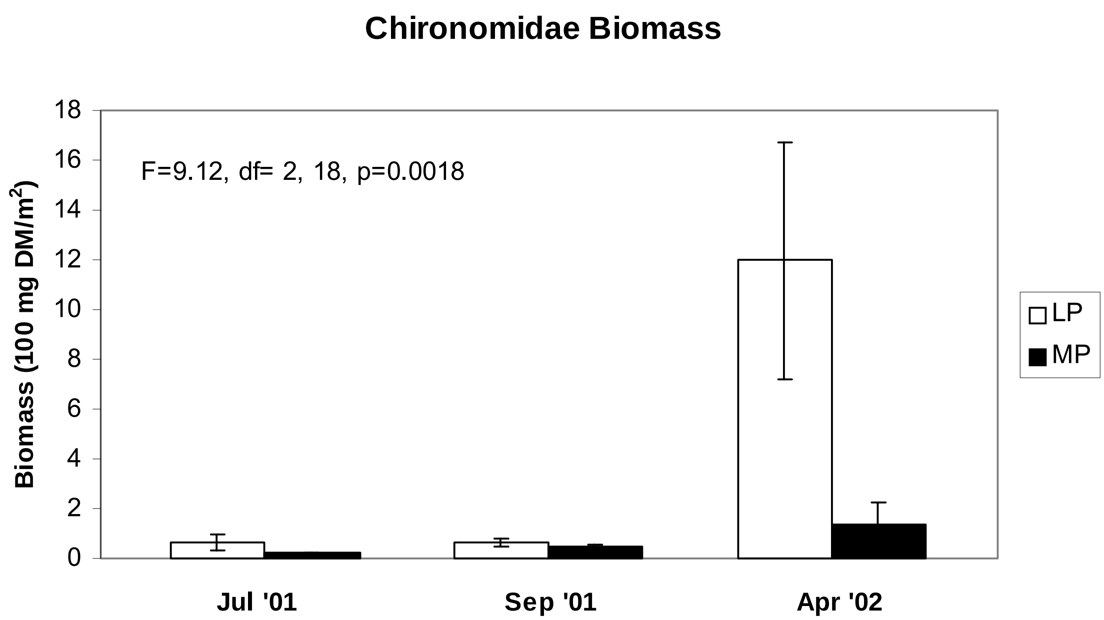

Figure 5-16. Zooplankton abundance in midpool (MP, n=3) and lower pool (LP, n=4) sites in Mississippi River Pool 25 during 2001 and 2002. Bars represent zooplankton no./L +/- 1 standard error.